DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY FOR INSTREAM FLOW ASSESSMENTS, AND SUPPORTING RESEARCH ON THE EFFECTS OF DIFFERENT MAGNITUDE FLOWS ON RIVERINE ECOSYSTEMS

R.E. THARME AND J.M. KING

Freshwater Research Unit
Zoology Department
University of Cape Town
Rondebosch, 7700

With contributions from D. Louw, J. Hollands, J. Cambray, C. Bruwer, D. Impson, M. de Kock, R. Simpson, A. Emery, P. Kuun and M. de Villiers.

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EXECUTIVE SUMMARY

E1. INTRODUCTION

South African rivers are deteriorating in condition, to a large extent due to general catchment degradation and to increasing manipulation of flow regimes to provide water for a burgeoning population. In this respect, the country mirrors a worldwide trend. The emerging science of instream flow, or environmental flow, assessments is one response to this trend. Such assessments provide recommendations on flow regimes for rivers targeted for water-resource developments. The recommended flow regimes are designed to contain, mitigate, or in some cases partially reverse, the impacts of such developments on river ecosystems. Such guidance is necessary as, until recently, virtually all ecological and social impacts have been externalised in the cost-benefit analyses of water-resource developments. The costs to the nation of poorly-functioning rivers is nevertheless undoubtedly exceedingly high, and many of the most substantial impacts are felt by those least able to withstand them – rural populations dependent on riverine ecosystems for subsistence.

In the late 1980s, and with strong support from the national Department of Water Affairs and Forestry (DWAF) and the Water Research Commission (WRC), the national body of river scientists began to develop expertise in the science of instream flow assessments. Much of this early development is documented in a WRC report (King and Tharme 1994), that described the learning and assessment of internationally-available instream flow methodologies and recommendations on ones for use in South Africa. In that report it was concluded that the country needed to develop a local instream-flow methodology that would reflect the realities of water-resource development in South Africa. Quantitative recommendations on flows for river maintenance needed to be produced quickly and would therefore probably have to be based largely on available data and understanding. All flows required for river maintenance would have to be explicitly justified, due to the strong competition for water in this arid land. By the completion of the above report, such a local methodology had begun development, the Building Block Methodology (BBM).

The project reported on here was designed to continue development of South African expertise in instream flow assessments. Its main objectives were to continue development of the BBM and, in parallel, to initiate research designed either to test assumptions within the BBM or to produce supporting scientific data to improve its application.

E2. PROJECT AIMS

The project ran for the three calendar years, from 1993-1995. Its aims and objectives were laid out in the original contract between the University of Cape Town and the WRC and later refined by the Steering Committee for the project. They are summarised below.

The overall aims of the project were to enhance the ability of river scientists to advise on sustainable development of the water resources of the country's rivers by:

- continuing the development of scientifically acceptable methodologies for assessing the water-quantity requirements of riverine ecosystems;

- increasing understanding of the effects of different magnitudes of flow on riverine biotas, through focused research on South African rivers and, in doing so, enhancing the efficacy of such instream flow methodologies.

The objectives were to:

- continue development of the BBM for use in instream flow assessments;

- improve estimates of water-quantity requirements made using the BBM, through focused research on South African rivers;

- concentrate such research on the flow-related requirements of the instream biota, acknowledging that flow-related aspects of river channels and banks, floodplains and wetlands would be addressed by other research groups in the country.

It was recognised that research on only a few of the potentially ecologically important flows could be addressed within a three-year project.

In the following report, a general introduction is given in Chapter 1 and development of the BBM is described in Chapters 2-6. This is followed by reports on original research into a number of different flow components, namely, dry-season low flows, dry-season freshes and wet-season floods (Chapters 7-15). These kinds of flows are routinely incorporated in some form into the recommended flow regimes emerging from instream flow assessments. The research projects were designed to enhance understanding of the ecological significance of these flow components, so that there would be greater certainty of them achieving the desired results when included in a recommended flow regime. Chapter 16 provides a general conclusion and recommendations for future research.

E3. THE BUILDING BLOCK METHODOLOGY

The need for specially designed flow regimes for maintenance of regulated rivers is now commonly recognised. Chapter 2 provides a simple and generalised explanation of the functioning of undisturbed river systems and the effects that flow manipulations have on them. This explanation has been routinely included in all DWAF reports on instream-flow workshops. The essential message is that rivers are dynamic systems, with the physical, chemical and biological characteristics of the channel, riparian and floodplain areas, as well as the related estuarine and nearshore ecosystems, endlessly changing in adjustment to prevailing environmental conditions. In undisturbed rivers, such adjustments tend to be around some dynamic equilibrium, with this equilibrium itself shifting over very long time spans as the system ages.

Manipulations of virgin flow regimes represent unnatural disturbances to rivers, with these disturbances increasing in severity the further the flow regime is altered from what lies within the realm of "normal" for a river. Responses of the river become more extreme as disturbance increases, and the whole system may shift toward a new equilibrium (such as smaller, sandier, less perennial, and so on) or endlessly swing away from and back towards its old equilibrium state. Many responses of a river are possible, with most of them fairly unpredictable, at least in detail. The costs to the country of such responses are undoubtedly enormous but largely non-quantified. It makes sense to minimise them, however, when targeting a river for water-resource development. This can be done by designing a modified flow regime that will help prevent or mitigate expected impacts.

Early investigations into the kinds of flows required for maintenance of regulated rivers centred on a questionnaire survey and a workshop at a national conference (Chapter 2). Specialists from many river-related disciplines did not know what kinds of flow regimes were needed for river maintenance. However, one group – the ecologists – did, and produced a remarkably consistent message. They specified that:

- the modified flow regime should mimic the virgin one, so that the natural timing of different kinds of flows was adhered to;
- the river's natural perenniality or non-perenniality should be retained;
- most water should be harvested from the river during the wet months and least during the dry months;
- the seasonal pattern of higher baseflows in the wet season than in the dry season should be retained;
- floods should be present during the natural wet season;
- flood durations could be shortened, but within limits that allowed flood-related activities, such as fish spawning in flooded areas and movement of larval fish back into the channel, to reach completion;
- whole floods could be removed but others should be left at full magnitude, rather than having all the naturally occurring floods but at lower magnitudes;
- the first (or one of the first) flood of the wet season should be retained.

The reasons for these requirements are given in Chapter 2.

Building upon this knowledge, the conceptual basis of the BBM was derived (Chapter 3). It is that some flow characteristics of the total flow regime of any river are more important for maintenance of that river ecosystem, than others. These flows can be identified for any one river, and described in terms of their timing, duration, frequency and magnitude. Where a water-resource development is planned, the identified flow components (the so-called building blocks) can be combined to describe a recommended modified flow regime that is specific for the river of concern. The 'blocks' of flow usually fall into the following categories: dry-season baseflows (low flows); wet-season baseflows (low flows), wet-season floods, dry-season freshes and dry-season subsurface flows. The minimum volume of water required for each 'block' is described, thereby defining a minimum flow requirement for maintenance of the river.

Based on this concept, development of the BBM began in 1991. Starting as little more than a basic set of ideas and actions, it has become a well-structured methodology that has been contributed to by a large number of the country's senior river scientists (Chapters 4-6). It differs from more traditional instream flow methodologies in that it focuses on maintenance of the "health" of the complete river system rather than on the preservation of specific species of commercial or conservation importance. In a pragmatic way, it links in very strongly with the engineering, water-management and decision-making aspects of water resources.

In doing so, it provides quantitative answers on flow regimes for river maintenance in a form that can be used by planners and design engineers.

Using the BBM, the instream flow requirement (IFR) for the river is described by a team of experienced river scientists. These specialists are from a range of relevant fields of expertise, but always include specialists in aquatic ecology (fish, invertebrates and riparian vegetation), fluvial geomorphology, aquatic chemistry, hydrology, hydraulics and sociology. In a preliminary data-gathering phase, set procedures are followed to gather essential data for the workshop. For instance, the sociologist determines the dependence of rural communities on the river ecosystem for subsistence, and identifies the resources used, at a resolution that can be taken up for consideration by, say, the botanist or fish ecologist. The botanist records the flow-related distribution of riparian trees and in-channel plants, and how critical life-cycle events such as the timing of seed-set or flowering may be linked to flow. The hydraulic modeller establishes stage-discharge relationships at set points along the river, in order to be able to describe hydraulic conditions at any flow and to convert to discharges the required depth and velocity conditions described by the biologists. The hydrologist provides back-up information on the hydrological nature of the river, as detailed in Chapter 5. Several other specialists will have similarly defined roles, as described in Chapters 3 and 4.

With the data-gathering phase complete, each specialist provides a review of relevant literature and a summary of collected data as a chapter in a summary document. Together with the engineers and water managers linked to the project they then attend the IFR workshop, where the recommended flow regime is described using a structured process (Chapter 4). This flow regime (or IFR) is designed to sustain the river of concern in some pre-determined Desired State (or state of "health", DS). The workshop takes place in DWAF's Pre-feasibility phase, with the workshop output being used in forward planning through the Feasibility and further phases. Further development of the BBM through Feasibility, and into Design, Construction and Operation is in the early stages (Chapter 6).

A manual for using the BBM is being written by senior scientists involved in its creation, for production in 1998. This signals the end of phase one of its development. All components of the BBM should be enhanced by further focused and controlled development, but three important components perhaps stand out as least developed and in most urgent need of attention.

- *Use of the concept of Desired State.* Defining DS is not an easy task, as there is no obvious procedure, at the catchment level, for deciding who should decide the future state of any river and how. It requires, *inter alia,* a comprehensive public participation process run by specialists in the field. There needs to be strong guidance from experienced river scientists, however, as the DS decided upon must be described in terms that scientists can measure and monitor. In a recent development, the DS has come to be seen by some as reflecting the final decision on water allocation from a planned project rather than the objective to be addressed up front at an IFR workshop. The implications around this approach are beyond the scope of this report.
- *Physical-habitat modelling.* Tharme (1996, 1997c) recognised a three-tier hierarchy of instream flow methodologies based on effort involved and resolution of output. Holistic approaches such as the BBM operate at the second tier. The highest-resolution tier links an holistic approach with an habitat-modelling activity, whereby the changing availability of different physical habitats with changing discharge can be modelled. This modelling facility needs developing in order to improve on the two-dimensional information provided by stage-discharge curves. Whether or not such a modelling facility

is available, steps forward can be taken by introducing a habitat-mapping approach to IFR site description. Such an approach is presently being developed within a related WRC project.

- *Water-quality.* Most emphasis in the BBM is on water-quantity requirements for the river ecosystem, and the water-quality component is weak. Again, a modelling component needs developing to inform on probable future chemical conditions in the river. This would need to be linked with range-of-tolerance data for the biota, in order to predict the biological affects of the water-resource development.

Vital to the successful development of the BBM, were three sets of circumstances.

- The recognition by DWAF that instream flow assessments should be done for all proposed water-resource projects that affected river flow, and the strong support of the department for development of an appropriate methodology. This support included approval for, and funding of, almost all of the IFR workshops held to date, and approval for appropriate specialist staff in DWAF to take leading roles in developing aspects of the methodology.
- The funding and other support provided by the WRC, that allowed the two authors of this report to develop expertise in instream flow assessments and, together with other WRC-funded river scientists, to play major roles in the development of the BBM.
- The willingness of the national body of river scientists to support development of the BBM. This they did by producing reports for IFR workshops, by being willing to make inputs at the workshops to the limits of their knowledge, and by many of them producing new ideas on how the methodology could further be improved. Their enormous contribution cannot be too highly rated.

By the end of the present project, the BBM had attracted international attention, and outputs from the IFR workshops where it was applied were being used in DWAF's planning and design stages for specific developments. In the only two cases to date where proposed projects for which an IFR recommendation were made have reached the stage of dam construction, the water required for river maintenance, as described using the BBM, has played a prominent role. In one case, in the search for a solution between the conflicting water demands of offstream users and for river maintenance, the IFR provided the negotiating point for the local Nature Conservation body. With a compromise agreed upon, DWAF issued the private developers with a permit for dam construction that specified releases for maintenance of the downstream river. In the second case, a DWAF-driven project, the first Government White Paper has appeared which specifies an amount of water for river maintenance (DWAF 1997a).

E4. RESEARCH LINKED TO THE CONCEPTUAL BASIS OF THE BUILDING BLOCK METHODOLOGY

From the outset it was acknowledged by those developing the BBM that for the methodology to be able to cater for the water managers' needs for rapid answers on IFRs, it would have to be based on available or easily-acquired data and on expert opinion. This being the case, it was also strongly felt that research to improve the knowledge base had to be incorporated into the overall activities related to its use. Hence, within this project a programme of research was designed that would complement the development of the BBM.

The principal objective of the research programme was to improve understanding of the relationships between the magnitude, duration, timing, and frequency of different categories of flow and the structure and functioning of various components of riverine ecosystems. Individual research projects differed in length and intensity, from an intensive three-year ecological project, through two year-long engineering projects, to four short projects at the level of a B. Sc. (Honours) or fourth year B. Engineering degree. For each flow component studied, a combination of ecological and engineering projects was undertaken, to encourage cross-fertilisation of ideas and techniques, and multidisciplinary training. Chapters 7 to 12 deal with annual baseflows, specifically dry-season low flows; Chapters 13 and 14 with dry-season freshes; and Chapter 15 with wet-season floods. Four short student projects linked to the main project are outlined in Appendices 1.1-1.4.

E5. ASSESSMENT OF THE ANNUAL BASEFLOW REGIME OF WESTERN CAPE RIVERS, WITH PARTICULAR REFERENCE TO DRY-SEASON LOW FLOWS

E5.1 Identification of ecologically relevant low flows on the basis of assessments of the ecological effects of natural and unnatural dry-season baseflow regimes (Chapters 7-11)

Introduction to the low-flow study

Water-resource developments are responsible for a number of physical disturbances to riverine ecosystems, perhaps the most critical of which is a change in the natural flow regime. Prolonged reductions in annual baseflows have been identified as potentially detrimental to all components of riverine ecosystems, at a number of spatial and temporal scales. Within instream flow assessments, the extent to which the flow regime of a river can be altered from the natural condition, while still maintaining the integrity or an acceptable level of degradation of the riverine ecosystem is addressed. This requires a thorough knowledge of the flow-related conditions governing riverine ecosystem structure and functioning and of the potential impacts of altered flow patterns on such aspects of the system. This knowledge base provides an indication of which flows are perhaps more relevant from an ecological perspective than others, within the river's hydrological regime.

As part of the identification of a modified hydrological regime for a river, using an instream flow methodology like the BBM (Chapters 3-6), a detailed assessment needs to be made of the dry-season baseflow requirements of the river. Baseflow is the flow in the river contributed by the channel and banks at all times of the year, excluding higher flow events generated by surface runoff. It may be divided into dry-season baseflow and wet-season baseflow. Baseflows during the dry season are sometimes referred to as low flows by ecologists. However, the term low flows is used here to represent the range of flows within the dry season baseflow, that may be naturally or unnaturally low in magnitude, as specified.

Dry-season baseflows have numerous vital roles to play in rivers. These include, among others, defining whether the system is naturally perennial or non-perennial, provision of wetted habitat for instream biota that allows their survival during the dry months, provision of appropriate temperatures and chemical conditions as biotic cues, maintenance of communities of marginal and riparian vegetation, and provision of sources of water for consumption by terrestrial and riparian wildlife. However, much of the existing

scientific literature on the effects of natural and unnatural low flows on riverine ecosystems is devoted to phenomenological accounts of the impacts of drought or on research into the effects of drying and the development of intermittent conditions, mostly in temporary rivers. There is currently little research focusing on the effects of either natural or unnatural dry-season low flows, or on cease-to-flow conditions, on perennial rivers.

In this section, the results of a low-flow study (Chapters 7-11) aimed at investigating the effects of different magnitude natural and unnatural dry-season baseflows on rivers are summarised. The work was aimed at identifying low flow events that constitute various degrees of physical disturbance to riverine ecosystems, to enable ecologically relevant definitions of low flows. This definition was expected to be a function of several factors, including the composition of the benthic macroinvertebrate assemblages (BMIs) as the target biota, biotope characteristics, water chemistry and hydrological history. It was anticipated that the results of the study would enable a more structured and scientifically based assessment of the low flow requirements of riverine biota and the possible effects of various low flow scenarios on rivers. This information could be used to further the development of the BBM, or a similar instream flow methodology and, particularly, to contribute to its scientific basis and provide support for its main assumptions.

The activities within the low-flow study were divided into three parts, as follows: a survey of hydrological gauging weirs (Chapter 7); a pilot study (Chapter 8); and an experimental low-flow study comprising the focus of the research (Chapters 9-11).

Survey of hydrological gauging weirs and pilot study

A survey of 33 rivers in the southwestern Cape with hydrological gauging weirs on them was undertaken in January and February 1994, to identify suitable study sites. The survey was required for this low-flow study and for the work reported on in Section E5.2. Site selection was primarily based on criteria aimed at minimising, as far as possible, inter-system biological, geomorphological and hydrological variability. There were no entirely suitable sites on any of the rivers at which flow reductions were already taking place. Hence, a decision was taken to artificially reduce flows at sites with gauging weirs recording near-natural flow conditions, using flow diversion techniques. Of the located 13 river reaches with DWAF gauging weirs recording near-natural flows, only eight sites fulfilled most of the required selection criteria for the low-flow study (Chapter 7).

A pilot study was undertaken to identify the four ecologically most similar of these eight rivers, for the field-based low flow experiment, using data collected on riffle BMI assemblages and water chemistry. The data also provided necessary information on intra-reach biological variability at two locations within each reach. The most ecologically similar group of rivers was the Molenaars (weir H1H018), Du Toits (weir H6H007), Elands (weirs H1H017 and H1H033) and Riviersonderend (weir H6H008) rivers, all of the Breede River system.

The nature of the experimental low-flow study

The low-flow field experiment conducted at the Elands, Molenaars, Du Toits and Riviersonderend study sites, was structured as a spatially replicated extension of a spatial non-replicated Before-After, Control-Impact (BACI) design. That is, there were replicated locations at multiple sites that were subjected to a

low-flow impact, the experimental or impact locations, and locations that were not subjected to any unnatural reduction in flow, which acted as controls. One of the four sites was selected as an overall control for the experimental reduction of flow using temporary diversion weirs.

At each site, two 60 m-long locations were identified. A control location was established upstream of the predetermined flow diversion point, and an impact location was located no more than *c.* 80 m further downstream. The locations were situated in close proximity to a DWAF hydrological gauging weir. The low flow experiment represented almost a full dry season, and extended from 3 December 1994 to 17 May 1995, a duration of 166 days. Independent temporal replication of samples was achieved by sampling both locations, at each of the four sites, twice before experimental, unnatural flow reduction, twice during the impact phase, and twice after the diversions were removed and natural baseflows were reinstated.

A graded series of reductions in discharge magnitude was applied to the three experimental sites during the February to March impact phase, where each reduction represented a proportion of the natural instantaneous baseflow. The Elands River was assigned a 0% reduction in dry-season baseflow, as the control site. Flow reduction at the three experimental sites was achieved by construction of flow diversion weirs, the designs of which were site-specific. For the Riviersonderend and Du Toits rivers, similarly constructed sandbag weirs were erected with several PVC water offtake pipes set within the wall and continuing along the bottom of the river channel. At the Molenaars River, the existing gauging weir was used as the diversion point, and offtake pipes were fitted to a modified scour gate. Approximately 36% of the baseflow was successfully abstracted in the Molenaars impact location, 86% in the Du Toits impact location, and a little more flow, though on average 86%, from the Riviersonderend impact location.

Of the total set of data collected for all eight locations on each of the six sampling trips (Section 9.5) the following subsets were analysed for the present report: 48 standard sets of water chemistry, for 19 physical and chemical variables; 96 sets of discharges; 48 sets of information on hydraulics, including depths, average (0.6-depth) velocities, and biotope proportions, for a riffle and run cross-section in each location; and 144 sets of stone dimensions for stones from which BMIs were collected. Three BMI samples from riffles and runs, for the December, February and April field trips, were examined at the taxonomic level of family. Each sample was divided into three parts representing the stone top, stone bottom and substratum underlying the stone, so a total of 144 samples and 432 individual sample parts were analysed. The additional data collected during the study are examined in a doctoral thesis (Tharme in prep.).

Preliminary findings of the low-flow study

A synopsis of the salient findings of the low flow study to date, specifically of the effects of unnaturally low flows on water chemistry, the character of physical biotopes and the composition of BMI assemblages, are given in this section (see Chapters 10 and 11). The implications of the various results for the development of the low flow component of instream flow methodologies and assessment of the potential impacts of low flows on riverine ecosystems are then summarised. Observed responses to unnaturally low flows were compared with recorded trends in the three components of the riverine ecosystem at natural dry-season baseflows. Such comparisons were found to be essential as, in a natural dry season, the four upper rivers exhibited considerable ranges of variability in all measured characteristics. An understanding of the extent of such variability both within and among rivers allowed responses directly associated with unnaturally low flows to be differentiated from natural changes during summer.

Natural and experimental dry-season baseflows in the context of the historical flow records

The following natural trends were observed in the perennial, seasonal baseflow regime from early summer in December to autumn (May) (Sections 10.2 and 11.2):

- On the basis of field measurements of instantaneous discharge, the summer month of lowest baseflow differed for the four sites. For the Elands site, baseflow was lowest in March to April, the Molenaars site experienced its lowest discharge in early March and February was the month of lowest flow for both the Du Toits and Riviersonderend sites.
- On the basis of average daily discharges from the near-natural historical flow record, the months of lowest and highest baseflow were February and May, respectively, for all four sites.

In order to identify links between the long-term hydrological history of each of the study sites and observed flow data from the present study, the discharge magnitudes represented by various hydrological indices derived from the near-natural, hydrological record first needed to be assessed. Flow percentiles were derived from 1-day FDCs for the months concerned. These and a series of other hydrological indices showed the following main trends in terms of baseflow regime for each of the sites, during the six month period corresponding with the low-flow study:

- For each site, (mean) mean, (median) median, median 7-day consecutive low and (minimum) minimum indices of flow were fairly similar to one another in terms of the corresponding discharges, for the months of December to March. However, these indices were increasingly divergent in April and May. In these latter months, higher flow events occur more frequently, with the transition to a winter rainfall climate.
- For all sites, average daily discharge was the hydrological index most influenced by the occurrence of fresh events, which tended to occur towards the very beginning and end of the dry season.
- For each site, Q_{95} values were highly similar in all months, except December and sometimes May when they were higher.
- The Q_{95} values were not markedly lower than Q_{75} figures, except in December and to a greater extent in May.

Natural instantaneous discharges corresponded with a range of the above hydrological indices, depending on both the month and the site, as follows:

- At the Elands site, recorded instantaneous baseflows approximated the long-term, median 7-day low flows, but were slightly higher than discharges corresponding with them in January and early March. Generally, the measured discharges were in the order of or slightly higher than the 90-95th percentile range of magnitudes.
- At the Molenaars site, measured baseflows were typically at lower than or similar magnitudes to the long-term median 7-day low flows, except in December when they were higher. For February, the instantaneous discharges fell between the median 7-day low flow and the absolute minimum. They approximated the Q_{75} for December to January, decreased to about the Q_{95} in February, and increased again to between the Q_{75} and Q_{95} values from March onwards.
- At the Du Toits site, instantaneous discharges were similar to long-term median flows and median 7-day low flows in December. Observed baseflows from January to April approximated the discharges corresponding with the median index, but were closer to the median 7-day low flow index in May. None of the discharges were as low as Q_{95} values.

- At the Riviersonderend site, observed baseflows were similar to median 7-day low flows in December and February, greater than the long-term median discharge in January and similar to the median discharges from March to May. None of the instantaneous discharges corresponded with Q_{95} values.

The experimental, unnatural reductions in baseflow during February and March (the impact phase of the low-flow study) represented two main hydrological impacts on the three experimental sites, namely a decrease in the magnitude of baseflow during two of the peak months of the dry season, and a duration of the period of flow reduction of about two months. These unnaturally low baseflows corresponded with various calculated long-term hydrological indices of natural flow, in the period February to March, as follows:

- The 36% flow reduction in the Molenaars impact location resulted in baseflows that were of lower magnitude than the long-term, natural Q_{95} values for the river in February and March, and similar in magnitude to the absolute minima on record for these months.
- An 86% reduction in discharge in the Du Toits impact location resulted in baseflows that were significantly lower in magnitude than both the Q_{95} values and absolute minima on record for the river in these two months.
- An 86% reduction in discharge in the Riviersonderend impact location resulted in discharges that were far lower than natural Q_{95} values and just below the absolute minima on record for the river in these two months.

Changes in water chemistry in relation to dry-season baseflows

There were several inherent limitations in the sampling of water chemistry that had implications for the reliability of the results obtained (Section 10.3). Although the focus of sampling was on still-flowing sections of river, during the impact phase of the experiment, in order to characterise the general condition of the river at natural and unnatural baseflows, there were areas of potentially altered water quality within the main channel that were not sampled, especially isolated pools.

The main preliminary findings of the assessment of changes in water chemistry at natural baseflows were as follows:

- Although each of the four rivers exhibited a unique chemical character and natural variability was often high, broad similarities among sites were shown.
- Only a few of the recorded physical and chemical variables, such as temperature, TA and conductivity, among others, showed characteristic increases to peak levels in midsummer (February), and subsequent decreases in concentration with the onset of higher flows in April.
- Differences in values of many chemical variables between summer months were often fairly low, indicating high overall stability during a dry season.
- Several variables showed considerable natural inter-reach variability at some sites, for instance, minimum temperature, TA, phosphate, and potassium.
- The main changes in chemistry appeared to occur at the very end of the dry season, in late March, with the initiation of a natural shift to autumn conditions. However, they were not necessarily associated with large-scale increases in baseflow in April.

Unnaturally low baseflows in February and March had the following implications for the riverine ecosystem in terms of water chemistry (Sections 10.3.2-10.3.7 and 11.3.1):

- An unnatural 36% reduction in peak dry-season baseflow did not appear to markedly alter any of the measured water chemistry variables. Furthermore, although an extremely high reduction in baseflow magnitude of 86% resulted in possible changes in a few variables such as temperature, TA and pH, *inter alia,* the changes were generally not marked relative to the observed natural ranges. Moreover, the increases or decreases in variables were not consistently evident at both sites at which similarly high magnitudes of flow were diverted.
- A clear change in water chemistry, however, was observed for the Riviersonderend impact location. Specifically, an increase in conductivity was recorded in isolated pools that were previously linked within the main channel. Moreover, flushing of the main channel with the reinstatement of natural baseflows, in April, resulted in pulses of highly elevated levels of the correlated variables conductivity and TDS, relative to natural concentrations. These two variables did not show pronounced increases in the still-flowing section of the main channel, during the flow reduction phase of the experiment.
    - The impacts of unnaturally low discharges on the above variables were short-lived, with a recovery to natural values soon after the reinstatement of natural flows in April.
    - The direction of change in conductivity at unnaturally low flows was the same as under the natural baseflow regime at this site, but was exaggerated.
- Several indirect effects of unnaturally low flows on water chemistry were observed, though not quantified, which probably altered water quality in the affected reaches, including:
    - Accumulation and subsequent flushing out of high levels of humic acids leached from desiccated, decomposed instream plant matter, and associated darkening of water colour (Riviersonderend).
    - Proliferation of green filamentous algae in slow- and non-flowing areas (Riviersonderend).
    - Desiccation of epilithic algae on exposed substrata (Du Toits).

Effects of dry-season baseflow regimes on physical biotopes

Natural and unnatural baseflow regimes were shown to have several effects on the hydraulic character and dynamics of biotopes (Sections 10.4 and 11.3.2). In many cases, extreme low flows principally resulted in an exaggeration of the trends recorded in midsummer with natural decreases in baseflow magnitude. The major trends in hydraulic character observed with changes in flow were usually sustained, irrespective of observed morphological differences within and between biotopes at each site, as well as among sites. The two naturally dominant biotopes, examined in detail, were riffles and runs. At natural baseflows, riffles were typically shallower, had a wider range of velocities and depths, higher maximum velocities, greater turbulence, and an increased number of exposed bed elements than runs.

The following effects on biotopes were observed, with an increasing degree of change occurring with decreasing baseflow magnitude, as indicated. In several instances, similar changes occurred, though to a far lesser degree, under conditions of naturally low flow during February and March:

- A loss of wetted width, and hence it can be assumed of wetted perimeter, occurred that tracked natural and unnatural changes in discharge at all sites, although the effects of large changes in discharge magnitude were dampened as a result of cross-section shape.

- At natural and 36% reductions in baseflow, there was no whole-scale transformation in cross-section biotope type for riffles or runs. However, at reductions in discharge of about 86% of natural baseflow, runs showed a tendency to change to pool/run transitional biotopes, primarily where depth was high. At similarly high reductions in flow, riffles either transformed partly to become riffle/run transitional biotopes, where depths were high, or transformed entirely to trickle 'runs' in shallow reaches. The latter biotope was characterised by no turbulence and trickling shallow flow between mostly exposed cobbles.

- Under conditions of natural flow, in the peak of the dry season, riffles and runs typically remained the two dominant biotopes, although the relative proportions of these and other biotopes were noticeably altered. In midsummer, with a 36% reduction in baseflow, there was no major shift from one of natural dominance by runs, and although riffles decreased in availability they remained the second most abundant biotope. Although at the Du Toits impact location, runs remained dominant with an 86% reduction in flow, there was a change in the subdominant biotope from riffles to pool/run transitional biotopes and severe to total loss of riffle habitat. In the Riviersonderend impact location, the shift in dominance was even more marked with a similar decrease in discharge, from naturally common riffles to a dominance by pool/run transitional biotopes. There was also a severe to total loss of riffles, while pools and standing waters increased markedly in their proportions. At all unnaturally low baseflow magnitudes slow-flowing, transitional and alternative types of biotopes, primarily pool/run transitions, isolated standing pools, trickle 'runs', and patches of partially and fully exposed substrata, assumed a greater role in the provision of physical habitat. However, these biotopes were possibly only partially suitable, if at all, for the biota.

- Reductions in average and maximum depth were recorded at unnaturally low flows, relative to natural depths. With a 36% reduction in baseflow (Molenaars), decreases in average depth of 52% and 36% were evident in runs and riffles, respectively. With 86% reductions in discharge, run depths decreased by 50% (Du Toits impact location) and 55% (Riviersonderend impact location). Riffle depths decreased by 46% in the Du Toits, but those in the Riviersonderend did not markedly differ from natural depths. A pronounced increase in the proportion of extremely shallow areas (< 0.05 m) and complete loss or marked narrowing of the deepest sections of the channel, also occurred at unnaturally low flows. This was especially prevalent with an 86% reduction in discharge.

- Reductions in average (0.6-depth velocities) and maximum velocities were often observed at low flows. With an unnatural baseflow reduction of 36%, average velocities in control and impacted runs remained similar, but average riffle velocities were 17% lower than natural values. With 86% reductions in baseflow, the average velocity in runs decreased by 43% in the Du Toits impact location and drastically by 81% for the Riviersonderend impact location, relative to natural magnitudes. At the same discharges, the average velocity in riffles was reduced by 66% in the Du Toits, but by only 18% in the Riviersonderend impact location. At extreme low flows particularly, there were also marked increases in the cross-channel proportions of non-flowing sections.

- With unnatural degrees of flow reduction, increased exposure of the substratum was recorded, both with shrinkage of the wetted channel margins and instream with increased dissection of the wetted channel. As a result, both edge and instream habitat was lost through drying, there was a marked loss of connectivity of biotopes, and total isolation of some marginal and instream pools took place.

- The natural physical character of riffles rendered them more susceptible to large-scale losses in connectivity of wetted habitat than runs and to increases in the proportion of wetted width subjected to flow cessation, with a major decrease in discharge.
- After the reinstatement of natural baseflows, recovery to physical biotope conditions hydraulically similar to those in the control locations was immediate.

Effects of dry-season baseflow regimes on benthic macroinvertebrate assemblages

From preliminary assessments of the effects of natural and unnatural baseflow magnitudes on the BMI assemblages (Section 10.5), it is clear that the relationships between discharge and biotic response are highly complex. In most instances, the types of analyses conducted did not reveal distinct or consistent changes in the composition of BMI assemblages in relation to either natural or unnatural, dry-season baseflows. Possibly, more sensitive kinds of analyses, including species-level analyses, are required to elucidate such patterns. However, for the purposes of this study, the emphasis was on the identification of any broad-scale responses of invertebrates to flow reduction, using relatively simple biotic measures of response. Classification and ordination analyses of patterns in the composition of the BMI assemblages, at the four sites, were generally inconclusive at several levels of resolution (Section 10.5.5).

The observed main responses of BMI assemblages to natural dry-season baseflow regimes were as follows:

- At the four sites, numbers of BMI families and higher taxa were fairly stable over the range of baseflows, although some natural variability was apparent. There was no consistent pattern among rivers of an overall increase or decrease in the numbers of taxa with a natural decrease in baseflow magnitude. However, numbers of taxa were typically lowest at the end of summer or in early autumn (March to April). With natural decreases in baseflow from December to February, there were no clear overall responses in the numbers of taxa inhabiting riffles or runs. Moreover, changes in numbers among the various substratum components of each biotope were highly varied.
- For all four sites, there was a high degree of natural variability in numbers of individuals. For all sites except the Riviersonderend, there was a decrease in the numbers of individuals in riffles and runs from December to lowest numbers of individuals, overall, in February. This was followed by an increase in abundances again in April. However, within the various substratum components of the two biotopes, there were some divergences from this characteristic pattern. At the Riviersonderend site, the numbers of individuals in riffles and runs exhibited opposite trends to those at the other three sites, under conditions of natural baseflow. Specifically, there were increases in numbers of individuals from December to February, overall and for riffles and runs. Further, there was a consistent natural decrease to lower abundances in April.
- Numbers of taxa and individuals were typically higher in riffles than in runs. In both instances, lowest numbers of taxa and individuals were recorded from the tops stones, while stone bottoms and the underlying stones exhibited similar, higher abundances and numbers of taxa.
- Comparison of all four sites, under natural baseflows, revealed that the numerically dominant families throughout summer were Baetidae, Chironomidae, and Simuliidae. However, there were considerable differences among the rivers in the percentage representation and by various other taxa recorded at abundances $\geq 5\%$ of the total assemblage. Natural variability in assemblage composition at the family level was also high within the same reach and between months.

- There were no entirely consistent trends in taxon dominance among the four sites or in riffles and runs, either with the natural decrease to low flows in February, or the subsequent transition to autumn. Trends were most frequently site-specific, but in all instances, with a natural decrease in flow, numerous small and larger-scale shifts in abundances of common taxa were observed (Section 10.5.3). For instance, *inter alia*, from December to February, Baetidae typically decreased in abundance while the numbers of Ephemerellidae increased.
- Typically, during a natural dry season, BMI assemblage diversity was greatest at lowest baseflow, in February, and lowest in April. However, the latter trend did not hold for the Riviersonderend mountain-stream site, where diversity was lowest at the onset of summer, in December.

Comparison of natural trends in BMI assemblages with observed patterns at unnaturally low baseflows in February, revealed the following kinds of effects of unnatural flow reduction:

- The numbers of families and higher taxa increased in all three impacted locations, although there was not a direct relationship between the proportion of flow abstracted and the observed responses. For the Molenaars and Riviersonderend sites, in February, with moderate and extreme baseflow reduction below natural levels, respectively, numbers of taxa increased in riffles but decreased in runs. However, the opposite response was observed in riffles and runs with an 86% reduction in flows in the Du Toits impact location. After the reinstatement of natural baseflows, in April, the site at which the least baseflow had been abstracted, the Molenaars, showed complete recovery in numbers of taxa. However, for both the Du Toits and Riviersonderend impact locations, numbers were still marginally elevated in April.
- There was no obvious effect of a 36% reduction in baseflow (Molenaars), in February, on the overall numbers of individuals. In contrast, with an 86% reduction in discharge, there was a pronounced increase in the total number of individuals per 0.1 m^2 by 291% above natural levels, in comparison with the natural response of a decline of 21% in abundance. However, this trend was only apparent at the Du Toits site. In the Riviersonderend impact location, overall numbers increased only marginally above natural levels during the impact phase. At the biotope level, the effects of unnatural baseflows on abundances in riffles and runs differed according to the percentage flow reduction below natural magnitudes. For riffles and runs, there was an overall decrease in the total number of individuals with 36% reduction in baseflow in February, where the decrease occurred primarily in the underlying substratum. Hence, the general direction of change in abundance was the same as the natural one. With *c.* 86% reduction in flow, however, there were distinct increases in numbers of individuals in riffles, especially on the tops of stones. In contrast, in impacted runs, only small increases in the numbers of individuals were recorded. Full recovery was recorded at all three impacted locations, with the reinstatement of natural baseflows in April, with the possible exception of the Du Toits impact location.
- With unnatural flow reduction in February at a location scale, there were few major shifts in taxon dominance that could be identified as markedly different from natural ones, and recovery seemed to have occurred in April (Section 10.5.3). In addition, observed changes in taxon dominance in riffles and runs, under unnatural baseflow regimes were not consistent and were most frequently site-specific. For example, with 36% and 86% reductions in dry-season baseflow, in the Molenaars and Riviersonderend impact locations respectively, there were no major alterations in taxon dominance. At the Du Toits site, taxon dominance was shared among a greater number of taxa under conditions of extremely low baseflow (86% reduction), than at natural baseflows.

- With a 36% reduction in dry-season baseflow in February, assemblage diversity increased marginally from natural levels, still following the natural trend. However, with extreme reductions in baseflow of 86% below natural magnitudes midsummer, diversity decreased markedly from natural levels in both impacted locations.

Conclusions, with reference to the implications of the results for development of the low flow component of instream flow methodologies

The results of the study of the effects of natural and unnaturally low (experimental) dry-season baseflows on four upper-river ecosystems (Chapters 10 and 11) lead to the following general conclusions, with their implications for development of the low flow component of instream flow methodologies.

To assess the potential impacts of abnormally low flows, the natural baseflow regimes at the time of the study and the experimental low flows needed to be placed in the context of the natural, long-term hydrological history of each river. This revealed that the impact locations on the three experimental rivers were subjected to unnaturally low discharges during the experiment, whilst their control locations and the fourth river, used as a control, were not. The experimental reductions represented marked hydrological disturbances to the three riverine ecosystems, being well beyond most that they had been subjected to historically over a known period of almost 30 years.

The magnitude of experimental flow reduction was shown to be an inadequate index of the degree of hydrological disturbance the rivers had experienced. Rather, this reduction needed to be placed into perspective by being expressed as a percentage of the natural low flow pertaining at the time. Even so, it became clear that it would be inappropriate to use a single low-flow index to identify suitable low flows for an entire dry season, and for all rivers. Each of the experimental rivers in each month of the dry season naturally exhibited an individual hydrological character. Only the peak dry month, February, showed a highly stable, low range of baseflows at all sites.

On a month-by-month basis, three hydrological indices routinely used internationally to identify low flows were shown to have merit for delineating the approximate discharge range below which flows probably have an unnatural impact. These were the Q_{90} and Q_{95} values from 1-day annual FDCs, and the median 7-day consecutive low flow. Each of these showed merit in that the natural flows measured during the study lay near their values, whilst the experimental low flows were closer to other indices representing lower flows, such as the absolute minima on record. However, there was no clear correlation between any one of these indices and the natural measured discharge of the rivers over the dry season. Thus, in the selection of appropriate low flow indices for instream flow studies, cognisance should be taken of which hydrological indices best represent the flows that a particular river usually experiences.

The natural and experimental dry-season baseflow regimes were assessed in terms of their effects on the following ecosystem components: water chemistry; biotope proportions and hydraulic character; and the composition and distribution of BMI assemblages.

Under natural baseflows, the four studied rivers exhibited considerable between-river, inter-reach and temporal variability in measured components. Differences in values of many chemical variables between summer months were often fairly low, indicating high overall stability during a dry season with no clear

relationships with baseflow magnitude. The main, natural changes in chemistry appeared to occur at the very end of the dry season, with the initiation of a natural shift to autumn climatic conditions. The proportions and hydraulic character of physical biotopes clearly changed in response to increases or decreases in discharge magnitude, although the dominant biotopes were always riffles and runs, each of which supported a distinct BMI assemblage. Changes in the distribution of BMIs were seldom consistent during the study period, however, suggesting that other factors in addition to discharge, particularly biotope hydraulics and proportions, and season, contributed to observed trends. An understanding of the extent of such variability is essential in order to differentiate between a river's natural functioning and responses directly associated with unnaturally low flows. It is therefore suggested that the emphasis in a low flow assessment should be especially on the presence of the overall combination of biotopes and their associated BMI assemblages, rather than only on those of riffles as has often occurred historically. Such knowledge would be a critical point of departure in the development of a recommended flow regime.

When assessing the effects of experimental low flows, quantification of the impacts was ecologically more meaningful when related to some combination of the components than to any one component. For instance, an increase in a species of BMI with reduced flows might be related to animals packing into a reduced area of suitable habitat rather than to an increase in abundance *per se*.

Experimental low flows had limited secondary impacts on water chemistry. With flow cessation over much of the location, as well as extremely low velocities and the creation of isolated standing waters in other areas, there was some deterioration in water quality, mostly in conductivity, TDS, TA and pH. The most pronounced effects were recorded in isolated pools in the Riviersonderend River, where the reduction in flow resulted in increased conductivities. A free-flowing main channel would undoubtedly have limited the extent of such changes. In slow moving or still waters, tertiary effects of flow reduction included the proliferation of algae and desiccation of instream/marginal vegetation, the biological implications of which have yet to be explored. Freshes occurring late in the dry season flushed the poor-quality water from the affected locations, rapidly restoring natural conditions. It should be noted that the study sites were near natural and that there might be additional or more significant impacts of unnaturally low flows on the water chemistry of polluted reaches.

With reference to biotope dynamics, experimental low flows mostly resulted in highly exaggerated forms of the changes observed with natural low flows. However, there were some major shifts in biotope dominance and hydraulic character. There seemed to be a redistribution of invertebrates between runs and riffles in response to such biotope changes, with the BMI composition of run assemblages thus growing to more closely resemble that of riffles. The responses were difficult to detect, probably due to various, poorly understood contributing factors. These could have included the extent to which the hyporheos acted as a refuge area; facultative versus obligate utilisation of biotopes other than riffles and runs; downstream drift of insect species; and species' life-cycle generation times and emergence patterns. Factors such as the inherent resistance and resilience of the BMI assemblages in relation to flow-related physical disturbance may also have played a role. For instance, a lesser response by BMIs to extreme low flows was evident in the Riviersonderend than the Du Toits impact location, despite similar experimental reductions in discharge in both rivers. This could perhaps be attributed to factors such as the naturally, very low midsummer baseflows and highly heterogeneous composition of physical biotopes in the Riviersonderend, relative to the other sites. Finally, BMI response might have been occurring within families, which would not have been

detected in the analyses done for this report. Extreme low flows might also be resulting in longer-term, cumulative stresses on the biota, not necessarily detectable using the types of data collected in this study or within its timeframe. Overall, the changes were variable and not necessarily proportional to the degree of flow reduction. It is worth noting that the above analyses were based only on data from the first impact month. Data for the second impact month, collected but not yet analysed (Tharme in prep.), might reveal different levels of response. In particular, the intensity of biotic interactions might well increase during the longer period of extreme low flow.

Biotic variables such as diversity, the number of families per biotope, and the total number of individuals, showed merit as suitable measures of invertebrate response to low flows. However, due to the complexity of observed responses, it is likely that a suite of biological indices would need to be used to assess impacts of low flow, in conjunction with further, more sensitive multivariate analyses. Apparent trends in responses could also be due to factors other than, or in addition to, changes in the magnitude of baseflow, some of which might not be flow-related. As yet, the observed biotic responses to low flows and the implications thereof for the ecosystem remain poorly understood.

In conclusion, although various changes were observed in all ecosystem components at low flows, there were few clear trends. Furthermore, ecosystem responses to experimental low flows were not simply directly related to the graded percentage decrease in flow from natural magnitudes. Between-river differences in BMI assemblages were typically more pronounced than differences related specifically to natural or experimental dry-season baseflow regimes. River zone, season, biotope type and hydraulic character, and substratum component appeared to have at least as much influence on the composition of the BMIs as baseflow magnitude. This precluded the identification of a series of well-substantiated low flows, each strongly linked to a set of consequences for the river ecosystems. Hence, although at this stage the study has not revealed that experimental flow reduction had major impacts on the BMI assemblages of the rivers, broad-scale generalisations in instream flow assessments regarding suitable low flows would be simplistic.

Comparison of the results of the study with other international or local literature was limited by the dearth of similar published studies on the effects of abnormally low flows on perennial rivers. Additional similar research would be required before significant further development could occur of the scientific basis of the baseflow component of instream flow methodologies like the BBM (Section 16.2). Some of the required research, which follows on from the preliminary results of this study, is underway (Tharme in prep.).

E5.2 Investigation of the links between wetted perimeter and easily computed hydrological indices for studying baseflow characteristics (Chapter 12)

With development of a river's water resources, there may be a need to reduce the natural baseflow so that some of the water can be allocated to offstream users. Reducing baseflows without causing undue damage to the river ecosystem is a management challenge. When this management decision is needed for a stream which has not been monitored in terms of biota, channel morphology or flow patterns (as often happens in planning studies), the problem threatens to become intractable.

Chapter 12 describes the search for a guideline that has its roots in the abiotic realm, by focusing on the wetted perimeter-discharge characteristics of streams. Essentially, as the discharge of a river decreases,

there is a flow at which the loss of wetted perimeter accelerates as the wetted channel shrinks in from the banks (King and Tharme, 1994). The study was designed to identify this flow (called the Wetted Bed Flow or WBF) in several western Cape headwater streams and to attempt to link it to one or more easily computed hydrological indices. If such a link could be made, it could act as a guideline in future situations where the water requirements for river maintenance were being considered but fine-resolution (daily) hydrological data were unavailable. Hydrological indices of a common nature are often available on a regional basis or can be generated from regional relationships or via catchment models. It was hoped that the general availability of such indices would provide a key to making reasonable first-approximation inferences about critical low flows at unmonitored sites.

Four cross-sections (two riffles and two runs) were established on eleven relatively undisturbed headwater streams in the Western Cape. A good hydrological record was available for each site from a nearby gauging weir. Water surface elevations were surveyed for each cross-section on at least three occasions, at different discharges. A backwater hydraulic model was used to calculate the wetted perimeter over a range of discharges, and wetted perimeter-discharges (WP-Q) curves were generated. Low-flow inflection points were identified on these curves, that is, the points where the rates of change of WP with respect to Q were greatest. Research then focused on whether or not these inflection points, reflecting the discharge at which the wetted bed began to shrink in from the banks (i.e. the WBF), were correlated with one or more common hydrological indices.

The results revealed that the most promising indices in terms of consistency across the full sample of rivers were aspects of the MAR, the monthly flow duration curves and the 1:2 Year Maximum Average Daily Flow (MADF). The WBF lay between 29% and 42% of the MAR (expressed as an instantaneous flow), between the 40th and 53rd percentile on the annual flow duration curves; and between 2.0% and 2.9% of the 1:2 year flood (expressed as a maximum daily average). The relative consistency of the WBF links with these three indices suggests that the WBF in south-western Cape streams could be calculated from some coarser hydrological statistic.

Another insight into the use of wetted perimeter-discharge relationships was revealed by the research. It had been expected that the WBF would be a very low flow, which had tentatively been named the critical minimum. The critical minimum flow had been seen as a value below which, for ecological purposes, discharges should never drop. It had been expected that some higher flow would be more acceptable as the low flow necessary for ecosystem maintenance, with the WBF or critical minimum being the level that flow could drop to in drought years only. The research showed this not to be so, as the WBF approximates the median flow of each river (on average the 47th percentile of the FDC). Hence, the research reported on here showed that relatively undisturbed rivers experience flows lower than the WBF for about half the year, probably during the dry season. They must therefore experience wetted-bed shrinkage in the normal course of events and the aquatic species must be adapted for such seemingly adverse conditions.

It was concluded that the lower limit of flows for the dry season should be based not on the WBF, but on biological considerations. Nevertheless, the WBF is a useful first point when considering ecologically-acceptable low flows, as it provides an order-of-magnitude indicator of a low flow that is within the normal baseflow regime of an ungauged river.

E6. DRY-SEASON FRESHES AS TRIGGERS FOR FISH SPAWNING

Chapters 13 and 14 address the fact that one of many potential impacts of instream dams on downstream flow regimes is the eradication of many, if not all, of the small pulses of higher flow or freshes. These add variability to the flow regime during the dry season and cue many life-cycle activities of aquatic plants and animals. To investigate the importance of freshes, artificial ones were released from Clanwilliam Dam into the downstream reaches of the Olifants River in 1993 and 1994 (Chapter 13) and in 1995 (Chapter 14), during the suspected breeding season of the Clanwilliam yellowfish, *Barbus capensis*. The only known spawning ground of the species in the middle reaches of the river between Clanwilliam Dam and Bulshoek Weir, is situated about 1 km downstream of the dam. The objective of the research was to investigate whether or not the fish could be induced by the freshes to spawn at this site.

Over the period 14-18 December 1993, experimental freshes of magnitude 8.0-12.0 $m^3 s^{-1}$ and duration 4.0-12.5 hours were released during the day or overnight against a background baseflow release of 4.6 $m^3 s^{-1}$. No spawning was observed at the spawning grounds linked to these releases. This was possibly because of the activities of the survey team beside and in the river, and because only a limited time was spent observing the spawn site from an overarching bridge. Much of the evidence of what happened that year was therefore pieced together from analysis of the eggs, free embryos and larvae found and collected. The general evidence pointed to sporadic spawning, some of which had happened before, and some during, the experimental period. The 1993 study also suggested that no significant changes in water chemistry and temperature occurred in the river during the freshes, which is unsurprising as they were merely higher releases of hypolimnetic (bottom) water.

During the experiment in 1994 most time was spent viewing and filming fish activity from the bridge. On 1-2 December 1994, two experimental freshes were released overnight, of magnitude 17.0 $m^3 s^{-1}$ and 9.0 $m^3 s^{-1}$ and duration 12.3 hours and 12.0 hours respectively, against a background baseflow release of 2.0-3.0 $m^3 s^{-1}$. Approximately 100 fish were observed moving upstream during the two days beginning with the first experimental fresh and abundant spawning behaviour was observed. Two months later, "far more larvae and young juveniles than usual" were found at a downstream larval site, and the evidence pointed toward a far more successful season of recruitment. It cannot be said with certainty that the freshes triggered spawning, but all spawning activity occurred on the receding limbs of these high-flow pulses.

It seemed probable that *B. capensis* was being brought into spawning condition by the general spring increase in river temperatures, possibly linked to other factors such as photoperiod. Once in condition, spawning then appeared to be triggered by a small increase in discharge. Thus, a link between high-flow pulses and spawning seemed to be emerging. To validate this link, the successful experiment of 1994 needed to be repeated.

A replicate experiment was thus undertaken in 1995, with the same magnitude freshes planned for the same calendar days (Chapter 14). In addition, an extended programme of measurements of river water temperature was completed for the three months leading up to the experiment, in order to assess the effect of temperature on fish movements. On 28 November 1995, an hypolimnetic fresh of magnitude 14.0 $m^3 s^{-1}$ was released for 13.0 hours overnight, followed by two overnight epilimnetic (surface level) freshes on 29 and 30 November, of magnitude 11.0 $m^3 s^{-1}$ and duration 15.0 hours.

The cold (16-18 °C) hypolimnetic fresh failed to induce spawning. This appeared to be because the extension of winter conditions into spring, characterised by late snow on the mountains of the headwaters and floods, resulted in heavy flows of cold water arriving in Clanwilliam reservoir later than usual. As a result, water temperatures during the experimental period in 1995 were 3-4 °C colder than in 1994. Release of the two warm (19-21 °C) epilimnetic freshes correlated with fish moving onto the spawn beds and exhibiting pre-spawning behaviour. However, when each warm fresh was replaced by cold hypolimnetic baseflow releases, the fish moved downstream away from the spawn beds, without spawning. The presence further downstream of free embryos and larval *B. capensis* indicated, from their developmental stages, that spawning had occurred before the experimental releases began, probably when warm (21-23 °C) epilimnetic water was spilling over the dam. This spawning appeared to have halted once spillage stopped at the end of winter and hypolimnetic releases began. Indeed, the presence of dead and deformed young suggested that the cold hypolimnetic releases might have had a detrimental affect on those spawned during the earlier warmer spell. It was hypothesised for further testing that water temperature on the spawn beds for successful spawning of *B. capensis* has to be at least 19 °C and stable or rising. It was further hypothesised that if the fish are in breeding condition and these temperature conditions are met, then experimental freshes of the size described in these studies will trigger spawning.

Dry-season freshes, such as those released in these experiments, are recommended in most IFR studies using the BBM. Justification for them is usually that they will promote fish spawning. It was concluded from this research that providing the right hydraulic conditions at the right time probably will trigger spawning, but only if the fish are in breeding condition and the water temperature is suitable. The temperature of released water can be manipulated from dams with multiple-release structures. This strategy may have limited success, however, unless the temperatures are then maintained at levels that are suitable for the developing young.

E7. WET-SEASON FLUSHING FLOWS FOR MOBILISING SEDIMENTS

Fine sediments entering high-gradient, cobble-bed reaches of rivers are normally scoured out during wet-season floods. Damming of rivers and other manipulations of flow regimes often lead to a loss of floods. This in turn can result in a reduction in the scouring of fines. Many small aquatic animals, particularly the rare and endemic species, live on the underside of cobbles or in the substrata beneath the cobbles (Chapters 10-11 and Appendix 1.2). Sedimentation of these reaches can destroy their refuge areas, often seriously affecting the normal functioning of such ecosystems and possibly eradicating species from the river. In recognition of this and associated problems, flushing flows are usually included in recommendations for flow regimes for rivers targeted for water-resource development. This is done without clear understanding of whether such flows would scour fines from cobble beds.

Research was therefore undertaken to establish a relationship between the extent to which scour takes place within a cobble-bed river, and the magnitude of discharges that have occurred in the river. Using field data on cross-section river size and shape, the position of the top of cobbles and of the sand surface, sediment size and discharge, collected in the Molenaars River in Du Toit's Kloof, a stage-discharge relationship was obtained for the study reach and a Sediment Scour Model developed. The model links the extent to which

sediment is scoured (as revealed by the average diameter of the sampled sediments), with the maximum velocity of the water flowing over the area of scouring in time-spans from 24 hours to 20 days before the sediment was sampled. Thus, for instance, for any average-size substratum, the velocity that needed to occur the previous day to achieve that size could be identified. The higher discharges that would be needed at time-spans ranging from two to 20 days earlier, to achieve the same result, could also be identified.

At present, use of the full Sediment Scour Model is limited to applications to the Molenaars River in subsequent winters in which the same catchment conditions apply, as the relationship derived is strongly dependent on the rate of supply and deposition of sediments during the study. With changing conditions, the further removed the sampling day is from the day of maximum discharge, the more uncertain the derived relationship between flow and sediment size becomes. However, relatively little deposition usually occurs in the 24 hours after maximum flood discharge. Thus, the part of the Model that links sediment size with a required maximum flow 24 hours previously can be applied more confidently to the Molenaars for subsequent years, as well as to other rivers with similar bed and sediment input characteristics.

E8. OTHER RESEARCH

E8.1 The hydrological nature of freshes

A short research project, undertaken by a civil engineering student, focused on a hydrological analysis of freshes, in order to describe and quantify them and to illustrate the effects of eradicating them on the shape of flow duration curves. This research was done because flow durations curves are often presented by hydrologists to river ecologists in order to illustrate how a proposed water-resource development would affect the flow of a river. Without an understanding of how the loss or manipulation of various components of flow regimes is reflected in flow duration curves, however, ecologists are poorly equipped to interpret the ecological significance of changes in their shapes. The research provided a definition of a fresh as "any increase in flow that is large enough to ensure that the average flow of the next two days is 1.7 or more times that of the preceding two days". It was concluded that the replacement of freshes by baseflows alters only the middle part of a flow duration curve, causing it to decline.

E8.2 The effects of a flood on benthic macroinvertebrate communities

Additional research on floods was undertaken by an ecologist in a six week B.Sc. Hons. project, who studied the effects of a flood on benthic invertebrate species. It had been hypothesised that, in cobble-bed rivers that are not choked with fine sediments, the animals would migrate vertically down into the riverbed during spates (floods) to avoid harsh hydraulic conditions. If this were true, the siltation of cobble beds through catchment or bank erosion would result in a serious loss of refuge areas for this vital component of riverine ecosystems. The research was designed to investigate the extent to which vertical migration occurs during spates.

The differences between a riffle and a run in the Molenaars River, in terms of their hydraulic characteristics and their benthic invertebrate fauna, were determined under conditions of winter baseflow. The effects of a spate on these hydraulic characteristics and on the animals were studied using a sequence of samples taken during and after the spate. The riffle and run were distinctly different, hydraulically and biologically, under

conditions of winter baseflow. The hydraulic differences, in terms of Froude number and depth, but not in terms of velocity, remained distinct for the two biotopes through the course of the spate. The differences in terms of the species composition and abundance were destroyed by the spate and there was then a process of recovery over the next seven days that was detectable by the partial return of the invertebrate species to their original biotope.

There did not appear to be mass migration of invertebrates into the cobble bed during the spate. Some species appeared to move to the tops of the rocks, and there were also signs that some were washed downstream. However, other species – often rarer, endemic ones - did show a movement into the cobble-gravel substrata under the bed. Thus, for some species at least, the areas between and under the cobbles were essential refuges during spates. These areas would be unavailable in sediment-choked, cobble-bed rivers.

E8.3 The effects of a variety of anthropogenic perturbations on the benthic invertebrates of a river system

A B.Sc. Hons. study, undertaken by an ecology student, investigated the effects of a number of different anthropogenic disturbances on rivers, in order to place the manipulation of flow regimes within the spectrum of low to high impact. Invertebrates were sampled from the foothill zones of six tributaries of the Olifants River, Western Cape, and of the mainstem of the river. Samples were taken upstream of all obvious disturbance and, close by, in disturbed reaches. In all, 14 sites were sampled. Notes were made of environmental conditions at each site, and particularly of any signs of unnatural perturbation. Grouping of similar invertebrate samples indicated a set of relatively undisturbed sites, mostly correlating with the upper sites on each river, and a set of disturbed sites, correlating with the lower sites. In terms of the difference between invertebrate assemblages of upstream and downstream sites on any one river, bulldozing and extensive water abstraction emerged as the most extreme disturbances, followed by clearance of riparian vegetation. A camping site provided the least disturbance, while citrus farming with a well-established buffer zone of indigenous riparian vegetation had an impact that was noticeable in terms of water clarity, but minimal in terms of changes in invertebrate assemblages.

E.8.4 The implications of predetermined release policies and dam sizes on the natural flow regime of Western Cape rivers

This hydrological study, by a fourth-year civil engineering student, focused on the effects of predetermined release policies and dam sizes on the flow regimes of Western Cape rivers. Part of the work involved investigating the implications, in terms of dam location and yield, of dam release patterns designed to maintain the downstream river in some pre-determined state of 'health'. It was concluded that one-MAR dams are more suitable for sustaining water releases for river maintenance and downstream irrigation farmers than are smaller half-MAR dams. Furthermore, it was concluded that very small rivers should not be dammed, as dams on such rivers would not be able to store sufficient water to ensure that the required volume of water could be released at the required times for river maintenance.

E9. RECOMMENDATIONS FOR FUTURE WORK

E9.1 Recommendations for further development of the Building Block Methodology

Desired State versus Current Environmental Values

One of the concepts central to the BBM has been that the flow regime described for the river – the IFR – is designed to achieve and maintain some pre-determined Desired State of the river. In practice, this has been difficult to work with, as the DS has tended to be described in a somewhat general way. This has not guided the IFR team well in determining exactly what they are describing flows for. Nor has it provided the post-development monitoring team with specific targets for assessing the success of the IFR in maintaining that DS. Recent production of a protocol for defining the DS of riverine systems (Rogers and Bestbier 1997) will help improve this situation.

Australian colleagues (Dr Jane Doolan, Department of Conservation and Natural Resources, Melbourne pers. comm. and Prof. Barry Hart, Monash University, Melbourne pers. comm.) have employed a somewhat different approach, using the concept of Current Environmental Values (CEV). They have suggested that an IFR team could describe a flow regime for maintaining pre-identified CEVs, with each specialist thus having a clear objective for recommending certain flows. Other conditions of the river linked to other possible flow-release patterns could be addressed at the Scenario Meetings. Using this approach, targets for monitoring maintenance of the CEVs would also be clarified.

An alternative scenario-based approach has recently begun development in a collaborative effort between South African, Australian, North American, New Zealand and Lesotho scientists. This removes all aspects related to stakeholders value judgements from the IFR workshop deliberations, and places them in a different part of the process. The IFR workshop then concentrates specifically on the consequences to the river ecosystem of a range of possible flow regimes.

The implications of all these approaches, and the extent of their compatibility, need to be researched.

Habitat mapping and physical-habitat modelling

Scientists working on IFR assessments have expressed the need for information that places in context the surveyed cross-sections forming the basic information on an IFR site – a kind of bird's eye view of the site. There is also a need to be able to visualise how physical habitat changes across the entire site with flow and how this might affect biotic distributions. Both of these facilities exist in the Instream Flow Incremental Methodology (IFIM) (Bovee 1982). However, there are many drawbacks to applying IFIM, including its complexity and the uncertainty regarding the meaning of some of its central outputs (King and Tharme 1994).

Different kinds of approaches which could meet the scientists' requirements are being researched as part of the Kruger National Park Rivers Research Programme and at the University of Cape Town, in what have been named Abiotic-Biotic Link projects. These approaches, and any other potential ones, need assessment and further development. As the BBM is essentially a methodology for organising data and understanding, it has the potential to be able to use data from any such approach.

Water-quality link-up

Most emphasis in the BBM is on water-quantity requirements for the river ecosystem, and the water-quality component is weak. Strengthening the quantity-quality link could take several forms, but should probably include a modelling component. This should be able to inform on probable future chemical conditions in the river at a resolution relevant to ecological studies. There would need to be a parallel comprehensive programme of data collection on the range of tolerances of riverine biotas, in order to predict the biological responses linked to predicted future chemical changes. A current WRC project is examining ways of predicting changes in water chemistry resulting from changes in streamflow (Dr J.A. Day, Freshwater Research Unit, University of Cape Town pers. comm.).

Taking the Building Block Methodology into the Feasibility and later Phases of developments

After the IFR workshop, the water manager uses the recommendations on flows in yield analyses. These reveal the viability of the water-resource scheme with IFR flows in place. If a project of dam building is to proceed, the IFR is also considered in dam design and in any Scenario Meetings where flow-release options other than the IFR are considered. The ecological, economic and social consequences of each option are conveyed to interested and affected parties through a Public Participation Process. Environmental management plans may be drawn up for the construction phase and monitoring programmes set in place.

All of these activities take place at present, but no recognised set of procedures exists. The BBM urgently needs further development in these areas.

E9.2 Recommendations for flow-related research, with particular reference to its application in instream flow methodologies

Recommendations for further research on dry-season baseflows

A considerable body of further work is required to define low flows on an ecological basis and to identify ecological-relevant low flow indices. This work needs to include the development of a conceptual model of the possible effects of low flows on perennial riverine ecosystems, and the establishment of low flow modelling techniques, linking hydrology with habitat quality and quantity and the instream biota. These sets of information are required for the further development of the baseflow component of the BBM and other holistic instream flow methodologies. In this regard, apart from the continuing focused work on dry-season baseflows (Tharme in prep.) based on the findings of the current report, several main areas of research were identified that require further effort.

Linking present hydrological events with aspects of the historical flow record

Currently, links between present-day hydrological events and the longer-term flow record are seldom made, and usually only on a case-specific basis within instream flow assessments. Although low flow regionalisation is now well underway (Smakhtin *et al.* 1995; Smakhtin and Watkins 1997; Smakhtin and Toulouse 1998; *inter alia*), there is considerable further scope for the inclusion of more ecologically meaningful hydrological indices, specifically those describing flow variability and predictability, in the hydrological component of the BBM. As part of this process, analyses of flow percentiles are required to identify which of them (or other low flow indices) represent baseflows that are commonly experienced in different river zones, and various sizes and types of river during the different months of the dry season.

Additionally, research should be continued on the implications of flows derived on the basis of various low flow indices, for riverine ecosystems. From a field perspective, as a result of the natural variability in discharge as measured within a reach, many instantaneous measurements of dry-season baseflow are required before a relationship between present-day discharge and various flow indices derived from the long-term record can be adequately established. Thus, there is a need to develop a better understanding of the relationship of instantaneous baseflow to average daily discharge and other discharge variables for the same time period. In relation to such records of instantaneous discharge, accurate measurement at low flows is extremely difficult, and there is a high experimental error associated with the results. Such a margin of error could mean that small fluctuations in discharge magnitude that are potentially important from an ecological perspective might be masked. More accurate ways of measuring discharge and associated hydraulic variables at very low flows, therefore, need to be established to avoid this potential problem.

The importance of antecedent hydrological events also needs to be considered when assessing the impacts of low flows on rivers or developing instream flow recommendations for their baseflow component. For example, antecedent flows may allow the development of certain conditions favouring a particular state of the ecosystem. Although subsequent flow events may alter conditions of individual ecosystem components, it may be to a lesser degree or take a different amount of time for changes to be reflected, due to previous flow conditions. Research is required into ways in which antecedent flow events can be adequately incorporated or considered when developing instream flow recommendations. These various issues have been recognised as aspects of the BBM that require further research effort, with more intensive collaboration between freshwater ecologists and hydrologists (Tharme 1997c; this study).

Development of broad scale relationships between baseflow and ecosystem variables

The relationships between dry-season baseflow magnitude and concentrations of various chemical variables need to be established on the basis of different river zones, types of rivers and regions of the country, particularly with reference to observed natural ranges of variability. Work is also required to establish similar regional trends in wetted perimeter-discharge relationships, as well as patterns of biotope dominance, biotope diversity, and other relationships between hydraulic variables and dry-season discharge.

The importance of differences in physical biotopes at low flows

Further work is required to objectively classify riffles, runs and other biotopes, and to describe the extent to which various transitional biotopes differ from them, both hydraulically and in terms of their invertebrate assemblages, at low flows. Relationships between various combinations of substratum and flow type, which comprise physical biotopes, and the assemblages inhabiting them, are the focus of a current WRC project (King pers. comm.). Such knowledge might further assist with the identification of similar areas of river habitat for the biota.

It was evident from this study that additional work would be required to establish at what proportions, and with what ranges in hydraulics, various biotopes become critically limiting for the biota, as a result of flow reduction. Such an assessment would require knowledge of the specific biotope requirements of the various taxa comprising each invertebrate assemblage, and hence an indication of preferred and marginal biotopes for each. For instance, the relative suitability of fully immersed cobbles compared with partially exposed cobbles and entirely exposed substratum, as invertebrate habitat has yet to be examined.

Direct influences of low flows on instream biota

Research is required to explore the probability, as suggested in the international literature, that there is an increase in the importance of biotic interactions at very low flows. Additional related studies are required to address the implications of unnaturally low flows for aspects of the life cycles of individual species, such as generation times and growth rates. Currently, there is no research on these topics, yet it is urgently required if progress in understanding the fundamental requirements of instream biota for dry-season baseflows is to be achieved.

Comparative studies of the similarities and differences between biotic (and other) responses to low flows in perennial and temporary rivers, for different baseflow regimes, and under undisturbed and anthropogenically-altered conditions, are required. Such work would assist with establishing the degrees of resistance to disturbance and resilience in terms of recovery from impacts, associated with different types of river.

Recommendations for further research on dry-season freshes

Present knowledge is inadequate regarding the fundamental nature of freshes, and of their role in river ecosystem functioning.

Freshes are loosely defined as small pulses of higher flow during the dry season. There is still no clearer definition, or recognition of what such a definition should be based on. It seems probable that the most meaningful definition will eventually emerge from an enhanced understanding of the relevance of freshes to river ecosystem functioning. For instance, they appear to be extremely important from the perspectives of fish breeding and water quality. It is therefore recommended that basic research be designed to further understanding of how these and other lesser-researched components are affected by natural freshes. How important are freshes in terms of bringing relief to fish or other instream aquatic species, from extreme dry-season chemical or thermal conditions? What level of hydrological variability is encompassed in freshes, and what are the consequences of their loss in regulated rivers?

In parallel, research should continue to address management issues, such as how the operating rules for dams could help mitigate their effects on downstream reaches. In this respect, it is recommended that the experiments to trigger spawning of *Barbus capensis* in the Olifants River, reported on in Chapters 13 and 14, be repeated, in order to test the hypotheses developed. These hypotheses are that:

- the water temperature on the spawn beds for successful spawning of *B. capensis* should be at least 19 °C and stable or rising;
- if the fish are in breeding condition and these temperature conditions are met, then experimental releases from Clanwilliam Dam of the size used in this study will trigger spawning.

The research reported on in Chapters 13 and 14 has already stimulated plans for similar studies elsewhere in South Africa and in Australia. An opportunity exists to co-ordinate efforts and enhance collaborative endeavours.

Recommendations for further research on wet-season floods

The role of floods in maintaining silt-free cobble beds is poorly documented. With increasing levels of soil erosion in the country, and consequent increasing sedimentation of the rivers, river habitat for aquatic species must be deteriorating in both area and quality. Research is needed on the effects of siltation on river ecosystems, and the extent to which floods reverse or exacerbate this situation.

At the level of a student project, the Molenaars River Sediment Scour Model could be used to identify the discharge that would remove sand from the cobble bed. There was not time in the project reported on here to use the river's hydrological record to ascertain the return period of this discharge. This further step is essential, in order to compare it with the magnitude of floods normally recommended in instream flow assessments.

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TABLE OF CONTENTS

PAGE

EXECUTIVE SUMMARY ..... i-xxviii
ACKNOWLEDGEMENTS ..... xxix-xxxii
LIST OF TABLES ..... xxxvii-xxxviii
LIST OF FIGURES ..... xxxix-xliv
LIST OF PLATES ..... xlv-xlvi
LIST OF APPENDICES ..... xlvii-xlviii
TERMS OF REFERENCE ..... xlix-lii

1. INTRODUCTION 1
1.1 BACKGROUND 1
1.2 DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY 2
1.3 FLOW-RELATED RESEARCH LINKED TO THE BUILDING BLOCK METHODOLOGY 6

2. THE NEED FOR INSTREAM FLOWS AND EARLY INVESTIGATIONS OF THE FORM THEY SHOULD TAKE 7
2.1 INTRODUCTION 7
2.2 DEFINITION OF TERMS 7
2.3 THE NEED FOR INSTREAM FLOWS 8
2.4 EARLY INVESTIGATIONS INTO THE NATURE OF INSTREAM FLOWS REQUIRED FOR REGULATED RIVERS 10
2.5 THE WAY FORWARD 13

3. DEVELOPMENT AND NATURE OF THE BUILDING BLOCK METHODOLOGY 17
3.1 INTRODUCTION 17
3.2 THE ORIGIN OF THE BUILDING BLOCK METHODOLOGY 18
3.3 ASSUMPTIONS AND CHARACTER OF THE BUILDING BLOCK METHODOLOGY 19
3.4 THE SEQUENCE OF ACTIVITIES IN THE BUILDING BLOCK METHODOLOGY 21
3.5 THE POSITION OF THE BUILDING BLOCK METHODOLOGY IN THE DEPARTMENT OF WATER AFFAIRS AND FORESTRY'S PROCEDURE FOR WATER-RESOURCE DEVELOPMENTS 27
3.6 AN ASSESSMENT OF THE NATURE AND PRESENT STATUS OF THE BUILDING BLOCK METHODOLOGY 28
3.7 CONCLUSIONS 31

4. PRACTICAL APPLICATION OF THE BUILDING BLOCK METHODOLOGY 33
4.1 INTRODUCTION 34
4.2 SYNOPSIS OF PREPARATORY ACTIONS FOR AN IFR WORKSHOP 34
4.3 DETERMINATION OF THE EXTENT AND NATURE OF THE PROPOSED DEVELOPMENT 35
4.4 IDENTIFICATION OF RELEVANT SPECIALISTS WITH KNOWLEDGE OF THE RIVER 35

4.5 THE PLANNING MEETING FOR AN INSTREAM FLOW ASSESSMENT 41
4.6 THE WORKPLAN LEADING UP TO THE IFR WORKSHOP 44
4.7 DESIGN OF THE STARTER DOCUMENT 46
4.8 STRUCTURE AND PURPOSE OF THE SITE VISIT 47
4.9 THE IFR WORKSHOP 48
4.10 EXAMPLE OF AN IFR WORKSHOP OUTPUT: FOR IFR SITE 3 OF THE MOOI RIVER 56
4.11 POST-WORKSHOP ACTIVITIES 59

5. HYDROLOGICAL INFORMATION NEEDED AT AN IFR WORKSHOP 61
5.1 INTRODUCTION 61
5.2 THE GENERAL APPROACH TO PRESENTATION OF HYDROLOGICAL DATA 62
5.3 HYDROLOGICAL INFORMATION THAT SHOULD BE PRODUCED FOR THE IFR WORKSHOP STARTER DOCUMENT 62
5.4 VISUAL AIDS FOR THE TRANSFER OF HYDROLOGICAL INFORMATION AT THE WORKSHOP 86
5.5 DISCUSSION 86
5.6 CONCLUSIONS 88

6. SUMMARY OF PRESENT AND FUTURE DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY 89
6.1 INTRODUCTION 89
6.2 MILESTONES IN THE DEVELOPMENT AND APPLICATION OF THE BUILDING BLOCK METHODOLOGY 89
6.3 FURTHER DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY IN THE DEPARTMENT OF WATER AFFAIRS AND FORESTRY'S FEASIBILITY PHASE 94

7. INTRODUCTION TO THE STUDY OF ECOLOGICALLY RELEVANT LOW FLOWS, STUDY OBJECTIVES AND APPROACH 99
7.1 INTRODUCTION 99
7.2 STUDY OBJECTIVES 104
7.3 STUDY APPROACH 105

8. LOW FLOW PILOT STUDY 111
8.1 INTRODUCTION AND OBJECTIVES 111
8.2 METHODS OF DATA COLLECTION AND ANALYSIS 112
8.3 RESULTS 116
8.4 CONCLUSIONS 123

9. THE LOCATION, EXPERIMENTAL DESIGN AND PROCEDURES FOR IMPLEMENTATION OF THE LOW-FLOW STUDY 127
9.1 INTRODUCTION TO THE STUDY AREA 128
9.2 GENERAL DESCRIPTION OF THE STUDY RIVERS AND STUDY SITES 132
9.3 HYDROLOGICAL CHARACTERISATION OF THE STUDY SITES 148
9.4 EXPERIMENTAL DESIGN AND PROCEDURES FOR FIELD IMPLEMENTATION 161
9.5 METHODS OF DATA COLLECTION AND ANALYSIS 175

10. PRELIMINARY RESULTS OF THE LOW-FLOW STUDY 201
10.1 INTRODUCTION 202

10.2 HYDROLOGICAL EFFECTS AND LINKS WITH THE LONG-TERM HISTORICAL FLOW RECORD 202
10.3 EFFECTS OF NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS ON RIVER PHYSICAL AND CHEMICAL VARIABLES 217
10.4 EFFECTS OF NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS ON PHYSICAL BIOTOPE CHARACTER AND DYNAMICS 247
10.5 RESPONSES OF BENTHIC MACROINVERTEBRATE ASSEMBLAGES TO NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS 283

11. SYNOPSIS AND DISCUSSION OF THE PRELIMINARY RESULTS OF THE LOW-FLOW STUDY 329
11.1 INTRODUCTION 329
11.2 CHANGES IN DRY-SEASON BASEFLOW REGIME WITH SPECIFIC REFERENCE TO HYDROLOGICAL DISTURBANCE HISTORY 329
11.3 ECOLOGICAL RESPONSES OF UPPER RIVERS TO NATURAL DRY-SEASON BASEFLOW REGIMES AND REDUCTIONS IN BASEFLOW MAGNITUDES BELOW NATURAL LEVELS 333

12. A SEARCH FOR THE LINK BETWEEN WETTED PERIMETER AND EASILY COMPUTED HYDROLOGICAL INDICES OF BASEFLOW CHARACTERISTICS 345
12.1 INTRODUCTION 345
12.2 OBJECTIVES 346
12.3 DESCRIPTION OF THE DATA BASE 346
12.4 METHODS 348
12.5 RESULTS 350
12.6 DISCUSSION 354

13. SPAWNING BEHAVIOUR AND EARLY DEVELOPMENT OF THE CLANWILLIAM YELLOWFISH (*BARBUS CAPENSIS*; CYPRINIDAE), LINKED TO EXPERIMENTAL DAM RELEASES IN THE OLIFANTS RIVER, SOUTH AFRICA 355
13.1 INTRODUCTION 356
13.2 STUDY SITE 356
13.3 THE STUDY SPECIES 358
13.4 METHODS 360
13.5 RESULTS 363
13.6 DISCUSSION 376
13.7 CONCLUSIONS 380

14. LINKED EFFECTS OF DAM-RELEASED FRESHES AND WATER TEMPERATURE ON SPAWNING OF THE CLANWILLIAM YELLOWFISH *BARBUS CAPENSIS* 381
14.1 INTRODUCTION 381
14.2 STUDY SITE 382
14.3 METHODS 382
14.4 RESULTS 386
14.5 DISCUSSION 402
14.6 CONCLUSIONS 405

15. CRITICAL HYDRAULIC CONDITIONS FOR THE MOBILISATION OF SEDIMENT IN A CAPE COBBLE-BED RIVER 407
15.1 INTRODUCTION 407
15.2 OBJECTIVE OF THIS RESEARCH 408
15.3 APPLICATION OF A DISCHARGE-SCOUR RELATIONSHIP 409
15.4 A MATHEMATICAL MODEL DESCRIBING THE SCOUR OF SEDIMENT IN A RIVER 409
15.5 A SUMMARY OF THE CONSERVATION OF POWER LAW AND THE POWER APPROACH 409
15.6 DERIVATION OF THE MATHEMATICAL MODEL 410
15.7 THE RIVER SELECTED TO DETERMINE THE DISCHARGE-SCOUR RELATIONSHIP 413
15.8 FIELD TRIPS TO THE MOLENAARS RIVER 413
15.9 MEASUREMENTS TAKEN ON THE FIELD TRIPS 413
15.10 IDENTIFICATION OF THE SIGNIFICANT DISCHARGE 414
15.11 IDENTIFICATION OF POSSIBLE SIGNIFICANT DISCHARGE SCENARIOS 418
15.12 IMPLICATIONS OF THE SIGNIFICANT DISCHARGES 418
15.13 THE MOLENAARS RIVER SEDIMENT SCOUR MODEL 419
15.14 STEPS IN THE APPLICATION OF THE MOLENAARS RIVER SEDIMENT SCOUR MODEL TO OTHER RIVERS 421
15.15 COMMENTS ON THE DEVELOPMENT OF THE MODEL 423
15.16 CONCLUSIONS 423

16. CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH 425
16.1 INTRODUCTION 425
16.2 EFFECTS OF DIFFERENT MAGNITUDE FLOWS ON RIVERINE ECOSYSTEMS 428

REFERENCES 437-452

LIST OF TABLES

(Table captions are abbreviated) *PAGE*

Table 3.1	An example of a blank IFR table	24
Table 3.2	Department of Water Affairs and Forestry's engineering and environmental planning stages for a water-resource development, showing the positioning of the IFR workshop	27
Table 4.1	The ideal sequence of actions for an IFR workshop, compared to the actual programme for the Mooi River IFR workshop	36
Table 4.2	Preparatory actions for an IFR workshop	37-38
Table 4.3	Categories of information necessary for the IFR workshop	35
Table 4.4	Agenda for the Mooi River IFR planning meeting	41
Table 4.5	Considerations when delineating the extent of the study area	42
Table 4.6	River characteristics to identify when partitioning the study area into stretches of river, each of which will be represented by an IFR site	42
Table 4.7	Considerations when selecting IFR sites within the chosen IFR reaches	43
Table 4.8	Draft proposal on the IFR workshop objective and Desired State of the Mooi River in the study area	43
Table 4.9	Criteria for selection of the date and venue for the IFR workshop	44
Table 4.10	Standard table of contents of the starter document for an IFR workshop	47
Table 4.11	First draft of lists of aspects to consider, discuss or observe during an IFR site visit	51
Table 4.12	Programme for the Mooi River IFR workshop	53
Table 4.13	The IFR recommended for IFR site 3 on the Mooi River	54
Table 5.1	Patched record of flow volume at gauging weir V2H005	69
Table 5.2	Summary statistics of daily discharge for present conditions for the months July to December, for all the years of record, with monthly total volumes	78
Table 5.3(A)	The estimated frequency of virgin daily discharges in each month for selected class intervals	80
Table 5.3(B)	The estimated percentage of virgin daily discharges in each month for selected classintervals	81
Table 5.4	Daily average discharge for a wet month (February), recorded at gauging weir A5H004	82
Table 5.5	The recurrence intervals of different magnitude floods	84
Table 7.1	Selection criteria for sites with Department of Water Affairs and Forestry hydrological gauging weirs	108
Table 7.2	Eight potential sites for the main low-flow study	109
Table 8.1	Gauging weir co-ordinates and associated information on the historical flow record, for the eight pilot sites	113
Table 8.2	Physical description of the eight pilot sites and the two subreaches within each site	114
Table 8.3	Instantaneous discharges and wetted channel widths for the eight pilot sites, for the March 1994 sampling trip	123
Table 8.4	Summary of water chemistry conditions at the eight pilot sites, March 1994	124-125
Table 9.1	Summary of the Department of Water Affairs and Forestry historical, average daily discharge data, computed to date from raw gauging weir records at the four study sites, for use in the hydrological analyses	150
Table 9.2	Observed original and patched Mean Annual Runoff (MAR), estimated naturalised MAR (Mm3) and naturalised runoff coefficients for the four study sites	153
Table 9.3	The timing and duration of the pre-impact, impact and post-impact phases of the low-flow experiment	174
Table 9.4	Summary of the types of data and number of samples collected, and sampling dates, for all six sampling trips during the low-flow study	176-179

Table 10.1	Mean and modal natural average daily discharges, and average daily discharges corresponding with 5%, 25%, 50% (median), 75%, and 95% exceedance values derived from *daily flow duration curves*	210
Table 12.1	Characteristics of the study rivers	347
Table 12.2	Hydrological indices calculated for the eleven study rivers	352-353
Table 13.1	Fish species occurring in the Olifants River system, their distributions and Red Data Book status as of 1987	359
Table 13.2	Release pattern of experimental freshes from Clanwilliam Dam wall, Olifants River system, in November to December 1993 and 1994.	361
Table 13.3	Ranges in water quality variables measured at spawn bed B in the Olifants River system, before, during and after experimental freshes F1, F2 and F3 in December 1993	366
Table 13.4	Spot readings of water conditions at spawn bed B in the Olifants River and at the larval site, linked to the 1994 experimental freshes F1/94 and F2/94	367
Table 13.5	Movement of *Barbus capensis* at spawn bed A in the Olifants River in December 1994	368
Table 13.6	Locality, dates and times of various *Barbus capensis* developmental stages collected at spawn beds A and B below Clanwilliam Dam wall, Olifants River system	371-373
Table 13.7	Larval and early juvenile *Barbus capensis* developmental records, collected 2 km downstream of spawn sites A and B, Olifants River system, December 1993 and 1994.	374
Table 14.1	Characteristics of fish survey pools	389
Table 14.2	Fish counts at pools 1, 2 and 3 on three occasions before and one occasion after the first experimental fresh	390
Table 14.3	Fish counts on spawn beds A and B per 10 minutes and observed movements in two pools per 10 minutes	394-395
Table 15.1	The relationships tested between the square roots of the sediment diameters and the average velocities above the test section	417

LIST OF FIGURES

(Figure captions are abbreviated) *PAGE*

Figure 2.1 The natural flow regime of an hypothetical perennial river in central Natal, and those parts of it deemed most important for ecological maintenance of the river 14

Figure 2.2 The natural flow regime of an hypothetical ephemeral river in the northern Transvaal, and those parts of it deemed most important for ecological maintenance of the river 15

Figure 3.1 Focusing thought on (A) perceived important features of a river's natural flow regime and (B) which of these should be retained in an IFR 20

Figure 3.2 An hypothetical IFR created using the Building Block Methodology 23

Figure 3.3 Sequence of activities in the Building Block Methodology. 25

Figure 4.1 One example of the sequence of events from the IFR planning meeting to the IFR workshop 45

Figure 4.2 The sequence of events in an IFR workshop 49

Figure 4.3 Example of a surveyed IFR cross-section, showing the stage-discharge relationship and the distribution of vegetal species 50

Figure 5.1 Diagrammatic map of the study area for an IFR workshop, showing the location of IFR sites, relevant gauging weirs and extant and proposed dams 64

Figure 5.2 Naturalised total annual runoff at gauging weir V2H002 for all the years of simulation, to show annual variability 66

Figure 5.3 Cumulative deviation of total annual runoff about the virgin Mean Annual Runoff (MAR), for all years of record, to show wet and dry periods as defined by the hydrologist 67

Figure 5.4 Naturalised monthly flow volumes for all years of record 68

Figure 5.5 Monthly distributions of mean virgin flow volumes 70

Figure 5.6 Seasonal distribution of virgin and present mean monthly flow volumes for site IFR 1/96 on the Berg River 72

Figure 5.7 An hypothetical simulation of the seasonal reversal in mean daily discharge caused by the presence of a proposed dam 73

Figure 5.8 Simulated virgin and future daily hydrographs for the Olifants River (Western Cape) upstream of Clanwilliam Dam, to illustrate the delayed onset of winter floods caused by the proposed Rosendaal Dam 74

Figure 5.9 Simulated virgin and future daily hydrographs for the Olifants River (Western Cape) upstream of Clanwilliam Dam, to illustrate the higher baseflow in the dry season (January to April) than in the wet season (May to August), caused by the proposed Rosendaal Dam 75

Figure 5.10 Simulated virgin and future daily discharge hydrographs for the Olifants River (Western Cape) at Grootfontein, to illustrate that, with the proposed Rosendaal Dam in place, freshes would still occur in the dry season (November to January) 76

Figure 5.11 Box and whisker plots illustrating the distribution of virgin daily discharges at IFR site 1/96 on the Berg River 77

Figure 5.12 Virgin, present and simulated future flow duration curves for each calendar month, averaged over the period of record October 1960-September 1987 for gauging weir B7H009 on the Olifant-Sand River 83

Figure 5.13 Flood hydrographs at gauging weir H1H013 on the Koekedouw River, showing the magnitude, duration and hydrograph shape for "last floods of the wet seasons" selected from the overall hydrological record 85

Figure 6.1 Suggested flow path for Building Block Methodology-related activities during the Department of Water Affairs and Forestry's Feasibility Phase 98

Figure 8.1 Dendrogram showing hierarchical agglomerative classification, with group-average linking, of the eight pilot sites, based on average sample abundances (n = 6 samples) for all families of benthic macroinvertebrates per 0.1 m^2 of riffle 117

Figure 8.2 Ordination, using non-metric MDS, of the eight pilot sites, based on the same similarity matrix as Figure 8.1 117

Figure 8.3 Dendrogram showing hierarchical agglomerative classification, with group-average linking, of the eight pilot sites, based on average sample abundances for each subreach (n = 3 samples) for all families of benthic macroinvertebrates per 0.1 m^2 of riffle 119

Figure 8.4 Ordination, using non-metric MDS, of the eight pilot sites, based on the same similarity matrix as Figure 8.3 119

Figure 8.5 Dendrogram showing hierarchical agglomerative classification, with group-average linking, of all 48 benthic macroinvertebrate samples collected in the March 1994 pilot study. The samples represent family-level abundances, per 0.1 m^2 of riffle 120

Figure 8.6 Ordination, using non-metric MDS, of all 48 benthic macroinvertebrate samples collected in the March 1994 pilot study. The plot is based on the same similarity matrix as Figure 8.5 121

Figure 9.1 Study area indicating the locations of the study sites on the Elands, Molenaars, Du Toits and Riviersonderend Rivers, southwestern Cape 129

Figure 9.2 Characteristic seasonal flow patterns at the four study sites 155

Figure 9.3 Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Elands site, for the period 11/03/1969 to 26/02/1997 156

Figure 9.4 Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Molenaars site, for the period 27/02/1969 to 31/01/1997 157

Figure 9.5 Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Du Toits site, for the period 01/10/1964 to 31/08/1992 158

Figure 9.6 Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Riviersonderend site, for the period 01/10/1964 to 30/06/1992 159

Figure 9.7(A)-(C) Hypothetical abundances of "Species X" at the impact and control locations, and the difference of the abundances, as functions of time, in three versions of assessment of an impact (e.g. experimental flow reduction). 162

Figure 9.7(D) The three functions to be distinguished in the characterisation of the differences between the Control and Impact locations 163

Figure 9.8(A) Schematic of the Elands study site, showing the 60 m control and impact locations 164

Figure 9.8(B) Schematic of the Molenaars study site, showing the 60 m control and impact locations 165

Figure 9.8(C) Schematic of the Du Toits study site, showing the 60 m control and impact locations 166

Figure 9.8(D) Schematic of the Riviersonderend study site, showing the 60 m control and impact locations 167

Figure 9.9 Schematic illustrating the identification of physical biotopes at the transect and cell levels within a river reach 183

Figure 9.10 Schematic illustrating the division of each sample stone into three separate sample parts, from which benthic macroinvertebrates were collected 192

Figure 9.11 Schematic illustrating the three axes measured in the field for each sample stone in order to calculate surface area 192

Figure 10.1 Instantaneous discharges ($m^3 s^{-1}$) for all three phases of the low-flow study, at the four sites 204

Figure 10.2 Comparison of natural instantaneous discharges ($m^3 s^{-1}$) for the Elands control (Natural Instant (C)) and impact (Natural Instant (I)) locations with various hydrological indices,for the duration of the low-flow study 209

Figure 10.3 Comparison of natural instantaneous discharges ($m^3 s^{-1}$) for the Molenaars control (Natural Instant) and impact (Impact Instant) locations with various hydrological indices, for the duration of the low-flow study 212

Figure 10.4 Comparison of natural instantaneous discharges ($m^3 s^{-1}$) for the Du Toits control (Natural Instant) and impact (Impact Instant) locations with various hydrological indices, for the duration of the low-flow study 214

Figure 10.5 Comparison of natural instantaneous discharges ($m^3 s^{-1}$) for the Riviersonderend control (Natural Instant) and impact (Impact Instant) locations with various hydrological indices, for the duration of the low-flow study 216

Figure 10.6 Maximum, instantaneous and minimum temperatures (°C) recorded at the Elands (E) control and impact locations, during the three phases of the low-flow study 220

Figure 10.7 Maximum, instantaneous and minimum temperatures (°C) recorded at the Molenaars (M) control and impact locations, during the three phases of the low-flow study 221

Figure 10.8 Maximum, instantaneous and minimum temperatures (°C) recorded at the Du Toits (D) control and impact locations, during the three phases of the low-flow study 222

Figure 10.9 Maximum, instantaneous and minimum temperatures (°C) recorded at the Riviersonderend (R) control and impact locations, during the three phases of the low-flow study 225

Figure 10.10 Changes in pH at the four sites, during the three phases of the low-flow study 226

Figure 10.11 Changes in total alkalinity (mEq ℓ^{-1}) at the four sites, during the three phases of the low-flow study 227

Figure 10.12 Changes in conductivity (mS m^{-1}) at the four sites, during the three phases of the low-flow study 230

Figure 10.13 Changes in total dissolved solids (mg ℓ^{-1}) at the four sites, during the three phases of the low-flow study 231

Figure 10.14 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Elands site, during the three phases of the low-flow study 234

Figure 10.15 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Molenaars site, during the three phases of the low-flow study 235

Figure 10.16 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Du Toits site, during the three phases of the low-flow study 236

Figure 10.17 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Riviersonderend site, during the three phases of the low-flow study 238

Figure 10.18 Changes in Na^+ (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 239

Figure 10.19 Changes in K^+ (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 240

Figure 10.20 Changes in Ca^{2+} (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 241

Figure 10.21 Changes in Mg^{2+} (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 243

Figure 10.22 Changes in $C\ell^-$ (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 244

Figure 10.23 Changes in SO_4^{2-} (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 245

Figure 10.24 Changes in nitrate (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 246

Figure 10.25 Changes in nitrite (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 248

Figure 10.26 Changes in phosphate (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 249

Figure 10.27 Changes in silicon (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study 250

Figure 10.28 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 1) in the control location, Elands River, at three natural discharges 252

Figure 10.29 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 3) in the impact location, Elands River, at three natural discharges 253

Figure 10.30 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 2) in the control location, Molenaars River, at three natural discharges 254

Figure 10.31 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 1) in the control location, Du Toits River, at three natural discharges 255

Figure 10.32 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 2) in the control location, Riviersonderend River, at three natural discharges 256

Figure 10.33 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 3) in the impact location, Molenaars River, at three discharges 259

Figure 10.34 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 3) in the impact location, Du Toits River, at three discharges 260

Figure 10.35 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 4) in the impact location, Riviersonderend River, at three discharges 261

Figure 10.36 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 2) in the control location, Elands River, at three natural discharge. 262

Figure 10.37 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 4) in the impact location, Elands River, at three natural discharges 263

Figure 10.38 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 1) in the control location, Molenaars River, at three natural discharges 264

Figure 10.39 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 2) in the control location, Du Toits River, at three natural discharges 265

Figure 10.40 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 1) in the control location, Riviersonderend River, at three natural discharges 266

Figure 10.41 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 4) in the impact location, Molenaars River, at three discharges 269

Figure 10.42 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 4) in the impact location, Du Toits River, at three discharges 270

Figure 10.43 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 3) in the impact location, Riviersonderend River, at three discharges 271

Figure 10.44 Average wetted widths calculated from two cross-sections in each control and impact location, for the four sites 273

Figure 10.45 Riffle widths calculated from a single cross-section in each control and impact location, for the four sites. 274

Figure 10.46 Run widths calculated from a single cross-section in each control and impact location, for the four sites 275

Figure 10.47 Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Elands River 279

Figure 10.48 Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Molenaars River 280

Figure 10.49 Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Du Toits River 281

Figure 10.50 Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Riviersonderend River 282

Figure 10.51 Numbers of families and higher taxa recorded in the control and impact locations at the four sites, under conditions of natural and unnatural baseflow 287

Figure 10.52 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Elands site, at natural baseflows. 288

Figure 10.53 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow 289

Figure 10.54 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow 290

Figure 10.55 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow 291

Figure 10.56 Total numbers of individuals (0.1 m^{-2}) recorded in the control and impact locations at the four sites, under conditions of natural and unnatural baseflow 293

Figure 10.57 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Elands site, at natural baseflows. 297

Figure 10.58 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow 298

Figure 10.59 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow 299

Figure 10.60 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow 300

Figure 10.61 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) in the control and impact locations at the four sites, under conditions of natural and unnatural baseflow 303

Figure 10.62 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Elands site, under a natural baseflow regime....312

Figure 10.63 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow....313

Figure 10.64 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow....314

Figure 10.65 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow....315

Figure 10.66 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Elands site, under a natural baseflow regime....316

Figure 10.67 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow....317

Figure 10.68 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow....318

Figure 10.69 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow....319

Figure 10.70 Shannon-Weiner diversities (H') for the control and impact locations at all four sites, for the pre-impact (December), impact (February) and post-impact (April) phases of the low-flow study....326

Figure 10.71(A) Classification of benthic macroinvertebrate assemblages at all four sites, during the pre-impact (December, 12), impact (February, 02) and post-impact (April, 04) phases of the low-flow study....327

Figure 10.71(B) Ordination of the same data set, by non-metric multidimensional scaling....327

Figure 10.72(A) Classification of benthic macroinvertebrate assemblages of riffles and runs in the control and impact locations, at the Du Toits site, during the impact phase (February)....328

Figure 10.72(B) Ordination of the same data set, by non-metric multidimensional scaling....328

Figure 12.1 Wetted perimeter-discharge curves for four cross-sections of the Du Toits River, showing the inflection point of Wetted Bed Flow (WBF)....349

Figure 12.2 Wetted Bed Flow (WBF) as a ratio of: (A) mean annual runoff (MAR); average = 37.2%; (B) percentile of flow duration curve; average = 46.9%; (C) 1:2 year maximum average daily flow (MADF); average = 2.5%....351

Figure 13.1 Olifants River system (western Cape) showing the *Barbus capensis* spawn beds A and B, the road bridge and the larval fish site below the Clanwilliam Dam wall....357

Figure 13.2 (A) Movement of water through the study area below the dam in December 1993 is illustrated by the dam release pattern and measured discharge levels at the larval site. (B) Water level at spawn bed B. (C) conductivity at spawn bed B....365

Figure 13.3 Plot of hydraulic conditions at *Barbus capensis* larval fish site below Clanwilliam Dam....375

Figure 14.1 Sketch map of the spawn site for *Barbus capensis* in the middle reaches of the Olifants River, South Africa....385

Figure 14.2 Maximum and minimum water temperatures at the farm Hoopvol, 7 km downstream of Clanwilliam Dam on the Olifants River, from 4 September to 10 November 1995....387

Figure 14.3 Water temperatures at the spawn site and at Hoopvol, and water release and spillage patterns at Clanwilliam Dam, both in the middle reaches of the Olifants River, from 10 November to 28 December 1995....388

Figure 14.4 Water level, water temperature, air temperature and dissolved oxygen levels at the spawn site during the experimental hypolimnetic and epilimnetic releases from Clanwilliam Dam, related to the presence of *Barbus capensis* on spawn bed A....392

Figure 14.5 Vegetation and substrata maps (A), and contour maps of Froude numbers (B), at a dam release of 3.4 $m^3 s^{-1}$, for the channel area encompassing the designated spawn bed A in the Olifants River 398

Figure 14.6 Contour maps of water depths (A) and average column velocities (B) at a dam release of 3.4 $m^3 s^{-1}$, for the channel area encompassing the designated spawn bed A in the Olifants River. 399

Figure 14.7 Vegetation and substrata maps (A), and contour maps of Froude numbers (B), at a dam release of 8.0 $m^3 s^{-1}$, for the channel area encompassing the designated spawn bed B in the Olifants River. 400

Figure 14.8 Contour maps of water depths (A) and average column velocities (B) at a dam release of 8.0 $m^3 s^{-1}$, for the channel area encompassing the designated spawn bed B in the Olifants River 401

Figure 15.1 Diagrammatic representation of the similarity between cobble-bed and duned sand-bed rivers 411

Figure 15.2 Schematic view of a long section of a river showing the variables used to describe the conditions in the river 411

Figure 15.3 The profile of transect 4, as surveyed in May 1995 415

Figure 15.4 The relationship of wetted perimeter (wetted P), wetted area (xsect area), and hydraulic radius (hyd rad) to depth, for transect 4 415

Figure 15.5 Stage-discharge graph for transect 4 using measured data, and theoretical data predicted using the Chézy equation and the transect 416

Figure 15.6 Hydrograph of weir H1H018 on the Molenaars River over the period 10/05/95-15/10/95 416

Figure 15.7 Graph comparing average sediment sample diameter with maximum flow velocity above sample area over 3 days before sample collection 420

Figure 15.8 The Molenaars River Sediment Scour Model and the uniform sand bedded river equation 420

Figure 15.9 Critical discharges at transects 1, 3 to 6 in the Molenaars River 422

LIST OF PLATES

(Plate captions are abbreviated) *PAGE*

Plate 9.1	The control location of the Elands study site	170
Plate 9.2	The impact location of the Elands study site	170
Plate 9.3	The control location of the Molenaars study site	171
Plate 9.4	The impact location of the Molenaars study site	171
Plate 9.5	The control location of the Du Toits study site	172
Plate 9.6	The impact location of the Du Toits study site	172
Plate 9.7	The control location of the Riviersonderend study site	173
Plate 9.8	The impact location of the Riviersonderend study site	173

LIST OF APPENDICES

(Appendix captions are abbreviated)

Appendix 1.1 Analysis of the effect of freshes in river flow regimes.

Appendix 1.2 A preliminary study of the effect of spates on the species composition and abundance of benthic macroinvertebrate assemblages on rocks in two biotopes in a Cape mountain stream.

Appendix 1.3 The effects of anthropogenic perturbations on the aquatic invertebrate fauna of the Olifants River system.

Appendix 1.4 Researching the implications of pre-determined release policies and dam sizes on the natural flow regimes of Western Cape rivers.

Appendix 2.1 Specialists who responded to the questionnaire on flows for river maintenance in 1991 (see Section 2.4.1).

Appendix 2.2 Delegates attending the workshop "Toward a methodology for instream flow assessments for South African rivers" at Rand Afrikaans University on 7 July 1993 (see Section 2.4.2).

Appendix 8.1 Benthic macroinvertebrate abundances per 0.1 m^2 of riffle for the eight pilot sites, March 1994.

Appendix 9.1 Explanation of the codes used when surveying the cross-sections and biotope maps in the control and impact locations at each site.

Appendix 9.2 Benthic macroinvertebrate samples collected from the wetted channel edges of the four impact locations, during the flow reduction phase of the low-flow study.

TERMS OF REFERENCE

The overall aim of this study, in terms of the agreement between the Water Research Commission and the University of Cape Town, was to enhance the ability of river scientists to advise on sustainable development of the water resources of the country's rivers by:

- continuing the development of scientifically acceptable methodologies for assessing the water quantity requirements of riverine ecosystems;

- increasing understanding of the effects of different magnitudes of flow on riverine biotas, through focused research on South African rivers and, in doing so, enhancing the efficacy of such instream flow methodologies.

More specifically, the aims were divided into a series of primary and secondary objectives.

Primary objectives

- Compile literature syntheses on the responses of riverine ecosystems to low flows, and to higher flows and flood events. In each review, identify the ecosystem components that appear to be particularly susceptible to changes in flow.

- Initiate research on the effects of abnormally low dry-season baseflows, in order to identify flows below which substantial changes in ecosystem structure and functioning occur.

- Begin research on the requirements for high flows and floods, at both the ecosystem level and the level of aquatic habitat.

- From these research activities, derive definitions or descriptions of a series of ecologically meaningful flows, that is, flows that riverine biotas react to.

- Continue analyses of the hydrological records held by the Department of Water Affairs and Forestry (DWAF), to clarify regional differences in annual flow regimes.

- Using all of this knowledge, continue development of the Building Block Methodology (BBM) for determining the instream flow requirements of rivers.

Secondary objectives

These objectives were to be attempted only if time allowed.

- Create, in liaison with DWAF, a number of groundtruthing stations across the country for studying the losses and gains in aquatic habitat with changes in discharge.

- Compile a report on the long-term spatial and other effects of modified flow regimes on South African riverine ecosystems.

- Initiate a Southern Hemisphere newsletter on the topic of "Water and the environment".

- Promote, if feasible, an upgrading of the hydraulic component of the Instream Flow Incremental Methodology (IFIM) model Physical Habitat Simulation Version II (PHABSIM II).

Refinement of the principal objectives

During the course of the project, its direction was refined, and this was approved by the steering committee on 23 March 1994. Essentially, three principal objectives then covered virtually all of the work proposed for the project. It was agreed that the project staff would:

- continue development of the BBM for use for instream flow assessments;

- improve estimates of water quantity requirements made when using the BBM, through focused research on South African rivers;

- concentrate such research on the flow-related requirements of the instream biota, acknowledging that flow-related aspects of river channels and banks, floodplains and wetlands would be addressed by other research groups in the country.

It was recognised that investigation of only a few of the many potentially ecologically important flows could be investigated within a three-year project.

The contract objectives, listed in these terms of reference, were met to the following extent

In terms of the three amended principal objectives

- Development of the BBM continued through application and refinement in water-resource projects. The methodology is described in Chapters 2-6. Project funds are also being used to produce a comprehensive manual of the methodology. This should be completed during 1998.

- The second and third amended objectives refer to research designed to improve application of the BBM. This research is described in Chapters 7-15. Research focused on three flow components used in the BBM: summer low flows, summer freshes and winter floods. The research increased understanding of how such flows influence river ecosystem functioning, and highlighted further uncertainties, as documented in the various chapters.

Referring back to the initial primary objectives

- The literature review of the responses of riverine ecosystems to various magnitude flows exists in draft form, and will be completed during 1998.

- The research on high flows and abnormally low flows is documented in Chapters 7-15.

- Descriptions of a series of ecologically meaningful flows, derived from these chapters, are provided in Chapter 16.

- Continued analyses of DWAF hydrological records did not take place.

- Development of the BBM was completed to a point that was recognised as the end of Phase I. Most components of the methodology would benefit from further refinement. This is discussed in Chapter 16.

Referring back to the initial secondary objectives

- None of these objectives were addressed. However, in subsequent research after this project, the subject of hydraulic modelling of the habitats of instream biota is being given further attention by the authors of this report and colleagues at both the University of Cape Town and University of Stellenbosch.

1. INTRODUCTION

Written by J. King and R. Tharme

1.1 BACKGROUND
1.2 DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY
1.2.1 The Building Block Methodology in the context of international developments in instream flow assessment
1.2.2 The first stages in the evolution of the Building Block Methodology
1.3 FLOW-RELATED RESEARCH LINKED TO THE BUILDING BLOCK METHODOLOGY

1.1 BACKGROUND

For the past decade, several South African institutions, including the Department of Water Affairs and Forestry (DWAF), research funding bodies, academic institutions and conservation authorities have been involved in activities designed to contain, and hopefully in some cases reverse, deterioration in the flow-related condition of the country's rivers. Much of this deterioration has been caused by general catchment degradation and by an increasing demand for water from a rapidly growing population. In this semi-arid country of about 45 million people, 12 million still do not have access to an adequate supply of potable water, and nearly 21 million lack basic sanitation (DWAF 1994a). With the demand for water continually increasing, the last few decades have seen large-scale, direct abstraction of water from rivers and an extensive programme of dam building (DWA 1986). Parts of several large rivers in heavily populated areas have already virtually ceased to exist except as channels for floods or waste disposal. Those human communities living beside such rivers, and directly reliant on them, probably bear the most severe consequences of the loss of flow and of other attributes of the riverine ecosystems. Additionally, the riverine ecosystems themselves progressively lose their structural and functional integrity (Davies *et al.* 1993), which in turn leads to considerable long-term, long-distance costs to the country that are largely externalised in water-resource developments.

With escalating development of the country's water resources to meet human demand, and because of concern for the condition of rivers, there has been a concomitant intensifying pressure on the scientific community to advise on flows for sustaining riverine ecosystems. King and Tharme (1994) outlined the development of this recent field of research in South Africa, and concluded their report with a description of a South African approach to instream flow assessment that was in the first stages of development. This approach, based on the need to produce answers on the amount of water required for river maintenance, often using limited data, has been named the Building Block Methodology (BBM) (King and Louw in press). The project reported upon here was designed to continue development of the BBM and to initiate relevant research that would enhance both its scientific foundation and its application.

1.2 DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY

1.2.1 The Building Block Methodology in the context of international developments in instream flow assessment

Most instream flow methodologies reflect the fact that different magnitudes of flow affect the riverine ecosystem in various ways. Higher flows provide more living space for aquatic plants and animals than do lower ones, and so influence the overall productivity of a system (Davies *et al.* 1993). Floods scour river channels, increase bank storage of water, cleanse rocks and re-sort alluvial bed material, thus affecting the amount of physical habitat available for different kinds of aquatic and riparian biota (Gordon *et al.* 1992). Changes in the magnitude of flow may provide cues for seed setting of riparian trees (Van Coller and Rogers 1996), upstream fish migrations (Wootton 1990) or mass emergences of aquatic insects (Hynes 1970). In countless similar ways, the orderly working and efficient functioning of a river ecosystem is inextricably linked to the natural seasonal cycles of high and low flows in the river. The subject is re-visited in Chapter 2. Recognition of the vital role that flow regimes play in the functioning of healthy rivers has led to increasing attention being paid to the development of methodologies for assessing instream flow requirements (IFRs) for river maintenance.

Tharme (1996, 1997c) reviewed the international literature on the most commonly applied and best known instream flow methodologies, and found that the majority of them fall into four more or less distinct categories. The first and typically simplest type of methodology relies on the use of hydrological data, usually in the form of historical virgin and/or present-day daily or monthly flow records, for making instream flow recommendations. Hydrology-based methodologies include, among others, the 10th Percentile Method (Northern Great Plains Resource Program 1974, cited in Growns and Kotlash 1994), September Median Flow Method (Reiser *et al.* 1989), Texas Method (Matthews and Bao 1991), various forms of flow duration curve (FDC) analysis (see Tharme 1996, for examples), as well as the Tennant Method (Tennant 1976). The latter method is the second most widely used technique in North America (Reiser *et al.* 1989).

Generally, such office-based, standard-setting methodologies (Prewitt and Carlson 1980) provide only simplistic, low resolution minimum flow recommendations, such as percentages of MAR on a seasonal or monthly basis. Tharme (1996, 1997c) discusses further limitations inherent in such methodologies. Their application is recognised as being most appropriate at the reconnaissance level of a basic two-tier system of instream flow assessment, especially where regionalisation of instream flow allocations is required on a river ecotype basis (Petts *et al.* 1996; Tharme 1997c; Dunbar *et al.* 1998). Recently, methodologies based on hydrological data have been further extended or newly developed to incorporate professional judgement, and various hydraulic, biological and geomorphological criteria, in their foundation and/or application (Tharme 1997c). For instance, a modified version of the Tennant Method incorporating specialist expertise and fish periodicity data, has been developed for state wide use in Alaska (Estes 1996).

The majority of approaches based on hydrology are of low flexibility and do not address the dynamic nature of the flow regime or include ecological data. However, some recent techniques, like the Range of Variability Method (Richter *et al.* 1996, 1997), have particular merit for future application. This is because they incorporate flow variability in the form of several potentially ecologically relevant hydrological

indices. Tharme (1997c) suggests the use of these more advanced kinds of hydrological index methodologies locally, at the broadest level of instream flow assessment.

Hydraulic rating and habitat simulation methodologies form the second and third categories of instream flow methodologies respectively, with both utilising a quantifiable incremental relationship between instream habitat and discharge to calculate instream flow requirements (IFRs). Hydraulic rating methodologies are identified by Loar *et al.* (1986) as single cross-section methodologies that recognise changes in various individual hydraulic variables, such as wetted perimeter or maximum depth, as a surrogate for habitat factors that limit riverine biota. These are used to develop a relationship between habitat and discharge for instream flow recommendations. Commonly, such approaches are based on the simplistic and limiting assumption that riverine flow requirements can be directly related to wetted riffle area, and use various breakpoints on habitat-discharge curves to identify suitable instream flows. An example of such an approach is the Wetted Perimeter Method (see Gippel and Stewardson 1996), the most commonly used methodology of this type worldwide and the third most applied one in North America (Reiser *et al.* 1989).

The strengths and weaknesses of hydraulic rating methodologies are summarised by Tharme (1996, 1997c). In terms of their limitations, the methodologies are reliant on the highly simplistic assumption that a single hydraulic variable can adequately represent the IFRs of a target species for a particular activity. They cannot readily be used for certain components of the riverine ecosystem, such as riparian vegetation, and the IFRs of instream species are ascertained only by inference. Moreover, placement of the single cross-section is critical to the results obtained. Although the outputs are of fairly low resolution, hydraulic rating methodologies allow a rapid, focused and flexible assessment of the IFRs for one or a few species or activities, and for the maintenance of habitat areas for requirements such as invertebrate production.

Hydraulic rating methodologies are viewed by Tharme (1996) and Dunbar *et al.* (1998) as precursors of the more sophisticated category of habitat simulation or modelling methodologies. This latter category of methodologies are used to simulate the changing availability of physical habitat with changing discharge. Such microhabitat is modelled in terms of various hydraulic variables, commonly depth, velocity, substratum, cover and benthic shear stress, using data from multiple cross-sections. Changes in the simulated availability of hydraulic habitat are combined with biological information on the relative suitability and quantity of such microhabitat for particular target species, lifestages or species assemblages, under different flow regimes, to allow identification of discharges that provide optimum habitat. This information is used when setting recommendations for IFRs.

Tharme (1996) reviews some of the earlier habitat simulation approaches, including the Oregon Usable Width Method (Thompson 1974, cited in Stalnaker and Arnette 1976) and Multiple Transect Analysis (Richardson 1986), among many others. Currently, several computer-based habitat modelling methodologies that represent the state-of-the-art are in use or under development worldwide. Most notable of these is the Instream Flow Incremental Methodology (IFIM) (Milhous *et al.* 1989), which is still the most widely used methodology in the world (Gore and Nestler 1988; Reiser *et al.* 1989). Other very similar methodologies are the River Hydraulics and Habitat Simulation program (RHYHABSIM; Jowett 1989), the Evaluation of Habitat Method (EVHA; Ginot 1995, cited in Dunbar *et al.* 1998); the Computer Aided Simulation Model for Instream Flow Requirements in regulated streams (CASIMIR; Jorde 1996); the River

Simulation System (RSS; Killingtviet and Fossdal 1994, cited in Dunbar *et al.* 1998), and the Riverine Community Habitat Assessment and Restoration Concept (RCHARC; Nestler *et al.* 1994, cited in Hart in prep.). Tharme (1996, 1997c), Dunbar *et al.* (1998) and Gippel and Stewardson (1996) provide the most recent reviews of these methodologies.

Habitat simulation techniques are deemed more appropriate than hydrological or hydraulic rating methodologies in high profile cases, for rivers of exceptional conservation importance, or where negotiated decisions on instream versus offstream allocations of water are to be made (Lamb 1989; Tharme 1996; Dunbar *et al.* 1998). They provide for a dynamic assessment of changes in physical habitat with discharge, at a scale that is relevant to the instream biota, and the IFRs of several species, life stages and assemblages can be compared under various flow scenarios. Moreover, the models enable efficient processing of large amounts of data and often have sophisticated hydrological and habitat time series components, as well as outputs at a high degree of spatial and temporal resolution. However, there are several weaknesses inherent in such methodologies, including, *inter alia*, their single-species bias and the primary assumption that species are mainly limited by the availability of suitable microhabitat (Tharme 1996).

In recent years, international studies of the flow requirements for riverine ecosystems have increasingly indicated the necessity of maintaining the natural intra-annual and inter-annual regimes of flow variability. Thus, the focus of many instream flow studies is shifting away from recommendations based on single minimum flow figures or on only the physical-habitat requirements of individual species. Development has been toward assessments that address the ecological relevance of all components of the hydrological regime, including specific flow events and aspects of flow timing, duration and recurrence interval. This move has culminated in the fourth and most recently established type of methodology, the holistic methodology (Tharme 1996).

Such an holistic, whole-ecosystem approach to instream flow assessment addresses all components or attributes of the riverine ecosystem and their inter-relationships. It is heralded by many instream flow researchers as one of the main future directions in which the science is headed (Stanford 1989; Tharme 1996; Dunbar *et al.* 1998). Awareness of the need to adopt an holistic emphasis to instream flow assessment has grown most rapidly in Australia and South Africa, where there are real constraints to the use of other types of methodology. These constraints include limited data, time before development, manpower and finances. Additionally, the application of methodologies developed in the Northern Hemisphere may be limited or even inappropriate.

The BBM is one of three internationally recognised holistic methodologies to date, and currently the one used routinely in South Africa (Tharme 1996). The other two approaches are the Holistic Approach (Arthington *et al.* 1992, cited in Arthington *et al.* in press) and the Expert Panel Assessment Method (Swales and Harris 1995; Thoms *et al.* 1996) which have been developed and applied in Australia. The relative strengths and weaknesses of these three methodologies are explored in Tharme (1996, 1997c), and are also briefly reviewed in Growns and Kotlash (1994) and Dunbar *et al.* (1998). In essence, the main strengths of holistic methodologies are their holistic nature, where the IFRs of all ecosystem components can potentially be addressed; their strong clear links with the natural historical flow record; and their pragmatic use of available data. They are limited, however, in their strong reliance on specialist experience of the river system or other similar ones, and on adequate hydrological data. Additionally, they have not

been compared formally with other international instream flow methodologies and some procedures in the methodologies require further development.

1.2.2 The first stages in the evolution of the Building Block Methodology

Globally, holistic methodologies were no more than unstructured ideas when development of the BBM began in South Africa. A point of departure for its development had gradually evolved in the early work on instream flow assessments, that identified the need for a methodology that would be able to:

- focus on river health at the whole-river level rather than be concerned with the protection of single species or groups of similar species;
- operate in situations of few data and poor understanding of river structure and functioning, and thus rely heavily on best-available knowledge;
- provide comprehensive recommendations on flow regimes for regulated rivers;
- quantify these flows in space and time, reflecting the different functions of different magnitude flows;
- include an indication of confidence level in the recommendations;
- highlight the greatest and most important uncertainties requiring research.

It was acknowledged that in the continuing competition for water among various user sectors, it would be vital to be able to justify all flows required for the river.

Intensive development of the fledgling BBM began in about 1990, with an impetus for its development that is detailed in King and Tharme (1994). However, its origins can be traced to an instream flow workshop in 1987 (Ferrar 1989). It met the criteria listed above, and has evolved through application in real-life water-resource projects. Its evolution was made possible by three vital sets of circumstances.

- The recognition by DWAF, from the early 1990s, that instream flow assessments should be conducted for all proposed water-resource projects that affected river flow, and the strong support of DWAF for development of an appropriate methodology. This support included approval for, and funding of, almost all of the IFR workshops held to date, and approval for appropriate specialist staff in DWAF to help develop aspects of the methodology.
- The funding and other substantial support provided by the Water Research Commission (WRC), that allowed the two authors of this report and other river scientists to develop expertise in instream flow assessments, and to undertake research in flow-related aspects of their disciplines.
- The willingness of the national body of river scientists to support development of the BBM. This they did by attending and producing reports for flow-assessment workshops, by being willing to make inputs at the workshops to the limits of their knowledge, and by many of them producing new ideas on how the methodology could further be improved. Their enormous contribution cannot be too highly rated.

By 1995, the BBM was developed to a point that problems encountered in its application from river to river were being resolved and routine procedures were emerging. The conceptual basis of the methodology and these procedures are described in Chapters 2-6. The methodology was beginning to attract international attention, and outputs from the workshops where it was applied were being used in DWAF's planning and design stages for the developments concerned. A practical manual to guide on all aspects of its application, including scheduling and budgeting, is planned for completion in 1998. This manual signals the completion of phase one of the methodology's development. All of its components could benefit from further

development, and a second phase would now be appropriate, in which the transition is made from pioneer methodology to a comprehensive, scientifically defensible, decision-making tool.

1.3 FLOW-RELATED RESEARCH LINKED TO THE BUILDING BLOCK METHODOLOGY

From the outset it was acknowledged by those developing the BBM that if the methodology had to be able to cater for the water managers' needs for rapid answers on IFRs, it would have to be based on available or easily acquired data and expert opinion. This being the case, it was also strongly felt that research to improve the knowledge base had to be incorporated into the overall activities related to its use, in whatever way was possible. Hence, within this project a programme of research was designed that would complement the development of the BBM.

The research programme was founded on the conceptual basis of the BBM (Chapter 3), which can be summarised as follows: **Some flows within the total flow regime of any river are more important than others for maintenance of that riverine ecosystem. These flows can be identified for any river; and then described in terms of their timing, duration, frequency and magnitude.** It is explained in Chapter 2 that such flows will usually fall into one of the following categories: dry season baseflows (low flows); wet season baseflows (low flows), wet season freshes and floods, dry season freshes, or dry season subsurface flows. This being accepted, research projects were undertaken to address various aspects of some of these flow categories. Each project focused on one of the flow categories and was designed to test assumptions within the BBM and/or to produce supporting scientific data to improve its application.

The research projects differed in length and intensity, from a three-year ecological project, through two year-long engineering projects, to four short projects at the level of a B. Sc. (Honours) or fourth year B. Engineering degree. Some of the projects were funded by this parent project. Others were inspired by it, but then developed independently with no funding from the parent project; these did not necessarily provide answers directly linked to the parent project's aims. Further explanation on this is provided in the appropriate chapters.

For each flow component studied, a combination of ecological and engineering projects was undertaken, to encourage cross-fertilisation of ideas and techniques, and multidisciplinary training (Chapters 7-15). Chapters 7-12 report on investigations of dry-season baseflows, and include the three-year ecological project, which is the major research element of the whole parent project. Chapters 13 and 14 deal with dry-season freshes, and work on wet-season floods is described in Chapter 15. Chapter 16 provides a general summary and recommendations for future research. Four smaller research projects linked with the main project are briefly documented in Appendix 1. Appendix 1.1 summarises work on the analysis of the effects of freshes in river flow regimes. Appendix 1.2 presents the results of a preliminary study of the effects of spates on benthic macroinvertebrates. Appendix 1.3 documents the anthropogenic effects of a number of specific perturbations, including water abstraction, on rivers. Finally, Appendix 1.4 provides a summary of research on the implications of pre-determined release policies and dam sizes on the natural flow regimes of Western Cape rivers.

2. THE NEED FOR INSTREAM FLOWS AND EARLY INVESTIGATIONS OF THE FORM THEY SHOULD TAKE

Written by J. King

2.1 INTRODUCTION
2.2 DEFINITION OF TERMS
2.2.1 Baseflows
2.2.2 High flows
2.3 THE NEED FOR INSTREAM FLOWS
2.3.1 General characteristics of undisturbed river systems, with particular reference to flow
2.3.2 The effects of flow regulation on riverine ecosystems
2.4 EARLY INVESTIGATIONS INTO THE NATURE OF INSTREAM FLOWS REQUIRED FOR REGULATED RIVERS
2.4.1 The questionnaire
2.4.2 The workshop
2.4.3 Summary of results of the questionnaire survey and the workshop
2.5 THE WAY FORWARD

2.1 INTRODUCTION

Two main topics are dealt with in this Chapter, after a brief section on terminology (Section 2.2). Section 2.3 consists of a simple and generalised explanation of why instream flow assessments are necessary, which was written at the request of DWAF for inclusion in all IFR workshop proceedings. Section 2.4 reports on some enquiries made by the authors of this report early in the project on the kinds of instream flows that river scientists thought should be retained in regulated rivers for purposes of ecosystem maintenance.

2.2 DEFINITION OF TERMS

The following flow components have been defined from an ecological perspective for the purposes of this report.

2.2.1 Baseflows

Baseflow is the flow in the river contributed by the channel and banks at all times of the year, excluding higher flow events generated by surface runoff (Gordon *et al.* 1992). It may be divided into dry-season baseflow and wet-season baseflow. Baseflows during the dry season are sometimes loosely referred to as low flows by ecologists, sometimes with particular reference to drought events. However, the term low

flows is used in this report to represent the range of dry-season flows that may be naturally or unnaturally low in magnitude, as specified in each instance.

2.2.2 High flows

High flow events are periods of elevated flow that are variable in magnitude, duration, timing and return period (recurrence interval), and result mainly from surface runoff. High flows are typically divided into freshes and floods for ecological purposes. Although neither category of high flow event has been specifically defined in the literature, freshes are understood to be small elevated flow events or pulses, in the order of about two times the magnitude of the baseflow. They may occur during both the dry and wet seasons, but are of greatest ecological significance in the dry season. Floods may similarly occur at various times of the year, but are generally concentrated in the wet season. They are larger, longer duration events that differ in their return periods, such as the 1:2 and 1:50 year floods.

From a hydrological viewpoint, flood and fresh events can be separated from the underlying baseflows throughout the year, using a variety of recognised baseflow separation techniques (Gordon *et al.* 1992).

2.3 THE NEED FOR INSTREAM FLOWS

2.3.1 General characteristics of undisturbed river systems, with particular reference to flow

Various combinations of climatic, geological and topographical conditions at different latitudes have created a diversity of rivers in South Africa. Rivers in some areas flow most strongly in winter, in other areas in summer. There are perennial rivers, ones that flow continually during the wet season, and ephemeral ones that flow only briefly after rains. Their water chemistry differs from region to region, as do their water temperatures, the shape of their channels, the mosaic of substrata in their river beds, and the amount of silt that they naturally carry.

Plant and animal species have evolved that are characteristic of each kind of river, and that can thus cope with the related conditions, no matter how unpredictable or variable these may be. Together, the plants, animals and their abiotic environment form the structural and functional basis of riverine ecosystems. Different groups of species, adapted to cope with specific ranges of physical, chemical and biological conditions, occur in different geographical areas and, within each river, along the length of the system and through the year. Naturally, each riverine ecosystem exists in a state of dynamic equilibrium, responding to seasonal and annual fluctuations in environmental conditions. This can happen because its species have different tolerance ranges to the environmental conditions and so differ in their abundances as conditions change. Thus, at any time and point along a river, there is always a mix of well-represented species that can efficiently cope with prevailing conditions, while other species may be present in lower numbers or surviving as, for instance, eggs, seeds or spores, until more suitable conditions occur. Additionally, the broad range of environmental conditions shifts over longer time spans, as the river changes, and as this happens the species composition of the assemblages present also gradually changes. At any time, the abiotic environment and the species themselves jointly control the mix of species and numbers of

individuals present, usually resulting in assemblages where no one species proliferates to pest proportions. The river's natural flow regime is one of the dominant abiotic controlling factors, as discussed below.

The different magnitudes of flow that form part of the natural flow regime affect riverine ecosystems in a variety of ways. For instance, the higher flows dictate the general geomorphological character, shape and size of a river channel, as well as the mean size of the bed particles which generally decreases downstream. Boulders give way to cobbles, gravel, sand and finally mud, with the proportion of each differing at any point along the river. Upon this template, the local hydraulic, thermal and chemical conditions create a mosaic of areas of roughly uniform environmental conditions, often referred to as hydraulic biotopes (Wadeson 1995). Obvious hydraulic biotopes in rivers are the areas commonly called riffles, pools or backwaters. In general, the higher the diversity of hydraulic biotopes available, the higher the number of species that can exist in the river, and many species, particularly fish and aquatic invertebrates, need more than one such biotope to complete their life cycles.

The annual flow regime of any river affects these hydraulic biotopes in many ways, such as increasing water depth or velocity. For ecological purposes, the influence of the flow regime could be said to be felt through three or four main types of flow. These are: the baseflow; floods, or pulses of high flow, in the wet season; freshes, or smaller pulses of high flow, in the dry and wet seasons; and, in ephemeral rivers, periods of no surface flow.

The baseflow maintains the basic ephemeral, seasonal or perennial nature of the river, thereby influencing which species can survive in it. Species which need to spend several months in water in order to complete their life cycles are rare in ephemeral rivers, though certain riparian tree species may be able to live on such a river's banks if the groundwater conditions are favourable. The different magnitudes of baseflow in the dry and wet seasons create more or less wetted habitat and different hydraulic and chemical conditions, which directly influence how many individuals can survive at any time and what the balance of species will be.

Freshes have several important functions, such as stimulating spawning in serial-spawning fish species, flushing out poor quality water, mobilising sandy sediments, and creating the flow variability that seems to be such an important influence on the character and functioning of riverine ecosystems, particularly in arid areas. Floods reset a wide spectrum of conditions in the river, triggering and synchronising activities as varied as upstream migration of fish and germination of riparian seedlings. Floods also scour river beds and estuaries, ensuring, amongst other things, continued accessibility of the latter to marine fish dependent on them as nursery areas, as well as the maintenance of habitat diversity.

2.3.2 The effects of flow regulation on riverine ecosystems

Manipulations of virgin flow regimes represent unnatural disturbances to riverine ecosystems, with these disturbances increasing with severity the further the flow regime is altered from what lies within the realm of "normal" for that river. Responses of the riverine ecosystem become more extreme as disturbance increases, and can take many forms. For instance, with flow manipulation, hydrological cues that trigger fish spawning or seed germination may occur at the wrong time of the year or not at all, resulting in affected species perhaps failing to produce new generations of individuals and gradually dying out. Seasonal reversal of high and low baseflows could mean that hydraulic and thermal conditions, in particular, become mismatched with life-cycle requirements, again causing species to decrease in numbers and abundances.

Other species, many seen as pests, are often able to take advantage of such environmental conditions or the weakening of competition from the affected species, and increase in abundance. Riparian trees could die from rapid lowering of the water table, or young seedlings fail to grow because of lack of bank flooding, with a consequent weakening of bank stability and probable increase in soil erosion. The loss of floods could reduce scouring and "working" of the river bed, simplifying the mosaic of biotopes that previously would have supported a rich variety of species. Some marine fisheries could falter because estuaries are no longer accessible to the fish species dependent on them for nursery areas. Many other responses by the riverine ecosystem are possible, with costs to the country that are largely externalised when decisions on water-resource developments are made and thus, to a large extent, are non-quantified.

In accordance with the developing policy of DWAF regarding protection of the resource base, an assessment of the IFR for river maintenance is now routinely done for all rivers targeted for water-resource development by that department (DWAF 1992a). An instream flow assessment, through which the IFR is defined, can be described as "The identification of those fundamental components of the flow regime of a riverine ecosystem considered essential for ensuring perpetuation of its valued features, and the combination of these to create a recommended modified annual flow regime for river maintenance". Most commonly used in South Africa for these instream flow assessments is the BBM, which is being developed by DWAF specialists and members of the national community of river scientists. Using the BBM, the assessments are designed to produce, using the best available knowledge and expertise, a recommended modified flow regime (or IFR) for the targeted river that will facilitate its maintenance in some pre-determined desired future condition. Usually, the assessment has to be made in limited time, using limited data. Although it may be years or decades before the full efficacy of these recommendations is known, the BBM represents the best option presently available for producing a rapid holistic recommendation on flows that will minimise the impacts that a proposed development might have on the downstream river. In many cases, recommendations could also be supplied that would mitigate, to some extent, damage already done through earlier manipulations of the flow regime.

2.4 EARLY INVESTIGATIONS INTO THE NATURE OF INSTREAM FLOWS REQUIRED FOR REGULATED RIVERS

Two preliminary investigations took place to ascertain the perceptions of river scientists regarding the nature of instream flows required for regulated rivers. The first consisted of a questionnaire survey sent to a number of specialists with river-related knowledge, and the second consisted of a workshop held at a conference. In the following sections, the two investigations are described and general conclusions are then drawn from them.

2.4.1 The questionnaire

In August 1991, a questionnaire was sent to 46 ecologists, engineers, geographers, hydrologists and water managers with specialist knowledge of South African rivers. Twenty-eight replies were received (Appendix 2.1). The questionnaire was designed to discover whether it might be possible to distil the years of experience, and expert opinion, of those who conserve, manage, study or otherwise have substantial knowledge of, South African rivers, into a series of statements regarding river flow, which could be researched or used for management purposes. Further, it was wished to determine if there is any consensus

among the specialists within any one region, as to what might be the vital components of the annual flow regime necessary to maintain the integrity of the rivers in their region.

Three questions were asked:

- **For which region(s) of South Africa do you know well the rivers, their flow patterns and how these patterns might be influencing the character of the riverine ecosystems?** A map of the country was provided so that such regions could be indicated by shading.
- **What is the characteristic seasonal flow pattern for the rivers of the region(s)?** A blank graph of mean monthly discharge (Y axis) versus months of the year (X axis) was provided.
- **What are the essential features of that seasonal flow pattern for ecosystem maintenance - that is, which parts of the flow regime would you least like to see removed?** The same blank graph as above was used to allow them to depict the modified flow regime.

The 28 responses fell into three categories:

- those who could not answer the questionnaire for a variety of reasons, including because they had no idea what might be the essential features of a river's flow regime for ecosystem maintenance;
- those who answered wrongly, usually by mistakenly offering the present-day modified flow regime rather than an ecologically-acceptable modified flow regime;
- those who answered correctly.

All but one of the specialists who answered the questionnaire correctly were river ecologists. Their responses were similar to those recorded in the workshop, and have been amalgamated below into a joint summary (Section 2.4.3).

2.4.2 The workshop

In July 1993, a half-day workshop was held at the Annual Conference of the Southern African Society of Aquatic Scientists, at Rand Afrikaans University. The workshop was convened by the authors of this report, and was entitled "Towards a methodology for instream flow assessments for South African rivers". Its purpose was stated as "To assess the knowledge available in the national community of river scientists regarding the importance of different magnitude flows for maintenance of riverine ecosystems". Seventeen delegates attended the workshop (Appendix 2.2), almost all of whom were ecologists.

The workshop participants were asked to group themselves according to the region for which they had best knowledge and understanding of riverine ecosystems. As with the questionnaire, each group was then asked to describe, using a graph of mean monthly discharge versus calendar months, the natural flow pattern of an hypothetical river that was typical of its region. The participants were then asked to delete half the area under the plot of the annual hydrograph, that is, half of the annual discharge of the hypothetical river. It was requested that this be done by entering on the graph, in order of importance until half the annual discharge was spoken for, those components of the annual flow regime that were considered most important for maintenance of the riverine ecosystem. For each component entered, motivation had to be provided for its inclusion from either an ecological or geomorphological perspective.

2.4.3 Summary of results of the questionnaire survey and the workshop

In either the questionnaire or the workshop, generalised annual hydrographs for instream flows for regulated rivers were drawn for the southwestern Cape, the Eastern Transvaal lowveld, the Eastern Cape coastal area, Qwa Qwa, central Natal, the Limpopo and neighbouring rivers, and ephemeral rivers in the Northern Transvaal. Annual hydrographs for central Natal (Figure 2.1) and the Northern Transvaal (Figure 2.2) illustrate the kinds of modified flow regimes respondents saw as essential for maintenance of riverine ecosystems and the kinds motivations they provided for requested flows. Accepting that most participants had not given much structured thought to the subject before, the general messages emerging from the responses were remarkably consistent.

- The modified flow regime should mimic the virgin one, so that the natural timing of different kinds of flows is adhered to.
- The river's natural perenniality or non-perenniality should be retained.
- Most water should be harvested from the river during the wet months and least during the dry months.
- The seasonal pattern of higher baseflows in the wet season than in the dry season should be retained.
- Flood durations could be shortened, but within limits. This is because some flood-related activities, such as hatching of fish eggs in bank vegetation and movement of the larvae back into the main channel, require a certain number of days for completion.
- Whole floods should be removed and others left at full magnitude, rather than having all or most floods but at lower magnitudes.
- The first (or one of the first) flood of the wet season should be retained.

The reasons for requiring different magnitude flows at specific times of the year were given as listed below.

First floods

- Provide ecological cues for fish spawning and migration.
- Stop fish kills related to low discharges, increasing water temperatures and decreasing concentrations of dissolved oxygen (summer rainfall area).
- Reset river to wet-season conditions, through improvement in water quality, scouring of river bed and flushing of fine sediments.

Peak flood flows

- Flush sediments, "work" bed particles and maintain the channel shape and form.
- Inundate marginal vegetation, providing refuge and new habitat for aquatic fauna.
- Recharge the groundwater stored within river banks.
- Inundate floodplains, allowing fish to move into these feeding and nursery areas.
- Control reed encroachment.
- Enhance habitat diversity.
- Control pests such as problem algae and blackflies.

Post-rain baseflows

- Enhance growth and productivity of aquatic species, through the provision of water and food.
- Enhance growth of riparian species and establishment of new seedlings, through high groundwater tables in banks.

Dry season freshes

- Trigger spawning in serial spawning fishes.
- Enhance water quality in slow-flowing areas.

Dry season baseflows

- Maintain aquatic habitat in perennial rivers for fish and other species with long life cycles.
- Help control proliferation of pest species through reduced discharge and aquatic habitat.

In summary, it was pointed out that while fluvial geomorphologists are mainly concerned with the magnitude, velocity and return period of flows, ecologists are additionally concerned about the timing and duration of such flows.

2.5 THE WAY FORWARD

The enquiries and responses described above provided a template for development of a South African approach to instream flow assessments. It was clear that the approach should allow comprehensive recommendations to be made on the annual flow regime for a regulated river, and should be able to use information on any component of the riverine ecosystem. Development of such an approach, the BBM, is described in Chapter 3.

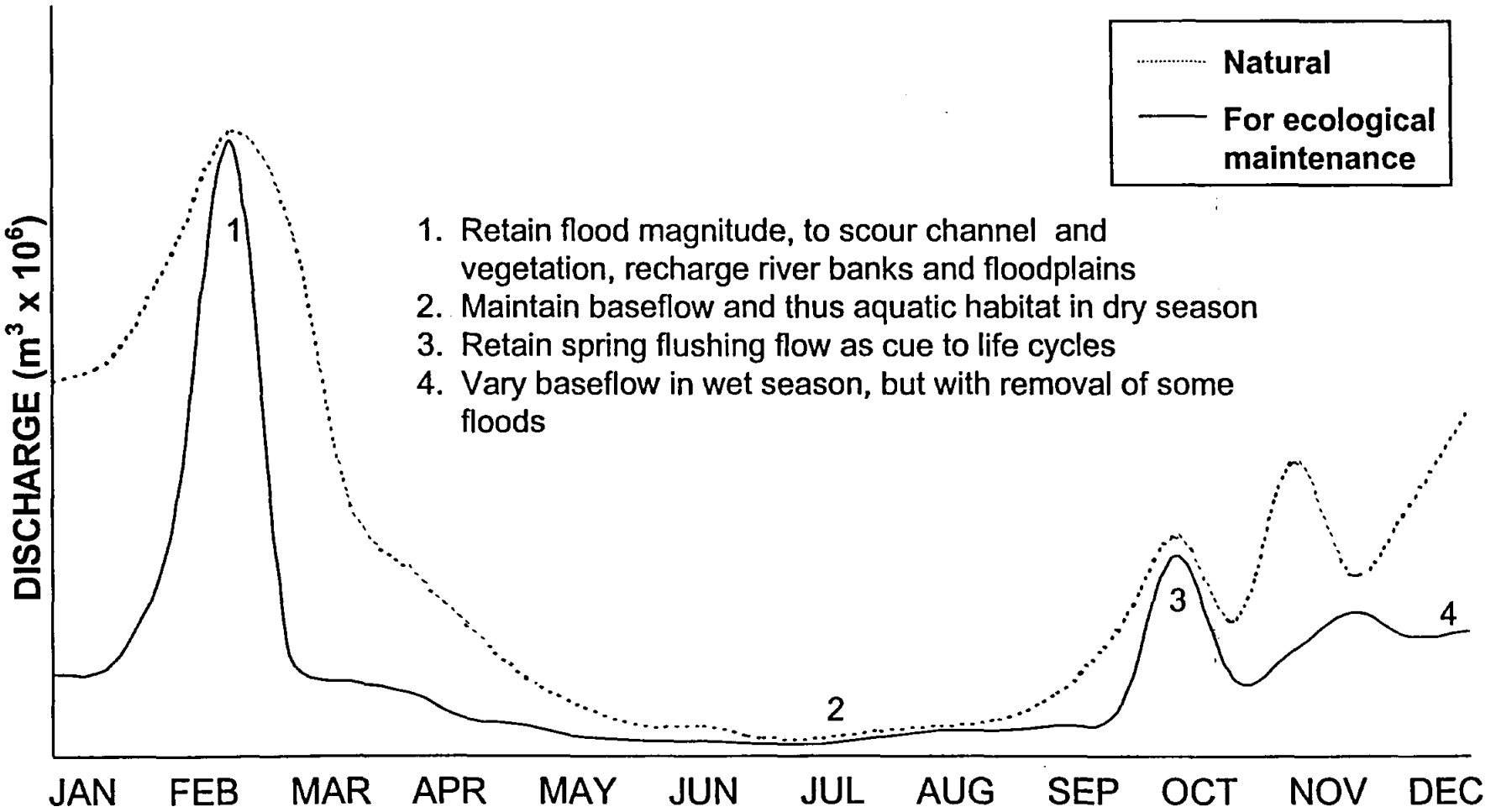


Figure 2.1 **The natural flow regime of an hypothetical perennial river in central Natal, and those parts of it deemed most important for ecological maintenance of the river. Results of a workshop held at the annual conference of the Southern African Society of Aquatic Scientists in July 1993.**

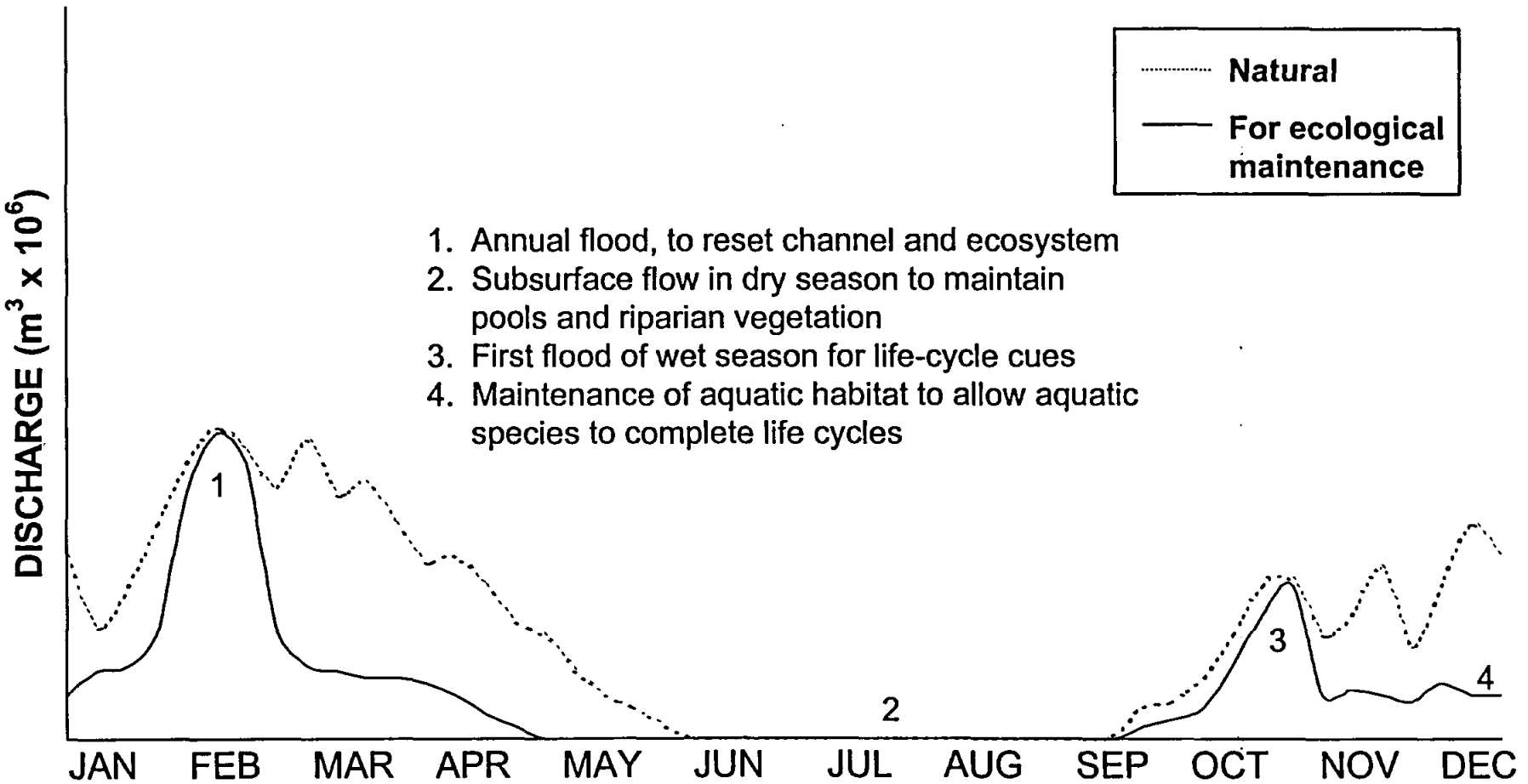


Figure 2.2 ***The natural flow regime of an hypothetical ephemeral river in the northern Transvaal, and those parts of it deemed most important for ecological maintenance of the river.*** **Results of a workshop held at the annual conference of the Southern African Society of Aquatic Scientists in July 1993.**

3. DEVELOPMENT AND NATURE OF THE BUILDING BLOCK METHODOLOGY

Written by J. King and D. Louw (then at DWAF)

3.1 INTRODUCTION

3.2 THE ORIGIN OF THE BUILDING BLOCK METHODOLOGY

3.3 ASSUMPTIONS AND CHARACTER OF THE BUILDING BLOCK METHODOLOGY

3.4 THE SEQUENCE OF ACTIVITIES IN THE BUILDING BLOCK METHODOLOGY

3.4.1 Part One of the Building Block Methodology - preparation for the IFR workshop

3.4.2 Part Two of the Building Block Methodology - the IFR workshop

3.4.3 Part Three of the Building Block Methodology - linking environmental and engineering concerns

3.5 THE POSITION OF THE BUILDING BLOCK METHODOLOGY IN THE DEPARTMENT OF WATER AFFAIRS AND FORESTRY'S PROCEDURE FOR WATER-RESOURCE DEVELOPMENTS

3.6 AN ASSESSMENT OF THE NATURE AND PRESENT STATUS OF THE BUILDING BLOCK METHODOLOGY

3.6.1 Strengths of the Building Block Methodology

3.6.2 Weaknesses of the Building Block Methodology

3.6.3 Aspects of the Building Block Methodology requiring priority attention

3.6.4 Summary of the Building Block Methodology and early criticism

3.7 CONCLUSIONS

3.1 INTRODUCTION

The conceptual basis of the BBM is simple and reflects the findings reported in Chapter 2. It is that some flows within the total flow regime of any river are more important than others for maintenance of that river ecosystem. These flows can be identified for any specific river, and described in terms of their timing, duration, frequency and magnitude. Where a water-resource development is planned, the identified flow components (the so-called building blocks) can then be combined to describe a recommended modified flow regime that is specific for the river of concern. These ideas are expanded upon in Section 3.3.

It was recognised from the outset that, because of time constraints, the BBM would have to rely on best available knowledge and expert opinion. The core of the BBM has thus become, for any one river, an instream flow requirement workshop (IFR workshop) in which senior river scientists representing a range of relevant fields of expertise reach consensus on the modified flow regime that should sustain that river in some pre-determined Desired State (DS). The water managers and engineering and social consultants linked to the proposed water development are also important participants at the workshop, contributing

knowledge on hydrological, hydraulic and social aspects, and gaining knowledge on why particular flows are important from ecological or geomorphological perspectives. However, they do not influence the decisions reached by the river scientists on the recommended flows. Around the core activity of the workshop has developed a structured process for compiling the material required for the workshop and using the output of the workshop for guiding negotiations on the future flow regime for the river.

3.2 THE ORIGIN OF THE BUILDING BLOCK METHODOLOGY

The BBM originated in two major South African workshops on instream flow assessments in 1987, where parts of it began evolving in the form of the 'Cape Town' and 'Skukuza' approaches (King and O'Keeffe 1989; Bruwer 1991). It was added to during meetings with Australian colleagues (Arthington *et al.* 1992), where it was termed "The Holistic Method". Its major development then took place during practical applications of the embryonic methodology in the new kind of workshop mentioned in Section 3.1. By arrangement with DWAF, one of the authors of this report (JMK) attended the first twelve of these workshops, initiating some analyses of the process and aiding other analyses. She acted as advisor to DWAF on the data needed for an IFR workshop and on workshop structure, and took the role of group leader during workshop discussions. Both authors of this Chapter and the other author of this report (RET) continually updated the description of the BBM, for inclusion in IFR workshop starter documents, and worked closely with DWAF personnel and colleagues in the scientific community, on many aspects of the methodology. These included:

- identifying the hydrological data needed for an IFR workshop;
- developing a method to transfer to workshop participants large amounts of data on a river in such a way that they could absorb the information quickly and use it effectively in their deliberations;
- developing the use of river cross-sections and associated hydraulic data as the vital link between those who think in terms of water depth or velocity (mostly the ecologists) and those who work mostly with water volumes (water managers and engineers);
- guiding river scientists in the production of workshop starter documents that focused primarily on flow-related aspects of their knowledge;
- developing the most effective format for the written proceedings of an IFR workshop.

The workshops, each designed to produce a rapid first estimate of the IFR for a river targeted for water-resource development, were mostly convened by the Environment Studies sub-directorate of DWAF, and involved many of the country's most experienced river scientists. Between 1991 and 1996, IFR workshops were held for, *inter alia,* the following rivers: the Lephalala (King 1993), Berg (Ninham Shand Consulting Engineers (NSI) 1993a, b; NSI 1994a; Brown 1996b), Olifants (Western Cape) (Tharme 1994), Olifants (Transvaal) (NSI 1993c, 1995a), Letaba (Steffen, Robertson and Kirsten Consulting Engineers (SRK) 1994), Luvuvhu (Luvuvhu Development Consultants 1995; DWAF 1997a), Lomati (NSI 1994b), Koekedouw (Henderson and Ractliffe 1995; SRK/Southern Waters (SW) 1995), Senqu (Lesotho) (DWAF 1995a), Mooi (NSI 1995b, c), Tugela system (DWAF 1995b), Mogalakwena (Ractliffe 1996a, 1996b), Mvoti (Tharme 1997a) and Sabie-Sand system (Tharme 1997b). Starter documents and proceedings for all these workshops are housed at Environment Studies, DWAF.

It was in these IFR workshops; in a few other ones held independently of DWAF (e.g. SRK/SW 1995); in associated research initiated by the scientists taking part in them; and through thorough analysis of the inadequacies of each workshop by the authors of this Chapter and by certain river scientists, that the BBM developed to its present form. The process of refinement and improvement continues with each new IFR workshop, and through incorporation of the results of more detailed scientific studies such as those documented in this report.

3.3 ASSUMPTIONS AND CHARACTER OF THE BUILDING BLOCK METHODOLOGY

In the methodology it is assumed that the biota associated with a river can cope with those baseflow (low flow) conditions that naturally occur in it often, and may be reliant on higher flow conditions that naturally occur in it at certain times (e.g. specific floods). This assumption reflects the thinking that the flows that are a normal characteristic of a specific river, no matter how extreme, variable or unpredictable they may be, are ones to which the riverine species characteristic of that river are adapted and on which they may be reliant. On the other hand, flows that are not characteristic of that river will constitute an atypical disturbance to the riverine ecosystem and could fundamentally change its character. In the methodology, it is further assumed that identifying what are felt to be the most important components of the natural flow regime and ensuring that they are incorporated as part of the modified flow regime will facilitate maintenance of the natural biota and natural functioning of the river. Finally, it is also assumed that certain kinds of flow influence channel geomorphology more than other flows, and that incorporating such flows into the modified flow regime will aid maintenance of the natural channel structure and diversity of physical biotopes. In total, the flows incorporated into the modified flow regime will constitute the IFR for the river. As the minimum acceptable value will have been entered for each flow component incorporated, the IFR describes, in space and time, the minimum amount of water that it is felt will facilitate maintenance of the DS of the river.

The IFR flow components are identified and their magnitudes, timing, frequency and duration decided upon in the IFR workshop. Initially, thought is focused on the characteristic features of the natural flow regime of the river. The most important of these are usually: its seasonality or perenniality; the different magnitude baseflows in the dry and wet season; the magnitude, timing and duration of floods in the wet season; and the small pulses of higher flow, or freshes, that occur in the drier months (Figure 3.1). Attention is then given to which of these features are considered most important for maintaining or achieving the DS of the river, and thus should **not** be eradicated during development of the river's water resources (Figure 3.1).

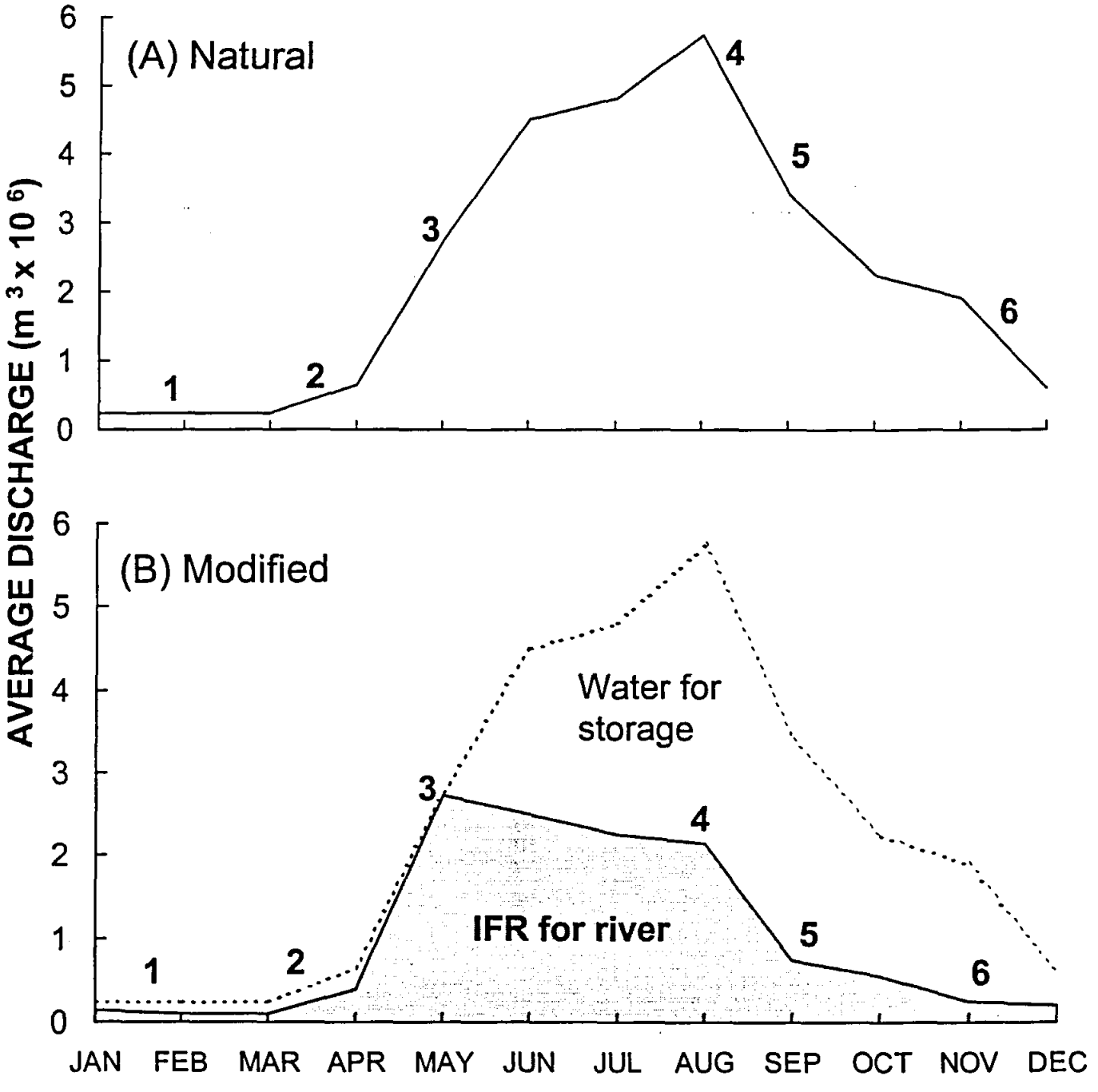


Figure 3.1 **Focusing thought on (A) perceived important features of a river's natural flow regime and (B) which of these should be retained in an IFR. For instance, features 1 and 6 may recognise the perenniality of the river (A) and the need to retain this (B); features 2, 4 and 5 may recognise the need to retain the fundamental difference between wet-season and dry-season baseflows; and feature 3 may recognise the timing of the first major flood of the wet season and the need to retain this.**

The identified essential parts of each flow component are the building blocks that will create the IFR, each being included because it is understood to perform certain ecological or geomorphological functions (Figure 3.2). The details of the flows identified are added one by one to a blank IFR table of discharge (rows) versus calendar months (columns), with each addition being described in terms of four criteria: magnitude, timing, frequency and duration (Table 3.1), and with relevant motivation being supplied by each contributing specialist. Entry lines are completed for the IFR for normal, river maintenance, the upper limit of baseflows (or capping flows) that should not be exceeded on a continual basis, and the IFR for drought conditions. Usually, each entry remains within the limits of the virgin hydrograph, with the completed IFR thus representing a skeleton of the virgin flow regime. In order for ecologists to become familiar with some relevant hydrological terminology, each entry is also identified as a volume of water and a percentile on its month's flow duration curve. Finally, the baseflow and high-flow components of the IFR are expressed as a percentage of the Mean Annual Runoff (MAR) of the river. Results of the workshop are available immediately for planning purposes and the workshop report is written up within about six weeks of its closure.

The planners use the IFR for dam-design purposes and to "block-book" water for the river during the planning process. There is an understanding by DWAF that some adjustment of the IFR might become necessary as more information becomes available.

It should be noted that the BBM is restricted to stating the case for the riverine ecosystem, and does not directly take into account the requirements of any offstream water users. Indirectly however, it acknowledges offstream users in three ways. It is recognised that water-resource development of some kind is almost always inevitable and so constructing an IFR that is the same as the original flow regime would usually be inappropriate. Direct dependence on the riverine ecosystem by, for instance, rural dwellers who gather resources such as food or medicines from the riparian vegetation, is also recognised by attempting to cater for this with appropriate flows. Lastly, the overall requirements of Interested and Affected Parties (IAPs) are encompassed in a statement of DS for the river, which guides the recommendation of flows. Other kinds of specific water users, such as irrigation boards or municipalities, are dealt with later, in scenario meetings held during DWAF's Feasibility stage (Section 3.5), where their representatives will present their own water requirements; these are then considered in light of the IFR. Subsequently, DWAF will act as the decision-maker, deciding on how the water will be allocated.

3.4 THE SEQUENCE OF ACTIVITIES IN THE BUILDING BLOCK METHODOLOGY

The BBM has three main parts, which encompass preparations for and running of the IFR workshop, and follow-up activities linking the workshop with the engineering and planning procedures. The practical details of all the activities are detailed in Chapter 4, while an overview is given below.

3.4.1 Part One of the Building Block Methodology - preparation for the IFR workshop

Part One consists of a structured set of activities designed to collect and display the best available knowledge of the river, for consideration by the workshop participants. Presently, the time limit for completion of this part is usually not more than six months, with a small budget available for very limited collection of new

data. The six-month time frame is a reflection of how soon busy specialists can include extra work in their schedules, rather than of the amount of work done. Guidelines are presently being written for the various activities in Part One, which cover 14 main topics. These are: instream and riparian habitat integrity; conservation importance; geomorphological character; study area; fish studies; invertebrate studies; vegetation studies; social dependence; water quality; DS; location of IFR sites; hydrological and hydraulic analyses for the IFR sites; and planning of the IFR workshop. This list reflects one of the major features of the methodology, namely, that it can incorporate and use **any relevant information** on the river. Co-ordination of the activities takes place early in the process through an IFR planning meeting, and Part One of the BBM culminates with production of a starter document for the workshop. This document incorporates reports on all the activities as well as other relevant background information.

The activities in Part One are detailed in Louw and King (1995), where each one is explained briefly and motivated, the expertise to complete it is detailed, and the sequence in which the activities (Figure 3.3) should be done is explained. Running parallel with the activities is the ROIP (Afrikaans acronym for Relevant Environmental Impact Prognosis) or impact assessment report, which is continually updated, usually by DWAF, during the course of the water-resource development as more information becomes available. The main activities in Part One are outlined below, with reference to Figure 3.3.

- Determination of the present habitat integrity of the river, through an aerial survey by helicopter (*what is the present condition of the river, in terms of available instream and riparian habitat for riverine plants and animals?*). **Point 2**

- Determination of the importance of the river, at the local, regional, national and international level (*how important is the river in terms of its conservation importance?*). **Point 2**

- Assessment of the geomorphological characteristics of the river (*which reaches of the river are different in terms of geology, channel shape, substrata and diversity of physical biotopes*?). **Point 3**

- Identification of the study area (*which part of the whole river system will be dealt with in the instream flow assessment?*). **Point 4**

- Completion of biological studies, in order to update knowledge of the ecological status of the river and to determine its longitudinal zonation (*which reaches of the river are different in terms of the biota, and what is the characteristic biota of each? Are there any sites, species or communities of special importance?*). **Point 5.** In addition, any information on life history data, ranges of tolerance to environmental variables, or the water quantity or water quality requirements, of any of the species that occur in that river, will be presented in the report for the starter document. Ecosystem components usually reported on are the fish, the riparian and instream plant communities and the aquatic invertebrates.

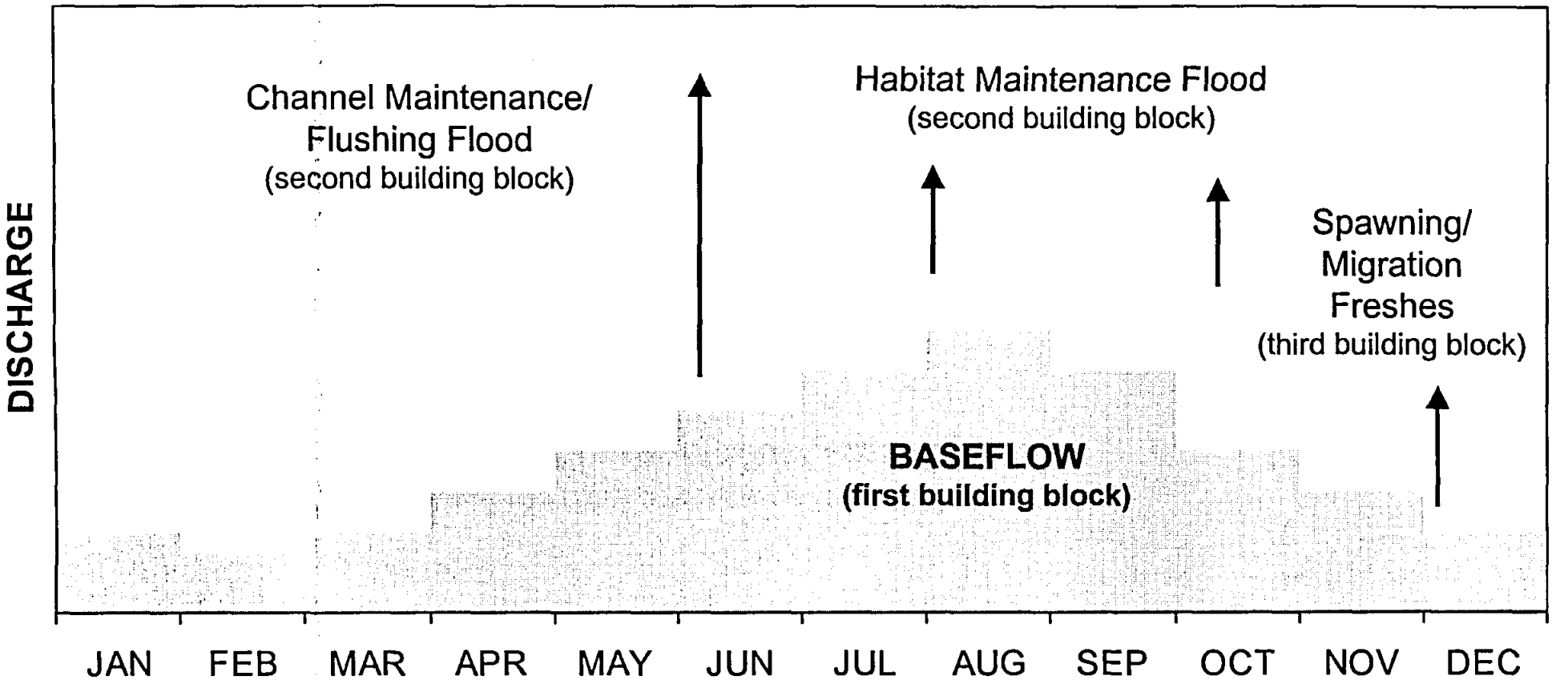


Figure 3.2 An hypothetical IFR created using the Building Block Methodology.

Table 3.1 **An example of a blank IFR table.**

Site: _____ **River:** _____ **MAR:** _____ $m^3 \times 10^6$

IFR BUILDING BLOCKS		OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	VOL ($m^3 \times 10^6$)	% MAR
CAPPING of LOW FLOWS ($m^3 s^{-1}$)															
IFR for maintenance	LOW FLOWS ($m^3 s^{-1}$) 1. Flow percentile (%)														
	FLOODS 1. Magnitude ($m^3 s^{-1}$) 2. Duration (d) 3. Return period (y) 4. Flow percentile (%)														
IFR for drought conditions	LOW FLOWS ($m^3 s^{-1}$) 1. Flow percentile (%)														
	FLOODS 1. Magnitude ($m^3 s^{-1}$) 2. Duration (d) 3. Return period (y) 4. Flow percentile (%)														

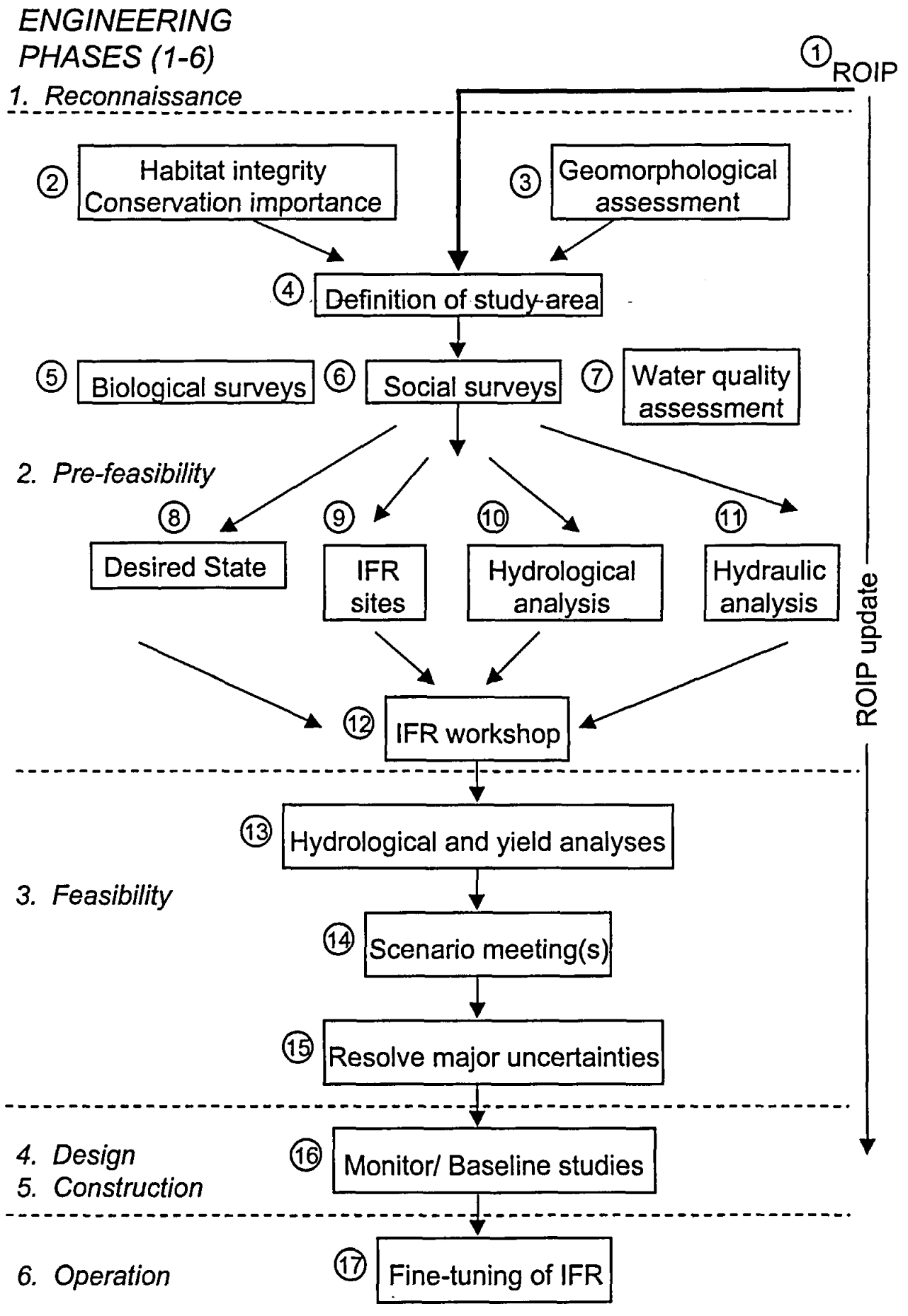


Figure 3.3 **Sequence of activities in the Building Block Methodology. ROIP - Relevant Environmental Impact Prognosis.**

- Completion of a social survey (*are there rural communities, or any other group(s) of people, directly dependent on the riverine ecosystem?*). **Point 6**

- Assessment of the past, present and required future water quality of the river (*bearing in mind the DS of the river (see next point), at what level should future water quality be maintained?*). **Point 7**

- Determination of the DS of the river (*in what condition should the river be maintained, in the future?*). **Point 8**

- Identification of the IFR reaches and IFR sites along the river. These will form the focus for most of the collection, simulation and analyses of new data required specifically for the IFR workshop, and each site will have an IFR described for it (*knowing the longitudinal biological zonation and special attributes of sections of the river, which parts need an individual assessment of their instream flow requirement and which sites within these parts will be used for those assessments?*). **Point 9**

- Analysis of the hydrological data, simulated if necessary and at the resolution of daily average discharges if possible, for the selected IFR sites along the river, to determine their virgin and present flow regimes (*what are the essential natural and present characteristics of the various BBM flow components at the selected IFR sites?*). **Point 10**

- Analysis of stage-discharge rating curves, channel cross-section profiles, and other relevant data on hydraulics, channel morphology and biotopes, gathered at cross-sections established at each of the selected sites (*what is the shape of the channel at each site, how deep is the water at any specific discharge and where does the water lie in the channel at those discharges? What biotopes are present and how are these likely to be affected by changes in discharge?*). **Point 11**

The starter document encompassing the above reports is sent to workshop participants about three weeks before the IFR workshop, and all are expected to read it thoroughly before arrival at the workshop venue. Apart from its main role of informing participants about the river, the starter document has two other values. It encourages the specialists who undertake each activity to focus more thought on the river than if they did not have to produce a written report, and thus enhances their inputs at the workshop, and it has value in its own right as a detailed synthesis of knowledge on a specific river at a specific time.

3.4.2 Part Two of the Building Block Methodology - the IFR workshop

The workshop (Point 12, Figure 3.3) generally involves about 20 people, with those present previously having been DWAF water managers and engineers, the consulting engineers appointed for the development, hydrologists, hydraulic modellers and scientists with a range of expertise in river structure and functioning or with first-hand knowledge of the river under consideration. The workshop typically takes place in an isolated conference venue, with the first part of it being an information-gathering and familiarisation field trip to the river of concern. Over a period of two to four days, a chairperson and group leaders experienced in the methodology then guide the workshop participants to consensus on a recommended modified flow regime for the river. Isolating the participants in a secluded venue in the catchment of concern encourages focused thinking and a vibrant work ethic.

The structure of the workshop is described in Section 4.9.

3.4.3 Part Three of the Building Block Methodology - linking environmental and engineering concerns

Part Three of the BBM occurs after the IFR workshop. It constitutes a series of activities that link environmental flow considerations to the engineering activities also taking place (Points 13-17; Figure 3.3; and see Section 3.5). It is the least developed part of the BBM to date, with the sequence and nature of necessary activities gradually being devised as the various water-resource projects for which the BBM has been applied evolve through planning to design, construction and operation. It is dealt with more fully in Chapter 6.

3.5 THE POSITION OF THE BUILDING BLOCK METHODOLOGY IN THE DEPARTMENT OF WATER AFFAIRS AND FORESTRY'S PROCEDURE FOR WATER-RESOURCE DEVELOPMENTS

The IFR workshop links in, via the sub-directorate of Environmental Studies, with the main engineering phases followed by DWAF during a water-resource development (Table 3.2; Figure 3.3).

From the engineering perspective, a situation assessment takes place during the Reconnaissance Phase, often in the form of a catchment study or a regional systems analysis. The most probable options for the water-resource development are then identified and investigated during the Pre-feasibility Phase. In the Feasibility Phase, major uncertainties are attended to and, if there are no major problems, one option is then chosen for development. Design, Construction and Operation Phases usually then follow.

Table 3.2 Department of Water Affairs and Forestry's engineering and environmental planning stages for a water-resource development, showing the positioning of the IFR workshop. IEM - Integrated Environmental Management.

ENGINEERING/IEM STAGE	ENGINEERING ACTIVITY	ENVIRONMENTAL ACTIVITY
Phase 1: Reconnaissance	Catchment/systems analysis	Issues assessment
Phase 2: Pre-feasibility	Possible developments identified	ROIP completed
		IFR workshop
Phase 3: Feasibility	Detailed investigation of selected option	Scenario meetings
Phase 4: Design	Creation of management plan	In-depth base-line studies for monitoring purposes
Phase 5: Construction	Implementation of management plan	Monitoring
Phase 6: Operation	Audit	Monitoring and fine-tuning of IFR

Proposed developments may remain at the Reconnaissance Phase from months to years, before moving rapidly through the Pre-feasibility and Feasibility Phases and into the Design Phase. From the environmental perspective, funding for the IFR workshop presently only becomes available during the Pre-feasibility Phase. The IFR process then has to be implemented rapidly, to provide the information on the IFR for the river that is needed for the Feasibility and Design Phases (Chapter 6).

3.6 AN ASSESSMENT OF THE NATURE AND PRESENT STATUS OF THE BUILDING BLOCK METHODOLOGY

3.6.1 Strengths of the Building Block Methodology

- The BBM is a methodology for rapidly assessing the IFR for a river, which provides a quantitative answer that can be used in the planning and design stages of a proposed water-resource development.
- It can be used where time and data are limited, but there is a clear lower limit to the amount of each needed for successful application of the methodology. At the very least, experienced freshwater ecologists, fluvial geomorphologists and hydrologists with knowledge of that, or a similar, river should be involved, as should a social scientist with first-hand knowledge of the use made of the river by rural communities. Additionally, the workshop chairperson and group leaders should be experienced in the methodology, and the latter, at least, should be experienced river ecologists. A realistic time frame for successful application of the BBM is four to six months.
- It is simple in concept, pragmatic, and easily understood by the non-scientist.
- It is designed to aid water managers, by producing a preliminary quantitative answer on the flow requirements for a river, with appropriate motivation. This answer may be adjusted as more data are collected.
- It draws on specific or regional knowledge of rivers, which is gleaned in a structured fashion from the specialists by scientists trained in the methodology.
- It is a transparent process.
- It develops cross-disciplinary communication and understanding, so that all workshop participants feel able to contribute, and in turn benefit by being able to talk of the process in their own work life with some understanding and conviction.
- It does not have to be linked to a water-resource development. Thus, it could also be used to define a less damaging flow regime for an impacted river, or even used for regional planning purposes to assess how much water is still available for development within a region or catchment.
- It incorporates scenario meetings after the IFR workshop, where potential offstream water users meet with the water managers and the IFR experts to search for resolution of conflicting needs.
- It has been contributed to by most senior river scientists within the country through their participation in the workshops.

3.6.2 Weaknesses of the Building Block Methodology

- The statement on DS (point 8 in Figure 3.3) tends to be general and descriptive rather than specific and quantitative, because generally little is known of the river. Success in achieving the DS thus cannot easily be measured.
- The methodology largely assumes a steady-state river ecosystem, judging the required future flows from sparse knowledge of present conditions. However, a component for managing change is incorporated via the monitoring feedback loop and adjustment of the IFR.
- The BBM was designed to consider required water quantities. The incorporation of required water quality into the methodology is not well developed, although it is generally recognised that the IFR cannot be effective in terms of the DS unless water quality conditions are also suitable. At present,

dilution flows to solve water-quality problems are not seen as environmental flows, but are superimposed on the IFR.

- The IFR sites need to be as natural as possible, so that clues on required flows can be gleaned from, for instance, the position of riparian species and the degree of inundation of various biotopes. Finding such sites that also represent the parts of the river that will be affected by the proposed development is often a problem.

3.6.3 Aspects of the Building Block Methodology requiring priority attention

- There is presently no clarity on who should decide the DS for any river, or how it should be done. The present approach is that DWAF selects a group of scientists, water managers and other interested and affected parties, all with specialist knowledge of the specific catchment and river but not necessarily fully representative, to recommended a **realistic DS.** The IFR is defined with reference to this state and, in the scenario meetings, the consequences for this desired state of alternative flow regimes are described, albeit in general rather than specific terms. There is a need for more development of the process or, perhaps, a change of approach.
- Analysis of the results of most IFR workshops to date by one of the authors (JMK) has revealed a possible trend related to where the IFR site is situated in the catchment. It may be that for sites high in the catchment a greater percentage of the MAR at that point is being asked for in the IFR than for sites lower down. This had led to considerations that the methodology might be biased against large or mature rivers. It has been suggested by Dr K. Rowntree (Rhodes University, Grahamstown) that the methodology is not inappropriate, but that the level of understanding of the ecological and geomorphological significance of large bodies of moving water in lower rivers is poor, and so suitable statements on IFRs for such reaches might not be made.
- The confidence in the IFR output differs from workshop to workshop, depending on the amount of available data on the river and the level of understanding of the functioning of that or similar rivers. At present, this IFR is used in the design and planning stages with no consideration of its confidence level, which could create problems if it were later found that the water needs for the river had been severely underestimated. Thus, it might be useful to incorporate a weighting factor into the IFR that increases the amount of water asked for as the confidence level decreases; research on areas of major uncertainty would reduce the weighting factor.
- Floods constitute one of the blocks of flow incorporated into the IFR. One of their major functions is seen as the provision of flushing flows for maintenance of channel form and biotope diversity. However, classic studies in fluvial geomorphology from the Northern Hemisphere (e.g. Wolman and Miller 1960) do not necessarily apply in semi-arid areas. For instance, the linkage of channel maintenance flows with bankful discharge and with floods of a return period of about two years may not apply in systems where rains may fail in one or more consecutive years. A better understanding of the nature and function of channel-maintenance flows in semi-arid environments is needed.
- Recently, the first IFR assessment was done for a sand-bed river with only sub-surface flow for much of the year (Ractliffe 1996a, b). Several such ephemeral or seasonal rivers are being dammed or at least having water abstracted, and an increased understanding is required of the water needed for their maintenance and of how to approach groundwater considerations within the BBM.

3.6.4 Summary of the Building Block Methodology and early criticism

The BBM provides a rapid approach for assessing the IFRs of rivers, based on best available knowledge and expert opinion. It is designed to cope with water-resource planning where time and data are limited, but has the facility of being able to use any relevant data. It is designed to meet, within time and other constraints, developers' needs for a quantitative recommendation on a comprehensive modified flow regime, with appropriate motivation.

As a rapid assessment methodology, the BBM was specifically designed to rely mostly on data that are already available in some form in the country, although some are collected or derived specifically for the workshop. The IFR workshop for the Sabie-Sand River system (Tharme 1997b) moved on from this concept, by using the considerable data base of the Kruger National Park Rivers Research Programme (KNPRRP) to apply the BBM in a data-rich situation. It is now felt that the BBM can operate equally well in either a data-rich or data-poor situation, with the main difference being in the confidence level of the resulting flow recommendations. Whatever the level of data available, these are presented in an easily understood format that has been systematically developed to enable the specialists at the workshop to understand them quickly and to use them effectively. Participants unused to the process soon understand the approach and feel confident to join in discussions, because the conceptual foundation of the methodology is simple.

Some critics of the methodology are uneasy with a pro-active approach that requires flow components to be incorporated, one by one and each with motivation, into the IFR. They feel that important flows may be excluded simply through lack of understanding of the functioning of the river, and would prefer a reactive approach, whereby response is made to the developer's plans. However, pro-active statements such as that made in the IFR are recommended in the guidelines on Integrated Environmental Management issued by the Department of Environment Affairs and Tourism (DEA 1992). Also, reaction to a developer's plan implies an understanding of the importance of various components of a flow regime and so some version of the activities described above must presumably be gone through anyway. Certainly, such knowledge would be needed by DWAF's IFR consultants, for the department requires detailed motivation of why specific flow regimes are recommended, which the BBM is designed to produce.

Rogers (1995) criticised what he felt to be the methodology's two-dimensional approach. This refers to the use of cross-sections and stage-discharge relationships to describe the hydraulic attributes of biotopes at each IFR site, and to allow ecologists used to working with water depths or velocities to communicate with engineers who need answers in terms of volume of water. He suggests that a three-dimensional modelling approach should be adopted, whereby each IFR site is described more comprehensively in terms of its local hydraulics. This could be similar to the facility offered by the Physical Habitat Simulation Model (PHABSIM II) in IFIM (Milhous *et al.* 1989), and is certainly an option to be explored if enough hydraulic modellers skilled in low-flow modelling become routinely available. At present, however, the BBM does incorporate a more modest three-dimensional approach in two ways.

Firstly, usually three to five IFR reaches are considered at an IFR workshop, and the IFR for each is determined independently. Within each reach, an IFR site is chosen that has a diversity of biotopes including ones that are critical for completion of life cycles of important species, all in as near as possible natural conditions. Secondly, typically three to four cross-sections describing different biotopes such as

riffles and pools are surveyed in at each IFR site, and stage-discharge relationships prepared for each. This allows some understanding of the loss and gain of different hydraulic biotopes with changes in discharge. In addition, a biotope-mapping exercise is presently being developed for use at each site to enhance the information provided on loss and gain of biotopes (King pers. comm.).

Although hydraulic measurements must be linked to an assumption of steady state, in order to be usable, inevitable changes in cross-sections are addressed elsewhere in the methodology. Firstly, channel-flushing flows are included in the IFR to minimise channel change. Secondly, post-construction monitoring of the river allows the IFR to be adjusted where necessary. It is acknowledged that more research on flow-related abiotic-biotic links is needed to increase the ability to predict long-term morphological and other change in rivers.

3.7 CONCLUSIONS

In five years, the BBM has developed to the stage where an experienced group of practitioners, working with local river scientists, can make a rapid and structured assessment of the IFR for any river. The level of confidence in the workshop output, however, is directly related to the amount of relevant ecological data available, the availability of river ecologists and fluvial geomorphologists with first-hand experience of the river of concern or similar rivers, and the extent to which data for the workshop have been prepared according to the required IFR format.

The BBM has gained institutional acceptance and support from DWAF, and its results are now used by national and provincial conservation institutions during negotiations that lead to decisions on water allocations. For some proposed developments that have already moved to the Design Phase, the BBM has successfully been used to reserve parts of the hydrological regime for river maintenance.

There have been many other beneficial spin-offs. For instance, the BBM has provided many river scientists with their first structured use of a combination of hydrological, hydraulic, biological, chemical, geomorphological and social data. It has also identified many shortcomings in knowledge and triggered a growing number of focused, medium-term and long-term research projects specifically designed to help improve both the methodology and its output.

Development of the methodology has gained a momentum of its own, through the support of DWAF and many voluntary initiatives by the scientific community. In this way, *inter alia,* guidelines have been drawn up both for the way that hydrological data should be presented for an IFR workshop (Chapter 5) and for the social consultants to use in rural areas. An hydrological model has been produced to help convert the IFR table to dam operating rules that are linked to current rainfall-runoff patterns (Chapter 6). Expansion of the structured approach into the Feasibility Phase is already underway, as is development of a protocol for the later monitoring programme (Chapter 6).

4. PRACTICAL APPLICATION OF THE BUILDING BLOCK METHODOLOGY

Written by D. Louw (then at DWAF) and J. King

4.1 INTRODUCTION

4.2 SYNOPSIS OF PREPARATORY ACTIONS FOR AN IFR WORKSHOP

4.3 DETERMINATION OF THE EXTENT AND NATURE OF THE PROPOSED DEVELOPMENT

4.4 IDENTIFICATION OF RELEVANT SPECIALISTS WITH KNOWLEDGE OF THE RIVER

4.4.1 The geomorphological reach analysis and aerial survey
4.4.2 Habitat integrity
4.4.3 River importance
4.4.4 Fluvial geomorphology
4.4.5 Biological surveys
4.4.6 Assessment of existing and required water quality
4.4.7 Social survey
4.4.8 Collection, simulation and analyses of the hydrological data for the IFR sites
4.4.9 Collection and analysis of hydraulic data for the selected IFR sites
4.4.10 In summary

4.5 THE PLANNING MEETING FOR AN INSTREAM FLOW ASSESSMENT

4.5.1 Determination of the study area (item 5)
4.5.2 Determination of the number and location of IFR sites (item 8)
4.5.3 Desired State (item 9)
4.5.4 Date and venue for the IFR workshop

4.6 THE WORKPLAN LEADING UP TO THE IFR WORKSHOP

4.7 DESIGN OF THE STARTER DOCUMENT

4.8 STRUCTURE AND PURPOSE OF THE SITE VISIT

4.9 THE IFR WORKSHOP

4.9.1 Purpose
4.9.2 Design
4.9.3 Describing the IFR for a site
4.9.4 Necessary personnel and aids

4.10 EXAMPLE OF AN IFR WORKSHOP OUTPUT: FOR IFR SITE 3 OF THE MOOI RIVER

4.10.1 General approach

4.11 POST-WORKSHOP ACTIVITIES

4.1 INTRODUCTION

DWAF follows the IEM Procedure of the Department of Environmental Affairs and Tourism (DEA 1992), which it has adjusted for application to all its proposed water-development projects. DWAF's version of the IEM process consists of incremental phases with related environmental actions (Table 3.2). To ensure that these actions are executed according to the IEM Procedure, an Environmental Task Group (ETG) is appointed to oversee the process. This group consists of people who contribute factual information regarding the ecological and social impacts of the proposed project, and is therefore not fully representative of all IAPs. IAPs may serve on the ETG, or on the Steering Committee which is guiding the total study and to which the ETG reports.

The first of DWAF's IEM phases is the Reconnaissance Phase, which is linked to the related engineering Systems Analyses or Catchment Studies (Section 3.5). Both the ETG and the Steering Committee are usually formed at the beginning of the Reconnaissance Phase. The general terms of reference for an ETG may be as follows: "To supervise application of the IEM procedure during all phases of the proposed water-resource development, thereby ensuring that any impacts on the natural and social environment are considered during the decision-making process."

For each phase of the IEM procedure environmental reports at the appropriate level of resolution are produced. For example, an initial environmental assessment is completed during the Reconnaissance Phase, and an impact assessment or ROIP (Louw 1995a) is initiated later in the same phase and updated at specific intervals during subsequent phases. The various reports emanating from these actions describe the range of expected impacts of a proposed development, suggest mitigatory actions where appropriate, and make recommendations for further work. Options that are strongly undesirable from a social or environmental perspective are usually abandoned early in the process.

Where the flow regime of the rivers would be changed by a proposed water-resource development, it is usually recommended in the ROIP report that an instream flow assessment (IFA) be undertaken during the Pre-feasibility Phase. Since 1990, the BBM has been used in DWAF's Pre-Feasibility Phase (Table 3.2) for these IFAs. This chapter details the practical application. The IFR workshop for the Mooi River, which took place in May 1995, is used as an example where necessary (King 1995b).

4.2 SYNOPSIS OF PREPARATORY ACTIONS FOR AN IFR WORKSHOP

Although the BBM has been designed for use where time is limited, there is a lower limit to the time in which it can successfully be applied. A six-month schedule works well, beginning when the decision is taken to perform an IFA, and culminating with the IFR workshop. For recent workshops, however, less time has sometimes been available, driven by engineering concerns, or more time has been required because adverse flow conditions at the river affect the ability to collect relevant data (see entries marked with asterisk in Table 4.2).

The ideal sequence of actions compared to the reality (Table 4.1) reveals how the main activities gradually proceed in better order as preparations get underway. Table 4.2 provides further details on the main sequence of actions. The activities are under the control of an IFR co-ordinator.

Details of the various activities are given in the following sections.

4.3 DETERMINATION OF THE EXTENT AND NATURE OF THE PROPOSED DEVELOPMENT

Discussion takes place between the IFR co-ordinator, the developer and the consulting engineer regarding the nature and extent of the proposed development. Of particular interest are the number and nature of the options under consideration, the number and lengths of rivers targeted by such options, and the similarity or lack of similarity between the virgin, present and proposed MARs at a number of points pertinent to the proposed development.

4.4 IDENTIFICATION OF RELEVANT SPECIALISTS WITH KNOWLEDGE OF THE RIVER

Having gained some knowledge of the proposed development, specialists are sought who can contribute to the IFA. As the BBM relies to a large extent on expert opinion and best available information, specialists with a good working knowledge of at least one of the necessary categories of information (Table 4.3) are employed.

Table 4.3 Categories of information necessary for the IFR workshop.

CATEGORIES OF INFORMATION	
Aerial video survey of river	Fish
Habitat integrity	Water quality
River importance	Social dependence on ecosystem
Fluvial geomorphology	Hydrology
Aquatic invertebrates	Local hydraulics
Riparian vegetation	

Full details of the techniques being developed to collect any necessary new data, and to present all data, will be included in an IFR manual scheduled for completion in 1998 (Chapter 6). A summary of what has to be done for each category of information, with respect to the IFR workshop, is given below.

4.4.1 The geomorphological reach analysis and aerial survey

A reach analysis of the full length of the river(s) in the identified study area is done by the geomorphologist on the team as a desk-top exercise, in order to provide initial information of different parts of the river in terms of catchment characteristics, geology, soil erodibility, vegetal cover and channel slope. The river is

Table 4.1 The ideal sequence of actions for an IFR workshop, compared to the actual programme for the Mooi River IFR workshop.

ACTIONS		MONTHS					
		1	2	3	4	5	6
Budget approval and breakdown	Planned timing						
	Actual timing						
Habitat integrity and fluvial geomorphology	Planned timing						
	Actual timing						
Arrange IFR planning meeting	Planned timing						
	Actual timing						
Preliminary workshop arrangements	Planned timing						
	Actual timing						
IFR planning meeting	Planned timing						
	Actual timing						
Draft programme	Planned timing						
	Actual timing						
Contact authors re: Terms of Reference	Planned timing						
	Actual timing						
Venue	Planned timing						
	Actual timing						
Site visit to choose IFR sites	Planned timing						
	Actual timing						
Invitations to participants	Planned timing						
	Actual timing						
Arrangements re: starter document	Planned timing						
	Actual timing						
Liaison re: hydrology	Planned timing						
	Actual timing						
Finalise programme	Planned timing						
	Actual timing						
Liaison re: hydrology	Planned timing						
	Actual timing						
Contact with authors re: papers	Planned timing						
	Actual timing						
Posters	Planned timing						
	Actual timing						
Travel arrangements	Planned timing						
	Actual timing						
Receive specialist reports for the starter document	Planned timing						
	Actual timing						
Distribute starter document	Planned timing						
	Actual timing						
Final logistical arrangements	Planned timing						
	Actual timing						
WORKSHOP	Planned timing						
	Actual timing						

Table 4.2 Preparatory actions for an IFR workshop.

PRELIMINARY ACTIONS FOR BUDGETING PURPOSES

- Determine extent and nature of proposed development.
- Identify relevant specialists with knowledge of the river.
- Enquire into the general condition and importance of the river and the extent to which relevant data are available.
- Determine gaps in the data and the work needed to rectify this.
- Assess the probable extent of the study area to be used for the IFA.
- Assess the probable number of IFR sites to be used in the IFA.
- Seek approval for draft budget.

MONTH 1

- Create diagrammatic map of the study area, showing rivers, gauging weirs, proposed and extant dams, IFR sites, the virgin and present MAR and the virgin and present median annual runoff.
- Liaise closely with the engineering consultants during all steps of the IFA, to ensure that current knowledge on the proposed development is being used.
- Arrange for geomorphologists to complete reach analysis for the river(s) in the study area and make this available to the team doing the aerial survey for the assessment of habitat integrity.
- Arrange for team consisting of, at least, a camera operator, a navigator and the person responsible for the Habitat Integrity report to fly the study area by helicopter, in order to make a video recording of the river(s). This will be used for assessments of instream and riparian habitat integrity, for the geomorphological analysis of the channel and for the initial identification of the IFR sites*. (Ideally, this flight should take place during the Reconnaissance Phase as part of the initial assessment, but at the latest during the IFR assessment).
- Arrange for a preliminary assessment of relevant hydrological data, for presentation at the IFR planning meeting.
- Organise and attend the IFR planning meeting, inviting key specialists who will be involved in the IFR workshop. Topics to be dealt with are:
    - approval of the suggested extent of the study area
    - approval of the suggested number of IFR sites and their locations
    - preliminary discussions on the DS of the river(s) in the study area
    - discussion on, and approval of, the extent of necessary biological, social, hydrological, hydraulic and other studies to be completed before the IFR workshop
    - determination of any additional participants for the IFR workshop
    - determination of a date for the IFR workshop that is suitable for key persons*
- Organise the site visit to choose the IFR sites within the selected reaches. Those who should attend the site visit include the land surveyor, the hydraulic modeller, and the fish, invertebrate, geomorphological and riparian-vegetation specialists.
- Inform all potential workshop participants not present at the IFR planning meeting of the date of the workshop.

Table 4.2 Continued.

MONTH 2

- Undertake the IFR site visit and finalise selection of sites*.
- Guide selection of cross-sections at the chosen sites, ensuring that appropriate data are recorded and site photographs taken that show the relationship between discharge and wetted habitat*.
- Inform all specialists of the latitudes and longitudes of the IFR sites and ensure adequate liaison between those dependent on each other's work*.
- Organise first meeting between key ecologists and the hydrological modeller for the workshop, to discuss the preparation of appropriate hydrological data.
- Identify venue for workshop.
- Compile a first draft of the programme for the IFR workshop.
- Contact author of each report regarding their terms of reference and deadline for their contribution to the IFR workshop starter document.
- Book the venue.
- Prepare skeleton of IFR workshop starter document.
- Initiate travel arrangements for participants to attend the IFR workshop.

MONTH 3

- Prepare document for the site visit that will take place at the beginning of the IFR workshop and finalise the programme for the visit.
- Decide on chair, group leaders and composition of groups, for the IFR workshop.
- Prepare draft statement of DS, and send to all workshop participants for comment.
- Finalise IFR workshop programme.
- Continue meetings of hydrological modeller and ecologists, as necessary.
- Remind workshop participants regarding submission of contributions for workshop starter document.

MONTH 4

- Receive all reports from specialists and incorporate into IFR workshop starter documents.
- Distribute starter documents to workshop participants at least three weeks before the workshop begins, so that they can be read before the workshop.
- Prepare wall poster of draft DS, for discussion at IFR workshop.
- Prepare wall posters of sites, showing features of channel at different discharges, location of site and other pertinent information.
- Prepare wall posters of "near-natural" daily hydrographs for as many years as available.
- Finalise site visit document.
- Make final travel arrangements.
- Make final arrangements regarding the venue.

MONTH 5

- Run the IFR workshop*.

then flown by helicopter, and a continuous video recording made of each 5 km segment of the channel and riparian surrounds. The technique for the reach analysis is being developed by Dr K. Rowntreee and Mr R. Wadeson of the Geography Department, Rhodes University, and for the aerial survey by Dr N. Kleynhans, Institute for Water Quality Studies (IWQS), DWAF.

4.4.2 Habitat integrity

Using the information gained during the aerial survey, an assessment is made of the habitat integrity of the instream channel and the riparian zones. The approaches used are being developed by Dr N. Kleynhans (instream channel) and Mr N. Kemper, then at NSI, (riparian zone). In some smaller rivers, the assessment can be made from the ground, using Kleynhans' Method as modified by Dr B. Gale, then at Cape Nature Conservation (CNC).

4.4.3 River importance

Assessment of river importance is usually done in an *ad hoc* fashion by specialists who know the river and region, and the topic has only recently (1995) been represented by a formal document in the workshop starter documents. A structured way of assessing the ecological, social and economic importance of any river is being developed by, *inter alia,* Dr N. Kleynhans, Mr S. Eekhout then at SW, University of Cape Town, and Prof. J. O'Keeffe, Institute for Water Research (IWR), Rhodes University, Grahamstown.

4.4.4 Fluvial geomorphology

Following the reach analysis, a detailed assessment is made of the geomorphology of the channel in the study area. Use is made, among others, of the reach analysis and the video recording taken during the aerial survey, and the specialist usually takes part in a data-gathering trip to the IFR sites. Features of the sites, such as critical habitats or unique ecosystem components are noted, as are changes that are likely to occur when flow patterns change. Information on downstream changes in channel morphology, major zones with different sediment regimes, and features of the channel such as secondary channels, meandering and braiding, is given, as are details of biotopes at the IFR sites. Comments are made on the likely responses of the different reaches to impoundment. The approach is being developed by Dr K. Rowntree and Mr R. Wadeson, Geography Department, Rhodes University. Dr G. Heritage, until recently at the Centre for Water in the Environment (CWE), University of the Witwatersrand, has contributed ideas on alternative approaches.

4.4.5 Biological surveys

Before the IFR workshop, literature reviews and field surveys of the riparian, aquatic invertebrate and fish communities are undertaken by local specialists, to collect data on as many as possible of the following:

- the distribution of species along the river;
- the presence of rare and endangered species or communities;
- longitudinal zonation of the river in biotic terms;
- the range of tolerance of species to specific flow-related environmental variables;
- habitat requirements for maintaining populations;
- life history data;
- any other relevant and useful data that can be collected within the known time and financial constraints.

Methods for effectively presenting these data in the workshop starter documents are being spearheaded by Dr N. Kleynhans (fish) and Mr N. Kemper (riparian vegetation).

4.4.6 Assessment of existing and required water quality

At present, there is no formal procedure for combining the water quantity and water quality requirements when compiling a recommended modified flow regime. In recent IFR workshops, a report on the existing and required future water quality of the river has been presented, but the method for incorporating that in the assessment still needs to be developed. Dr C. Palmer, IWR, Rhodes University, Grahamstown, is investigating ways of developing this component.

4.4.7 Social survey

Before the IFR workshop, an assessment is made of the dependence of local rural communities on the riverine ecosystem for subsistence purposes. The importance of river attributes such as, *inter alia,* run-of-river flow, the riparian species used for thatching, medicines or food, and floodplains and pools as food sources, is assessed. Initial development of an approach is being done by Ms S. Pollard of Wits Rural Facility, University of the Witwatersrand. An additional assessment of others who depend on the riverine ecosystem, such as tourist industries and nature reserves, is also made where appropriate, and is usually reflected in the statements on River Importance and DS.

4.4.8 Collection, simulation and analyses of the hydrological data for the IFR sites

Identification of suitable hydrological data for use at an IFR workshop has been a difficult and protracted effort. Initially, hydrologists were producing the standard kind of engineering analyses used for water management purposes. The ecologists did not understand these data or, if they did, usually could not react to them because they were presented as monthly volumes rather than as descriptions of instantaneous flow conditions in the river. Over the last four years, the kinds of data needed and, more importantly, the way they are presented, have become clearer. A description of the data presently seen as necessary is given in Chapter 5. One of the authors (JMK) has spearheaded this activity.

4.4.9 Collection and analysis of hydraulic data for the selected IFR sites

Information on the cross-section shape of the channel and on the characteristic hydraulics at each selected site allow workshop participants to match details from the hydrological analyses with their own assessment of the flows of various magnitudes that would facilitate specific ecological and geomorphological functions. The hydraulic data provide information that is intuitively understood, appreciated and used by workshop participants (see Section 4.9.3). Mr W. Rowlston (Subdirectorate of Strategic Planning, DWAF), Ms A. Jordonova (Subdirectorate of Hydraulic Studies, DWAF) and Mr A. Birkhead (CWE, University of the Witwatersrand) have been prominent in guiding the use of hydraulic data in IFR workshops.

4.4.10 In summary

Bearing in mind the above requirements for IFR workshops, gaps in knowledge are identified at the IFR planning meeting (Section 4.5) and the work needed to rectify this agreed upon.

4.5 THE PLANNING MEETING FOR AN INSTREAM FLOW ASSESSMENT

The IFR planning meeting is designed to bring together, for the first and only time before the workshop, most of those who will play a significant role in the IFA. Participants are picked for their specialist knowledge of the area or of IFAs, or for their involvement in the planned water-resource development.

The Planning Meeting for the Mooi River IFR workshop was typical of its kind. It was held on 16 January 1995, and was chaired by the Head of Environment Studies (DWAF). Twenty people attended, including representatives from DWAF (Environment Studies, Hydraulic Studies, Project Planning, IWQS and Hydrology), Umgeni Water (the water supplier for the region that includes the Mooi catchment), Natal Parks Board and the engineering consultants for the development. Also present were specialist scientists familiar with IFAs, and a social scientist providing input on the importance of the river to rural populations in the area. Initial presentations made and topics discussed (Table 4.4) concerned the background to the development, the present condition of the river and state of development of its catchment, and a summary of the extent of the relevant information available for the IFR workshop together with recommendations of further necessary work.

Table 4.4 Agenda for the Mooi River IFR planning meeting.

ITEM	TOPIC
1	Welcome
2	Background to Mooi/Mgeni water transfer scheme
3	Description of the IEM process followed for the scheme to date
4	Proposed methodology for determination of instream flow requirement for the Mooi River
5	Proposal on extent of study area for IFR assessment for the Mooi River
6	Overview of present condition of the Mooi River
7	Quality and extent of available hydrological data on the Mooi River
8	Proposal on location and number of IFR sites for consideration in IFR workshop
9	Discussion to determine the DS of the Mooi River
10	Available information relevant to the IFR assessment
11	Further work/information needed for the IFR workshop
12	Co-ordination with the Tugela River development study
13	Workplan leading up to the Mooi IFR workshop
14	Date and venue of the workshop
15	Participant list for the workshop

Items 5, 8 and 9 (Table 4.4), which are structured activities forming part of the BBM, are discussed below.

4.5.1 *Determination of the study area (item 5)*

At the IFR planning meeting, a decision is made on the extent of the study area. Knowledge of the location and nature of all proposed development options will be needed, as well as the location of all rivers potentially included in the proposed development and the general character of the riverine ecosystem as a whole (Table 4.5).

Table 4.5 Considerations when delineating the extent of the study area.

CRITERIA
1 Upstream extent of study area will be delineated by the most upstream development option
2 Downstream extent of study area will be delineated by the furthest that flow could be managed by the options
3 Downstream reaches beyond where flow could actively be managed should still be considered if they contain areas of high ecological or social importance

4.5.2 Determination of the number and location of IFR sites (item 8)

Early IFR workshops produced lower-confidence recommendations on IFRs than later ones. One of the principle reasons for this was the unstructured process of selection of the IFR sites. These sites are used for all hydrological modelling and, perhaps more important, for all hydraulic modelling for the IFR workshop. The cross-section information on stage-discharge relationships emanating from the hydraulic modelling is the fundamental "language of communication" between those working with water volumes (managers, engineers) and those working with depths, velocities and biotopes (river scientists). If the sites are not representative of local hydraulic conditions, the conversion of required on-site hydraulic conditions to water volumes could be inappropriate, thereby affecting the whole planning process.

Ms D. Louw and Mr N. Kemper are developing the process of site selection, with input from specialists in several disciplines. This process begins with the partition of the study area into 'IFR river stretches' in a desktop exercise (Table 4.6). The chosen parts of the river(s) will have different hydrological, geomorphological and/or biological characteristics.

Table 4.6 River characteristics to identify when partitioning the study area into stretches of river, each of which will be represented by an IFR site.

	RIVER CHARACTERISTICS
1	Each stretch of the main channel that receives a major tributary that could materially alter channel size, shape and flow conditions
2	Each part of the study area with a different DS
3	Longitudinal biological zones with different biotopes and biota
4	Major parts of the study area where the flow regime could be drastically changed in future
5	Stretches with a high proportion of natural biota, as the BBM uses information on the distribution and habitat requirements of such species

Within each chosen stretch, one or more IFR site(s) is/are tentatively located from maps and with reference to the aerial video (Table 4.7), and then one per stretch is chosen during a site visit.

Table 4.7 Considerations when selecting IFR sites within the chosen IFR reaches.

	CONSIDERATIONS
1	Close proximity to gauging weirs with good quality, near-natural, daily flow data
2	Locality of the proposed development options
3	Locality and characteristics of tributaries
4	Areas with high social dependence
5	Suitability for follow-up monitoring
6	High diversity of instream and riparian biotopes
7	Suitability for accurate hydraulic modelling, especially of low flows
8	Accessibility
9	Presence of critical biotopes and those with critical depth relationships, e.g. riffles

4.5.3 Desired State (item 9)

Bearing in mind the condition and importance of the river, discussions take place in the IFR planning meeting on its DS. These discussions, together with the Habitat Integrity assessment, lead to a draft statement on DS that is distributed to the workshop participants for comment. The summary statement is then presented at the IFR workshop for discussion, refinement and the reaching of consensus. The approach is still being developed, but essentially it involves an attempt to identify a **specific** and **realistic** DS for the river (Table 4.8). At present, the team that produces the statement consists mainly of managers, planners, those with specialist scientific knowledge of the river or of some aspect of river functioning, nature conservation officials and those with knowledge of the social importance of the river. The recommended modified flow regime for the river, compiled in the IFR workshop, will be designed to facilitate maintenance of the river at the recommended DS.

Table 4.8 Draft proposal on the IFR workshop objective and Desired State of the Mooi River in the study area.

DRAFT PROPOSAL
WORKSHOP OBJECTIVE: To determine a modified flow regime for the Mooi River, between the proposed Dartington and Springrove Dams to the confluence with the Tugela River, that will promote the attainment and/or maintenance of the following DS.
DS: • Perennial flow regime that is a muted version of the virgin flow regime • Sustainable resource for those dependent on the presence of an healthy river ecosystem (sustainable defined in terms of run-of-river flow, subsistence fisheries, subsistence use of the riparian vegetation for grazing, building materials, food and medicines; and cultural and religious activities) • Maintenance of the present diversity of species, habitats and processes • All of the above are described in more detail for each IFR reach in the starter document

In a recent development, Prof. K. Rogers of the CWE, University of the Witwatersrand, has researched the whole approach to determining DS, as part of the KNPRRP (Rogers and Bestbier 1997).

4.5.4 Date and venue for the IFR workshop

Approval is sought at the Planning Meeting for the suggested date for the IFR workshop (Table 4.9).

Table 4.9 Criteria for selection of the date and venue for the IFR workshop.

DATE	
1	Need to recognise the developer's planning deadlines (often weeks to months only)
2	At least six months run-in time preferable for IFR workshop, in order to allow key specialists to complete their investigations, write reports and reserve time to attend the workshop
3	Run-in time should span a season when the river has low flow if possible, in order to obtain good hydraulic data from chosen IFR sites
VENUE	
1	Should be in the catchment of concern, near to one or more of the chosen IFR sites
2	Should have conference facilities
3	Should be isolated, so all participants stay day and night through the full workshop
4	Should be pleasant, with reasonable food and facilities for socialising in the evenings

4.6 THE WORKPLAN LEADING UP TO THE IFR WORKSHOP

The workplan for the procedures occurring between the IFR planning meeting and the IFR workshop consists of two kinds of action: logistical arrangements and scientific preparation. These two actions are related and dependent on each other (Figure 4.1).

As soon as possible after the Planning Meeting:

First step

- The budget is adjusted as necessary.

Second step

- The IFR sites are agreed upon on the ground, as several other necessary studies can only be undertaken once the researchers know the site localities.
- Workshop logistics, including dates and venues are finalised, bearing in mind that the number of participants will influence the choice of venue, and the identity of the participants will influence acceptable dates for the workshop.

Third step

- Liaison takes place with all researchers that are undertaking scientific studies. Their terms of reference are agreed upon and the deadline for contributions to the workshop starter document given.
- The draft programme for the IFR workshop is written.

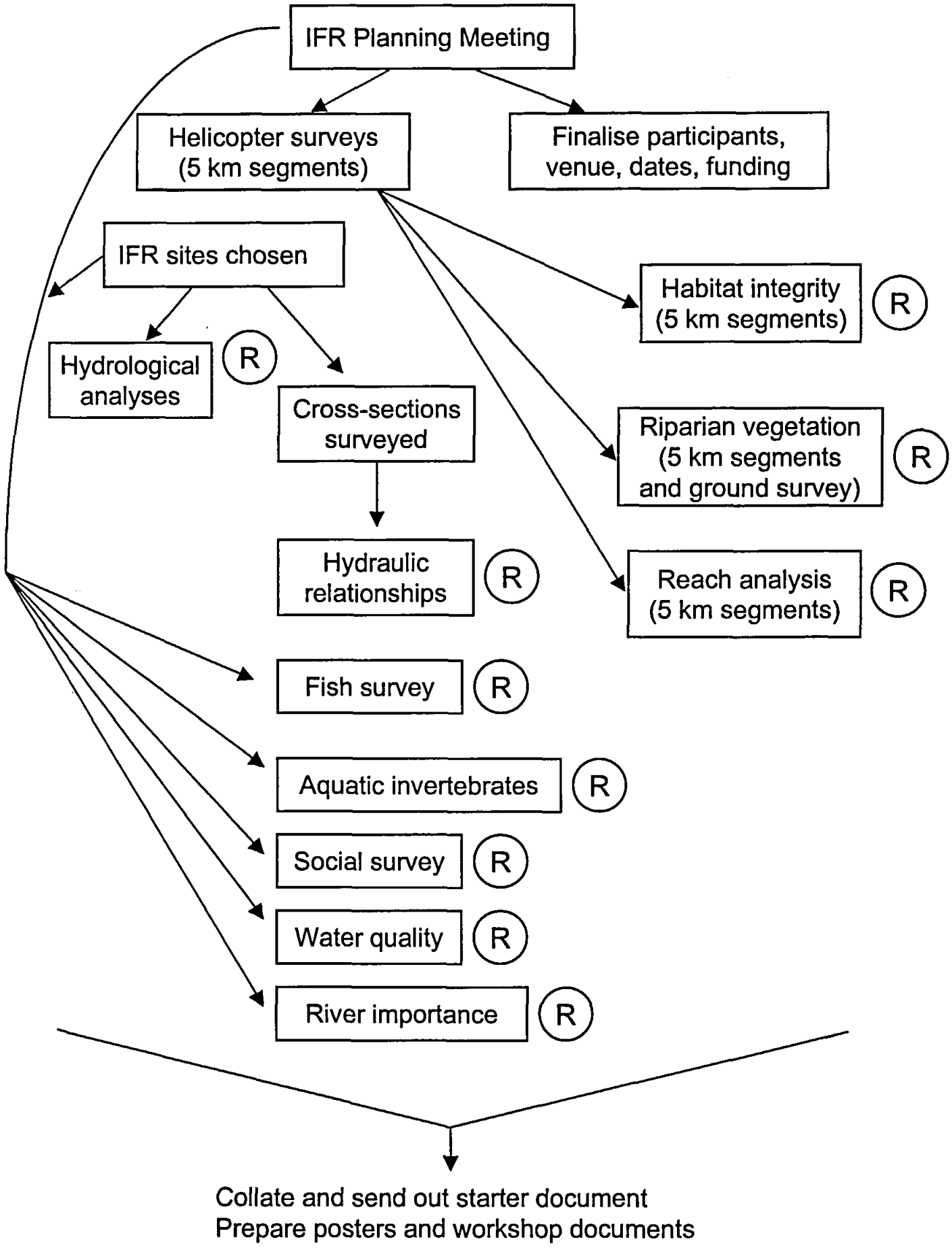


Figure 4.1 **One example of the sequence of events from the IFR planning meeting to the IFR workshop. R - report written for the IFR workshop starter document.**

Fourth step

- The starter document for the IFR workshop is prepared, which includes all reports from the specialists.
- The information and photographs taken during site visits are used to prepare the workshop posters on site information.
- Liaison with the hydrologist takes place to ensure the preparation of suitable hydrology posters for the IFR workshop.
- Final travel arrangements are made, as well as all logistical arrangements with the venue.

Fifth step

- The IFR workshop.

4.7 DESIGN OF THE STARTER DOCUMENT

Reports of the various activities, each limited to about ten pages but containing as much specific and relevant detail as possible, are collated into a starter document by the workshop co-ordinator and sent to all participants at least three weeks before the IFR workshop (Table 4.10). It is required that the document be read thoroughly before the workshop, as limited time is available for this exchange of information at the workshop. Additional data may be brought to the workshop for reference purposes, and submitted as Appendices, except those on the hydrology, which might be extensive. Instead, selected summary hydrological data are included, and more comprehensive information is available at the workshop on computer.

For many if not most rivers, it is felt at the beginning of the IFR exercise that little is known of its ecological and geomorphological character, but a surprising amount can be learnt about the system in a short time. The BBM has been developed to use the restricted amounts of time, money and specialist availability to their best advantages. Thus, specialists have been guided into honing their reports to only those details related to the task at hand; rapid biomonitoring techniques for the aquatic invertebrates are used to give a snapshot assessment of the ecosystem health of the river; reports draw attention to any species that are indicative of specific flow conditions and which might thus aid understanding of the system; general knowledge of the rivers of the area is used where river-specific knowledge is missing; and hydraulic cross-section figures link information on channel shape, discharge, depth, average velocity, wetted perimeter, substratum particle size and the distribution of species (particularly riparian) with respect to extent of inundation and flow patterns. The graphic plots of the hydraulic information have been found to be the essential link which allow ecologists and water engineers/planners to communicate.

An additional benefit of the starter document is that it is a permanent, albeit less than comprehensive, record of the ecological and geomorphological character of the river at the time. For the majority of rivers, this could be the only such record. It also contains information on species distributions and channel shape that is gradually helping build a comprehensive database at the national level.

Table 4.10 Standard table of contents of the starter document for an IFR workshop.

1.	INTRODUCTION
1.1	Programme
1.2	Participant list
2.	BACKGROUND TO THE PROPOSED WATER-RESOURCE DEVELOPMENT
3.	THE DETERMINATION OF THE IFR
3.1	The BBM
3.2	Application of the BBM to the XXX River
3.3	The purpose, selection process, location and characteristics of the IFR sites
4.	ENVIRONMENTAL OVERVIEW
4.1	Habitat integrity
4.2	Present water quality and potential future problems
5.	DESCRIPTION OF ECOSYSTEM COMPONENTS FOR USE IN THE IFA
5.1	Geomorphological characteristics of the river and the impact of flow decrease on these
5.2	Status of fish populations and their flow-related habitat requirements
5.3	Status of riparian communities and their flow-related habitat requirements
5.4	Status of the aquatic invertebrates and their flow-related habitat requirements
5.5	Hydrology
5.6	Hydraulics
5.7	Social dependence on the riverine ecosystem
6.	DS OF THE RIVER(S) - for discussion and agreement
6.1	River importance
6.2	DS
7.	SUMMARY OF IMPACT ASSESSMENTS OF DEVELOPMENT OPTIONS

4.8 STRUCTURE AND PURPOSE OF THE SITE VISIT

A site visit by all workshop participants, to all the IFR sites if possible, has become the essential first part of an IFR workshop (Figure 4.2). Attempts to run the workshop without such a visit have proved unsatisfactory for several reasons. Firstly, a considerable amount of local information and understanding is gained simply by driving through the catchment and talking on the banks of the river. Secondly, within the workshop, discussions by the participants and guidance by the facilitators are both improved by having seen the river, and so the level of confidence in the output is higher than without this opportunity. Thirdly, activities at the site visit are gradually becoming more structured, allowing river scientists who might have little experience with hydraulics or hydrology to assess the sites from these perspectives. The site visit now centres around a specially prepared **site-visit document** for each participant which consists of the items listed below.

- A map, showing with colour coding the route travelled during the site visit; all river reaches within the study area; location of the proposed development(s) and any towns; the numbered 5 km sections of river

as delineated for the video survey and then used in some of the starter document reports; and the IFR sites.

- A diagrammatic map, showing the virgin and present-day MAR at points of interest throughout the study area, including at the IFR sites.
- Selected summary hydrological data.
- A list of aspects for each participant to consider or observe, so that they develop a mental image of the character of each site (Table 4.11). An improved checklist for the site visit is being developed.
- Detailed graphic plots of the surveyed IFR cross-sections at each site, with the distribution of riparian and fringing species shown as well as the water surface elevations (WSEs) of different magnitude flows (Figure 4.3). Usually two such figures are provided for each cross-section: one showing the WSE of all larger flows, and one focusing in on the WSE of the lower flows. These allow each person to walk the cross-sections, note the zonation of the various vegetal species and the character of the channel, and ponder the implications, in terms of wetted habitat and levels of inundation, of different size flows. Graphs of the stage-discharge and other hydraulic relationships are also provided.

The structuring of the site visit has received widespread approval, and clearly leads to more confident discussions in the workshop. The workshop co-ordinator manages the site visit, which typically is the first item on the workshop programme and finishes with the participants' arrival at the workshop venue.

4.9 THE IFR WORKSHOP

4.9.1 Purpose

By its existence, the workshop reflects the fact that retention of a river's virgin flow regime cannot usually coincide with development of its water resources. Accepting this and bearing in mind the DS for the river, the purpose of the IFR workshop is to make a recommendation of the minimum flow requirements for maintenance of the river; **without consideration at this stage** of the present and future water demands of offstream users. In other words, the task is to describe how much water it is believed is required to maintain the river in the DS, and **not** to consider how that flow regime will be provided.

This recommendation is made by using the available information in the starter document and the understanding gained in the site visit to compile, in a structured way and by consensus, a comprehensive flow regime for the river. The flow regime so compiled for each IFR site represents, to the best of available understanding, the minimum quantity of water considered necessary in space and time to facilitate attaining and/or maintaining the DS of the river.

4.9.2 Design

The workshop consists of four main sessions (Table 4.12). Many different manipulations of the sequence and time allowed for each activity have been tested, with the overriding factors being that some participants will know very little of the river and have to gain a working knowledge quickly, while most cannot take longer than four days from their other responsibilities and often would prefer to take less.

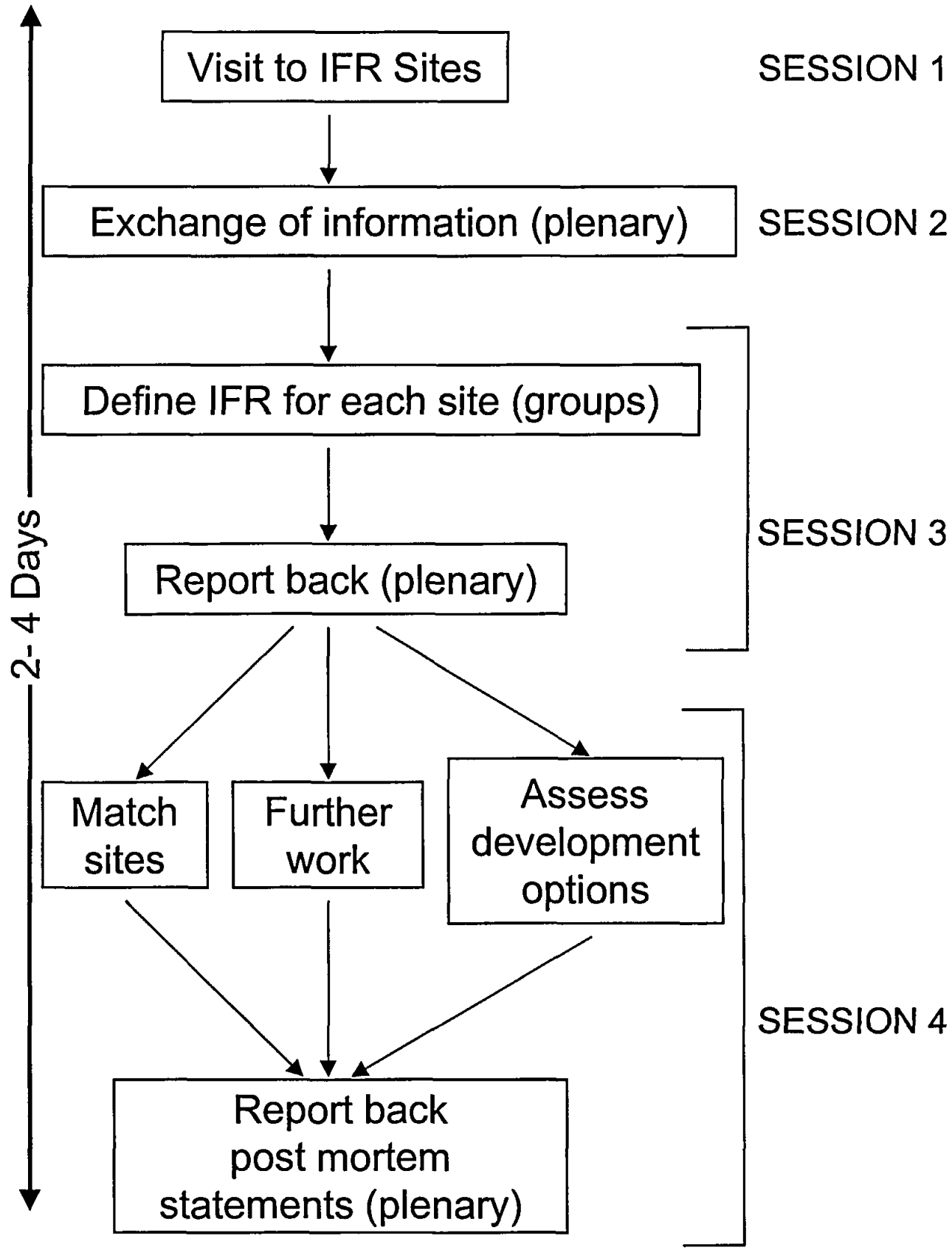


Figure 4.2 **The sequence of events in an IFR workshop.**

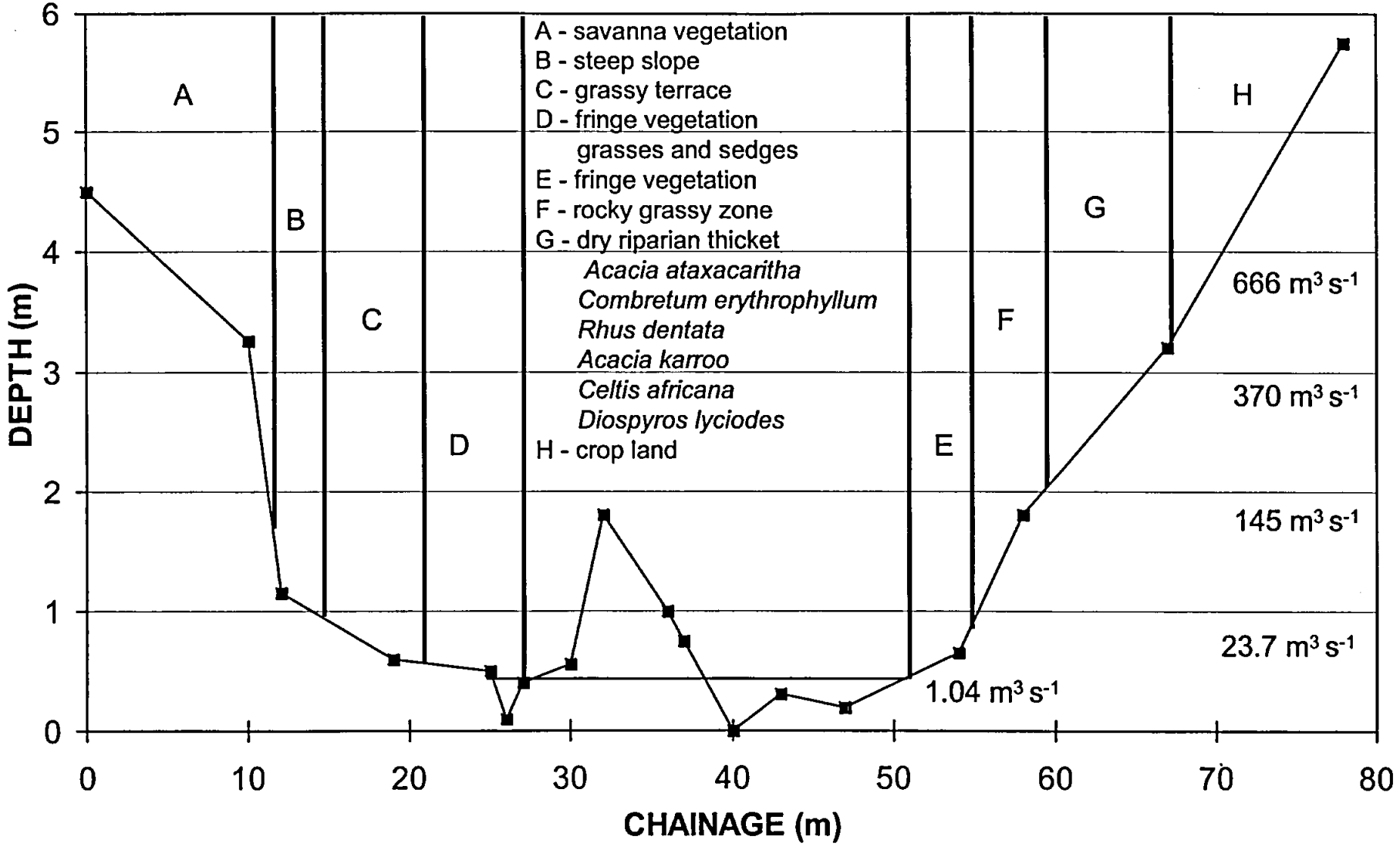


Figure 4.3 **Example of a surveyed IFR cross-section, showing the stage-discharge relationship and the distribution of vegetal species.**

Table 4.11 First draft of list of aspects to consider, discuss or observe during an IFR site visit.

	LIST OF ASPECTS
	MAKE NOTES ON THE FOLLOWING ON YOUR SITE DOCUMENTS
1	Try to establish the width and depth of the wetted channel, and the discharge. Come to group decision on discharge pertaining at the day of the site visit.
2	Note the mosaic of substrata: their particles sizes; degree of embeddedness in fines; the proportions of bedrock, boulders, cobbles, gravel, sand and mud.
3	Note the mosaic of flow types: still water, smooth flow, turbulent flow, chaotic flow, as well as isolated areas of water.
4	Do specific flow types only occur over specific substrata (substrata as listed in 2)?
5	Are there any riffles (areas of shallow, turbulent flow over cobbles)? Are they covered by water?
6	Does the water reach the marginal vegetation, creating aquatic habitat there? If not, how much higher must it be to do this?
7	What is the nature of the riparian vegetation? Is there marked vertical zonation within it reflecting inundation patterns? At what heights are the zones?
8	Is there an obvious tree line made by flows controlling channel encroachment? Where is it on the cross-section diagrams?
9	Is there flood debris in the trees? How high is it, and when might it have been deposited?
10	Is there evidence of newly forming sand banks, perhaps with young vegetation? Is the channel single, braided or anastomosing?
11	Are the banks stable or eroding? What might be causing any erosion?
12	Are the rocks clean, or coated with sediments, algae or other material? Are there marks on the rocks, indicating grazing by fish or recent inundation levels? Note where such marks are in relation to today's flow.
13	Are there spoor showing that terrestrial animals visit the river, and signs of crabs or other aquatic life being eaten?
14	Are there secondary channels, oxbow lakes or wetlands? Are they wet? How would they receive water?
15	Does the water quality appear acceptable?
16	Are rural people using the river as a resource in any way?
17	What would be an acceptable baseflow in the dry season?
18	What would be an acceptable baseflow in the wet season?
19	Can you identify where 'bankful' or the 'channel maintenance' discharge would reach on the cross-section plot?

The present apportioning of time to sessions works well:

- Session 1: The site visit. This usually occupies half a day, on the same day that participants travel into the area from many parts of the country.
- Session 2: The exchange of information. This usually takes half a day at most. Short presentations (mostly five minute summaries) are given of each paper in the starter document, in plenary session. Participants are expected to be familiar with all the material presented, through having read the starter documents, and in this session may clarify uncertainties through questions.
- Session 3: Compilation of the IFR. Initially, the biological, geomorphological and chemical specialists group by discipline and record in general terms the kinds of flows that might be relevant from their perspective and the motivations for those flows. Participants then split into pre-chosen groups, usually two, each of which contains at least one specialist from each of the relevant river sciences (fish, riparian

and aquatic vegetation, aquatic invertebrates, geomorphology, hydrology, hydraulics, aquatic chemistry) and facilitated by a river scientist experienced in the methodology. The IFR sites are allocated to the two groups in some logical way, and each group then focuses on the IFR for one site at a time. Identification and description of the recommended modified flow regime for each site is done in a specific way (Section 4.9.3). Plenary report-back sessions are convened when appropriate. Typically, it will take two groups one to one and a half days to compile the IFR for four sites (two sites each). Probably at least two-thirds of that time will be spent on the first of the two sites that each group considers, as the participants familiarise themselves with the river and the method, and begin to work as a team.

- Session 4: The wrap-up session of the workshop contains five main activities spread over about half a day. The recommended flow regimes for all the IFR sites are compared, to check there are no major mismatches in the flows that have been asked for. The various development options are considered and statements made regarding their environmental acceptability. Further necessary work is identified and usually falls into three categories: short-term work required to refine the IFR output - this usually takes place in the Feasibility Phase; medium-term research required to improve the methodology; and long-term fundamental research on subjects about which little is known. There is no money automatically available for any of this work. A post mortem of activities takes place, where constructive criticism is invited on any aspect of the workshop or the methodology used. Finally, any other statements that participants wish to make individually or collectively are noted and discussed if necessary. Reports on all these activities form part of the workshop proceedings.

4.9.3 Describing the IFR for a site

A group concentrates on one IFR site at a time. The group leader appoints a scribe, with the writer of the workshop proceedings fulfilling that role in one group. The leader guides general discussion, often for one or two hours, on the kinds of flow that should be retained in the river to promote the DS. Gradually, discussion is focused on actual amounts for specific times, with the aim being to fill in the blanks in the IFR table (Table 4.13). To aid discussions, the large posters of hydrological and site information are displayed on the wall. The hydrological posters display the best hydrological data available; ideally, these consist of daily hydrographs for many whole years, ranked from wettest to driest years and with the months lined up vertically and clearly marked. The hydrologist who prepared the data is on hand, with a computer and copious additional supporting data, to answer any enquiries. The site poster for any one site consists of large upstream and downstream colour photographs of the channel, with the surveyed cross-sections and the discharge at the time of the photograph clearly marked. If possible, there are photographs of the site taken at several different known discharges. The site poster also contains a map, to remind participants of the location of the site and other relevant places, and a detailed drawing of the cross-section profile.

Using the methodology and bearing in mind the DS, the river scientists are encouraged to reach consensus on the lowest baseflow that they consider acceptable for each month (building block 1) and the smallest number of freshes and floods (building blocks 2 and 3) that they consider to be necessary. All freshes and floods are described in terms of the required duration, magnitude, frequency and month of occurrence. To guide them, the scientists use their general or specific knowledge of the river and of river functioning, as well as the graphics on cross sections and hydraulic relationships (depth, width and wetted perimeter versus discharge - Figure 4.3). Those group members who are not river scientists participate fully, and soon understand the process, but do not contribute to the figures entered into the IFR Table. When consensus is reached on a figure, it is entered into the IFR table, each specialist records his or her motivation for that

Table 4.12 Programme for the Mooi River IFR workshop. All presentations in Session 2 are made by the authors who contributed specialist reports to the starter document and pertain specifically the to Mooi River (NSI 1995c).

WORKSHOP PROGRAMME	
DAY 1 SESSION 1	SITE VISIT
12h00 - 18h00	Visit IFR sites 1, 2 and 3
DAY 2 SESSION 1 (cont.)	
07h00-09h30	*Visit IFR site 4*
DAY 2 SESSION 2	PRESENTATIONS ON BACKGROUND INFORMATION
10h00	Workshop objective
10h05	Video demonstrating habitat integrity
10h35	Geomorphology
10h45	River importance
10H55	Status of fish species and their flow requirements
11h05	Status of aquatic invertebrates and their flow requirements
11h15	Status of the riparian vegetation and its flow requirements
11h25	Present and future water quality
11h35	Hydrology
12h05	Hydraulics
12h15	*Social dependence on the riverine ecosystem*
12h25	Lunch
13h25	Discussion and consensus on draft statement on DS
14h15	Use of the methodology to compile an IFR for each site
14h30	Group sessions (2 groups) to compile an IFR for each site
17h00	Close
DAY 3 SESSION 2 (cont.)	
08h00 - 19h30	Group sessions continue, with plenary sessions as appropriate
DAY 4 SESSION 2 (final)	
08h00	Final plenary report-back, with the recommended IFR for each site
09h00	Presentation on development options
09h50	Matching IFRs for all sites (group 1)
	Assessing development options (group 2)
	Further work (group 2)
11h50	Lunch
12h50	Plenary session to report back, with input from other group
	Post mortem and statements
13h50	Closure

Table 4.13 The IFR recommended for IFR site 3 on the Mooi River (NSI 1995c). Flow percentiles were not determined. A more updated IFR table is provided in Tharme (1997b).

Site: 3 **River:** Mooi River **MAR:** 330 $m^3 \times 10^6$

IFR BUILDING BLOCKS		OCT	NOV	DEC	JAN	FEB			MAR	APR	MAY	JUN	JUL	AUG	SEP	VOL ($m^3 \times 10^6$)	% MAR
CAPPING of LOW FLOWS (m^3s^{-1})											4	4	3	1	4		
IFR for maintenance	LOW FLOWS (m^3s^{-1})	2	3	3.5	5	5			4	3	2.5	2	1.5	1.5	1.5	91	27
	1. Flow percentile (%)	-	-	-	-	-			-	-	-	-	-	-	-		
	FLOODS																
	1. Magnitude (m^3s^{-1})		10	10	10	40	100	145	10								
	2. Duration (d)		5	5	5	8	8	8	5							23	7
	3. Return period (y)					1:1	1:3	1:5									
	4. Flow percentile (%)	-	-	-	-	-			-	-	-	-	-	-	-		
IFR for drought conditions	LOW FLOWS (m^3s^{-1})	0.4	0.7	0.8	1.5	1.9			1.7	1.2	0.8	0.6	0.5	0.3	0.3	114	34
	1. Flow percentile (%)	-	-	-	-	-			-	-	-	-	-	-	-		
	FLOODS																
	1. Magnitude (m^3s^{-1})		5		8				5								
	2. Duration (d)		5		5				5								
	3. Return period (y)																
	4. Flow percentile (%)	-	-	-	-	-			-	-	-	-	-	-	-		

discharge, and the hydrological record is referred to, in order to check that it is within the bounds of the virgin flow regime for that month over the entire period of the record.

It is easiest to consider the baseflow regime first, filling into the blank table a baseflow discharge for each month. Usually, the baseflows for the wettest and driest months of the year are filled in first, with extrapolation or independent assessment for the other months, as appropriate. Then specific floods are added, after intense consideration of such criteria as channel shape, return periods of different magnitude floods, zonation of the riparian vegetation and the flow requirements of the biota, including, for example, serial-spawning fish. The total flow regime so described is deemed, to the best of available knowledge, to be the minimum flow that would sustain the system at the DS. All flows entered are motivated for in ecological or geomorphological terms, so that the decision-maker can understand the implications of non-adherence to the IFR.

Following this, capping figures are determined for some or all months (Table 4.13), which will place an upper limit on the baseflows that should not be exceeded on a continuous basis. Such figures are necessary, in order to alert managers to the damage that can be done by continuous, high-volume irrigation releases or inter-basin transfers of water. Finally, the situation of drought years is considered, when both the offstream users and the river will face stress. Such stress is an important natural feature of South African rivers, which by nature have highly variable flow regimes, and its inclusion in a managed flow regime is considered essential. The baseflow and flood figures for drought years are usually a muted version of those for the maintenance IFR, often with some floods completely omitted.

The volumes of water encompassed in the maintenance IFR (Table 4.13) are calculated and expressed as percentages of the virgin and present MAR. This links the IFR to the realities of the proposed development, providing quantities that the project planners can use for design and planning purposes. It also allows comparisons with the results from IFR workshops of other rivers so trends can be sought. Additionally, if possible, the percentiles of all discharges requested are determined on monthly flow duration curves of daily virgin discharge. This also allows comparison with the results from other IFR workshops.

4.9.4 Necessary personnel and aids

With a methodology based on expert knowledge, the most important personnel are clearly the specialists of various aspects of rivers. Accepting this, the key personnel who will bring the workshop to successful fruition are the BBM co-ordinator and the group leaders in the workshop, all of whom should have considerable experience in the methodology. The co-ordinator is responsible for the integration of all activities leading up to the workshop, including preparation and dissemination of the starter documents. The group leaders, who should be river ecologists, have the task of guiding their groups to consensus within strict time limits. Other important persons are the chairperson of the workshop, who should have participated in other similar ones, an assistant who ensures that all aspects of the workshop, including timing of meals and use of facilities, run smoothly, and a hydrologist who provides new analyses and computes results as appropriate during the workshop. The author of the proceedings of the workshop, who should be an experienced river scientist, plays a vital role in accurately recording the nuances of the workshop as well as the final figures.

4.10 EXAMPLE OF AN IFR WORKSHOP OUTPUT: FOR IFR SITE 3 OF THE MOOI RIVER

IFR site 3 (IFR3) was generally agreed to be the most important of the sites considered at the Mooi River IFR workshop, with many rare and valuable attributes, including an upstream waterfall that is a designated Natural Heritage Site and a high diversity of aquatic biotopes and biotic communities (NSI 1995c). Rare aquatic invertebrates occur in the system, including an endemic mayfly species that is the only known representative of the genus *Ephemera* in Africa. The greatest diversity of fish in the Mooi River occurs in the river stretch represented by IFR3.

4.10.1 General approach

- Baseflows for maintenance were considered first.
- These baseflows were anchored by first by making recommendations for the wettest and driest month, and then for other key months. Remaining months were dealt with by interpolation.
- High flows for maintenance were considered next.
- Capping flows were considered next.
- Baseflows and high flows for drought years were considered last.
- For any flow entered, the sequence of consideration leading to this was
    - general discussion of flows that would facilitate the DS
    - referral to the information on the site questionnaires
    - specific depths or velocities of flows described and motivated for, with reference to cross-section data, and consensus reached on the values required
    - these descriptions converted to discharge, using the hydraulic relationships
    - the discharge checked with the hydrological record, to ensure it is within the bounds of the virgin flow regime.

The motivations for all numbers entered in Table 4.13 follow.

The baseflows

August - the driest month: 1.0 $m^3 s^{-1}$

- Water just links to marginal vegetation, ensuring modest use of this habitat.
- The depths and widths allow survival for fish and aquatic invertebrates.
- Minimum wetted width is provided for those species confined to "large" rivers.
- If discharge is increased to 1.5 $m^3 s^{-1}$, dilution of *E. coli* levels is provided to (acceptable?) levels.

February - the wettest month: 5.0 $m^3 s^{-1}$

- Fish passage is provided.
- Fish feeding areas are provided.
- Marginal vegetation is inundated, providing ample habitat and refuge areas for young fish and their food species.
- Wetted habitat is increased, so that fish abundances can increase.
- Contaminants are diluted, thereby enhancing the quality of drinking water.

- Need strong baseflow for important upstream conservation feature (waterfall) with rare biota restricted to adjacent spray/humid areas.
- In terms of hydrological record, the figure asked for is at lower end of baseflows for February.

Interpolation of baseflows for other months

- January: same as February.
- March and April: discharge dropped only gradually from level for wettest month, because spawning still occurring; juvenile fish need nursery areas in marginal vegetation; insects are emerging by climbing stems of emergent vegetation; religious activities centred around Easter require clean, flowing areas for baptisms.
- May and June: Straight interpolation between April and July, with reference to hydrological record.
- July and September: same as August.
- October: discharge increased because of increasing temperatures and evaporation potential, but not yet up to wet-season levels; interpolation from September.
- November and December: discharges increased further because of active fish spawning and use of nursery areas; wetted habitat increased to allow standing stocks to increase.

In summary, the basic character of the river is catered for, with perennial flow that is higher in the wet season (November to April) than in the dry season (June to September). The percentage of virgin MAR encompassed in the required flows is 27%, of which 2.4% is for the increased dry-season flows for water quality.

The higher flows

The many high flows that would have occurred in each wet season performed many functions of ecological and geomorphological importance, such as scouring clean rocks, controlling channel encroachment by riparian or emergent vegetation, flushing fines from cobble beds, maintaining the channel form, controlling pest species and providing biological cues for such activities as migration of fish and growth of riparian seedlings. In a river regulated by a major dam, only the largest of floods may spill, with all smaller pulses of high flow being completely eradicated from the downstream system. Probably, only a fraction of the number of high flows in the virgin flow regime can be re-instated through dam releases, and each of these has to be placed to its best advantage. Such considerations guide the description of high flows in an IFR assessment.

February - the major flood:

- A requirement of 40 $m^3 s^{-1}$ annually, 100 $m^3 s^{-1}$ every third year and 145 $m^3 s^{-1}$ every fifth year, in an irregular sequence.
- Duration of eight days.
- Normal rising limb of hydrograph.
- Delayed falling limb of hydrograph to prolong intermediate flows, to allow young fish to move back from edges into main channel.
- The floods would limit sediment build-up.
- The different magnitude floods would inundate the riparian strip to different extents.
- Floods increase habitat diversity.
- The larger floods will stress the system, which is felt to be necessary.

November, December, January, March and April

- A small fresh each month of magnitude 10 $m^3 s^{-1}$ (range 8-20 $m^3 s^{-1}$ for variability).
- Duration of five days.
- Normal hydrograph shape for that time of year.
- The April fresh only required during years of the 100-145 $m^3 s^{-1}$ floods, to allow some fish recovery through late spawning.
- The freshes are necessary for fish movement and spawning; and for inundation of hydromorphic zone and fringe vegetation (fish nursery and breeding areas). They also help maintain water quality through dilution and flushing and introduce flow variability to control pests.

In summary, a minimum number of high-flow pulses is identified, each designed to achieve specific purposes. The percentage of virgin MAR encompassed in the high flows is 6.7%, with the total IFR, including baseflows, being 34% of virgin MAR.

Drought situations

Drought is considered an important phenomenon affecting South African riverine ecosystems, and it is felt that it should continue to play a role in regulated rivers. Such drought conditions should be managed in a way that mimics natural droughts, however.

Baseflows June - September

- 0.6 - 0.3 $m^3 s^{-1}$; to put the system under natural stress. It was recognised that flows should be higher than this for water quality purposes, but the recommended ecological strategy would be to put the system under stress and supply drinking water in some other way.

Baseflows January - February

- 1.5 - 1.9 $m^3 s^{-1}$; consensus was not reached, because this would still not provide adequate dilution for drinking water purposes. The same argument was made as in the previous point.

Baseflows for remaining months

- Interpolated.

It was recognised that if sufficient flow is not provided to allow fish to move around, most may not evade capture and the resource could quickly be fished out. It was also noted that no floods were identified as necessary in the drought situation, with the recognition that these kinds of drought flows should not be allowed to run for more than two years consecutively without consultation with river ecologists, and that flows should be increased again as soon as the drought breaks.

Use of the drought IFR (Table 4.13) is not yet formalised. Prof. D. Hughes, IWR, Rhodes University, Grahamstown, has recently written a hydrological model to link the IFR workshop output (Table 4.13) to current rainfall-runoff patterns (Hughes *et al.* 1997). This model appears to solve the problem of when to use the drought IFR, and also provides hydrological scenarios to which ecologists have been able to react (see Chapter 6).

One of the points highlighted above is whether or not to add to discharges that would be acceptable from the perspective of water quantity, a dilution amount to ameliorate poor water quality. Many participants in the

IFR workshops feel that this confuses the basic message of the IFR, and also attributes to environmental flows volumes of water that would really be used to cope with previous poor catchment management practices. Others feel that realistically, poor water quality will not be managed in any other way in the near future, and so should be asked for in the IFR. The issue has not been resolved yet.

4.11 POST-WORKSHOP ACTIVITIES

Currently, the BBM has been developed to be used for IFA in DWAF's Pre-feasibility Phase. As the proposed development progresses to Feasibility, further engineering actions take place that should be matched by environmental actions related to the targeted river's flow regime. Development of the BBM to structure these environmental actions has begun recently. This topic is dealt with in Chapter 6.

5. HYDROLOGICAL INFORMATION NEEDED FOR AN IFR WORKSHOP

Written by J. King

5.1 INTRODUCTION

5.2 THE GENERAL APPROACH TO THE PRESENTATION OF HYDROLOGICAL DATA

5.3 HYDROLOGICAL INFORMATION THAT SHOULD BE PRODUCED FOR THE IFR WORKSHOP STARTER DOCUMENT

5.3.1 Source information

5.3.2 Summary of the approach used to provide the hydrological statistics

5.3.3 Formats of required hydrological data

5.4 VISUAL AIDS FOR THE TRANSFER OF HYDROLOGICAL INFORMATION AT THE WORKSHOP

5.5 DISCUSSION

5.5.1 Daily versus monthly data

5.5.2 Virgin versus present-day data

5.6 CONCLUSIONS

5.1 INTRODUCTION

Since IFR workshops were initiated, a considerable amount of effort has been spent on identifying the kinds of hydrological data that are most successful at the workshops (King and Tharme 1993; Louw 1995b). Many river ecologists and geomorphologists do not regularly work with hydrological data and cannot relate to the kinds of hydrological analyses routinely done by engineering hydrologists. The following is a list of hydrological information that has been found by the river scientists to be most helpful. Some of the items might seem to be duplicate presentations of the same knowledge, but should still be provided as they may be used in different ways, or individual scientists may be more comfortable with one form of presentation than another. This then serves the purpose of the exercise, which is to facilitate rapid use of relevant data by the river scientists at the IFR workshop, through provision of clear, simple graphics and tables. The data are included as a chapter in a workshop starter document, which is read by all participants before the workshop, and so should be kept brief. Further data are brought to the workshop by the hydrologists as reference material, either as documents or as computer files, in order to deal with queries that might arise during discussions.

5.2 THE GENERAL APPROACH TO THE PRESENTATION OF HYDROLOGICAL DATA

At the IFR workshop, the scientists will be looking for information on the natural character of the river's flow regime, as well as its present character. In particular, they will need the best possible information on its virgin flow regime, which months are climatically wet and dry, which years were wet and dry years, if the river is or was perennial, and the general variability and predictability of flow. The pattern of natural flooding will be needed, in terms of the timing, duration, frequency and magnitude of different size floods. Participants will also need to understand the kinds of modifications of the flow regime that have occurred in the past, through information on, for instance, the presence, location, capacity and date of construction of dams on the system. Data for as many years as possible, covering wet and dry years, should be used in the analyses, and the data should be presented in as clear and concise a form as possible.

For many purposes, the scientists cannot use monthly average discharge data, and work best with daily hydrographs and statistics derived from these. Long-term data on natural, daily discharge are ideal for three main reasons.

- Long-term data provide comprehensive information on the timing, magnitude, frequency and duration of flow conditions that occur **often** in the river or that are characteristic of it in some other way - that is, conditions that the natural biota of the river must be adapted to cope with.
- Data on natural flow are considered essential, because the natural flow regime is one of the driving forces that has sculptured both the geomorphological character of the river channel and the character of its biological communities. Maintaining something resembling the natural flow pattern in a regulated river reduces the likelihood of costly ecological or geomorphological repercussions.
- Daily discharge data are considered essential, because these come closest to describing the instantaneous discharge that the riverine biota experiences and reacts to. To explain further, a figure for monthly average discharge could represent a similar discharge each day of the month, or 29 days of dry river bed and a flood, or any variation in between. In terms of total yield these may produce the same amount of water, but they represent widely different conditions for riverine species; an understanding of these conditions is essential when undertaking an IFA.

It is acknowledged that such hydrological data are often not available. This problem is discussed further in Sections 5.5 and 5.6.

5.3 HYDROLOGICAL INFORMATION THAT SHOULD BE PRODUCED FOR THE IFR WORKSHOP STARTER DOCUMENT

Where appropriate below, an example of the kind of data required is provided from a starter document for a recent IFR workshop; the examples may not contain all the information detailed in the text.

5.3.1 Source information

Before the source information can be provided, it is necessary to confirm from the IFR workshop co-ordinator:

- which part of the river(s) is to be included in the study area
- locations of the IFR sites

Following this, information should be prepared on:

- the presence and location of all gauging weirs within the study area, and any outside of the study area, which are to be used for simulating hydrological data for the study area
- length of historical record for each weir
- which part of each record represents "near-natural" flow
- which years of record are for dry years and which for wet years
- accuracy assessment for each gauging weir, particularly in terms of very low flows and flood peaks
- which weirs were actually used for preparation of the workshop hydrological document, and why

Some of the above information is probably best presented in a brief written summary with tables. However, two maps should be provided. The first should be a topographical map, showing the complete study area and as much as seems relevant of its catchment, together with some information on contours, place names and a scale line in km. This allows those unfamiliar with the area to orientate themselves, understand the length and gradient of the river, and locate key towns, tributaries and other landmarks. The gauging weirs mentioned above, and the IFR sites, should be shown on this map. The other map should be a diagrammatic one (Figure 5.1) that shows:

- the rivers and their names;
- the location and identity of all relevant gauging weirs;
- the IFR sites;
- the virgin and present MAR at each weir, at each IFR site and for both rivers immediately upstream of a river confluence;
- the virgin and present median flow at each weir, at each IFR site and for both rivers immediately upstream of a river confluence;
- the present MAR and median flow as percentages of virgin at each weir, at each IFR site and for both rivers immediately upstream of a river confluence;
- location of all existing and proposed dams;
- dates of construction of existing dams, and their capacities in volumes and as a percent of the MAR.

5.3.2 Summary of the approach used to provide the hydrological statistics

There should be a summary, in simple language, of the approach used to produce the hydrological data. This should include details of:

- the hydrological model used to prepare the data;

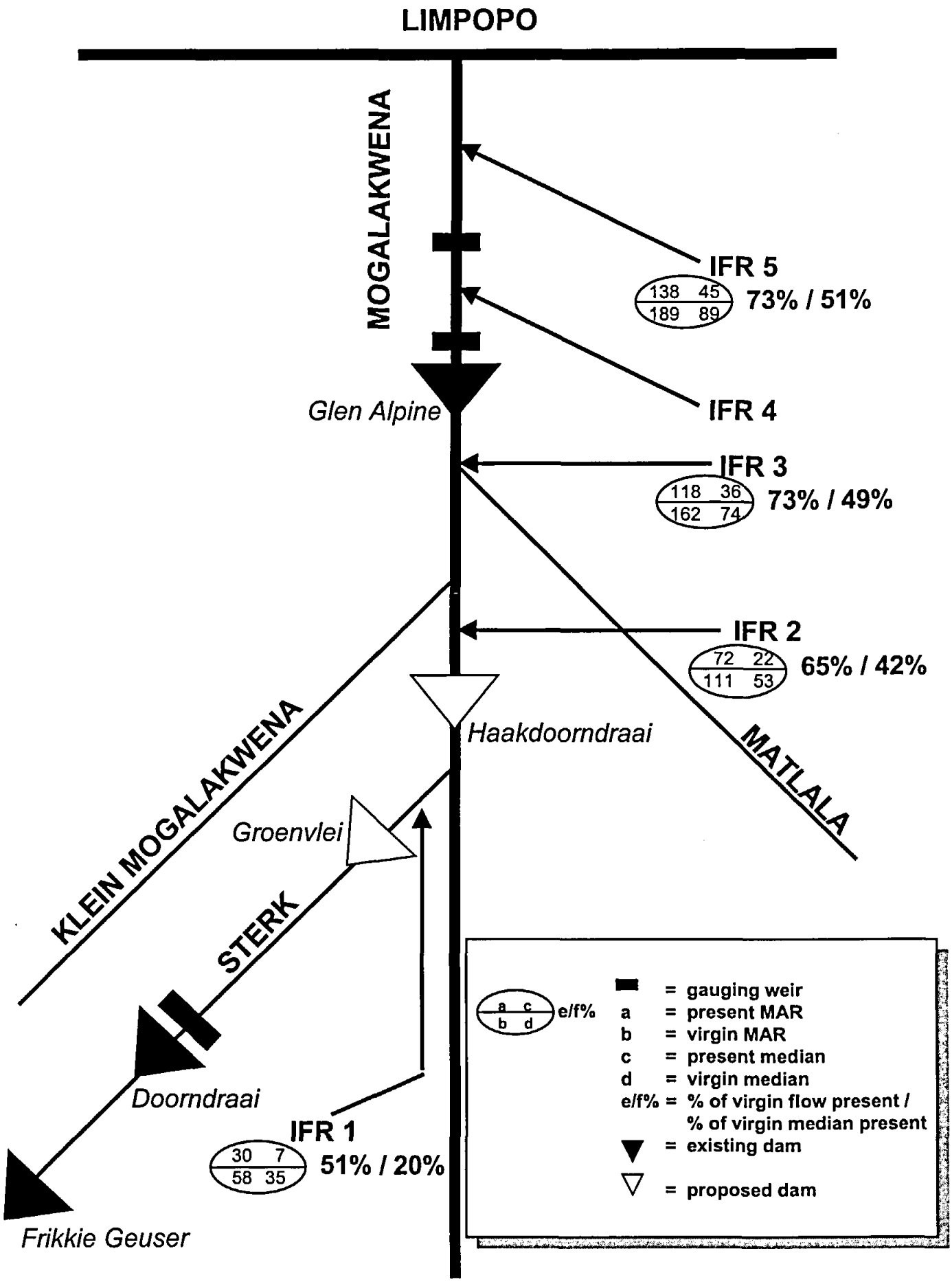


Figure 5.1 Diagrammatic map of the study area for an IFR workshop, showing the location of IFR sites, relevant gauging weirs and extant and proposed dams. Various statistics of the flow regime are also shown for each IFR site (modified from Ractliffe 1996a).

- why that model was used;
- what the model does;
- how the results were obtained;
- the level of confidence in the accuracy of each set of data produced;
- clear statements on which data are real and which simulated.

5.3.3 Formats of required hydrological data

Ideally, the following information is needed for each IFR site.

Annual flow variability

A bar graph of virgin (or naturalised) total annual runoff over all years of record should be provided, to show annual variability (Figure 5.2). Included should be any relevant comments about wet and dry years or cycles. Also needed is a plot of cumulative deviation of total annual runoff about the virgin MAR for all years of record (Figure 5.3). These kinds of data are used to establish an understanding of how variable river flow is from year to year, and whether dry and wet cycles have been included in the data presented.

Visual assessment of inter-annual patterns of runoff

Small bar charts, many to a page, of total monthly virgin or naturalised flow volumes, allow the inter-annual variability of the runoff for any calendar month to be assessed at a glance (Figure 5.4). At least one dry and one wet cycle should be included in the figure, and the extent of these should be shown on the figure.

Monthly volumes of flow over total record

For virgin (or naturalised) conditions, a chronological table of total monthly volumes over all years of record (months in columns) should be presented, together with a summary of the percentage of the total volume represented by each month (Table 5.1). This is used in the workshop to reach consensus on the dry, wet and transition seasons, as well as to assess variability in any one month from year to year.

Mean monthly flow volumes - virgin conditions

A full-page bar graph of the above monthly distributions of mean flow volumes (Figure 5.5). This allows a quick understanding of the seasonal distribution of flow.

Mean monthly flow volumes- present conditions

The same graph (Figure 5.5) could be used to depict present conditions, if different from virgin, with an indication of when the change from a natural flow regime occurred. This latter could also be presented as a plot of runoff versus rainfall, to show non-stationarity (King and Tharme 1994).

Differences between virgin and present flow regimes

Clear graphics and an accompanying short description should be provided of how flow has changed from virgin to present. These should provide a quick understanding of the following:

- how dry season baseflows have changed;
- how wet season baseflows have changed;
- if a seasonal reversal of wet season and dry season baseflows has occurred;

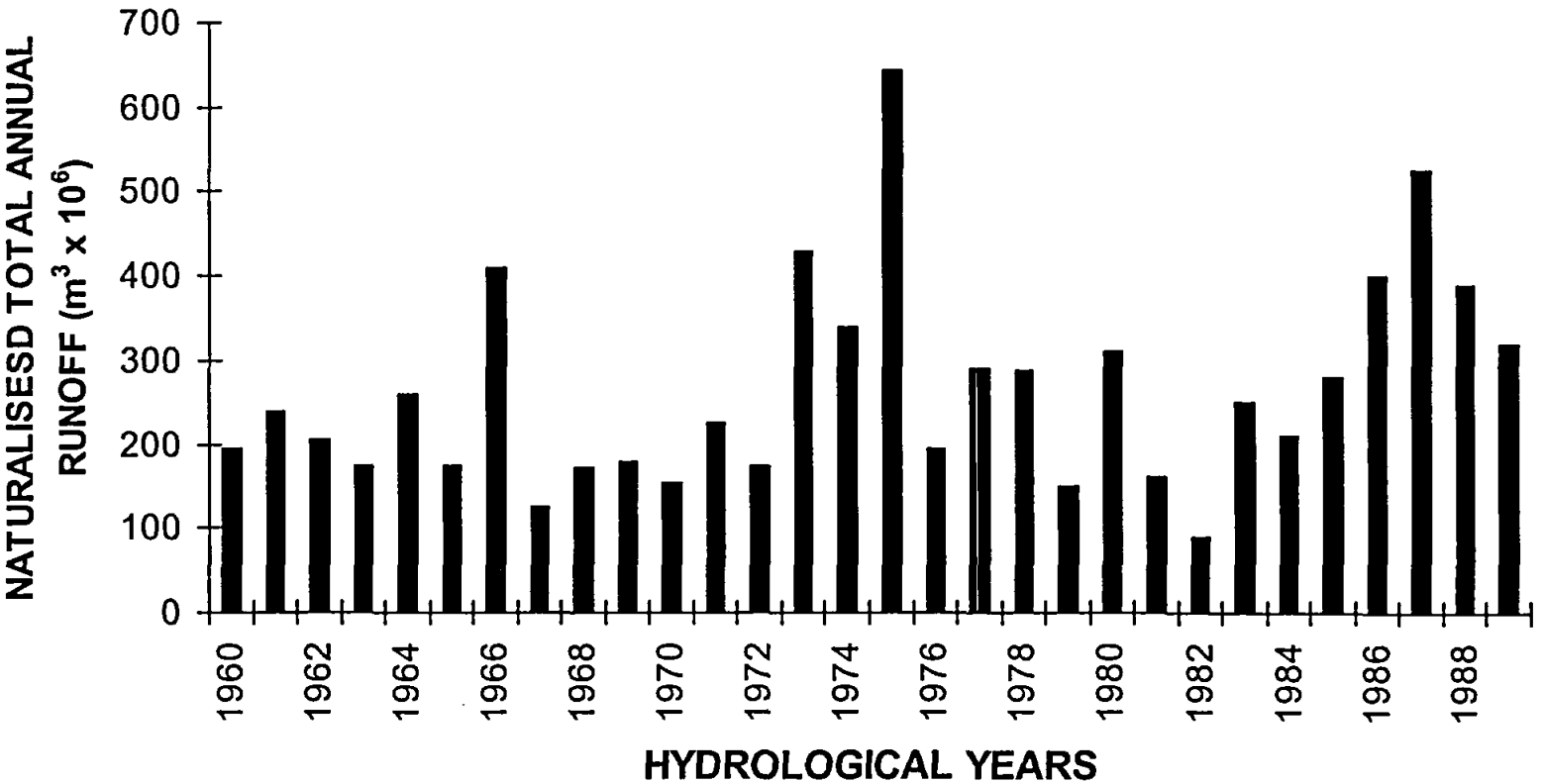


Figure 5.2 Naturalised total annual runoff at gauging weir V2H002 for all the years of simulation, to show annual variability (modified from the starter document for the Mooi River IFR workshop).

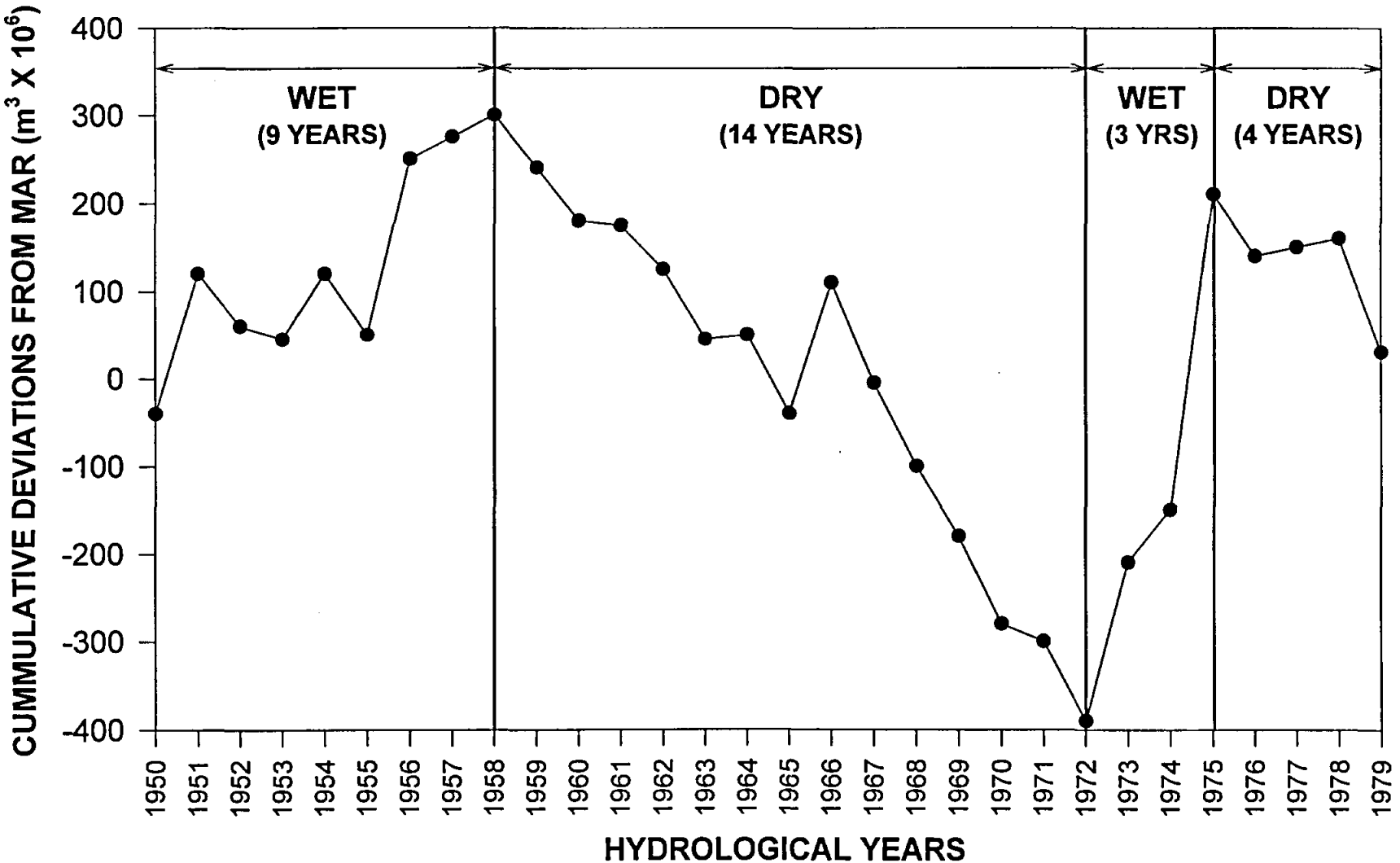


Figure 5.3 **Cumulative deviation of total annual runoff about the virgin Mean Annual Runoff (MAR), for all years of record, to show wet and dry periods as defined by the hydrologist (modified from the starter document for the Mooi River IFR workshop).**

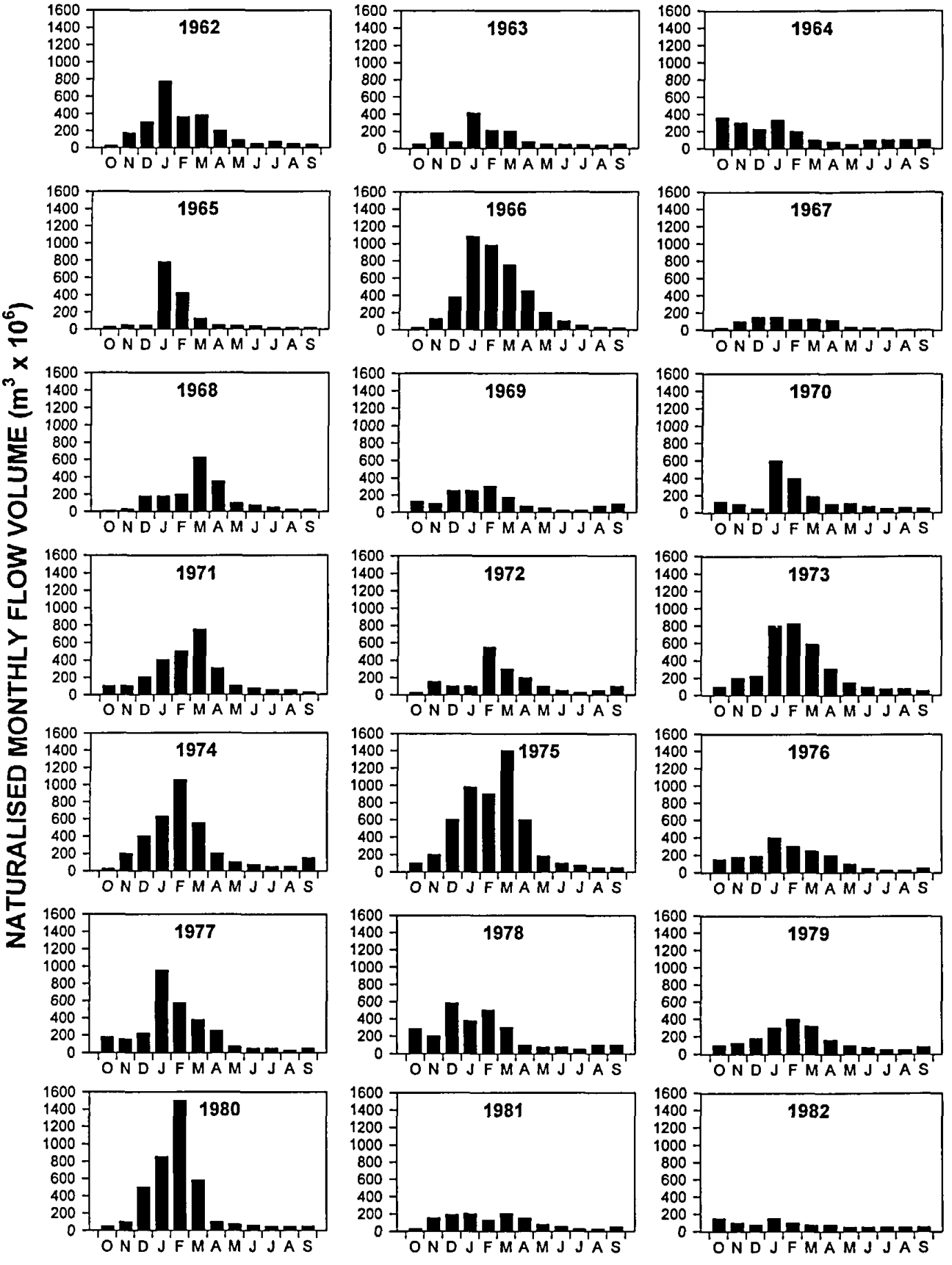


Figure 5.4 **Naturalised monthly flow volumes for all years of record (modified from DWAF 1995b).**

Table 5.1 **Patched record of flow volume at gauging weir V2H005 (modified from the starter document for the Mooi River IFR workshop). Mean annual runoff = 108.25 m^3 x 10^6. Standard deviation = 48.12 m^3 x 10^6. Catchment area = 260 km^2.**

YEAR	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	TOTAL	ACCUMULATED
1972/73	1.75	4.99	6.12	4.28	16.17	9.43	18.37	4.52	2.20	1.58	2.42	1.49	73.32	73.32
1973/74	7.65	10.47	9.40	35.32	41.68	37.05	19.08	6.11	3.99	2.91	1.95	1.33	176.94	250.26
1974/75	1.29	4.24	9.85	29.35	38.52	19.84	9.86	3.41	2.05	1.61	1.17	2.93	124.12	374.38
1975/76	2.52	8.23	30.46	46.30	41.41	62.45	16.02	6.29	3.30	2.33	2.09	2.18	223.58	597.96
1976/77	9.45	7.24	3.08	7.27	14.59	16.04	10.68	3.44	1.99	1.50	1.12	1.16	77.56	675.52
1977/78	1.91	4.40	4.85	21.65	19.08	15.32	13.65	5.51	2.77	1.91	1.61	1.91	94.57	770.09
1978/79	4.66	7.88	32.67	15.20	12.90	10.84	4.11	4.79	2.24	2.18	3.36	2.52	103.35	873.44
1979/80	2.17	2.85	4.57	11.21	11.83	15.37	6.16	2.34	1.49	1.13	0.86	2.98	62.96	936.40
1980/81	3.84	3.12	17.50	15.20	45.20	19.14	4.97	2.84	2.08	1.46	2.32	4.62	122.29	1058.69
1981/82	2.32	2.75	5.49	8.30	3.37	23.03	9.88	3.56	2.13	1.55	0.97	1.06	64.41	1123.10
1982/83	1.11	3.55	4.27	6.42	3.03	3.00	2.85	1.31	0.91	0.90	0.80	0.66	28.81	1151.91
1983/84	3.35	9.24	20.40	16.92	7.28	16.37	11.73	3.30	2.21	1.84	1.36	0.37	94.37	1246.28
1984/85	2.04	1.82	2.59	12.22	41.01	16.73	5.82	3.77	2.63	1.15	0.74	0.58	91.10	1337.39
1985/86	2.79	12.70	22.51	23.26	19.30	9.69	6.61	3.46	2.44	1.52	1.84	1.99	108.11	1445.50
1986/87	4.86	12.98	17.52	20.06	9.06	22.69	7.50	2.91	1.93	1.47	2.87	35.93	139.78	1585.28
1987/88	26.50	16.12	15.98	13.53	43.44	46.21	15.07	5.05	4.48	6.45	3.44	2.66	198.93	1784.21
1988/89	3.08	9.38	32.40	33.17	34.91	25.49	7.07	4.45	2.76	2.43	1.44	0.94	157.52	1941.73
1989/90	1.11	9.12	18.53	14.01	9.14	28.61	13.23	4.49	2.44	1.83	1.71	1.97	106.19	2047.92
1990/91	2.35	2.16	8.34	26.39	41.13	12.26	4.93	2.44	1.75	1.25	0.92	1.00	104.92	2152.84
1991/92	9.43	9.13	12.92	8.03	6.92	7.86	2.48	1.23	0.90	0.75	0.57	0.56	60.78	2213.62
1992/93	0.52	1.66	3.29	6.03	17.68	11.41	6.39	2.23	1.37	1.06	0.78	0.56	52.98	2266.60
1993/94	7.71	6.26	9.37	45.78	23.73	9.65	4.34	2.53	1.64	1.49	1.43	1.00	114.93	2381.53
% TOTAL	**4.30**	**6.31**	**12.27**	**17.63**	**21.05**	**18.41**	**8.43**	**3.36**	**2.09**	**1.69**	**1.50**	**2.96**	**100.00**	

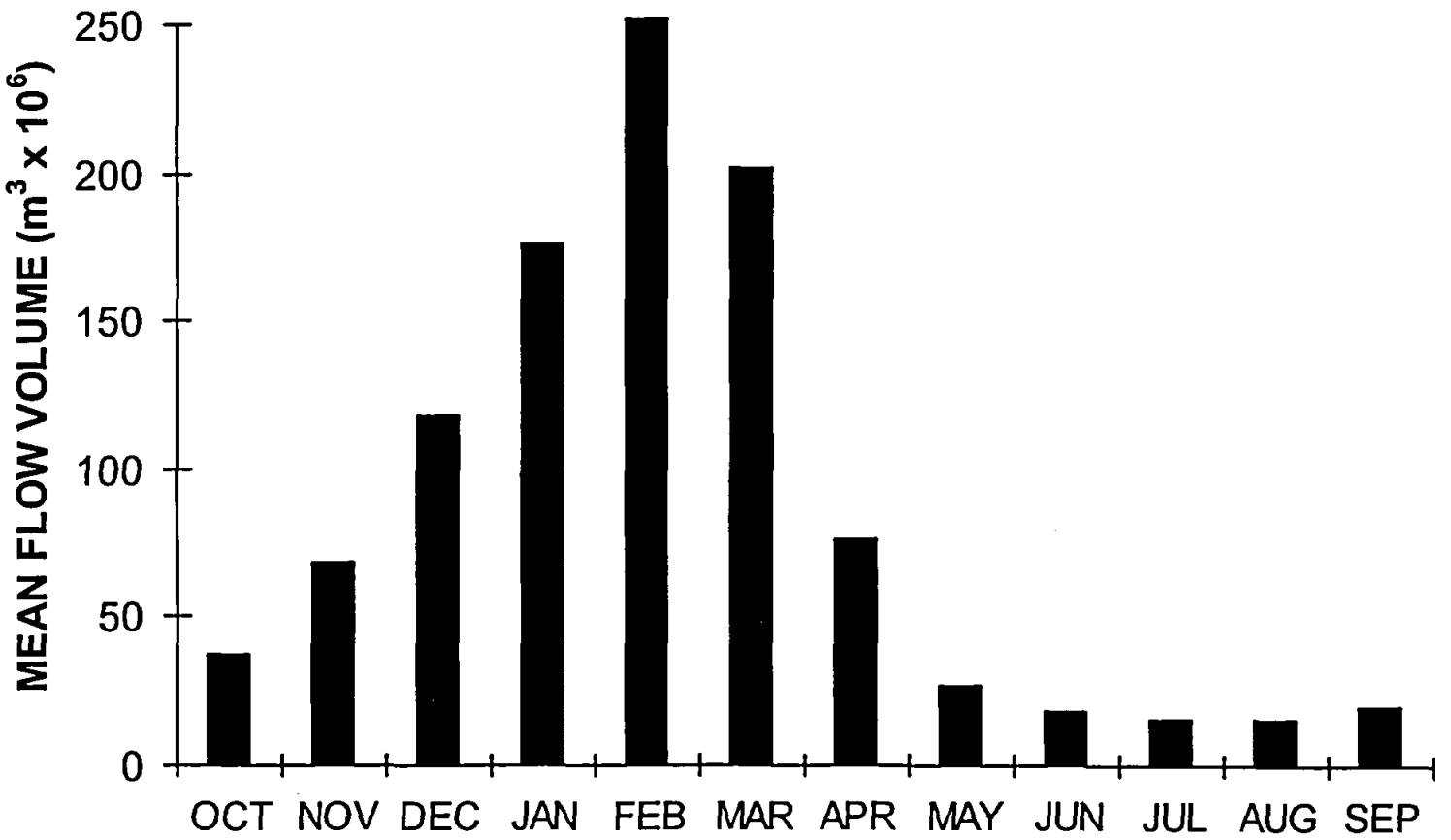


Figure 5.5 **Monthly distributions of mean virgin flow volumes (modified from DWAF 1995b).**

- if the timing of flooding has changed, and particularly if the first floods of the wet season have been delayed;
- if the magnitudes and frequencies of floods have changed and, if so, in what way;
- if the small pulses of higher flows in the dry season (freshes) still occur.

Possible ways of summarising the above include the following:

- graph of seasonal distributions of virgin and present flow volumes (Figure 5.6);
- bar graph of seasonal distributions of virgin and present discharge (Figure 5.7);
- selected examples of daily hydrographs based on real or simulated data, which show, for instance:
    - the delay of wet-season floods caused by a dam. Figure 5.8 illustrates how the onset of floods would be delayed from April to June 1982;
    - higher low-flows in the dry season than in the wet season due to irrigation releases. Figure 5.9 illustrates that discharges caused by high irrigation releases in the dry season (January to April 1987) would be followed by lower low-flows leading up to and in the wet season, because irrigation releases would have stopped (May to August 1987);
    - the retention of freshes in the dry season, although of reduced magnitude. Figure 5.10 illustrates that small pulses of higher discharge would still occur in the dry season (December 1984 to January 1985).
- box and whisker plots, with flows for each month shown in $m^3 s^{-1}$ (Figure 5.11). These plots provide daily average discharge values for the mean, mode and median, as well as the 5, 25, 50, 75 and 95 flow percentiles. Virgin conditions could be shown as a full-page plot and present conditions as a slightly offset transparent overlay, so that past and present distributions of daily discharge values can be compared closely for each month;
- a table of summary statistics (Table 5.2), provides much the same information as the box and whisker plots, but allows accurate flow values to be identified.

Indices of flow variability and predictability

Indices of flow variability and flow predictability, for virgin, present and simulated future conditions, are beginning to be used by river ecologists. There is little understanding yet as to which would be most useful. As a suggested starting point, the intra-annual and inter-annual coefficients of variation, and Colwell's predictability index (King and Tharme 1994, page 294), could be derived. Gordon *et al.* (1992) give further examples of indices that could be derived.

Frequency of occurrence of virgin daily discharges

Tables of frequency of occurrence of different size virgin daily discharges ($m^3 s^{-1}$), allow quick visual assessment of how common specific discharges were in the virgin flow regime (Table 5.3). The frequencies should be listed by month (columns), with the years used for the analysis stated. If there are clear wet and dry cycles encompassed in the record being analysed, they should be analysed separately.

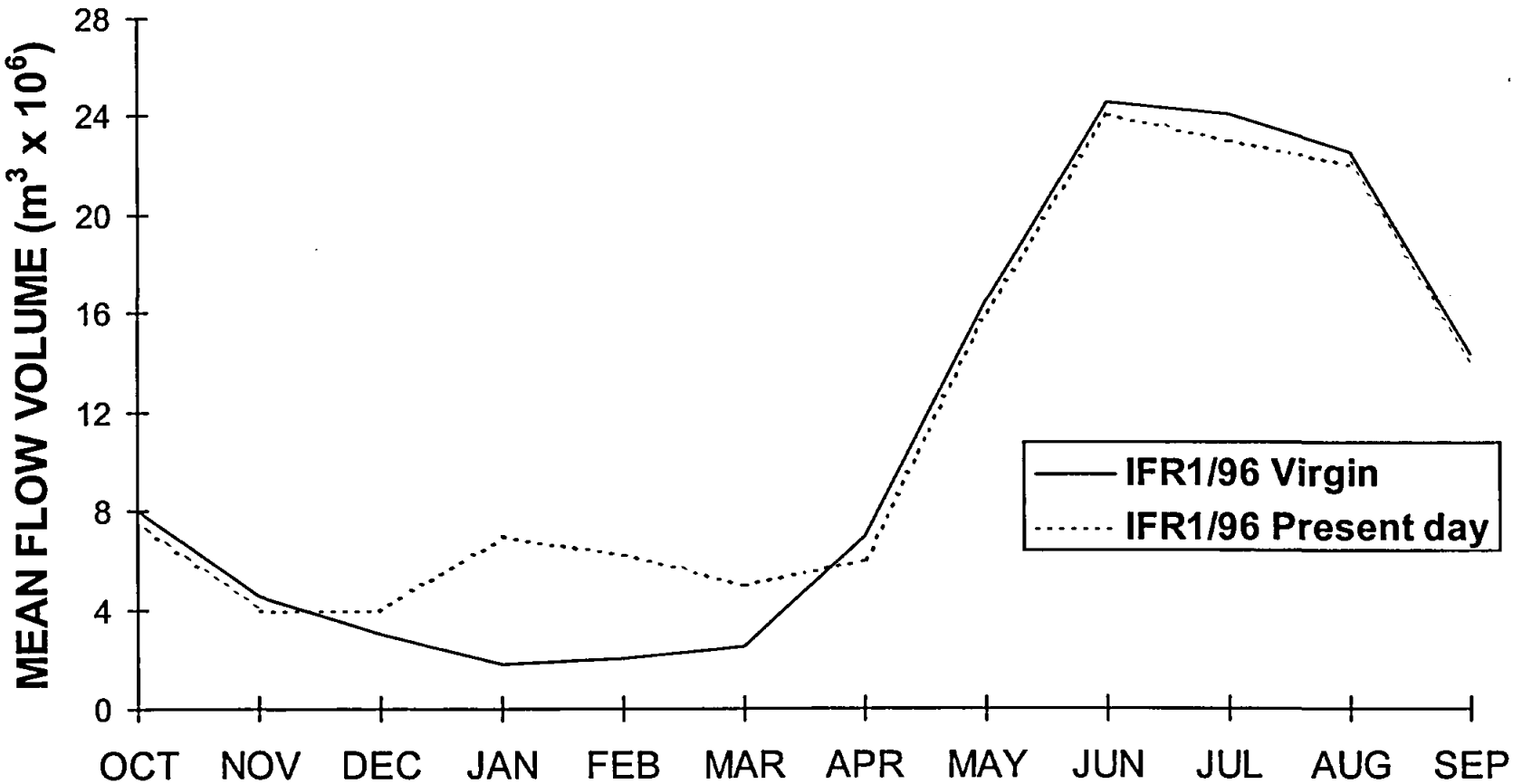


Figure 5.6 **Seasonal distribution of virgin and present mean monthly flow volumes for site IFR 1/96 on the Berg River (modified from the starter document for the Berg River feasibility study).**

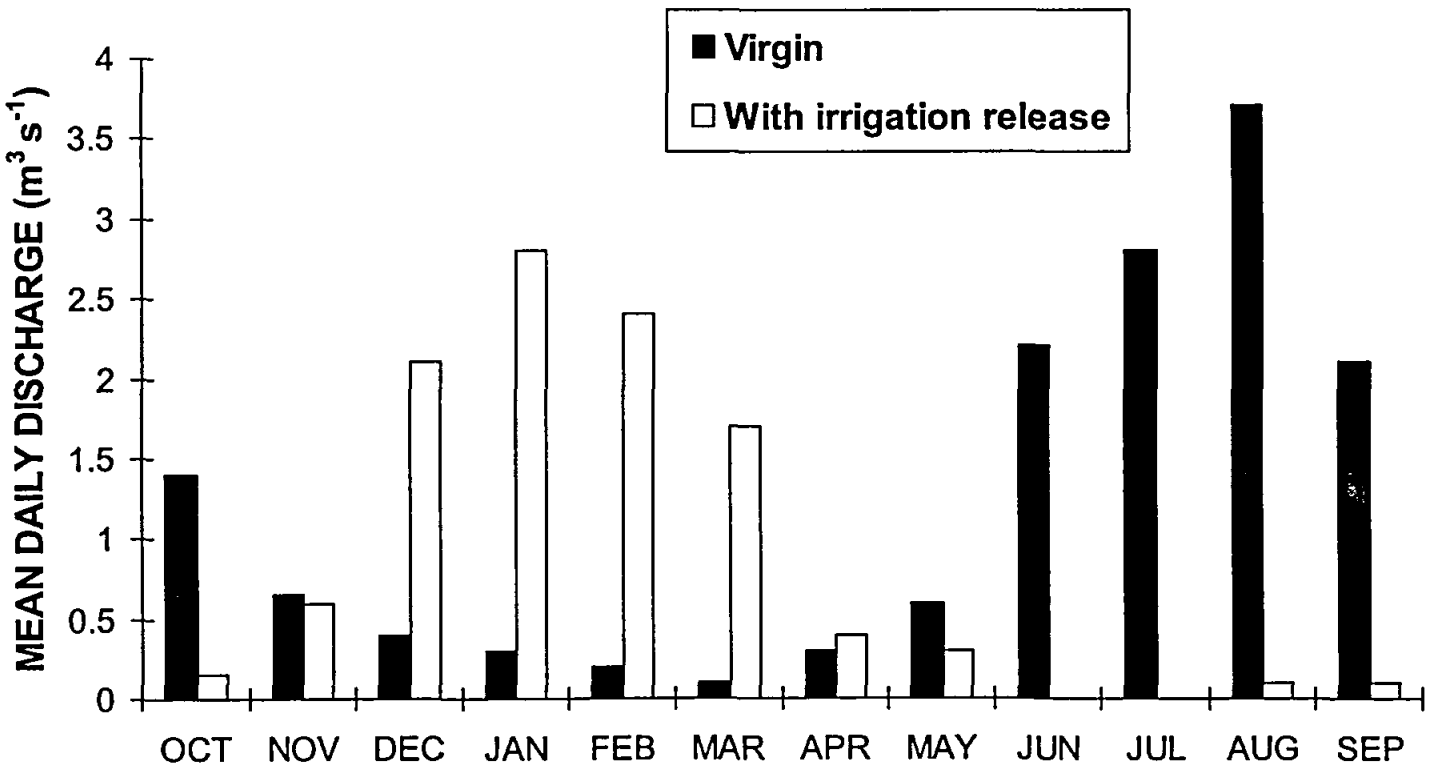


Figure 5.7 **An hypothetical simulation of the seasonal reversal in mean daily discharge caused by the presence of a proposed dam.**

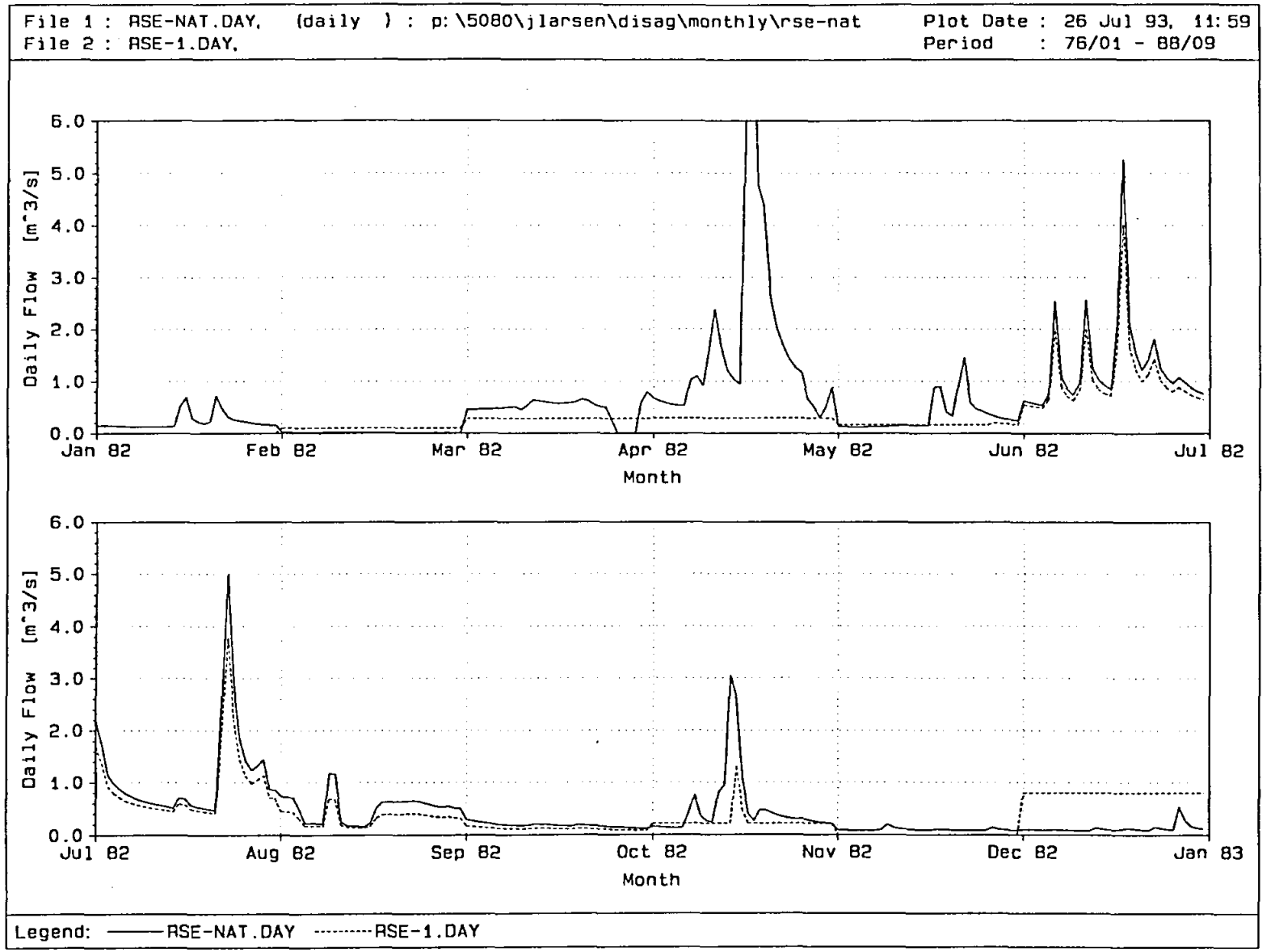


Figure 5.8 **Simulated virgin (RSE-NAT.DAY) and future (RSE-1.DAY) daily hydrographs for the Olifants River (Western Cape) upstream of Clanwilliam Dam, to illustrate the delayed onset of winter floods caused by the proposed Rosendaal Dam. Model printout supplied by Ninham Shand Consulting Engineers (from the hydrology starter document for the Olifants River IFR workshop).**

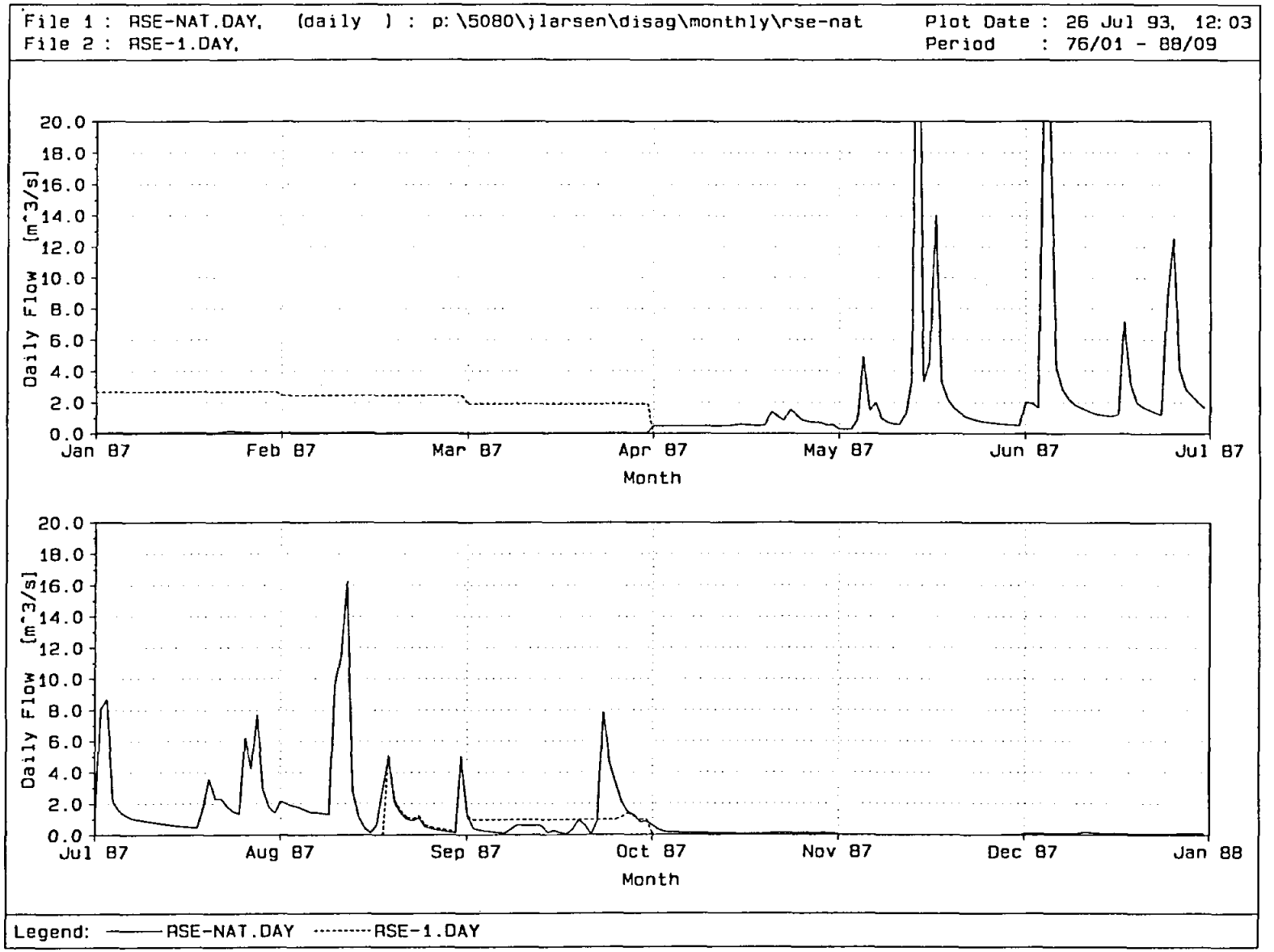


Figure 5.9 **Simulated virgin (RSE-NAT.DAY) and future (RSE-1.DAY) daily hydrographs for the Olifants River (Western Cape) upstream of Clanwilliam Dam, to illustrate the higher baseflow in the dry season (January to April) than in the wet season (May to August), caused by the proposed Rosendaal Dam. Model printout supplied by Ninham Shand Consulting Engineers (from the hydrology starter document for the Olifants River IFR workshop).**

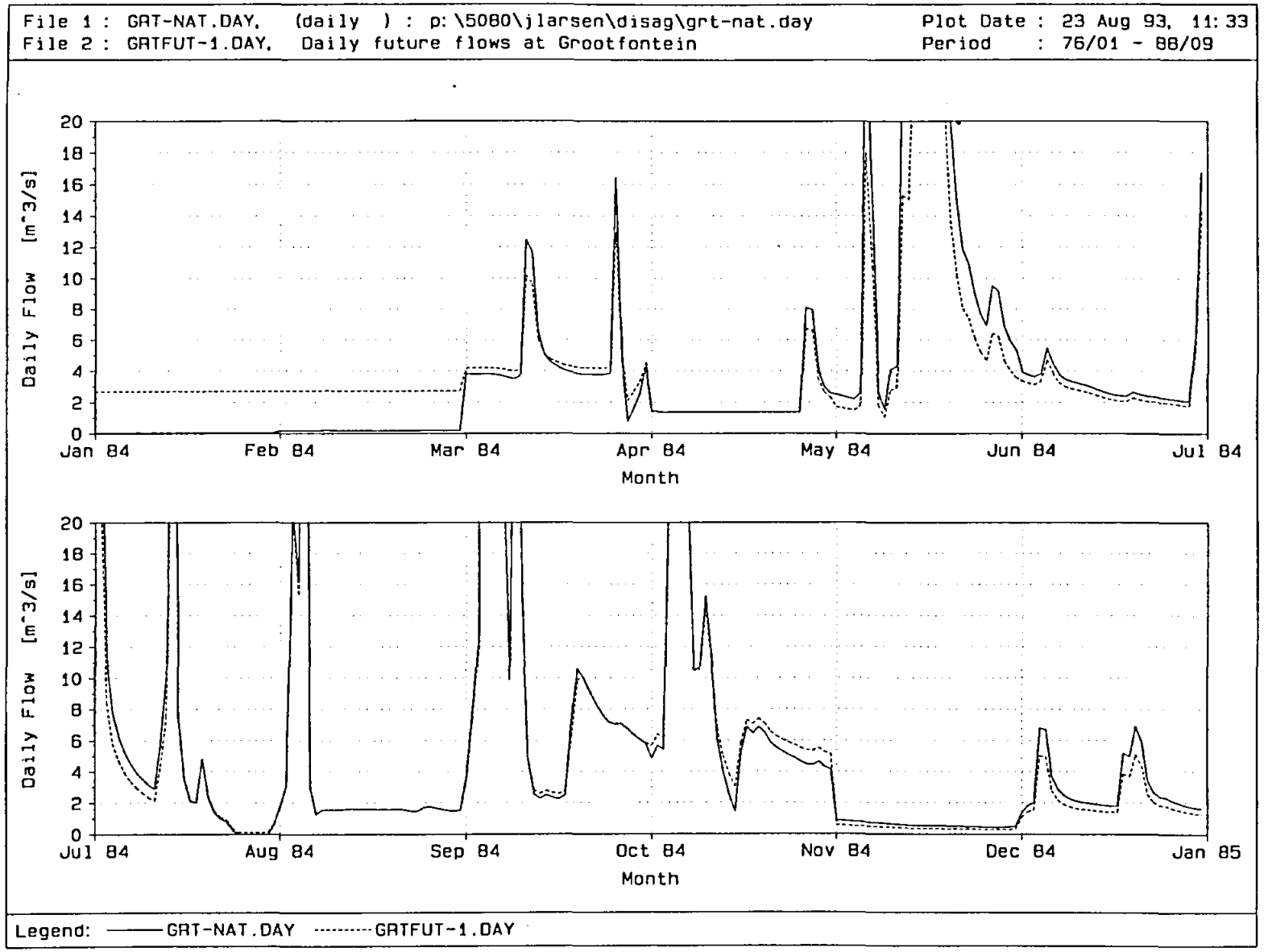


Figure 5.10 **Simulated virgin (GRT-NAT.DAY) and future (GRTFUT-1.DAY) daily discharge hydrographs for the Olifants River (Western Cape) at Grootfontein, to illustrate that, with the proposed Rosendaal Dam in place, freshes would still occur in the dry season (November to January). Model printout supplied by Ninham Shand Consulting Engineers (from the hydrology starter document for the Olifants River IFR workshop).**

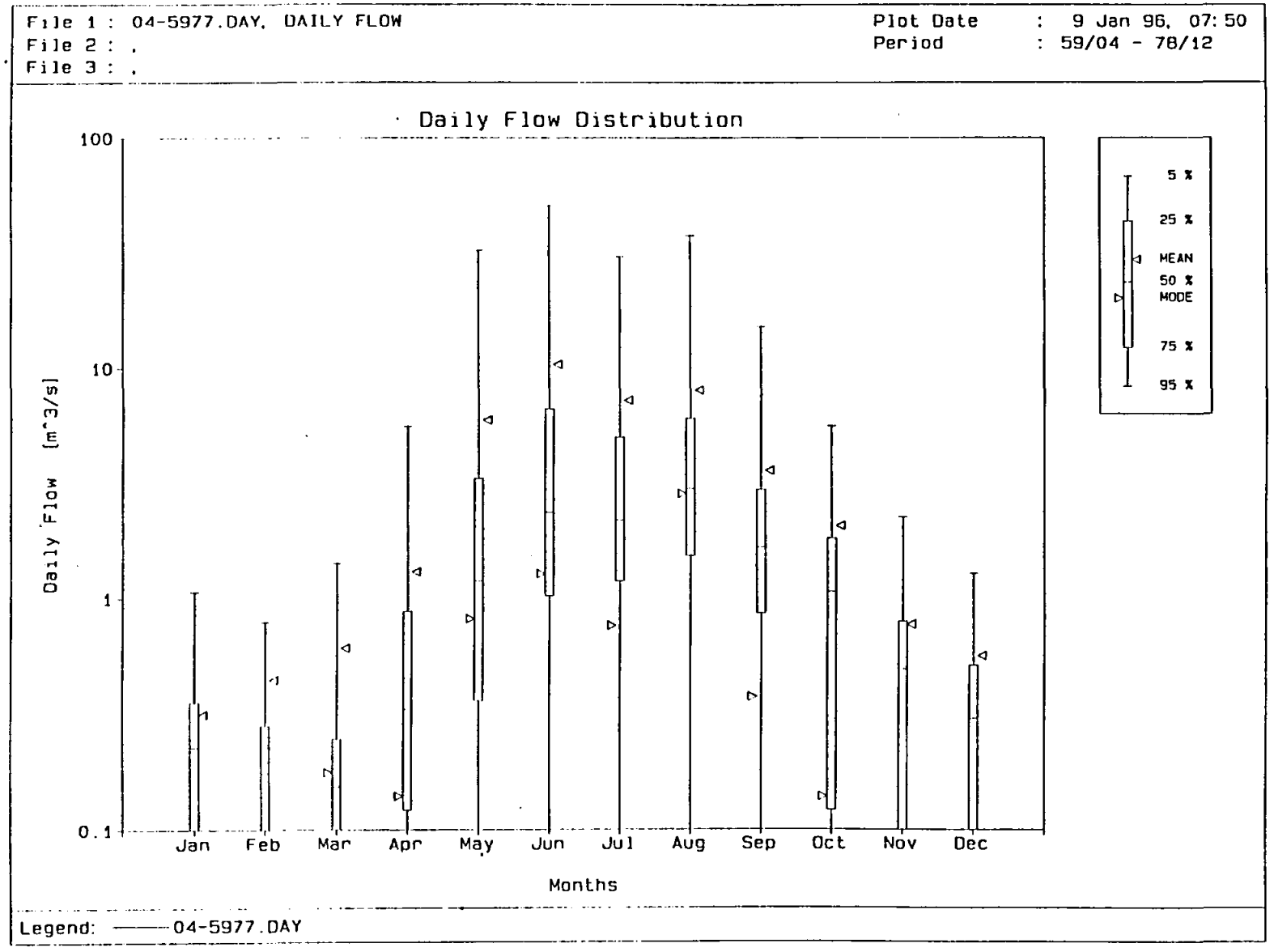


Figure 5.11 **Box and whisker plots illustrating the distribution of virgin daily discharges at IFR site 1/96 on the Berg River. Model printout supplied by Ninham Shand Consulting Engineers (from the starter document for the Berg River feasibility workshop).**

Table 5.2 **Summary statistics of daily discharge ($m^3 s^{-1}$) for present conditions for the months July to December, for all the years of record, with monthly total volumes ($m^3 \times 10^6$) (modified from the starter document for the Olifant-Sand River IFR workshop; NSI 1993c).**

	YEAR	1960	1961	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1960-87
JUL	*min*	.	*7.32*	*4.00*	*4.53*	*2.58*	*3.20*	*2.13*	*6.13*	*5.15*	*5.81*	*1.80*	*6.04*	*12.54*	*5.91*	*11.30*	*0.00*	*10.46*	*5.63*	*6.42*	*3.78*	*7.50*	*7.68*	*3.68*	*3.30*	*3.25*	*0.00*	*4.07*	*4.31*	*0.00*
	max	.	21.47	5.63	13.37	4.78	4.67	4.30	15.11	7.10	14.60	5.81	11.27	16.58	8.55	12.81	10.17	16.45	11.96	9.37	5.47	9.88	11.45	6.19	6.38	19.61	14.00	8.49	8.69	21.47
	mean	.	14.77	4.52	8.33	4.27	3.88	3.20	10.85	6.31	8.21	3.89	8.32	13.68	7.16	11.56	8.57	13.38	7.28	8.13	4.61	8.60	9.51	4.54	4.55	8.86	9.67	6.11	5.85	7.73
	mode	.	7.67	4.12	9.17	4.72	3.46	2.19	12.64	6.56	6.02	3.91	7.21	12.64	7.16	11.42	9.41	12.41	6.74	8.26	4.50	8.75	7.77	4.62	3.38	3.66	10.15	7.50	5.95	4.83
	median	.	18.02	4.44	8.66	4.56	3.78	3.16	11.31	6.45	7.67	3.91	7.63	13.54	7.23	11.40	8.75	13.00	6.76	8.43	4.63	8.56	9.43	4.44	4.41	6.05	10.54	6.23	5.78	6.98
	7 day	.	7.91	4.13	5.17	3.16	3.38	2.61	7.47	5.86	5.81	2.64	6.88	13.09	6.23	11.39	7.74	11.33	6.27	6.72	4.11	7.91	7.87	3.98	3.89	3.72	4.70	4.59	5.05	5.81
	total	.	39.60	12.10	22.30	11.40	10.40	8.60	29.10	16.90	22.00	10.40	22.30	36.60	19.20	31.00	22.90	35.80	19.50	21.80	12.40	23.00	25.50	12.10	12.20	23.70	25.90	16.40	15.70	.
AUG	min	.	8.68	1.49	3.60	0.98	1.17	0.80	0.00	0.00	3.93	0.33	2.29	7.06	2.00	5.31	3.20	4.86	3.56	3.41	2.82	3.42	5.01	1.69	1.13	2.76	3.87	1.93	0.00	0.00
	max	.	12.82	3.67	9.30	3.64	2.83	5.46	11.10	5.33	6.89	13.21	8.32	11.51	6.26	9.38	10.62	10.69	7.50	5.68	9.53	7.16	7.25	3.75	12.46	13.25	10.24	7.26	32.59	32.59
	mean	.	9.94	2.67	5.59	2.27	2.06	1.84	7.17	3.81	4.86	2.43	4.87	8.20	4.23	7.18	5.18	7.65	4.47	4.62	4.18	5.04	6.08	2.59	3.30	5.86	5.61	3.73	4.36	4.81
	mode	.	8.99	3.40	4.31	2.51	2.04	1.15	6.38	3.60	4.89	1.29	2.74	7.17	5.95	5.41	4.13	6.17	3.86	5.17	3.66	4.63	5.40	3.18	1.41	6.17	4.03	2.06	0.81	4.07
	median	.	9.75	2.94	4.89	2.30	2.04	1.66	6.59	3.74	4.86	1.61	4.99	7.79	4.03	7.20	4.54	7.37	4.04	4.70	3.78	4.68	5.95	2.63	2.34	5.47	4.99	3.72	0.51	4.32
	7 day	.	8.45	1.65	3.93	1.29	1.61	1.38	4.90	2.68	3.93	0.85	2.86	7.06	2.25	5.65	3.57	5.39	3.70	3.68	3.39	3.71	5.29	1.87	1.51	3.22	4.12	2.73	0.13	3.39
	total	.	26.60	7.10	15.00	6.10	5.50	4.90	19.20	10.20	13.00	6.50	13.10	22.00	11.30	19.20	13.90	20.50	12.00	12.40	11.20	13.50	16.30	6.90	8.80	15.70	15.00	10.00	11.70	.
SEP	min	.	4.17	0.01	0.97	0.00	0.04	0.00	1.98	0.00	0.00	0.00	0.01	3.41	1.26	3.41	1.02	2.08	2.09	1.87	0.05	2.55	1.88	0.18	0.00	0.04	0.44	0.06	1.65	0.00
	max	.	13.39	2.03	5.14	1.45	12.76	6.77	11.21	3.59	7.84	8.52	21.59	5.63	310.88	6.08	5.90	5.59	18.13	3.21	11.96	11.26	8.57	2.27	3.88	40.37	16.07	5.54	43.82	310.88
	mean	.	7.09	0.95	2.89	0.64	1.36	1.00	4.15	1.69	3.92	1.18	3.36	4.52	13.30	5.35	2.93	3.85	5.28	2.49	2.21	5.20	4.28	1.14	1.52	4.71	3.24	2.04	7.02	3.60
	mode	.	7.63	0.06	2.95	0.04	0.36	0.17	4.06	2.24	3.72	0.21	0.55	3.47	5.13	6.01	1.39	2.69	3.29	2.78	0.35	4.51	3.05	1.18	0.10	1.05	0.83	0.20	4.82	3.89
	median	.	6.90	1.04	2.95	0.67	0.06	0.00	3.83	0.00	1.14	4.49	2.60	6.08	2.49	3.81	4.05	3.81	4.05	2.55	1.11	4.59	3.74	1.18	1.39	1.77	1.62	1.94	4.94	2.55
	7 day	.	4.73	0.08	1.59	0.03	0.04	0.00	2.17	0.84	1.45	0.00	0.01	3.41	1.80	4.18	1.27	2.38	2.51	2.17	0.22	3.17	2.89	0.59	0.33	0.19	0.56	0.94	2.95	1.27
	total	.	18.40	2.50	7.50	1.70	3.50	2.60	10.80	4.40	10.20	3.10	8.70	11.70	34.50	13.90	7.60	10.00	13.70	6.40	5.70	13.50	11.10	3.00	3.90	12.20	8.40	5.30	18.20	.
OCT	min	0.82	0.63	0.00	0.00	0.00	0.00	0.00	1.87	0.71	0.15	0.00	0.00	1.16	3.36	0.00	0.24	1.03	0.89	0.00	0.13	0.00	1.83	0.00	0.00	0.00	0.01	0.00	.	0.00
	max	5.20	55.79	14.18	20.58	157.47	4.05	21.19	34.56	3.92	29.82	6.21	37.99	21.74	65.24	4.97	2.41	5.04	8.64	11.07	26.78	28.20	11.67	7.39	13.89	40.24	18.33	16.70	.	157.47
	mean	2.39	5.64	0.78	2.70	16.99	0.72	1.42	4.67	1.13	8.14	0.69	4.47	5.74	10.89	2.84	1.24	2.18	3.32	3.98	4.46	3.81	4.33	0.87	2.11	9.38	3.68	2.34	.	4.11
	mode	2.02	2.01	0.35	0.51	1.97	0.10	0.53	2.68	0.79	0.89	0.16	0.95	6.82	4.90	2.86	0.84	1.33	1.47	0.28	0.80	0.70	3.06	0.18	0.35	5.03	2.30	0.42	.	1.97
	median	2.08	4.29	0.00	0.00	0.00	0.18	0.00	2.82	0.71	2.52	0.00	1.00	5.08	9.08	2.81	1.21	1.84	3.05	2.55	0.20	0.00	3.69	0.00	0.00	6.79	2.10	0.00	.	1.71
	7 day	1.49	0.90	0.00	0.29	0.01	0.03	0.39	2.16	0.71	0.33	0.48	0.01	2.44	3.99	2.10	0.53	1.27	1.27	0.09	0.14	0.04	2.15	0.03	0.02	3.15	0.89	0.07	.	0.48
	total	6.40	15.10	2.10	7.20	45.50	1.90	3.80	12.50	3.00	21.80	1.90	12.00	15.40	23.20	7.60	3.30	5.80	8.90	10.70	11.90	10.20	11.60	2.30	5.70	25.10	9.90	6.30	.	.
NOV	min	0.61	2.02	0.10	0.00	7.95	0.00	0.50	1.04	0.00	0.00	0.90	0.00	0.00	7.88	2.33	0.76	0.00	3.90	2.66	4.89	7.13	2.10	0.10	0.01	0.00	0.00	0.00	.	0.00
	max	51.22	16.55	19.00	42.54	104.70	103.43	32.47	42.26	27.23	30.27	32.36	89.04	55.77	31.54	43.59	15.15	31.30	21.97	23.56	257.95	643.15	28.32	19.82	327.95	40.20	41.61	145.56	.	643.15
	mean	17.52	6.38	49.68	7.81	26.00	11.58	5.72	15.91	8.90	12.07	8.11	23.50	13.01	17.56	9.35	3.40	8.71	7.89	8.77	53.17	114.20	6.38	2.83	89.12	16.99	4.48	16.75	.	20.96
	mode	1.88	4.56	2.83	1.06	10.36	1.29	1.30	6.90	3.40	6.81	1.69	15.58	6.97	14.39	3.37	1.12	2.35	4.35	3.19	8.05	15.08	2.76	0.60	4.11	1.00	1.04	1.82	.	8.04
	median	18.58	5.15	43.34	5.23	20.26	6.01	2.63	10.03	7.09	10.38	3.00	18.45	7.83	17.08	5.34	2.62	5.45	5.86	7.88	30.77	29.58	5.28	1.69	71.59	19.83	2.25	4.47	.	7.95
	7 day	0.78	3.60	0.04	0.03	13.35	0.23	0.57	5.31	2.12	5.07	0.46	8.22	2.28	11.26	2.29	0.87	0.85	2.81	3.56	7.20	8.22	2.42	0.70	1.00	0.78	0.02	0.41	.	2.12
	total	45.40	16.50	128.80	20.20	67.40	30.00	14.80	41.20	23.10	31.30	21.00	60.90	33.70	45.50	24.20	8.80	22.60	20.40	22.70	137.80	296.00	16.50	7.30	231.00	44.00	11.60	43.40	.	.
DEC	min	29.56	7.88	18.62	2.97	19.71	0.00	16.29	3.17	19.74	8.10	0.42	9.56	2.42	9.78	7.07	0.00	3.18	6.87	3.66	14.67	39.82	0.00	0.00	40.67	0.00	0.00	4.22	.	0.00
	max	366.88	59.65	100.54	41.21	48.24	91.25	128.55	65.55	91.31	306.14	207.88	75.82	62.05	80.56	20.90	275.35	50.75	108.34	19.64	115.11	166.49	47.11	21.89	397.39	56.47	77.31	132.13	.	397.39
	mean	193.60	25.14	51.05	11.04	32.81	10.45	34.39	17.10	25.56	78.48	23.87	26.72	8.69	29.95	11.27	85.83	10.37	30.01	9.56	45.69	94.33	14.78	3.07	122.48	15.69	16.61	29.26	.	39.18
	mode	219.30	19.53	37.06	5.84	26.13	2.28	17.69	4.73	21.53	11.83	8.20	17.84	3.91	11.55	8.10	30.98	4.37	15.75	4.86	25.97	92.07	1.18	0.55	125.39	7.06	1.93	5.82	.	4.97
	median	204.22	20.75	46.17	7.31	30.75	3.76	25.58	12.01	19.89	72.98	7.54	19.11	3.64	25.93	7.98	74.34	7.22	20.35	8.49	36.17	90.79	11.99	1.04	123.03	11.93	7.40	17.36	.	19.74
	7 day	22.30	8.62	26.26	5.16	15.01	0.15	9.14	5.02	10.80	12.27	1.43	12.23	2.95	13.20	7.84	4.03	3.49	9.49	5.35	21.27	52.69	0.49	0.47	43.65	3.41	2.22	5.50	.	7.84
	total	518.50	67.30	136.70	29.60	87.90	28.00	92.10	45.80	68.50	210.20	63.90	71.60	23.30	80.20	30.20	229.20	27.80	80.40	25.60	122.40	252.60	39.60	8.20	328.00	42.00	44.50	78.40	.	.

Table 5.3 could be presented in two ways: as the number of occurrences per discharge class each month (Table 5.3A) , or as percentages of different classes within each month (Table 5.3B). The discharge class intervals (in $m^3 s^{-1}$) could be:

- 0.0-0.1 up to 0.9-1.0 (11 classes, with a summary line)
- 1–2 up to 9–10 (last summary line plus 9 classes, with a summary line)
- 10-20 up to 90-100 (last summary line plus 9 classes, with a summary line)
- > 100

Frequency of occurrence of present daily discharges

The same presentation as above, but for present-day discharges, allows a first assessment of how historical changes in the flow regime might be linked to historical changes in the riverine ecosystem or channel morphology.

Average daily discharge for virgin or near natural conditions

A table of several years of virgin or near-natural daily discharge ($m^3 s^{-1}$), if available, allows workshop participants to discuss the characteristics of periods of high and low flow (Table 5.4). Data for selected months in the wet, dry and transitional seasons could be presented, with the years displayed chronologically as columns and the days of the month as rows.

Zero and low-flow conditions

The number of days of no or very low flow should be provided, as a percentage of all the days of record, for each calendar month, for both virgin and present conditions. This helps workshop participants to visualise the ephemerality of the river, and the months of lowest flow. The need for this information was suggested by Prof. A. Arthington (Griffith University, Brisbane pers. comm.), and details of how to present it have not been decided upon yet.

Daily flow duration curves

Flow duration curves should be provided of daily average flows, for each month, for virgin conditions, present conditions (if different) and predicted future conditions (Figure 5.12). These allow the workshop leaders to assess if requested IFR flows lie within the virgin flow regime, and also to search for trends between the IFR requested at the workshop and those requested at other workshops. Flows should be shown as $m^3 s^{-1}$, and the length of record used in the analysis should be stated, as well as whether or not wet and dry years were included. The 10th and 90th percentiles for each month should be tabulated, and in addition, each plot should be sufficiently large for percentiles related to various flows to be read off accurately. Alternatively, the percentiles could be provided directly from computer by the hydrologist at the workshop.

Flood recurrence intervals

A table of the virgin (or naturalised) and present recurrence intervals of different magnitude floods allows the ecologists to check that flows of specific magnitudes are not being requested more often than naturally occurred (Table 5.5). Additionally, the geomorphologists will be interested in floods of a range of magnitudes and return periods. These include the so-called bankful discharge, which is assumed to play an important role in channel maintenance and is thought to have a fairly specific recurrence interval.

Table 5.3(A) The estimated frequency of virgin daily discharges ($m^3 s^{-1}$) in each month for selected class intervals (modified from the hydrology starter document for the Koekedouw River IFR workshop).

DISCHARGE CLASS INTERVAL ($m^3 s^{-1}$)	FREQUENCY OF OCCURRENCE												
	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	TOTAL
0 - 0	0	0	0	2	4	114	55	15	2	3	2	0	**197**
0 - 0.1	33	69	513	1649	1534	1678	1082	500	153	37	35	11	**7294**
0.1 - 0.2	183	326	979	329	271	209	466	455	284	194	121	145	**3962**
0.2 - 0.3	276	385	334	110	69	76	205	238	248	206	146	189	**2482**
0.3 - 0.4	341	257	163	40	35	35	106	162	193	207	162	206	**1907**
0.4 - 0.5	340	400	79	20	32	36	49	138	177	220	188	250	**1929**
0.5 - 0.6	233	239	36	11	12	17	29	85	131	122	141	201	**1257**
0.6 - 0.7	197	137	27	8	7	8	21	80	109	129	150	235	**1108**
0.7 - 0.8	115	80	23	6	4	5	17	63	69	97	114	160	**753**
0.8 - 0.9	95	61	9	1	6	7	8	49	76	96	120	142	**670**
0.9 - 1	69	30	6	1	3	3	8	39	49	78	83	83	**452**
Total 0 - 1	**1882**	**1984**	**2169**	**2177**	**1977**	**2188**	**2046**	**1824**	**1491**	**1389**	**1262**	**1622**	**22011**
1 - 2	219	111	15	5	20	3	38	148	273	395	422	294	**1943**
2 - 3	29	14	2	1	0	0	12	82	132	179	223	82	**756**
3 - 4	11	3	1	2	0	1	0	38	67	88	90	34	**335**
4 - 5	2	1	0	7	1	0	0	28	30	35	42	10	**156**
5 - 6	0	1	0	1	0	0	3	9	20	14	15	9	**72**
6 - 7	1	0	0	5	1	1	2	4	10	7	9	3	**43**
7 - 8	0	1	0	0	1	0	2	0	7	1	7	0	**19**
8 - 9	1	1	3	0	0	0	6	1	3	3	4	6	**28**
9 - 10	2	0	0	0	2	0	1	0	2	1	11	5	**24**
Total 1 - 10	**265**	**132**	**21**	**21**	**25**	**5**	**64**	**310**	**544**	**723**	**823**	**443**	**3376**
10 - 20	39	7	9	3	2	5	19	19	34	22	51	31	**241**
20 - 30	10	4	1	0	1	3	0	20	26	30	32	22	**149**
30 - 40	2	2	1	0	0	0	0	12	13	18	16	7	**71**
40 - 50	2	0	0	0	0	0	0	11	19	14	14	5	**65**
50 - 60	1	1	0	0	0	0	0	4	2	2	1	0	**11**
60 - 70	0	0	0	0	0	0	0	1	0	1	1	0	**3**
70 - 80	0	0	0	0	0	0	0	0	0	1	1	0	**2**
80 - 90	0	0	0	0	0	0	0	0	0	0	0	0	**0**
90 - 100	0	0	0	0	0	0	1	0	0	0	0	0	**1**
Total 10 - 100	**54**	**14**	**11**	**3**	**3**	**8**	**20**	**67**	**94**	**88**	**116**	**65**	**543**
100 - ∞	0	0	0	0	0	0	0	0	1	1	0	0	**2**
Total	**2201**	**2130**	**2201**	**2201**	**2005**	**2201**	**2130**	**2201**	**2130**	**2201**	**2201**	**2130**	**25932**

Table 5.3(B) The estimated percentage of virgin daily discharges ($m^3 s^{-1}$) in each month for selected class intervals (modified from the hydrology starter document for the Koekedouw River IFR workshop).

DISCHARGE CLASS INTERVAL ($m^3 s^{-1}$)	PERCENTAGE OF OCCURRENCE												
	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	TOTAL
0 - 0	0.0	0.0	0.0	0.1	0.2	5.2	2.6	0.7	0.1	0.1	0.1	0.0	**0.8**
0 - 0.1	1.5	3.2	23.3	74.9	76.5	76.2	50.8	22.7	7.2	1.7	1.6	0.5	**28.1**
0.1 - 0.2	8.3	15.3	44.5	14.9	13.5	9.5	21.9	20.7	13.3	8.8	5.5	6.8	**15.3**
0.2 - 0.3	12.5	18.1	15.2	5.0	3.4	3.5	9.6	10.8	11.6	9.4	6.6	8.9	**9.6**
0.3 - 0.4	15.5	12.1	7.4	1.8	1.7	1.6	5.0	7.4	9.1	9.4	7.4	9.7	**7.4**
0.4 - 0.5	15.4	18.8	3.6	0.9	1.6	1.6	2.3	6.3	8.3	10.0	8.5	11.7	**7.4**
0.5 - 0.6	10.6	11.2	1.6	0.5	0.6	0.8	1.4	3.9	6.2	5.5	6.4	9.4	**4.8**
0.6 - 0.7	9.0	6.4	1.2	0.4	0.3	0.4	1.0	3.6	5.1	5.9	6.8	11.0	**4.3**
0.7 - 0.8	5.2	3.8	1.0	0.3	0.2	0.2	0.8	2.9	3.2	4.4	5.2	7.5	**2.9**
0.8 - 0.9	4.3	2.9	0.4	0.0	0.3	0.3	0.4	2.2	3.6	4.4	5.5	6.7	**2.6**
0.9 - 1	3.1	1.4	0.3	0.0	0.1	0.1	0.4	1.8	2.3	3.5	3.8	3.9	**1.7**
Total 0 - 1	**85.5**	**93.1**	**98.5**	**98.9**	**98.6**	**99.4**	**96.1**	**82.9**	**70.0**	**63.1**	**57.3**	**76.2**	**84.9**
1 - 2	10.0	5.2	0.7	0.2	1.0	0.1	1.8	6.7	12.8	17.9	19.2	13.8	**7.5**
2 - 3	1.3	0.7	0.1	0.0	0.0	0.0	0.6	3.7	6.2	8.1	10.1	3.8	**2.9**
3 - 4	0.5	0.1	0.0	0.1	0.0	0.0	0.0	1.7	3.1	4.0	4.1	1.6	**1.3**
4 - 5	0.1	0.0	0.0	0.3	0.0	0.0	0.0	1.3	1.4	1.6	1.9	0.5	**0.6**
5 - 6	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.4	0.9	0.6	0.7	0.4	**0.3**
6 - 7	0.0	0.0	0.0	0.2	0.0	0.0	0.1	0.2	0.5	0.3	0.4	0.1	**0.2**
7 - 8	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.3	0.0	0.3	0.0	**0.1**
8 - 9	0.0	0.0	0.1	0.0	0.0	0.0	0.3	0.0	0.1	0.1	0.2	0.3	**0.1**
9 - 10	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.5	0.2	**0.1**
Total 1 - 10	**12.0**	**6.2**	**1.0**	**1.0**	**1.2**	**0.2**	**3.0**	**14.1**	**25.5**	**32.8**	**37.4**	**20.8**	**13.0**
10 - 20	1.8	0.3	0.4	0.1	0.1	0.2	0.9	0.9	1.6	1.0	2.3	1.5	**0.9**
20 - 30	0.5	0.2	0.0	0.0	0.0	0.1	0.0	0.9	1.2	1.4	1.5	1.0	**0.6**
30 - 40	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.5	0.6	0.8	0.7	0.3	**0.3**
40 - 50	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.9	0.6	0.6	0.2	**0.3**
50 - 60	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.1	0.0	0.0	**0.0**
60 - 70	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**
70 - 80	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**
80 - 90	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**
90 - 100	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**
Total 10 - 100	**2.5**	**0.7**	**0.5**	**0.1**	**0.1**	**0.4**	**0.9**	**3.0**	**4.4**	**4.0**	**5.3**	**3.1**	**2.1**
100 - ∞	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**
Total	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**

Table 5.4 **Daily average discharge for a wet month (February), recorded at gauging weir A5H004 (modified from the starter document for the Lephalala River IFR workshop).**

DAY	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981
1	1.86	3.11	3.07	1.77	14.30	0.18	0.62	1.51	3.46	5.22	0.14	21.10	17.60	14.90	1.06	6.66	0.20	5.83	9.40	1.59
2	1.69	2.16	3.06	5.20	18.30	0.16	0.57	1.43	3.09	4.76	0.14	15.40	14.20	12.10	1.35	5.61	0.22	6.02	7.37	1.41
3	1.53	1.76	6.10	5.48	96.10	0.15	0.52	1.57	2.91	4.36	0.14	12.80	12.10	10.60	1.54	5.03	0.30	5.94	11.50	1.25
4	1.39	1.51	4.05	3.32	73.80	0.13	0.46	2.01	2.60	3.98	0.14	10.50	10.30	8.50	1.91	4.60	0.37	5.05	13.20	1.16
5	1.29	1.33	3.96	2.09	47.70	0.12	0.42	1.66	2.44	3.64	0.13	8.98	8.96	7.42	3.20	4.52	0.32	4.75	14.00	1.02
6	1.18	1.20	3.35	1.60	24.50	0.10	0.39	1.50	2.49	3.38	0.12	11.50	8.28	6.63	3.66	4.85	0.30	5.09	11.10	0.96
7	1.09	1.16	2.76	1.33	25.90	0.09	0.57	1.66	2.02	3.09	0.12	13.90	10.50	10.00	3.11	5.64	0.36	4.18	9.00	0.90
8	1.00	1.07	2.33	5.40	20.40	0.08	0.55	1.84	1.86	3.12	0.15	30.10	17.90	17.10	2.54	5.41	0.51	3.73	8.04	0.85
9	1.01	0.97	2.06	5.16	47.70	0.07	0.45	1.74	1.81	3.15	0.40	17.40	15.00	11.00	2.24	4.82	3.78	3.38	14.80	0.83
10	0.95	0.90	1.87	3.27	97.80	0.11	0.54	1.60	3.69	3.20	0.36	12.40	26.60	8.70	2.08	4.66	6.54	5.26	16.50	0.80
11	0.87	0.84	1.80	2.22	48.50	0.11	0.44	1.46	4.26	3.35	0.30	10.50	21.20	10.00	1.88	4.38	7.46	9.90	12.70	0.76
12	0.81	0.79	1.63	1.77	23.90	0.09	0.54	1.36	4.24	2.87	0.25	9.40	20.40	154.00	1.87	4.16	6.46	7.36	10.30	0.71
13	0.76	0.70	1.49	1.48	17.30	0.09	1.04	1.30	4.25	2.73	0.22	8.21	21.30	58.90	1.71	4.59	4.46	5.26	10.10	0.70
14	0.70	0.64	1.40	1.33	15.10	0.08	0.65	1.25	3.95	2.36	0.22	7.61	15.00	26.10	1.51	6.02	3.12	4.83	8.70	0.70
15	0.64	0.62	1.34	1.21	15.70	0.08	0.49	1.51	3.20	2.14	0.21	7.19	41.20	17.60	1.37	10.10	2.28	3.81	7.59	0.61
16	0.60	0.58	1.28	1.11	15.00	0.13	0.41	2.40	2.76	2.10	0.20	6.83	23.60	13.80	1.24	8.22	2.48	4.19	6.92	0.57
17	0.56	0.65	1.24	1.01	11.00	0.16	0.45	3.16	5.67	2.17	0.19	6.51	18.70	11.30	1.16	11.50	3.47	15.30	6.31	0.56
18	0.53	0.72	1.21	0.91	25.70	0.13	0.62	5.44	9.10	2.08	0.17	8.27	26.60	9.70	1.11	11.70	3.49	16.40	5.76	0.62
19	0.50	0.63	1.93	0.87	25.10	0.12	1.23	2.97	6.11	2.18	0.16	11.90	17.10	9.00	1.14	9.40	2.96	19.60	5.57	0.78
20	0.48	0.59	1.58	1.17	19.90	0.11	1.20	2.19	4.82	5.48	0.15	9.40	20.00	8.24	1.22	8.46	2.21	31.10	5.85	1.08
21	0.46	0.55	1.30	1.13	14.40	0.10	1.03	1.87	4.18	6.52	0.14	7.75	28.10	7.69	1.41	8.24	1.85	75.50	5.11	1.00
22	0.44	0.51	1.18	0.96	12.00	0.11	0.98	1.70	4.09	6.14	0.14	6.93	29.80	9.00	1.18	12.10	1.62	26.00	4.79	0.82
23	0.42	0.48	1.14	1.77	10.00	0.10	1.45	1.59	4.11	12.40	0.13	6.52	34.70	24.50	1.10	9.80	1.41	18.10	4.60	0.75
24	0.69	0.44	1.12	2.26	8.85	0.10	1.90	1.49	4.56	11.30	0.12	6.13	19.60	29.30	1.04	8.36	1.26	33.50	4.41	0.70
25	0.52	0.41	0.98	1.50	8.03	0.09	1.59	1.39	4.80	9.80	0.12	5.93	24.40	20.90	0.99	7.27	1.14	79.40	4.22	0.69
26	0.54	0.38	0.91	1.26	7.53	0.09	1.29	1.30	4.72	11.20	0.12	5.81	15.40	19.50	0.95	6.41	1.04	40.40	4.09	0.82
27	0.48	0.35	0.84	1.08	6.98	0.08	1.15	1.19	3.95	9.40	1.19	5.62	12.90	14.00	0.91	5.80	0.95	21.60	4.34	4.47
28	0.47	0.33	0.83	1.00	6.94	0.08	1.05	1.14	3.88	16.90	1.30	5.56	17.30	11.30	0.86	5.80	0.89	17.1	4.20	4.03
Average	**0.84**	**0.91**	**1.99**	**2.10**	**27.09**	**0.11**	**0.81**	**1.83**	**3.89**	**5.32**	**0.26**	**10.36**	**19.60**	**20.06**	**1.62**	**6.93**	**2.19**	**17.09**	**8.23**	**1.11**

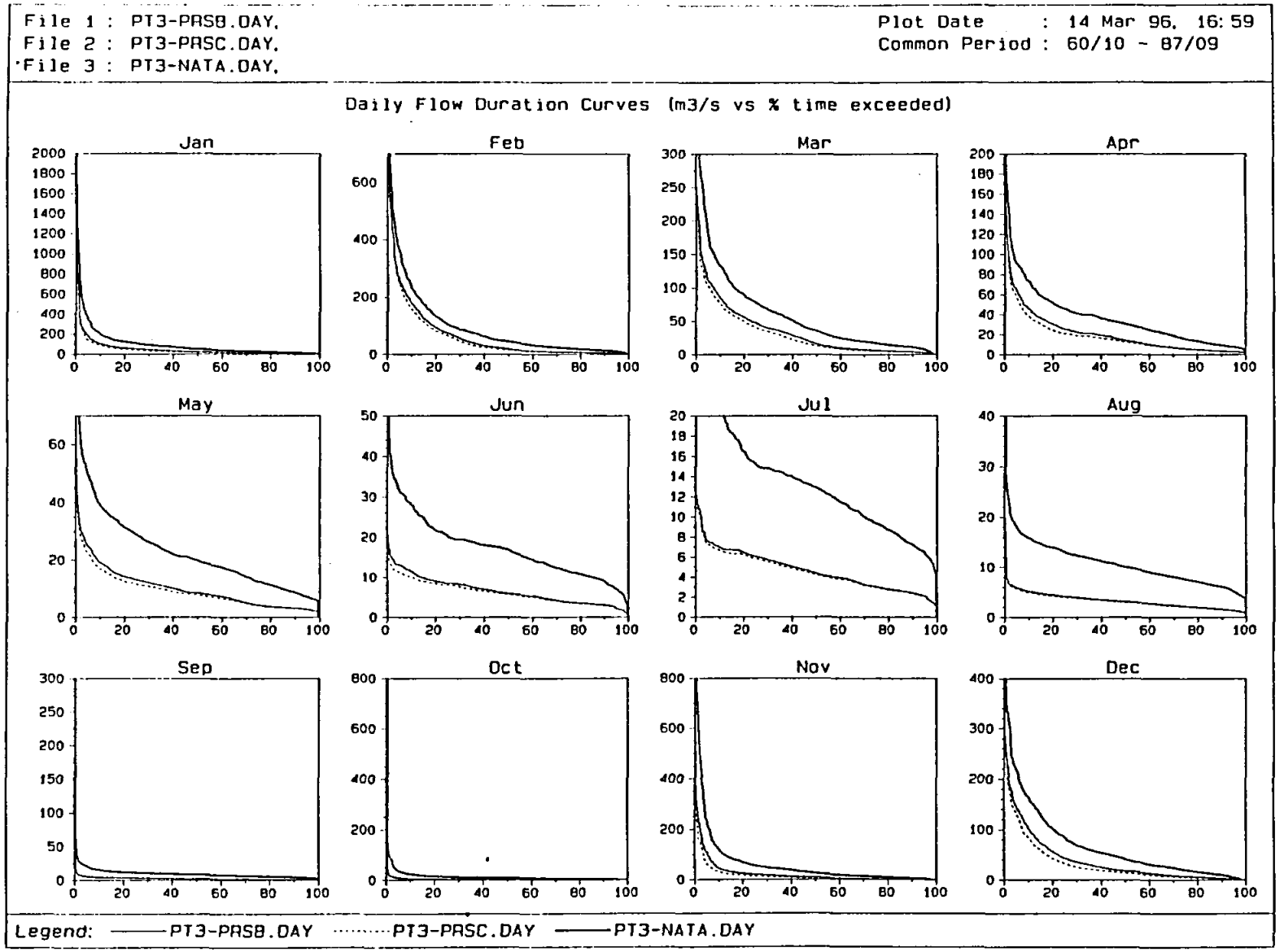


Figure 5.12 **Virgin (PT3-NATA.DAY), present (PT3-PRSB.DAY) and simulated (PT3-PRSC.DAY) future flow duration curves for each calendar month, averaged over the period of record October 1960 - September 1987 for gauging weir B7H009 on the Olifant-Sand River. Model printout supplied by Ninham Shand Consulting Engineers (from the starter document for the Olifant-Sand River IFR workshop; NSI 1993c).**

Table 5.5 The recurrence intervals of different magnitude floods (modified from the starter document for the Berg River (Skuifraam Dam) feasibility study). * - restricted by rating limit.

IFR SITE	PERIOD	PEAK DISCHARGE ($m^3 s^{-1}$)	RECURRENCE INTERVAL				
			1:1 y	1:2 y	1:5 y	1:10 y	1:20 y
IFR 1/96	1959-77	Average daily	85	115	163	163	200
		Instantaneous	220	260	360	360	400
	1981-94	Average daily	50	60	80	80	86*
		Instantaneous	86*	86*	86*	86*	86*
IFR 3/96	1967-77	Average daily	180	290	450	535*	535*
		Instantaneous	Not available (no observed flow data)				
	1980-94	Average daily	140	240	300	310*	310*
		Instantaneous	200	240	300	360	390

Dr K. Rowntree (Geography Department, Rhodes University) has advised on ways to improve this data input for the workshops. She suggests that to produce the requisite data on floods, the hydrological record should be analysed with respect to peak discharges. These should be instantaneous discharge peaks, but average daily peaks will suffice. A partial flood duration series (Gordon *et al.* 1992), incorporating all monthly peak flows above a specified threshold level, should be derived. The threshold level should be decided upon in consultation with the geomorphologists prior to the workshop: it could be the minimum annual flood, or preferably some threshold determined by channel morphology or sediment transport thresholds. A flood frequency curve should be presented, and the magnitude of floods with certain recurrence intervals extracted from this. From a geomorphological perspective, these recurrence intervals would be 0.9, 3 and 5 years (Dr K. Rowntree, Geography Department, Rhodes University pers. comm.). Ecologists additionally routinely require the magnitude of floods with 2, 10, 20 and 50 year recurrence intervals, which they will link to such factors as the degree of reed encroachment in channels and the positioning of certain components of the riparian vegetation on the banks.

Shape of the flood and fresh hydrographs

In the IFR workshop, high flows of various magnitudes are commonly asked for, usually together with the recommendation that they adhere to "the normal hydrograph shape for that time of the year". This recommendation is given as several biological activities related to freshes or floods, such as fish spawning and movement of fish larvae out of flooded areas and back into the main channel, could be expected to be successfully completed during a "normal high flow period". They might, however, be unsuccessful if a shorter period of high flow occurred. At the IFR workshop, the hydrologist describes the shape of such "normal" floods and freshes for various times of the year. Descriptions are in terms of time to peak, magnitude at peak, hydrograph shape, total duration, total volume of water, and dates of examples used. Ilustrations of selected flood hydrographs are presented (Figure 5.13).

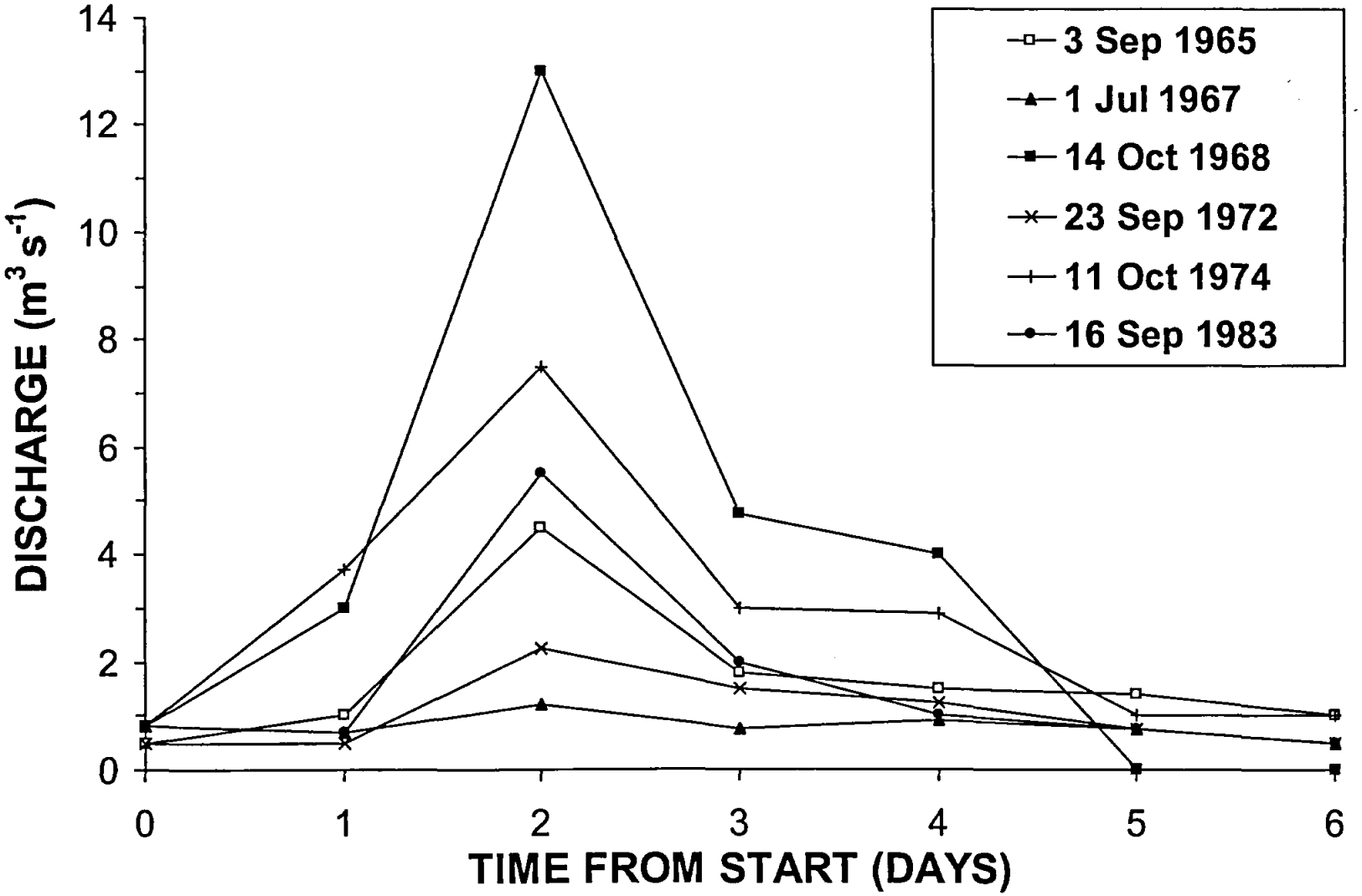


Figure 5.13 Flood hydrographs at gauging weir H1H013 on the Koekedouw River, showing the magnitude, duration and hydrograph shape for "last floods of the wet seasons" selected from the overall hydrological record (modified from the starter document for the Koekedouw River IFR workshop).

5.4 VISUAL AIDS FOR THE TRANSFER OF HYDROLOGICAL INFORMATION AT THE WORKSHOP

Posters of hydrological data are an essential feature of IFR workshops. Ideally, these consist of daily hydrographs of virgin discharge ($m^3 s^{-1}$) for whole years, with three years to a plot, each in a different bright colour, and ranked from wettest to driest year. The plots cover the full width of a poster board so that numbers can be read from them and the plots can be seen well from a distance. These posters allow workshop participants unfamiliar with hydrological data to quickly gain an essential feel for the variation of the river's flow, from day to day, season to season, and year to year. Where daily flow data are not available, the hydrologist consults with the IFR workshop co-ordinator regarding the most suitable information to include on the hydrology wall posters.

For ease of use in the workshop, it is also recommended that coloured adhesive tabs be provided for workshop participants, so that they can flag the various parts of their hydrological documents in a way that suits them.

5.5 DISCUSSION

It has become increasingly apparent during the last four years of IFR workshops that long-term data on natural, daily flow for IFR sites are rarely available. Gauging weirs may not exist near selected sites or, if they do, may have short-term or unreliable data. Hydrological data can be simulated for such sites, but usually at the level of resolution of monthly average runoff. Daily flow data can be generated, but generally with a very low confidence level; even simulations of monthly average runoff have a wide possible margin of error and many hydrologists are reluctant to attempt to disaggregate these figures down to daily flows. Another problem has been that of whether the historical flow regime is a good point of departure for assessing the IFR for a riverine ecosystem which already has a highly modified flow regime. The argument has been offered that as such a riverine ecosystem will already have changed, in response to historical changes in the flow regime, the virgin flow regime is no longer relevant. It has thus been suggested that in such a situation, the modified flow regime should be the basis of discussions on a future IFR for the river. Both of these points are discussed below.

5.5.1 Daily versus monthly data

The conceptual basis of the BBM relies on identification of the natural flow regime of the river (Chapter 3). Thus, it is necessary for workshop participants to be informed regarding **the flows that are a normal characteristic of a specific river, no matter how extreme, variable or unpredictable they may be, for these are ones which the riverine species are adapted to cope with and perhaps are reliant on. Flows that are not characteristic of that river will constitute an abnormal disturbance to the riverine ecosystem and could fundamentally change its character.** An accurate, long-term hydrological record of daily flows seems to be the best way of statistically defining the flows that are characteristic of a river (Section 5.2, last bulleted point). Such data are often not available, however, so hydrological consultants have asked the river scientists to describe the main features of the annual flow regime, or the main kinds of hydrological data, that they consider when determining an acceptable modified flow regime. The

hydrologists will then search for ways of extracting this information from whatever hydrological data are available. In this Chapter several formats for presenting the required hydrological data at the required resolution have been detailed.

5.5.2 Virgin versus present-day data

The river under consideration in an IFR workshop may already have had a considerable proportion of its MAR abstracted, or its natural flow pattern altered in some other way. The riverine ecosystem may have changed over time in response to this, to the point where attempts to revert it to its natural character are no longer possible or desirable in terms of the DS. Under such conditions, it is essential to question the necessity of considering the natural flow regime when compiling the recommended modified flow regime. The following points are pertinent to such considerations.

River channels and their biotas tend to differ from region to region within the country because of, *inter alia*, regional differences in flow regimes. Some consideration of these regional differences seems essential if building similar modified flow regimes for rivers throughout the country is to be avoided. Rather, attempts should be made to understand and retain regional differences in even highly modified rivers, in order to minimise potentially costly ecological repercussions.

Such repercussions are possible because the historical flow regime of any river helped to sculpture both the original river channel and its original biota. Parts of that flow regime, even if only the largest floods, still will be apparent in all but the most modified of rivers, and still will be influencing the character of the channel and biota to some unknown extent. Thus, it might not be possible, for instance, to create a narrower river with a new stable riparian zone, because the large floods still will define the size of the channel. Additionally, components of the original biota still may be present in the river and potentially able to proliferate to pest proportions; recent blackfly swarms from the Vaal River (De Moor 1994) and massive blue-green algal blooms in the Darling River (Australia) (Murray-Darling Basin Commission 1994) present good examples of ecological repercussions when natural flow regimes are ignored. Thus, an understanding of the regional character of the river, and the difference between its virgin and present flow regimes will help workshop participants in two ways. It will provide a guideline of the flow regime most suited to the river, an insight that is essential for specialists not familiar with the rivers of the region. It will also allow assessment of the extent to which that most-suited flow regime has been lost. Workshop participants then will need to decide the extent to which the modified flow regime that they compile needs to adhere to the historical one.

Arguments have been offered that when the desire is to maintain an already modified river in its present condition, all that is needed is to maintain the present altered flow regime. However, this holds true only when it can be shown that the riverine ecosystem has reached a new equilibrium and is not still changing in response to an altered flow regime that was introduced some years previously.

To aid the above considerations, two major procedures still need developing. These are: identification of which data on the virgin flow regime, if any, would be most useful when considering a new flow regime for an already highly modified river; and what investigations should be done on such a riverine ecosystem in order to assess whether or not it has achieved some new equilibrium state.

5.6 CONCLUSIONS

Descriptions have been given on the kinds of hydrological data that river scientists can use at IFR workshops. Where specific kinds of data cannot be produced, it is suggested that the hydrologist consider the options available, decide what can be provided at a reasonably high level of accuracy and reach agreement with the IFR workshop co-ordinator on the procedure to follow.

6. SUMMARY OF PRESENT AND FUTURE DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY

Written by J. King and D. Louw (then at DWAF)

6.1 INTRODUCTION

6.2 MILESTONES IN THE DEVELOPMENT AND APPLICATION OF THE BUILDING BLOCK METHODOLOGY

6.2.1 Pre-feasibility Phase

6.2.2 Feasibility Phase

6.3 FURTHER DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY IN THE DEPARTMENT OF WATER AFFAIRS AND FORESTRY'S FEASIBILITY PHASE

6.3.1 Suggested principles for the application of the Building Block Methodology in Pre-feasibility

6.3.2 Suggested principles for the application of the Building Block Methodology in Feasibility

6.3.3 Suggested sequence of actions related to the Building Block Methodology for completion during the Department of Water Affairs and Forestry's Feasibility Phase

6.1 INTRODUCTION

Chapters 2 to 5 have described the development and nature of the BBM. In this Chapter, some of the main milestones and achievements linked to its development are summarised (Section 6.2). Additionally, as some of the earliest proposed projects for which IFAs were done have now moved from DWAF's or other developers' Pre-feasibility to Feasibility Phase, the further development of the BBM to respond to this is addressed (Section 6.3).

6.2 MILESTONES IN THE DEVELOPMENT AND APPLICATION OF THE BUILDING BLOCK METHODOLOGY

6.2.1 Pre-feasibility Phase

- Mid to late 1980s: The Environment Studies sub-directorate of DWAF developed an internal process of Integrated Environmental Management, which identifies, *inter alia*, that instream flow assessments should be done for all proposed water-resource developments involving rivers (Louw 1995a). Early approaches to IFAs were initiated by DWAF (e.g. DWA 1989, 1990a).

- 1987: DWAF and the Foundation for Research Development (FRD) ran the first two major workshops dealing with instream flow requirements (Bruwer 1991, and Ferrar 1989, respectively)

- 1990-1993: The WRC funded a project to assess IFIM and to initiate development of alternative methodologies if deemed necessary (King and Tharme 1994). This project ended in a first description of what was to become the BBM.

- November 1991: A group of South African twelve water managers and aquatic scientists visited Australia on a scientific tour. As a result of presentations and workshop discussions at Griffith University, Brisbane, the conceptual basis of an holistic approach, later to become the BBM in South Africa and the Holistic Approach in Australia, was devised by South African and Australian river scientists. Prof. Angela Arthington (Griffith University) reported on this development two days later in a presentation at an international seminar and workshop at Armidale (Arthington *et al.* 1992).

- February 1992: IFR workshop for the Lephalala River (King 1993). DWAF appointed one of the authors of this Chapter (JMK) as a consultant for advising on the IFR for the river. As early indications from the above WRC project were that international approaches did not seem suitable for a variety of reasons, an approach based on a combination of hydrological, hydraulic and biological data was suggested by JMK and Mr C Bruwer (DWAF). At this stage the approach was still called the Holistic Method (Chapter 3.2). JMK arranged with the head of the sub-directorate of Environment Studies (DWAF) to be able attend future IFR workshops to aid development of the methodology.

- November 1992: IFR workshop for the Berg River (NSI 1993a, b). The need for input from a geomorphologist was recognised, in order to guide any instream requirement for floods. The need to define the objective to be achieved with the IFR (later to become known as the DS) was also recognised as essential in guiding the amount of water asked for. By now the name Building Block Methodology was being used, to distinguish the approach from the Holistic Approach which continued independent development in Australia.

- August 1993: IFR workshop for the Olifants River (Western Cape) (Tharme 1994). The topic of River Importance was introduced for the first time, to aid workshop participants to reach consensus on their recommendation for DS. A study area was delineated and reaches were recognised in this, for each of which a separate assessment of the IFR was done. IFR sites were established in each reach. Surveyed cross-sections were used for the first time, specifically to record critical and representative aquatic biotopes, and detailed stage-discharge data for low flows were provided for each cross-section. All of these data were fortuitously available because of another project (King and Tharme 1994), but served to illustrate their usefulness to workshop participants. This was the first IFR workshop financed by a private developer (Citrusdal Irrigation Board), and the first (and only one) at which a post-workshop assessment was made of the degree to which the IFR would be met on a daily basis by the proposed future flow regime.

- September 1993: IFR workshop for the Olifant-Sand River (NSI 1993c, 1995a). The need for input from a social consultant was recognised. This was because the social dependence of rural populations on the use of resources from riverine ecosystems would affect the DS and thus the IFR for that river. This was also the first IFR workshop for which a video recording of the river was made during a helicopter flight,

so that an assessment of the conservation status of the river could be made, and the first one to make use of wall posters for site and hydrological information.

- June 1994: IFR workshop for the Letaba River (SRK 1994). Procedures for the delineation of the study area and selection of specific IFR sites were formalised, and presentation of cross-section information with stage-discharge curves for these sites became a standard input to the IFR workshops. Reacting to demands from consulting engineers who were now having to address IFAs, guidelines were written on the sequence of necessary actions (Louw and King 1995).

- July 1994: IFR workshop for the Luvuvhu River (LDC 1995). This was generally acknowledged as the most successful IFR workshop to date, and provided the template on which all future ones were based. Its success was largely due to the analysis of previous workshops, mostly done by Ms D. Louw, which led to a constantly improving process and focussing of effort. Also of importance was the high standard of hydrological data presented, largely through close contact between the DWAF planning team and the hydrological consultant. Further success was due to the presence at the workshop of experienced hydrologists who interpreted the IFR recommendations in terms of MAR and provided much advisory input and guidance during the workshop discussions. Other features of this workshop were that the focus of the assessment of river condition changed from conservation status to habitat integrity, and the quality of the starter document, and wall posters of sites and hydrological data, was very high. The present format of the workshop was established by splitting participants into two groups. Each group consisted of at least one of each kind of specialist and was led by a group leader experienced in the BBM. Each group concentrated on the full range of flows required for the IFR sites allocated to them.

- September 1994: IFR workshop for Driekoppies Dam (Lomati and Komati rivers) (NSI 1994b). This was the first workshop where it was recognised that one of the rivers (the lower Komati) was so modified (by weirs) that an IFR for river maintenance would probably be inappropriate. A very small discharge, to maintain water quality in, and fill, the weirs, was all that could be motivated for.

- October 1994: DWAF held a one-day meeting with BBM specialists and consulting engineers involved in IFR workshops to try and resolve the problem of what hydrological data were needed for the workshops (Louw 1995b). It was partially successful, but further assessment on which data were used by specialists at each workshop led to Chapter 5 of this document.

- March 1995: IFR workshop for the Koekedouw River (SRK/SW 1995). This was the first IFR workshop organised and run completely outside DWAF.

- April 1995: IFR workshop for the Senqu River (Lesotho) (DWAF 1995a). This was the first IFR workshop for a river outside South Africa, and the first one where a formal explanation was given of the process of selection and character of IFR sites. The confidence of the workshop output was lower than for ones such as the Luvuvhu River. This was mainly because there was little biological and geomorphological knowledge and experience of this or similar rivers, and there were few clues to appropriate flows (such as vertical zonation of riparian plants species) provided by the river itself. Other causes for low confidence were that, because of budgetary constraints, the workshop was not held in the catchment of concern and there was no site visit for workshop participants. They, therefore, had no first

hand experience of the river, which in retrospect was seen as inadequate for application of a methodology based on expert opinion.

- May 1995: IFR workshop for the Mooi River (NSI 1995b, c). This was the first IFR workshop for a Natal river. It was of interest because a possible trend in IFRs seemed to be developing, whereby more of the virgin MAR was being asked for rivers with more strongly perennial flow regimes, than for more hydrologically flashy rivers. Most participants at the workshop were by now familiar with the process, and the quality of the starter document reflected this in that specialist chapters were becoming increasingly focused on flow-related information.

- June-July 1995: JMK visited Australia at the invitation of the Australian National Rivers Health Programme, to present papers on environmental flows at a national workshop on Environmental Flow Assessment techniques. She gave two presentations on the BBM to State water authorities and related papers at centres around Australia (King and Louw in press a; King and Tharme in press).

- September 1995: IFR workshop for the Tugela River (DWAF 1995b). Eight IFR sites were used in this workshop, double the number than at most other IFR workshops. As a result the workshop was longer than usual, and the two specialist groups only visited the sites they would be considering. This was the first IFR workshop where the specialists were asked to list in their own words their motivations for requested flows, so that the workshop scribe could record them more accurately. The workshop was attended by the first overseas observer, Prof. A. Arthington of Griffith University, Brisbane, Australia.

- October - December 1995: Activities related to the Mvoti IFR workshop (to be held mid 1996; Tharme 1997a) and the Skuifraam Dam Feasibility study (NSI 1996) led to a series of new procedures being established within the BBM. These included a preliminary reach analysis of the study site by geomorphologists before the aerial survey; identification of possible IFR sites from the air; preliminary site selection from the aerial video by a group of specialists; and incorporation of geomorphologists into the field team that makes the final selection of, and collects data for, the IFR sites.

- December 1995: Prof. D. Hughes (Rhodes University) created a model for linking the IFR workshop output to current climate (Hughes *et al.* 1997). This contributed to the solution of two problems that had dogged development of the BBM: a) how to guide when the 'maintenance IFR' or 'drought IFR' should be used (Table 3.1); and b) how to counteract the rigid set of numbers in the IFR tables and recreate some essential variability in the daily flow regime of the river.

- January 1996: Ms S. Pollard (Wits Rural Facility) wrote the first draft of a guideline to aid social consultants to prepare and present the appropriate data for an IFR workshop.

- January 1996: The Deputy Director-General of DWAF and its Head of Environment Studies spoke of environmental water requirements and the BBM on the national radio.

- February 1996: IFR workshop for the Mogalakwena River (Ractliffe 1996a, b). This was the first IFR workshop for a temporary river, and the first with a specialist input on subsurface flow. Lessons learnt were that the workshop site visit was of limited value when the river was in flood, except for providing a

general picture of conditions in the catchment. This was also the first IFR workshop that benefited from the discussions on "BBM into Feasibility" that took place in December 1995 (Section 6.2.2). As a result, an assessment was made at the end of the workshop as to the level of confidence in the workshop output, and vital additional work to improve the confidence level was identified. A programme was suggested for completion of that work, and persons were identified who would be responsible for taking the process forward.

- June 1996: The WRC held a one-day workshop, where the core group that developed the BBM discussed with other relevant specialists training courses for transferring the methodology to other potential practitioners (Brown 1996a).

- August 1996: An IFR workshop was held for the Sabie-Sand River system (Tharme 1997b). The Sabie is the main study river being studied within the KNPRRP. The workshop formally brought together for the first time the developers of the BBM and the researchers of the KNPRRP. In a one-day meeting after the workshop, there was a scientific assessment of the BBM and of the extent to which the KNPRRP has enhanced the process (King and Tharme in prep.).

- September 1996: The two authors of this Chapter travelled to Australia, funded by the Australian Land and Water Resources Land and Water Corporation, to help facilitate the first application of the BBM to an Australian river. The proposer, grant holder and organiser of the workshop was Prof. A. Arthington of Griffith University, Brisbane. The best features of the separately developed BBM and Holistic Approach were identified and the potential for their incorporation into the other methodology assessed.

- Mid-1998: A comprehensive manual on applying the BBM should be completed (King and Louw in prep.).

6.2.2 Feasibility Phase

- January 1994: The Olifant-Sand River (NSI 1993c). This was the first of DWAF's proposed projects for which an IFR workshop had been held, to move into the Feasibility Phase. A scenario meeting was held to present the results of the hydrological modelling exercise, including scenarios of daily river flow with and without the IFR being met.

- May 1995: The Koekedouw River (SRK/SW 1995). The first IFR assessment organised outside of DWAF moved quickly from the IFR workshop (March 1995) to Feasibility scenario meetings (May 1995). From an initial perspective of not being able to meet the IFR, a compromise was agreed upon by the developers and the conservation authorities, with advice provided by the IFR scientists. With consensus reached, DWAF was approached in July 1995 to issue a Section 9B permit for construction of the proposed dam. The permit contained details of the instream flow allocation and conditions for meeting it. The dam is presently being built, with an environmental team on site.

- August 1995: The Luvuvhu River. The output from the IFR workshop was refined, mostly through the taking of new hydraulic measurements and an on-site 'IFR refinement workshop' (Ms D. Louw, DWAF pers. comm.). The problems of, and procedures undertaken during, this process highlighted the need to formalise the development of the BBM in the Feasibility Phase (Section 6.3). This was also the first

Feasibility meeting where pre- and post-construction monitoring was discussed, led by Mr N. Kemper, then at NSI.

- December 1995: The first formal meeting of the core group of developers of the BBM was convened by DWAF to consider continued development of the methodology into DWAF's Feasibility Phase (Section 6.3).

- December 1995: A meeting was convened by DWAF to create a protocol for IFR monitoring, for rivers for which an IFR assessment has been made and the proposed water-resource development completed and operational.

- January 1996: The Luvuvhu River. At a further Feasibility meeting, the need was recognised, because of the difficulties of accurately modelling low flows, for calibration of the hydraulic model using at least two different discharges. Additionally, the need to carefully record all water depths asked for at the IFR workshop, and not just their conversions to discharge, was recognised, as this allowed the IFR to be recalculated, where necessary, with refined and amended hydraulic data.

- 1997: The first White Paper appeared which specified IFR amounts as produced through use of the BBM, together with their assurance of allocation. The White Paper dealt with the proposed construction of the Mutoti Dam on the Luvuvhu River (DWAF 1997a).

6.3 FURTHER DEVELOPMENT OF THE BUILDING BLOCK METHODOLOGY IN THE DEPARTMENT OF WATER AFFAIRS AND FORESTRY'S FEASIBILITY PHASE

The IFR workshop usually takes place during DWAF's Pre-feasibility Phase of a water-resource development, when various options for the locality and nature of the proposed development are assessed. In addition to producing the IFR for the river concerned, workshop participants are usually asked to make statements on which of the options would be preferred or not preferred from an ecological perspective. One or a few options are ultimately carried forward into the Feasibility Phase, at the end of which a final decision is made on whether or not to proceed, and if so with which option. Initiation of the chosen option is then announced in a government White Paper.

At present, the IFR process has only been formally developed to the end of the Pre-feasibility Phase. Recently, however, some of the proposed projects for which IFR determinations were made have moved into the Feasibility Phase (Section 6.2.2). This is now driving the need to examine how the BBM should be further developed into the Feasibility Phase. It is intended eventually to link the Pre-feasibility and Feasibility Phases of the IFR process with a follow-up monitoring programme that will be designed to cover the Design, Construction and Operation Phases of the relevant water-resource project (Table 3.2).

In December 1996, DWAF held a meeting involving most of the core group that has developed the BBM to discuss how the methodology should be further developed into Feasibility. The aim of the meeting was **To assess, through consideration of specific proposed water-resource projects that have entered the Feasibility Phase, what activities related to the BBM should take place between the IFR workshop and**

initiation of the monitoring programme. Through this, tentatively to describe the actions that should constitute the Feasibility Phase of the BBM, for testing and refinement in future proposed projects.

Those attending the meeting were the two authors of this chapter, Prof. J. O'Keeffe (Rhodes University), Mr N. Kemper (then at NSI), Dr K. Rowntree (Rhodes University), the other author of this report, RET, and Ms G. Ractliffe (FRU, University of Cape Town). A record of the meeting has been written and copies are held by all participants.

In the meeting, presentations were made on selected individual rivers and their associated proposed schemes. For each river, the BBM, as it was applied during Pre-feasibility and, if appropriate, Feasibility, was described. The degrees to which the Pre-feasibility process for each did not meet the present standards of application were identified. For those projects already moving through Feasibility, the good practice developed and problems encountered so far were discussed. Based on these analyses, discussions were directed as to what principles or guidelines could be extracted from the exercise pertaining to further development of the BBM.

6.3.1 Suggested principles for the application of the Building Block Methodology in Pre-feasibility

- The results of the IFR workshop held in Pre-feasibility should be used in whatever way is possible. It was recognised that earlier IFR workshops probably produced less useful results than later ones, because of the undeveloped nature of the BBM at that stage, and might need more follow-up work. Thus, after the workshop in Pre-feasibility, or at the latest in the early stages of Feasibility, the quality of the IFR workshop, including IFR sites selected, DS, input data and IFR output, should be formally assessed. This assessment should culminate in a written document that guides discussions on the work to be done in Feasibility.

- More comprehensive preparation of the IFR workshops, with the emphasis on accurate data inputs, will reduce expenses and effort at the Feasibility Phase and later. This is particularly true with regard to site selection, description of the hydrological character of the river, the choice of hydraulic cross-sections and low-flow hydraulic modelling. Good quality data of these kinds are a top priority, because the conversion of much environmental knowledge into recommended discharges is wholly dependent on the accuracy of the cross-section stage-discharge rating curves, while the recommendations for certain high flows, as well as the later checking that the IFR falls within the natural flow regime of the river, depend on accurate hydrological data. Poor quality hydraulic and surveying data inputs are often a major concern at the workshops, and are linked to a low confidence in the IFR produced and to requests that the work be repeated and assessed for accuracy at the Feasibility Phase. Such information inputs may lead to erroneous estimates of environmental flow requirements, which negate the workshop's purpose of providing an early responsible input to the Planning Phase of the water-resource development.

- Not all environmental uncertainties can be resolved during Feasibility, because of the scarcity of relevant data and the low level of knowledge and understanding of the functioning of specific river ecosystems. But the **major** uncertainties which impaired the ability of the river scientists at the IFR workshop to determine a recommended IFR, and which seriously and urgently need to be resolved before decisions on

ecological acceptability of the development are made, should be identified very clearly at the end of the IFR workshop.

- The scheduling of work to address these major uncertainties should also be made clear at the end of the IFR workshop, and specified delegates from the workshop should be made responsible for carrying these tasks forward in terms of acquiring DWAF's approval and ensuring that funds are available to meet the schedule. This formal link between Pre-feasibility and Feasibility is vital for two reasons. Firstly, it ensures continuity of the process and of relevant knowledge about the process. Secondly, the Feasibility Phase is usually too short for a year-round or longer survey to be organised and completed, if one is deemed essential, and so appropriate plans may need to be made for early funding and initiation of such a study.

6.3.2 Suggested principles for the application of the Building Block Methodology in Feasibility

- At the Feasibility Phase, environmental practice should mirror engineering practice. Thus, the remaining major ecological/environmental uncertainties linked to the remaining development options should be addressed, in order to guide the final IFR recommendation and statements on the acceptability of the proposed development.

- Hydrological scenario meetings take place after the IFR workshop, in Feasibility, where a range of development options to provide the IFR is considered. The objective of the meetings is to provide final statements from the ecological perspective on the acceptability of any of the presented scenarios. Participants at the meetings include the hydrologist modelling the scenarios, DWAF engineers and planners, representatives of the consulting engineers, and a selection of the river scientists who provided specialist input to the IFR workshop. Options presented can include those with a range of dam sizes, or ones where the water requirements of potential offstream users are reduced through changing the date to which the development can supply water at an assured yield. These meetings would be most effective in terms of ecological feedback if held after the major uncertainties identified in the IFR workshop had been resolved. However, for cases where the confidence in the IFR is high and there is an early awareness that it probably could not be provided under almost any circumstances, an alternative sequence of events needs to be developed.

- The appropriate presentations for hydrological scenario meetings need to be described. At present, the hydrologists are left to produce what they hope is appropriate, but their efforts may be negated by ecologists not being able to react to the scenarios presented.

- During Feasibility, the present and future demands for water in the whole river system, with the IFR included, should be assessed. This is because the study area for an IFR workshop is typically only that part of a river system that it is felt could directly be affected by the proposed development. There is an urgent need during Feasibility, however, to broaden the horizons of the IFR recommendation, so that water use and needs at the catchment level are brought into perspective before final recommendations on acceptability of the development are made.

6.3.3 Suggested sequence of actions related to the Building Block Methodology for completion during the Department of Water Affairs and Forestry's Feasibility Phase

Although the protocol for the IFR process in Feasibility is still being developed, its broad outline is becoming apparent.

It is felt that the three main activities that should be initiated before, or at latest by the early part of, Feasibility (Figure 6.1) are:

- formal assessment of the quality of the IFR (process, input data, output);
- completion of work to fill serious major data gaps, without which final recommendations cannot be made;
- hydrological yield analysis.

These should be supported or followed by a series of secondary activities related to:

- refinement of the IFR, using new and better quality data;
- hydrological scenario meetings;
- assessment of all the IFRs asked for in different IFR workshops for the total river system, to search for "red flags" indicating that water-resources are close to fully developed in specific parts of the catchment;
- if necessary, initiation of activities related to a decision not to meet the IFR. These should include clarity on the identity of the decision maker(s), the extent to which the public were involved and aware of the decision and its implications for the river, and a record of the decision;
- initiation of a monitoring programme to assess the efficacy of the IFR. The monitoring programme should provide the structure for a feedback loop that allows the IFR to be adjusted where necessary.

It is emphasised that the suggested sequence of IFR activities in Feasibility has not yet been formally recognised by DWAF and that Figure 6.1 should thus be seen as a preliminary template for discussion.

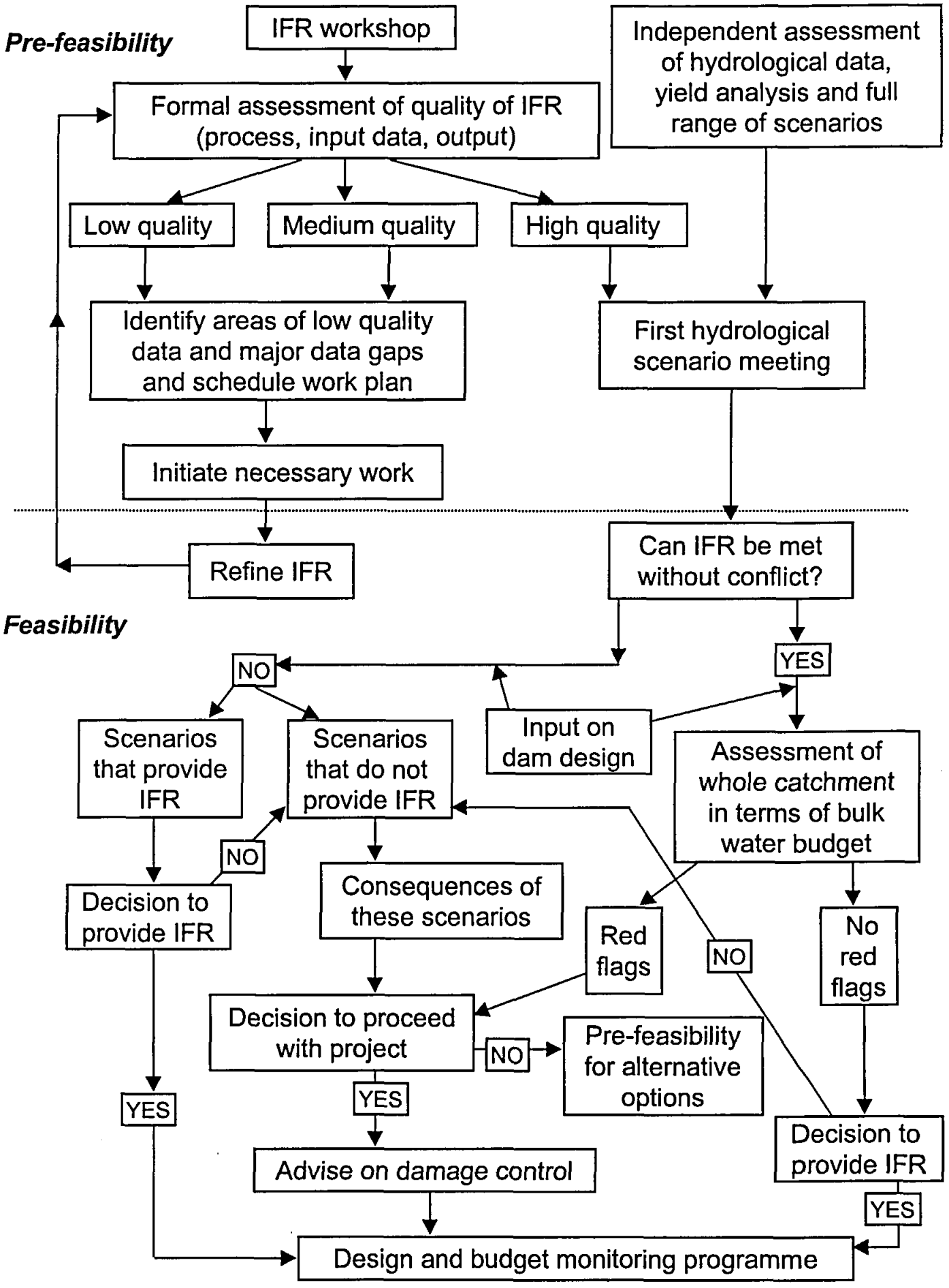


Figure 6.1 **Suggested flow path for Building Block Methodology-related activities during the Department of Water Affairs and Forestry's Feasibility Phase.**

7. INTRODUCTION TO THE STUDY OF ECOLOGICALLY RELEVANT LOW FLOWS, STUDY OBJECTIVES AND APPROACH

Written by R. Tharme

7.1 INTRODUCTION
7.1.1 Conceptual background
7.1.2 Instream flow requirements of rivers
7.1.3 Requirements within instream flow methodologies for information on low flows
7.1.4 The ecological importance of dry-season baseflows and the effects of natural and unnatural low flows on rivers
7.2 STUDY OBJECTIVES
7.2.1 Specific study objectives
7.3 STUDY APPROACH
7.3.1 Hydrological gauging weir survey
7.3.2 Conclusions

7.1 INTRODUCTION

7.1.1 Conceptual background

One of the dominant organising factors in riverine ecosystems is disturbance, including physical disturbance. In such ecosystems, disturbance is defined by Resh *et al.* (1988) as "any relatively discrete event in time that is characterised by a frequency (rate of occurrence of event), intensity (physical force of the event per time), and severity outside a predictable range, and that disrupts ecosystem, community, or population structure and changes resources or the physical environment". In recent years, there has been increased recognition of the importance of both natural and unnatural disturbances as determinants of the structure and function of riverine ecosystems, and of their role in shaping patterns and processes in rivers (Pickett and White 1985; Resh *et al.* 1988; Grimm and Fisher 1989; Stanley and Fisher 1992). Water-resource developments, including dams, diversion weirs and run-of-river abstraction, are responsible for a number of physical disturbances to riverine ecosystems, perhaps the obvious and most critical of which is a change in the natural flow regime. Although, in terms of hydrological events as disturbances, much of the emphasis has previously been placed on floods, attention is increasingly being paid to low flows.

Changes in the hydrological regime, such as prolonged unnatural reductions in baseflow, during all stages of the annual hydrological cycle, have been identified as potential forms of physical disturbance at a number of spatial and temporal scales (Ward 1982; Resh *et al.* 1988). However, this only applies when the flow conditions are outside the range of flows to which the various components of the riverine ecosystem have become adapted over time. Predictability can be considered of central importance in addressing disturbance

in river systems in that certain events may fall within the realm of "natural disturbance", hereby representing regular endogenous features of the riverine ecosystem rather than anthropogenic disturbances.

7.1.2 Instream flow requirements of rivers

Worldwide recognition of the critical need to establish the extent to which the flow regime of a river can be altered from the natural condition, while still maintaining the integrity, or an acceptable level of degradation, of the riverine ecosystem, has provided the impetus for the development of the concept of *instream flow requirements for rivers. Instream flows are those flows required for maintenance of a river's* structure and functioning. The determination of an instream flow requirement for a river requires a thorough knowledge of the flow-related conditions governing various aspects of riverine ecosystem structure and functioning. An understanding of the flow-related conditions that the instream biota is naturally subjected to is also essential. Moreover, some information on the potential impacts of altered flow patterns on such aspects of the system is necessary. This knowledge base provides an indication of which flows within the river's hydrological regime are perhaps more relevant from an ecological perspective than others for maintenance of the riverine ecosystem.

The development and application of formal methodologies for prescribing the IFRs of rivers is a rapidly developing field, both overseas and locally. Tharme (1996, 1997c) reviews the worldwide evolution of the field of instream flow assessment. Of the available instream flow methodologies, internationally, the North American Instream Flow Incremental Methodology (IFIM) is currently purported to be the most sophisticated and scientifically defensible methodology (Gore and Nestler 1988). As such, it is the most commonly used methodology worldwide (Reiser *et al.* 1989; Tharme 1996). However, a comprehensive evaluation of IFIM by King and Tharme (1994), for potential application of the methodology in South Africa, revealed several limitations. These deficiencies included the requirement for large databases, the absence of an ecologically predictive component, and the limited links that it allows between the historical hydrological regime of a river and the responses of the instream biota. This latter limitation is particularly significant in that, as a result, the methodology provides little indication of which regulated flows represent extremely high degrees of physical disturbance that are likely to perturb the riverine ecosystem. As documented in King and Tharme (1994) alternative methodologies to IFIM were sought. This resulted in the consolidation of the BBM as a formal and structured instream flow methodology for local application (Chapters 2-6).

7.1.3 Requirements within instream flow methodologies for information on low flows

In the process of compiling a recommended modified hydrological regime for a river, by means of an instream flow methodology such as the BBM (Chapters 2-6), a detailed assessment needs to be made of the baseflow (as defined in Section 2.2.1) requirements of the river of concern. The baseflow component of the flow regime, therefore, needs to be addressed for both the wet and dry seasons, for any river that is going to be subjected to a water-resource development. Of interest for the purposes of the present study (Chapters 7-11), were the potential impacts of water-resource developments on dry-season baseflows, that is on natural flow conditions during the low-flow season. Any activities that result in marked reductions in dry-season baseflow are likely to result in significant effects on a river, as the dry season is when water is typically most needed by the ecosystem. The extent to which the baseflow can be reduced during the dry season,

without undue damage to the riverine ecosystem, thus needs to be determined in such instances, within the IFR process.

In order to address the low flow requirements of rivers in any structured way, or to further knowledge about dry-season baseflows, it is essential to first have some understanding of the kinds of functions they perform in riverine ecosystems, and of the effects that both natural and unnatural low flows have on the systems. A synopsis of the natural importance of dry-season baseflows and some of the effects of unnaturally low flows on rivers is provided below (Section 7.1.4). A more detailed account of the topic is given in a literature review of the effects of various low flow conditions on all aspects of riverine ecosystems and their biota (Tharme in prep.).

7.1.4 The ecological importance of dry-season baseflows and the effects of natural and unnatural low flows on rivers

The ecological importance of dry-season baseflows

There appears to be little information in the literature that specifically documents the importance of dry-season baseflows for riverine ecosystems (Tharme in prep.). This is perhaps because many of the roles played by baseflows are intuitively understood and/or addressed on a component-specific basis. However, probably the most important function of dry-season baseflow is that it defines whether the river is naturally perennial or non-perennial, and all the implications deriving from this fundamental distinction. Its difference from wet-season baseflow is critical in defining the needs for various communities for different seasons. The other vital roles that dry-season baseflows fulfil include the provision of adequate wetted habitat for instream biota, such as invertebrates, macrophytes, algae and fish, that allows their survival during the dry months of the year. The baseflow regime ensures the availability and diversity of suitable physical or hydraulic biotopes as habitat (see Section 9.5.1 for the definition of a biotope and descriptions of the main biotope types). The flows also result in specific ranges of water temperature and chemical conditions, which may act as various biological cues to the biota. Baseflows also provide water to the river banks, via processes of lateral exchange, that enables the maintenance of communities of marginal and riparian vegetation. Dry-season baseflows are additionally important as sources of water for consumption by terrestrial and riparian wildlife. During the dry months, when the baseflows are typically more stable, many fish species initiate and complete their reproductive cycles (Skelton 1993). Moreover, aquatic invertebrates often undergo cycles of rapid development, growth, and in many instances emergence as aerial adults, during the low flow months. The baseflow regime also provides the basic volume of river flow, on top of which surface runoff contributions are able to generate freshes or pulses of higher flow. These freshes are recognised as having important ecological functions, such as flushing poor water quality from pools, triggering fish migrations and spawning, among others.

The effects of natural and unnatural dry-season baseflows on riverine ecosystems, with particular reference to benthic macroinvertebrates

Much of the body of literature on the effects of natural and unnatural low flows on riverine ecosystems is devoted to two main avenues of scientific research. Firstly, a large proportion of information is in the form of phenomenological accounts of the impacts of drought on perennial and temporary rivers, where the latter includes intermittent and episodic rivers (*sensu* Bayly and Williams 1973, cited in Boulton and Suter 1986). The documented drought events may be entirely natural or, as is nowdays more often the case, largely

induced by anthropogenic disturbances such as large scale river regulation. The second set of literature deals with the effects of drying and the development of intermittent conditions, mostly occurring naturally in temporary rivers.

There is currently very little research focusing on the effects of either natural or unnatural low flows, or on cease-to-flow conditions, in perennial rivers during any part of the annual hydrological cycle. The ways in which conditions in these rivers at naturally low flows might be exacerbated by natural droughts is seldom explicitly described. Furthermore, how unnatural low flows or unnatural droughts might be further affecting the situation is seldom documented. In the majority of instances, the terminology that describes the vast array of low flow conditions, ranging from entire drying of the river bed, through flow cessation, to low drought flows, is loosely employed. Moreover, in the literature there do not appear to be any clear or ecologically relevant definitions of what constitutes a natural or unnatural low flow event. Yet, research on responses of riverine ecosystems to both kinds of event is recognised as highly important, particularly as it is often difficult to distinguish low-level anthropogenic perturbations from normal variations imposed by long-term natural cycles (Schindler 1987, cited in Ward 1989). Moreover, the effects of natural, and especially, unnatural low flows on riverine ecosystems are highly complex, due to a wide array of factors. These factors include, *inter alia*: the natural inter- and intra-annual hydrological variability and degree of predictability of the flow regime (Poff and Ward 1989); altered water chemistry conditions (Chessman and Robinson 1987); the degree to which various species are inherently able to withstand low flow impacts such as desiccation or stranding (Fisher and Lavoy 1972; Corrarino and Brusven 1983; Perry and Perry 1986; Pollard *et al.* 1996); the nature of long-term and shorter-term instream conditions preceding the low flow impact (Fisher and Grimm 1988; Pollard *et al.* 1996); criteria such as the duration and magnitude of the impact, and the degree of flow intermittency experienced; and the inherent stability of the ecosystem (Meffe and Minckley 1987; Brooks and Boulton 1991).

Many of the kinds of effects mentioned here pertain directly to drought conditions and associated drying of the river bed, but some of these effects can probably be extrapolated, to various degrees, to conditions of extremely low flow. In some instances, the low flows reported on in the literature occurred during the wet season or in naturally temporary systems, while in others, they took place in perennial rivers and/or during the dry season. Most of the studies are limited in that the drought or unnatural flow conditions are not related to long-term, natural low flow patterns. Moreover, the responses of the biota are often not directly linked to concomitant changes in habitat quantity or quality, such as altered proportions of available wetted physical habitat, or physical and chemical conditions. This is despite the fact that such factors may have a direct bearing on biotic response.

Drying of the river bed, and thus in many instances low flow conditions, can change riverine population (Harrison 1966; Resh 1982; Cowx *et al.* 1984); community (Ladle and Bass 1981; Meyerhoff and Lind 1987; cited in Stanley and Fisher 1992; Pollard *et al.* 1996); and ecosystem properties (Hill and Gardner 1987; Stanley and Valett 1992; both cited in Stanley and Fisher 1992). It can change the physical and chemical environment (Williams and Hynes 1977; Chessman and Robinson 1987), and may alter the availability of habitat and resources, and/or the intensity of biotic interactions (Boulton and Lake 1992; Boulton *et al.* 1992). The majority of work on the effects of low flows on the components of riverine ecosystems has centred on the biotic responses of aquatic invertebrates.

The effects of drought on aquatic invertebrates, particularly, have been studied in conditions ranging from reduced discharge (e.g. Moth Iversen *et al.* 1978), to intermittent flow conditions (e.g. Hynes 1958; Larimore *et al.* 1959; Kamler and Riedel 1960; McElravy *et al.* 1989), to complete loss of aquatic habitat with the loss of surface water (e.g. Resh 1982). However, most studies of the response of riverine invertebrates to low flows have been concerned with the effects of cessation of flow or desiccation of the river bed, during droughts (Chessman and Robinson 1987; Pollard *et al.* 1996). In addition, they have tended to focus primarily on behavioural or physiological responses to drying, or have described changes in community structure associated with drying or rewetting (Boulton *et al.* 1992).

A vast array of different types of response to drought have been recorded for aquatic invertebrates. Various authors have reported increases, decreases and stability of populations of different invertebrate species (see Hynes 1958; Larimore *et al.* 1959; Kamler and Reidel 1960; Moth Iverson *et al.* 1978; Extence 1981; Ladle and Bass 1981; Canton *et al.* 1984; Cowx *et al.* 1984; Kownacki 1985; Pollard *et al.* 1996). For instance, some studies have shown that when flow is reduced, but not to the extent that the stream becomes dry or intermittent, the densities of some taxa, such as rheophilic taxa and passive filter feeders decreased. Under the same conditions, these studies have also shown that several other taxa increased in density, such as some filter feeders, detritivores and silt-tolerant organisms (Extence 1981; Pearson 1984, cited in McElravy *et al.* 1989). Drought and reductions in discharge below natural magnitudes have been shown to result in catastrophic decimation of the instream biota of perennial rivers and intermittent rivers, with associated recovery sometimes slow (Canton *et al.* 1984 and references therein; Larimore *et al.* 1959; Moth Iversen *et al.* 1978; Boulton 1988). The various effects of drying can be prolonged, and sections of river may become isolated over time. This isolation results in the concentration of predators (Boulton and Suter 1986), and increased likelihood of stochastic community change through strong biotic interactions within each isolated segment (Fisher and Grimm 1991). Significant reductions in numbers of species are frequently observed in instances where surface flow altogether ceases. Species that burrow into the substratum, that remain in resting stages such as diapausing eggs, or that leave the stream as aerial adults can persist, however, during such dry periods (Williams and Hynes 1977; Moth Iversen *et al.* 1978). The extreme case of complete loss of aquatic habitat, where the entire stream reach becomes dry deep into the substratum, has been shown to have devastating effects on aquatic invertebrates (and other biota), although recolonisation can be fairly rapid (Larimore *et al.* 1959; Gore 1959; cited in McElravy *et al.* 1989).

Other biotic responses to drought have included fish mortalities and changes in species composition and abundance, as a result of low oxygen concentrations, high temperatures and various other environmental factors (*inter alia*, Brochet 1977; Brooker *et al.* 1977; cited in Chessman and Robinson 1987; Chutter and Heath 1993; Pollard *et al.* 1996). Responses of riparian and instream vegetation to low flows appear to be poorly documented in the literature. However, drought conditions have been observed to have severe impacts on riparian and floodplain plant communities, for instance causing major tree mortalities (Birkhead *et al.* 1996).

The diversity of biotic responses has been attributed frequently to the numerous environmental changes associated with low flows and/or drought conditions. Some of these changes are adverse, such as the loss of wetted habitat and a decline in water quality for the biota. Other changes, however, might be considered beneficial, often favouring particular species, and include the accumulation of food resources and reduced scouring by higher flows. Although changes occur in the availability and quality of various instream

habitats with both natural and unnatural reductions in baseflow, few studies report specifically on such changes. Similarly, there is little information on geomorphological changes in response to low flows, even at the level of hydraulic biotopes. Chessman and Robinson (1987) note that the impacts of drought conditions on river water chemistry may be substantial. They cited typical effects such as: increases in total dissolved solids and constituent ions, increased levels of ammonia, elevated biochemical oxygen demand, and decreases in dissolved oxygen and suspended sediments (Anderson and McCall 1968; Anderson *et al.* 1972; Stefan and Combs 1978; Muchmore and Dziegielewski 1983; cited in Chessman and Robinson 1987); and increases or decreases in nitrate levels (Slack 1977; Walling and Foster 1978; Casey and Clarke 1979; Dickson *et al.* 1983; cited in Chessman and Robinson 1987).

7.2 STUDY OBJECTIVES

The low flow study detailed in the following Chapters 8-11 is aimed at investigating the effects of different magnitude dry-season baseflows on rivers. Specifically, it is designed to assist with identifying and categorising low flow events that constitute various degrees of physical disturbance to riverine ecosystems. It was hoped that this would provide some of the information necessary to produce an ecologically-relevant definition of low flow based on the instream physical habitat requirements of the identified target instream biota, in this case benthic macroinvertebrates (hereafter referred to as BMIs).

For the purposes of this study, emphasis was placed entirely on riverine BMIs, as an instream biotic component. Benthic macroinvertebrates were selected as the target instream biota for the low flow study, as they are relatively sessile organisms that are less able to evade environmental impacts than, for instance, fish. As a result of their relatively low mobility, their biotope requirements and various environmental conditions representative of their precise location can be well described. In addition, they are found in greater numbers and are more diverse in assemblage composition than the freshwater fish of the Western Cape, and therefore, have the potential to reflect a wider range of responses to flow reduction.

The measurement of flow-related environmental variables, specifically wetted habitat and physical and chemical variables, at different natural and unnatural dry-season baseflows, combined with information on the suitability of these variables for invertebrate assemblages and their constituent taxa, was expected to facilitate prediction of potential ecological consequences for the benthos of a given minimum discharge.

The knowledge gained on the low flow requirements was to be used to establish a link, if possible, between changes in the low flow hydrological regime and biotic response. This would enable a more structured and scientifically based assessment of the low flow requirements of riverine biota and the possible effects of specified low flow scenarios. It is hoped that this will ultimately be able to be used to further the development of the BBM, or a similar holistic instream flow methodology and, particularly, to contribute to its scientific basis and provide support for its main assumptions (Chapter 3).

7.2.1 Specific study objectives

The main aim of the low flow study is outlined above, while specific objectives were as follows.

- To identify those flow-related physical microhabitat variables that are most important in determining the instream, biotope-specific distributions and abundances of BMIs.

- To determine the tolerance ranges of BMIs to flow-related physical microhabitat variables, and to thereby identify taxa that are particularly sensitive to or tolerant of flow reduction. This objective was identified as potentially requiring further work outside the scope of the present study.

- To identify any characteristic responses over the dry season of BMIs, both to natural baseflow patterns and to baseflows of unnaturally low magnitudes and unnaturally extended durations.

- To establish the relationship both between changes in the proportions of suitable instream biotope areas for BMIs and unnatural low-flow events, and between such habitat changes and invertebrate response.

- To link various low-flow hydrological indices with invertebrate responses as described above, and so to produce a set of ecologically substantiated, hydrological descriptors of low flow events. This was identified as an objective that might require further research in order for it to be met with any success.

As indicated by these primary objectives, it was anticipated that the composition of the benthic macroinvertebrate assemblages would change over the dry season in response to both natural and unnaturally low baseflows. It was hoped that relationships between flow-related habitat and BMI abundances and diversities would be detected, that could then be used, in some way in future, to predict the responses of invertebrates to changes in dry season baseflow regime. Furthermore, it was thought that there would be a range of degrees to which an unnatural low flow constitutes a disturbance to the benthos. This was expected to be a function of several factors, including the composition of the invertebrate assemblages, the physical and chemical character of the river and its hydrological history.

It should be noted that several of the above objectives were noted as probably requiring further analysis beyond the scope of the present report. Consequently, more data were collected during the low flow study than were analysed in the following chapters; these data are indicated in the relevant sections. The additional data are to be presented in a doctoral thesis (Tharme in prep.) which was largely funded by the current WRC project.

7.3 STUDY APPROACH

The activities within the study of ecologically relevant, low flows were divided into three parts. The first of these was a general survey of hydrological gauging weirs located on southwestern Cape rivers, to identify suitable rivers to act as study sites. The survey was required for this low flow study and for the work by Mr J. Hollands (Civil Engineering, University of Stellenbosch) reported on in Chapter 12. Secondly, a pilot

study (Chapter 8) was required to select the most suitable study sites of all those identified, in preparation for a field-based low flow experiment, which formed the final and main phase of the study (Chapters 9-11).

The experimental low-flow field study was the means by which the above objectives (Section 7.2.1) were achieved to various extents. A series of flow reductions of predetermined approximate magnitude and duration were to be applied to the upper reaches of the selected southwestern Cape rivers in order to assess the effects of natural and unnaturally-low dry season baseflows.

The southwestern Cape is a particularly suitable region of South Africa for this study as it is a well-defined hydrological region. It exhibits a marked seasonal distribution of flow, with low flows in summer and high flow in winter (King and Tharme 1994). The invertebrates of upper rivers in this region also generally show clear seasonal changes in assemblage composition from summer to winter (King 1981), and are thus expected to be susceptible to disturbances of the flow regime, such as flow reduction, beyond the ranges of natural variability and predictability.

7.3.1 Hydrological gauging weir survey

A broad scale survey of rivers in the southwestern Cape with hydrological gauging weirs on them was undertaken, in order to identify a suitable core study area and associated study sites for the work on low flows. Rivers for which a historical flow record existed were considered most suitable, as the record would provide an indication of the long-term hydrological disturbance history of the river. Also, it was decided to limit the study to addressing the effects of low flows on the upper reaches of rivers, as a different suite of effects would be expected to occur in lower rivers. Consequently, only mountain-stream and foothill zones of rivers were considered in the survey.

Originally, it was hoped to establish study sites at rivers where point flow abstraction was taking place at a gauged diversion weir. There would then be an unimpacted reach upstream of the diversion weir and a downstream reach that has been subjected to the effects of flow reduction, over a long time period. The historical records for the weir and the offtake system could then be used to determine the quantities of flow abstracted during the summer months, as well as the duration of each phase of abstraction, its timing within the season, and the frequency and magnitude of various abstraction events. As this study and the one described in Chapter 12 were initiated at the same time, the survey included DWAF hydrological gauging stations in similar river reaches that were recording near-natural flow conditions. That is, the weirs were on reaches where there was no known abstraction of water or other form of flow regulation upstream on the river system.

As mentioned above, for both the rivers with diversion weirs and those with gauging weirs in their upper reaches that are recording near-natural flow patterns, the hydrological record of daily discharge provides information on the natural long-term low flow pattern of each river. This hydrological history is crucial in that low flow events that constitute a physical disturbance in one river may be a natural feature of another. Hence, the flow record enables the determination of the natural baseline hydrological variability and predictability of the system, and indicates characteristics of low flow events to which the riverine biotas have been responding in the long-term.

Methods

In January 1994, a list was compiled of all types of hydrological gauging weirs known on rivers throughout the southwestern Cape (DWAF 1990b). This was circulated to DWAF personnel, Western Cape Region, for comment on the location of the weirs, their physical condition and age, accuracy, length and quality of the historical record, the physical character of the river reach on which they were situated, and any other possibly significant information. In addition, the DWAF gauging weirs previously identified in King and Tharme (1994) as recording near-natural flows (by non-homogeneity tests) were noted.

From this preliminary list, a shortlist of 33 weirs was compiled. Mr J. Hollands and this author visited each of these weirs between January and February 1994. Evaluation of the weirs and river reaches immediately upstream and downstream of them was structured according to a set of predefined selection criteria, as listed in Table 7.1. For the purposes of the assessment, reaches were defined as sections of river less than 500 m long, while subreaches were smaller sections of each reach, of about 50-100 m long, that appeared highly similar in general character. Subreaches were equivalent to the locations (two per study site) selected for use in the low-flow experiment (see Chapter 9). The selection criteria were aimed at identifying sites that were suitable in terms of both the objectives of the critical low flow study (Chapter 12) and the one described here. Site selection was, thus, primarily based on criteria aimed at minimising, as far as possible, inter-system biological, geomorphological and hydrological variability. For instance, only mountain stream, mountain stream-foothill transition, and foothill river zones were considered, as these are usually the least anthropogenically impacted, and most diverse and sensitive to disturbance (King *et al.* 1986).

Results

Brief notes were compiled on all the hydrological gauging stations visited during the field survey (Tharme unpubl. data). Only three diversion weirs were located, namely: H2H003 on the Hex River; G2H008 on the Eerste River; and HIH006 on the Breede River. The sites of these weirs are briefly described below, and the main reasons for their lack of suitability for the low flow experiment are given.

Diversion weir H2H003

DWAF hydrological gauging weir H2H003, on the foothill section of the Hex River, De Wet, near Worcester, was visited on 02/02/94. It was considered a highly suitable diversion weir for the study, as it is the only weir visited where not all the summer baseflow was being diverted by the weir. On the day of the site visit, approximately 50% of the discharge at the weir was being diverted into a gated canal. The remaining flow over the weir formed a small, narrow flowing secondary channel on one side of the main channel. Small areas of riffle and run biotopes were observed in this reduced channel, and some sandy patches, while most of the river bed downstream of the weir was dry. Hence, it would have been possible to sample a small area of suitable cobble-bed riffles and run downstream of the weir. A subreach with suitable wetted biotopes existed upstream of the weir. However, the main problem with this site was that flow is only measured in the canal and not over the weir itself. Hence, there is no history of the hydrological character and thus disturbance of the river. A second problem was that the site was moderately to highly anthropogenically disturbed; for instance, the riparian vegetation was primarily alien.

Table 7.1 Selection criteria for sites with Department of Water Affairs and Forestry hydrological gauging weirs, for possible use in the low flow study (Chapters 8-11) and the study of relationships between wetted-perimeter and various hydrological indices (Chapter 12). See Sections 7.3.1 and 9.5.1 for definitions of a subreach and biotopes, respectively.

• Long-term historical flow record: greater than 5 y in length.
• Average daily discharge data: computerised and with the facility for updating the record to include the duration of the low flow experiment; gauge preferably still in operation.
• Accuracy of weir: specifically, suitability for measurement of low flows.
• Structure of the weir: preferably small size; limited upstream extent of effect of the weir pool.
• River zone: mountain stream, mountain stream-foothill transition, or foothill zone of river.
• Degree of anthropogenic disturbance: preferably zero to low disturbance; a high degree of naturalness, e.g. indigenous riparian vegetation.
• Subreach locations and lengths: suitable physical character (see below); preferably one subreach upstream and one downstream of the weir; short distances from subreach to weir, so that the discharge recorded at the weir is representative of conditions in the subreaches; subreaches at least 100 m long.
• Subreach biotopes: riffle, run and pool biotopes.
• Channel form and cross-section shape: natural or near-natural morphology.
• Substratum composition: predominantly cobbles and boulders.
• For diversion weirs: point abstraction at the weir; preferably no abstraction or water-resource developments upstream of the weir; presence of gauge recording abstraction volumes.
• For gauging weirs recording near-natural flow: no abstraction of water or other flow regulation of the river system upstream of the weir.

Diversion weir G2H008

This broad-crested diversion weir on the Eerste River, upstream of Kleinplaas Dam, was visited on 30/01/94. It diverts water for use in the environs of Stellenbosch during the summer months. There was no flow downstream of the weir at the time of the site visit, not even leakage, and drying of the entire river bed for some distance downstream of the weir was observed. The river was, however, flowing well upstream of the weir. The site was considered suitable for a low flow study, but there was a problem in that sampling would have needed to commence at the end of winter, when flows would start to decrease, but water would still be able to pass over the weir. In terms of its physical character, the reach was suitable, being a fairly unimpacted lower mountain stream with indigenous riparian vegetation, cobble bed, and riffle and run biotopes upstream of the weir. An historical flow record of average accuracy exists for the weir (accuracy rating of 2, representing an expected error of measurement of 20-30%), but the offtake pipeline did not appear to be gauged. This factor rendered the site unsuitable, as there is no record of the total flow in the river in the reach.

Diversion weir H1H006

This site on the Breede River at Witbrug was visited on 02/02/94. H1H006 is a diversion weir with an offtake canal to provide irrigation water. The canal is gauged reasonably accurately, but there is no accuracy rating for the weir. All water is abstracted from the river during summer, and there is no flow below the weir, except for slight leakage through the cobble bed. However, this flow only started forming a pool some 40-50 m downstream of the weir at the time of the site visit. Moreover, the entire bed is dry at the site for at least the month of February. Although the reach is suitable in instream character, both upstream and downstream of the weir, being a foothill zone and having cobble riffles and runs, it is lined

with alien riparian vegetation. The site was fairly suitable for a study of low flows, exhibiting a wide range of dry, damp, stagnant and flowing patches of water over cobbles in the downstream reach. However, the sampling area was limited in extent and in timing in terms of the availability of wetted habitat.

Hydrological gauges recording near-natural flows

A total of 13 river reaches were located that had gauging weirs on them that were recording near-natural flow conditions (Tharme unpubl. data), but of these, a subset of only eight fulfilled most or all of the required selection criteria (Table 7.1). The eight candidate rivers and their gauging weir reference numbers are listed in Table 7.2. These sites were visited between 30/01/94 and 02/02/94.

Table 7.2 Eight potential sites for the main low-flow study (see also Chapter 8). The corresponding Department of Water Affairs and Forestry hydrological gauging weirs (DWAF 1990b) are recording near-natural, daily hydrological discharge.

RIVER	DWAF GAUGING WEIR NO.
Zachariashoek	G1H014
Bakkerskloof	G1H018
Elands	H1H017 / H1H033
Molenaars	H1H018
Hex River	H2H006
Langrivier	G2H007
Du Toits	H6H007
Riviersonderend	H6H008

7.3.2 Conclusions

There were no entirely suitable sites on any of the rivers visited during the gauging weir survey at which flow reductions were already taking place, and only three diversion weirs of marginal suitability were found. However, 13 suitable river reaches with DWAF gauging weirs recording near-natural flow conditions were located. A decision was made to change the approach of the low flow study. It was decided that the method with the most potential was to artificially reduce flows at some of the sites with gauging weirs that are recording near-natural conditions, by means of some form of instream diversion technique, such as a temporary diversion weir. As this was likely to be a very difficult task that would be limited in the number of times it could be successfully applied, and as the overall number of samples from such an experiment that could be processed was also limited, it was decided to select only four rivers from the set of eight candidate rivers for the artificial manipulations of discharge.

In order to identify the most similar group of four rivers from the eight, and to gain some understanding of inter-river and intra-river biological variability, a pilot study was undertaken. The objectives of the pilot study, methods of data collection and analysis, and the results obtained, are documented in Chapter 8. A description of the main low flow study then follows in Chapters 9-10, and the overall results of the study are briefly discussed in Chapter 11.

8. LOW FLOW PILOT STUDY

Written by R. Tharme

8.1 INTRODUCTION AND OBJECTIVES
8.2 METHODS OF DATA COLLECTION AND ANALYSIS
8.2.1 Benthic macroinvertebrates
8.2.2 Physical habitat and discharge
8.2.3 Water chemistry
8.2.4 Historical flow records
8.3 RESULTS
8.3.1 Benthic macroinvertebrates
8.3.2 Physical habitat and discharge
8.3.3 Water chemistry
8.4 CONCLUSIONS

8.1 INTRODUCTION AND OBJECTIVES

As indicated in Section 7.3, the main purpose of the pilot study was to identify the four most similar rivers, from the total of eight candidate rivers selected during the gauging weir survey (Section 7.3.1), for use in a field-based low flow experiment. Data collected during the pilot study were principally used to determine the relative similarities in the composition of BMI assemblages, and chemical and physical characteristics, of the eight pilot river reaches. Furthermore, these two data sets were used to assess the degree of similarity between two subreaches (roughly equatable with the experimental control and impact locations to be used in the main study, see Chapter 9), and to provide some indication of the inherent variability of each of the different rivers.

The eight pilot sites are listed in Table 8.1, with their corresponding DWAF hydrological gauging weir reference numbers and locations. A brief description of each of the sites, with particular reference to the selection criteria employed during the weir survey (see Table 7.1), is provided in Table 8.2.

For each of the pilot sites, two subreaches were chosen, in close proximity to each other (less than about 100 m apart). For the pilot study it was assumed that the diversion of flow from the main river channel, during the low flow experiment, would take place at the DWAF gauging weirs, as there were already secure, physical structures in place. The two subreaches had to be located close to the gauging weir, to ensure that the historical hydrological record at the weir pertained to them, and so that the experimental flow reductions could be effective in the case of the locations to be impacted. It was the intention prior to the pilot field trip to identify the future experimental control location as a subreach immediately upstream of the proposed flow diversion point (the gauging weir), and the impact location as a subreach just downstream of the diversion point. However, this was not possible for four of the eight pilot sites. At the Du Toits site, both subreaches had to be located consecutively downstream of the weir. This was due to the merging of a tributary with the

mainstream a short distance upstream of the weir, so that there was too short a reach (less than 50 m) for sampling purposes where the flow at the weir pertained directly to the mainstream. Moreover, the section was unstable in terms of channel morphology. The two subreaches on the Hex River similarly were located downstream of the weir, as the area upstream was physically inadequate for sampling purposes, due to a long pool/run reach. In the case of the Riviersonderend, both subreaches were located upstream of the weir, as the downstream reach was physically unsuitable after only *c.* 20 m, with a change to deep, palmiet-lined channels reflecting the uppermost extent of Theewaterskloof reservoir. At Langrivier, the morphology of the reach downstream of the river was quite different from that upstream, so both subreaches were located above the weir. At the Molenaars, Zacharaiashoek and Elands sites, two suitable subreaches were identified upstream of the gauging weirs, while at the Bakkerskloof site, a very short subreach was identified upstream of the weir, and a second, longer subreach was identified downstream of both the weir and a road culvert.

8.2 METHODS OF DATA COLLECTION AND ANALYSIS

A general description was made of each site and the two subreaches (Table 8.2). The following types of data were then collected or collated (see the relevant sections) for each subreach per site during March 1994, a natural low flow period; collection dates are given in Table 8.3:

- benthic macroinvertebrate samples, only from riffles;
- physical habitat information, including microhabitat variables, wetted channel width and instantaneous discharge;
- water chemistry samples, for analysis in the laboratory;
- historical flow records.

The sampling methods and methods of analysis that were employed in the pilot study are detailed in this section only where they differ from the methods employed in the low flow experiment. Where similar methods were used in both cases, a full description is given in Section 9.5.

8.2.1 Benthic macroinvertebrates

Collection of benthic macroinvertebrate samples

In March 1994, six BMI samples were taken from cobble riffles at each of the eight pilot sites, representing an overall total of 48 samples, all of which were analysed for the present report (Appendix 8.1). Cobble/boulder riffle biotopes were selected as they typically exhibit the highest diversity of all river biotopes (Tharme and King 1994). For each site, three of the BMI samples were taken from riffles in one subreach and three samples from riffles in a second subreach. Benthic macroinvertebrates were sampled quantitatively, using a standard short, *c.* 0.34 m x 0.34 m benthic box sampler with an 80 µm-mesh terminal collecting bottle. The sampler covered a 0.1 m^2 surface area of river bed, and sampling was to a substratum depth of approximately 0.20 m.

Table 8.1 Gauging weir co-ordinates and associated information on the historical flow record, for the eight pilot sites (modified from DWAF 1990b, 1991a, 1994b, 1997b).

PILOT SITE	GAUGING WEIR REFERENCE NUMBER (PLACE NAME)	LATITUDE	LONGITUDE	CATCHMENT AREA ABOVE WEIR (km^2)	MAIN RIVER SYSTEM	START DATE - END DATE HISTORICAL FLOW RECORD	WEIR ACCURACY RATING/ DESCRIPTION
Bakkerskloof Stream	G1H018 (Zachariashoek)	33°49'21"	19°02'50"	3.40	Berg	07/06/1964 - ongoing	no accuracy rating; small weir pool; V-notch weir
Du Toits River	H6H007 (Purgatory Uitspan)	33°56'19"	19°10'17"	46.02	Breede	01/10/1964 – 31/08/92	poor accuracy; rating = 1, expected error 30-50%; broad-crested weir
Elands River	❖ H1H017 (old weir) (Hawequas Bos. Res.) ❖ H1H033 (new weir) 168 m upstream of H1H017	33°44'00	19°06'54"	61.20	Breede	❖ 01/03/1969 – 30/04/1991 ❖ 30/04/1991 – ongoing; data cross-correlated with H1H017 by DWAF	❖ moderate accuracy; rating = 2-3, expected error 20-30%/10-20% ❖ high accuracy
Hex River	H2H006 (Glen Heatlie)	33°34'39"	19°30'12"	703.0	Breede	02/09/1980 - ongoing	no accuracy rating; upstream of H2H003, weir pool backs up into long pool/run section
Langrivier	G2H007 (Jonkershoek)	33°59'13"	18°58'13"	2.50	Berg	01/10/1942 - ongoing	no accuracy rating, but probably high as well-maintained; V-notch for low flows and sharp-crest for higher flows
Molenaars River	H1H018 (Hawequas Bos. Res.)	33°43'24"	19°10'13"	109.85	Breede	27/02/1969 - ongoing	moderate accuracy; rating = 3, expected error 10-20%;
Riviersonderend River	H6H008 (Swarte Water Nuweberg Bos. Res.)	34°03'44"	19°04'23"	38.57	Breede	01/10/1964 – 30/06/1992	very good accuracy; rating = 4, expected error 5-10%; broad-crested weir
Zachariashoek River	G1H014 (Zachariashoek)	33°49'39"	19°02'08"	2.80	Berg	07/06/1964 - ongoing	no accuracy rating; V-notch weir; small, shallow weir pool

Table 8.2 Physical description of the eight pilot sites and the two subreaches within each site.

STUDY SITE	BIOLOGICAL RIVER ZONE	DEGREE OF ANTHROPOGENIC DISTURBANCE IN CATCHMENT AND AT SITE	PHYSICAL CHARACTER OF SUBREACHES
Bakkerskloof Stream	mountain stream	very low, primarily fynbos; pine plantation starts *ca* 50 m downstream of the second subreach, which has 20% *Pinus*; moderately less disturbed site than Zachariashoek	well-developed indigenous riparian fynbos, occasional pine tree in downstream subreach; boulder/bedrock substratum with some cobble; riffles, pools and few runs; narrow channel that rapidly becomes steep and bedrock-dominated upstream; upstream subreach very short, less suitable in this respect than G1H014; downstream subreach with more open canopy, but similar channel width to G1H014, downstream of road culvert pipe inflow; some instream *Isolepis* and marginal stands of *Prionium serratum*
Du Toits River	upper foothill, transitional to mountain stream	very low; primarily fynbos; large, natural wetland system downstream of site, in lower valley; some water abstraction near source	well-developed indigenous riparian fynbos; open canopy; primarily cobble bed; long sloping riffles and short runs, some interspersed riffle and run patches, few marginal or instream pools; occasional stands of *P. serratum*; some algae, predominantly red alga spp.; small wetland seeps
Elands River	lower mountain stream, transitional to foothill	moderately low; primarily fynbos; trout farm and some water abstraction further upstream; hiking trail and use of river for recreation and trout fishing	well-developed indigenous riparian vegetation; open canopy; cobble/boulder bed; patches of riffle and runs, some bars of riffle, occasional pools; occasional *P. serratum* and *Isolepis*
Hex River	middle to lower foothill	fairly high; mostly karroid vegetation, little indigenous fynbos	riparian vegetation primarily alien; cobble bed dominant, riffle and run biotopes; moderate amount of green algae
Langrivier	mountain stream	near-pristine; mountain fynbos; intensively sampled as long-term biological monitoring site	entirely indigenous riparian vegetation; boulder/cobble bed; step riffles, small cascades, pools and patches of run; high heterogeneity of biotopes; largely closed canopy
Molenaars River	foothill	moderate; trout farms further upstream; N1 road and bridge construction activities; some indigenous fynbos, *Acacia meamsii* dominant upstream of site	largely indigenous riparian vegetation; cobble/boulder bed; bar and slope riffles, runs, several pools largely upstream of weir; open canopy; *P. serratum* abundant
Riviersonderend River	mountain stream	very low; above pine plantation; scattered pines on mountain slopes; primarily fynbos	well-developed indigenous riparian vegetation; open canopy, cobble/boulder bed with some bedrock; riffles, runs and several pools; green algae present; abundant *Isolepis* and occasional *P. serratum*
Zachariashoek River	mountain stream	15-20 m upstream of start of pine plantation; moderately disturbed immediately downstream of weir; mountain fynbos	well-developed indigenous riparian fynbos; partly closed canopy; bed of gravel, cobble, boulders and bedrock, some sand; narrow step riffles, runs; marginal stands of *P. serratum*; little *Isolepis*

Field preservation and laboratory techniques were identical to those outlined in Section 9.5. For all samples, taxonomic identification was to the level of family or higher, and the numbers of individuals were counted.

Analysis of benthic macroinvertebrate samples

Non-parametric, multivariate methods of hierarchical, agglomerative classification and non-metric, multi-dimensional scaling ordination of the BMI samples (Appendix 8.1), performed for the pilot study using PRIMER Version 3.1b, are detailed in Section 9.5. For the analyses, the BMI data were analysed at three levels of resolution.

1) At the first level of resolution, the average numbers of individuals per family, of all six samples per site, were used to indicate the overall degree of similarity between the eight rivers.
2) At the second level of resolution, the average number of individuals per family of the three samples, from each of the two subreaches, was used to identify the degree of within-reach similarity in composition of faunal assemblages.
3) Thirdly, all six samples from each site were analysed independently, to provide a first indication of the natural biological variability in assemblage composition of each river.

8.2.2 Physical habitat and discharge

The following physical microhabitat data were recorded at the precise location where each individual BMI sample was taken:

- depth;
- average (0.6-depth from surface), 0.2-depth, 0.8-depth, and near-bottom velocities;
- substratum description: particle sizes, and proportions of both dominant and subdominant particles; surface heterogeneity of the river bed; general particle shape; substratum layering and sublayer composition; embeddedness; compactness;
- Statzner hemisphere indices of near-bed turbulence (see Statzner *et al.* 1988);
- percentages of overhead and instream cover;
- presence and species composition of algae.

Specific techniques for the collection of each type of physical microhabitat data are documented in Section 9.5. Forty-eight sets of data were collected, which were not analysed for the present report but will form part of the work reported on in Tharme (in prep.).

Depth and average velocity were also recorded at a series of points along four cross-sections established at each site to measure instantaneous discharge (dry-season baseflow) for this study (Table 8.3) and for the research described in Chapter 12. The field methods and equations for the calculation of discharge are given in Section 9.5.

The cross-section data were also used to provide an indication of typical wetted channel widths for each site during conditions of dry-season baseflow (Table 8.3), and hence to provide further information on river size.

8.2.3 Water chemistry

The following water chemistry variables were measured in each subreach of each site, on the same days that BMI samples were taken; sampling dates are indicated in Table 8.4:

- instantaneous, maximum and minimum daily water temperature;
- pH;
- conductivity;
- dissolved oxygen (DO);
- turbidity;
- major cations, e.g. calcium, and anions, e.g. sulphate;
- nutrients: nitrates, nitrites, phosphate and ammonia;
- total dissolved salts (TDS);
- total suspended solids (TSS).

A total of 48 sets of water temperature and chemistry data were collected (Table 8.4). Methods of collection are as per Section 9.5.

The temperature and chemistry data were visually assessed for differences. This level of examination was found to be sufficient to indicate which rivers were most suitable and which were entirely unsuitable for use in the low flow study.

8.2.4 Historical flow records

Historical daily flow records for seven of the eight gauging weirs were downloaded from DWAF, Pretoria, while data for Langrivier were obtained from Jonkershoek Forestry Research Station (FORESTEK). All these data were used by Mr J. Hollands (Chapter 12) and historical flow records for four of the sites were used by this author for the main part of the low flow study (see Section 9.3).

8.3 RESULTS

8.3.1 Benthic macroinvertebrates

The raw family-level BMI data are summarised in Appendix 8.1.

Results of level (1) analysis

The degree of similarity of all eight pilot sites is illustrated in Figures 8.1 and 8.2. The dendrogram (Figure 8.1) indicates that the Elands and Riviersonderend sites are most similar at 79.2% Bray-Curtis similarity, while the Molenaars, Elands and Riviersonderend rivers grouped at 75.2% similarity. Hence, the four most similar rivers are the Molenaars, Elands, Riviersonderend and Du Toits at 73.4% similarity. Zachariashoek, Langrivier and Bakkerskloof formed a separate group at 68.7% similarity of their BMI assemblages, with Zachariashoek and Bakkerskloof being most similar (74.0%). These three rivers grouped with the other four at a fairly low degree of similarity (66.3%). The Hex River split off from the other seven rivers at only 51.0% similarity in BMI assemblages.

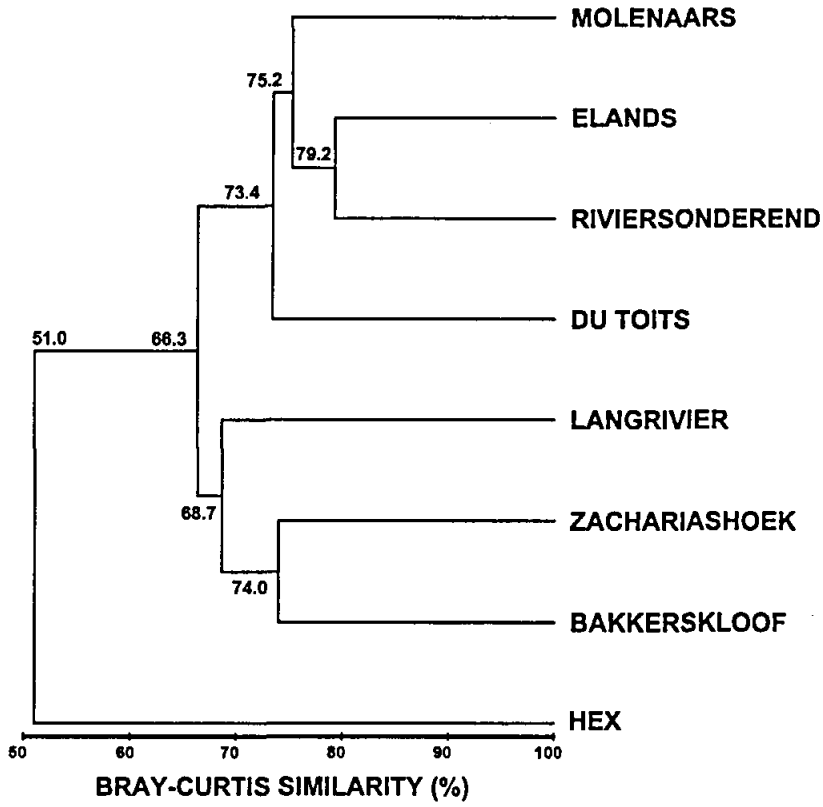


Figure 8.1 Dendrogram showing hierarchical agglomerative classification, with group-average linking, of the eight pilot sites, based on average sample abundances (n = 6 samples) for all families of benthic macroinvertebrates per 0.1 m^2 of riffle.

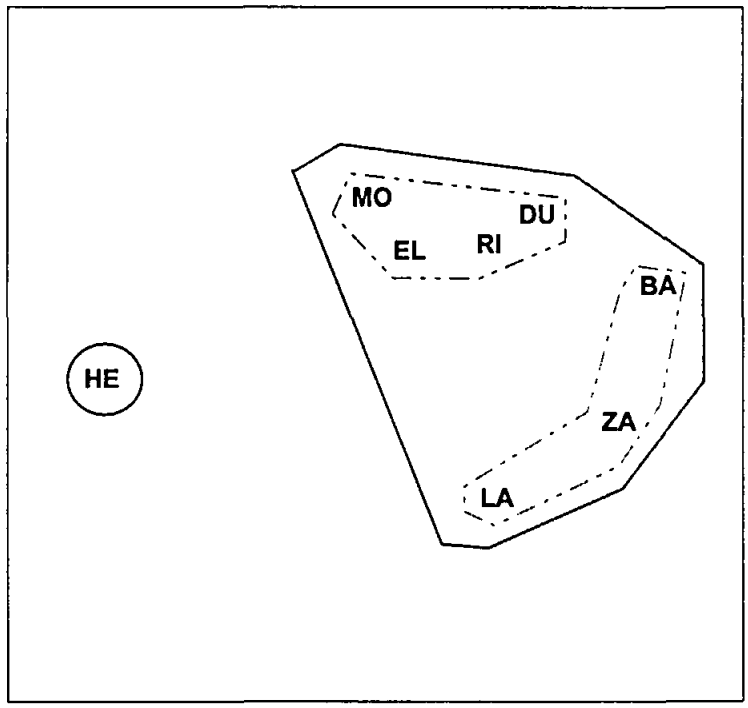


Figure 8.2 Ordination, using non-metric MDS, of the eight pilot sites, based on the same similarity matrix as Figure 8.1. Average sample abundances (n = 6 samples) for all families of benthic macroinvertebrates per 0.1 m^2 of riffle, were used. Stress = 0.04. Dendrogram clusters representing 51.0% similarity (solid line) and 66.3% similarity (dotted line) have been superimposed.

The main groupings from the dendrogram were supported by the ordination (Figure 8.2), at a stress level of 0.04. This low stress value, according to Clarke and Warwick (1994), indicates that the ordination is an excellent representation of the data, with no prospect of misinterpretation.

Results of level (2) analysis

Classification aimed at addressing variation between two subreaches within the same reach at each site, showed that each set of two subreaches grouped together (Figure 8.3). This result indicated that within-reach differences were lower than between-river differences in the composition of riffle benthos. The two subreaches of the Molenaars River were most similar at 88.8% similarity, followed by the Elands (87.6%), Du Toits (86.9%), and Riviersonderend (85.3%). The Elands and Molenaars grouped together at 76.0% similarity, while the Du Toits and Riviersonderend grouped together at 77.2%. Hence, at this level of analysis, the latter two rivers are marginally more similar than the former two. All four rivers grouped at 70.4% similarity in BMI assemblage composition. Zachariashoek, Langrivier and Bakkerskloof sites again formed a separate group at 65.7% similarity, with Zachariashoek and Bakkerskloof again being most similar (69.7%). These three rivers grouped with the other four at a fairly low degree of similarity (63.1%). The Hex River split off from all the other rivers at a low degree of assemblage similarity (45.6%).

The main groupings evident from the dendrogram were supported by the corresponding ordination, at a stress level of 0.08 (Figure 8.4). This stress level reflects a good ordination with no real prospect of a misleading interpretation, and no need for higher-dimensional solutions (Clarke and Warwick 1994). The Hex River has a clearly different BMI assemblage from the other rivers. However, the separation between the remaining seven rivers was less apparent. Consistency between subreaches within a reach ranged from highest to lowest in the following order: Molenaars, Elands, Du Toits, Zachariashoek, Riviersonderend, Langrivier, Bakkerskloof, and Hex. The four most similar rivers, the Elands, Riviersonderend, Molenaars and Du Toits, all had inter-reach similarities in BMI fauna greater than 85%.

Results of level (3) analysis

The results of the most detailed level of analysis provided an indication of the inherent biological variability of each river, as all six riffle samples per river were considered independently (Figures 8.5 and 8.6). As envisaged, percentage similarities were lower than for the level (1) and (2) analyses. The resultant dendrogram showed that the Elands, Molenaars, Riviersonderend, and Du Toits rivers formed a single group at 63% faunal similarity (Figure 8.5). The Elands samples exhibited the least variability in assemblage composition, with all six samples grouping at 77% similarity. Moreover, samples from the two subreaches were intermixed, indicating that the river reach was highly homogenous in terms of benthic composition. Similarly, all the Molenaars samples grouped at a high 72% similarity. The Elands grouped with the Molenaars at 70% faunal similarity, reflecting the fact that the Elands is a tributary of the Molenaars River. The Du Toits and Riviersonderend Rivers formed a group at a lower similarity of 64%. However, unlike the Molenaars and Elands which retained their own faunal identities, the Riviersonderend site was shown to be extremely similar, in terms of benthic composition and within-reach variability, to the Du Toits River (Figure 8.5). This is perhaps to be expected as the Du Toits River is (at least pre-Theewaterskloof Dam), a tributary of the Riviersonderend. The Bakkerskloof, Zachariashoek and Langrivier sites grouped at only 59% similarity (Figure 8.5). Bakkerskloof and Zachariashoek samples showed a high degree of similarity in benthic composition and degree of variability. In contrast, Langrivier retained its indentity, with a similarity

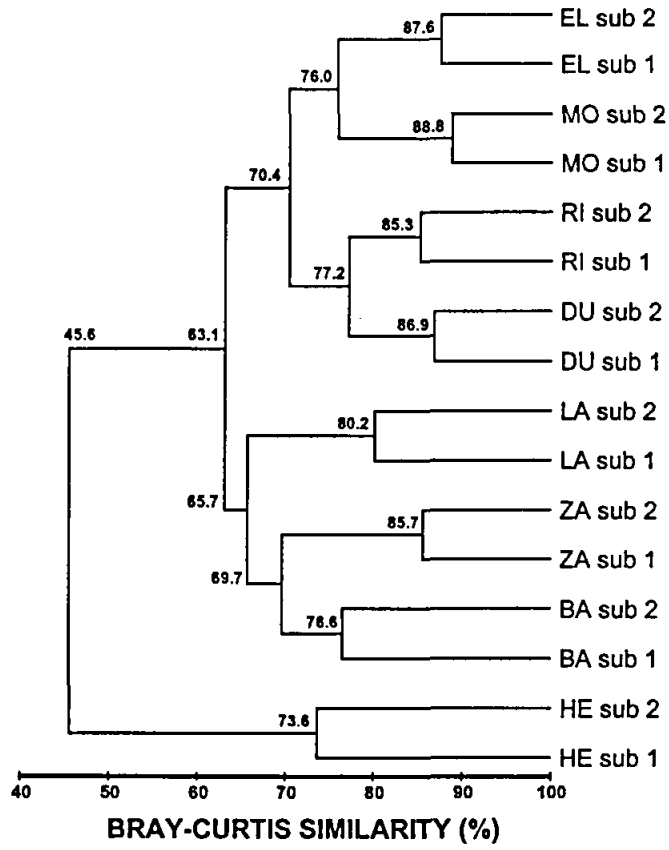


Figure 8.3 Dendrogram showing hierarchical agglomerative classification, with group-average linking, of the eight pilot sites, based on average sample abundances for each subreach (sub1 or sub2) (n = 3 samples) for all families of benthic macroinvertebrates per 0.1 m^2 of riffle.

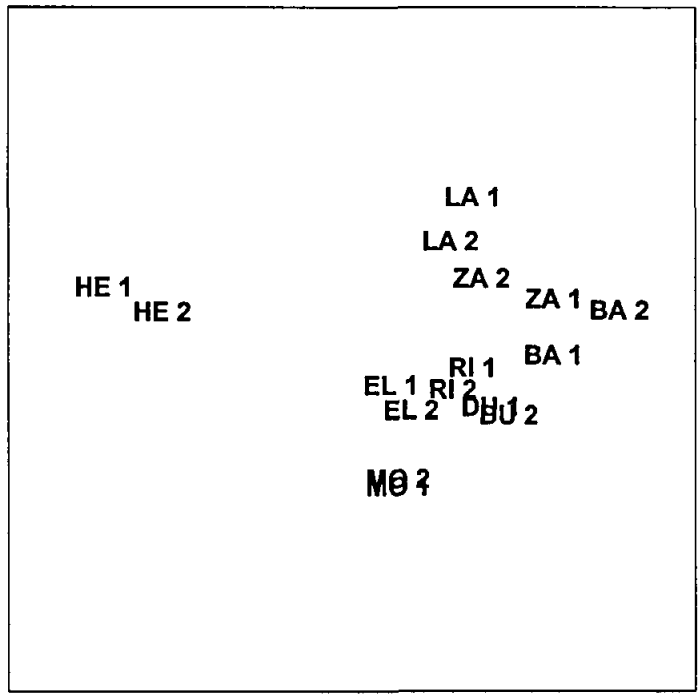


Figure 8.4 Ordination, using non-metric MDS, of the eight pilot sites, based on the same similarity matrix as Figure 8.3. Average sample abundances for each subreach (1 or 2) (n = 3 samples) for all families of benthic macroinvertebrates per 0.1 m^2 of riffle, were used. Stress = 0.08.

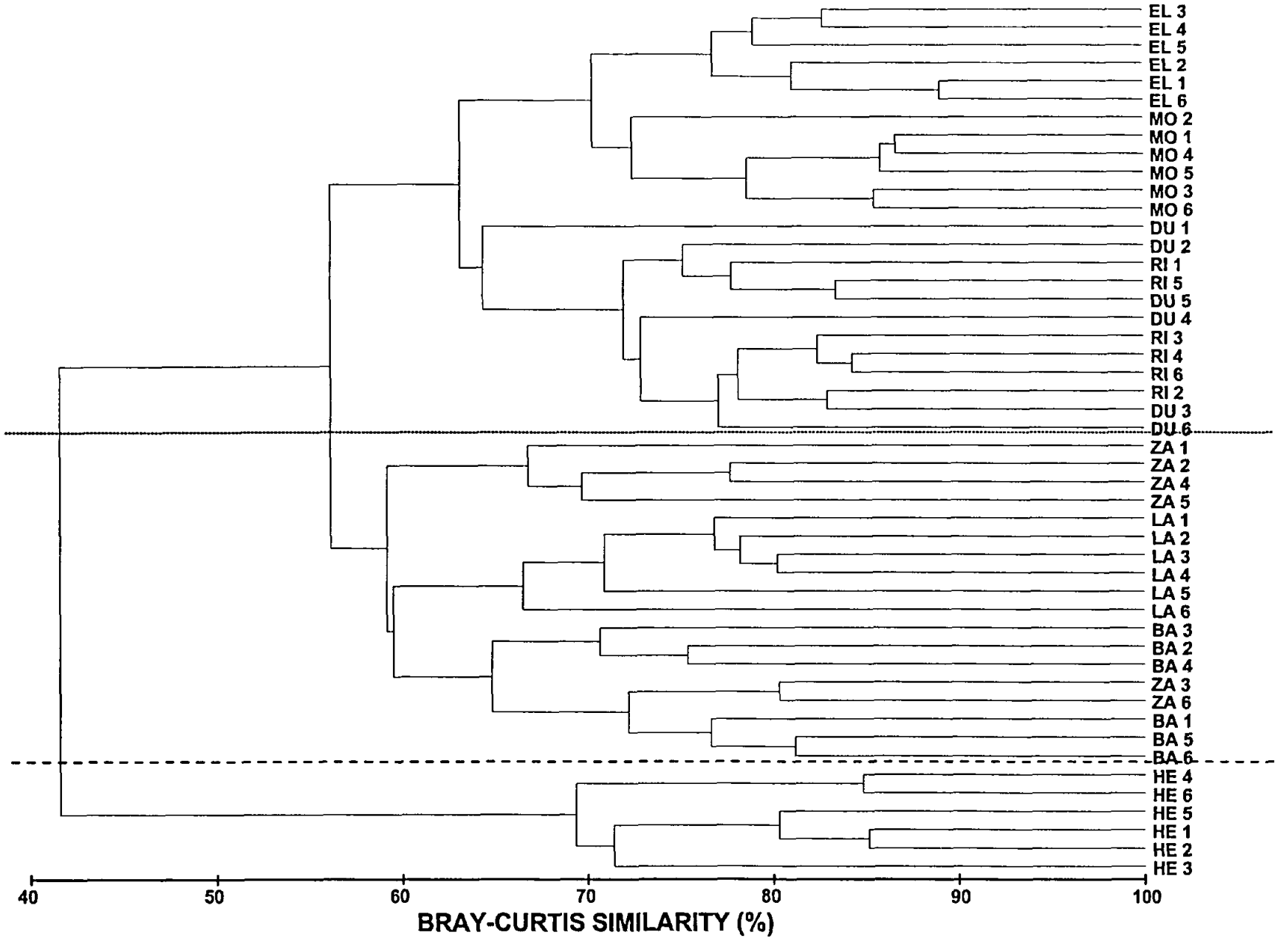


Figure 8.5 Dendrogram showing hierarchical agglomerative classification, with group-average linking, of all 48 benthic macroinvertebrate samples collected in the March 1994 pilot study. The samples represent family-level abundances, per 0.1 m^2 of riffle. The two main divisions into groups, at 42% similarity (dashed line) and 56% similarity (dotted line), are demarcated.

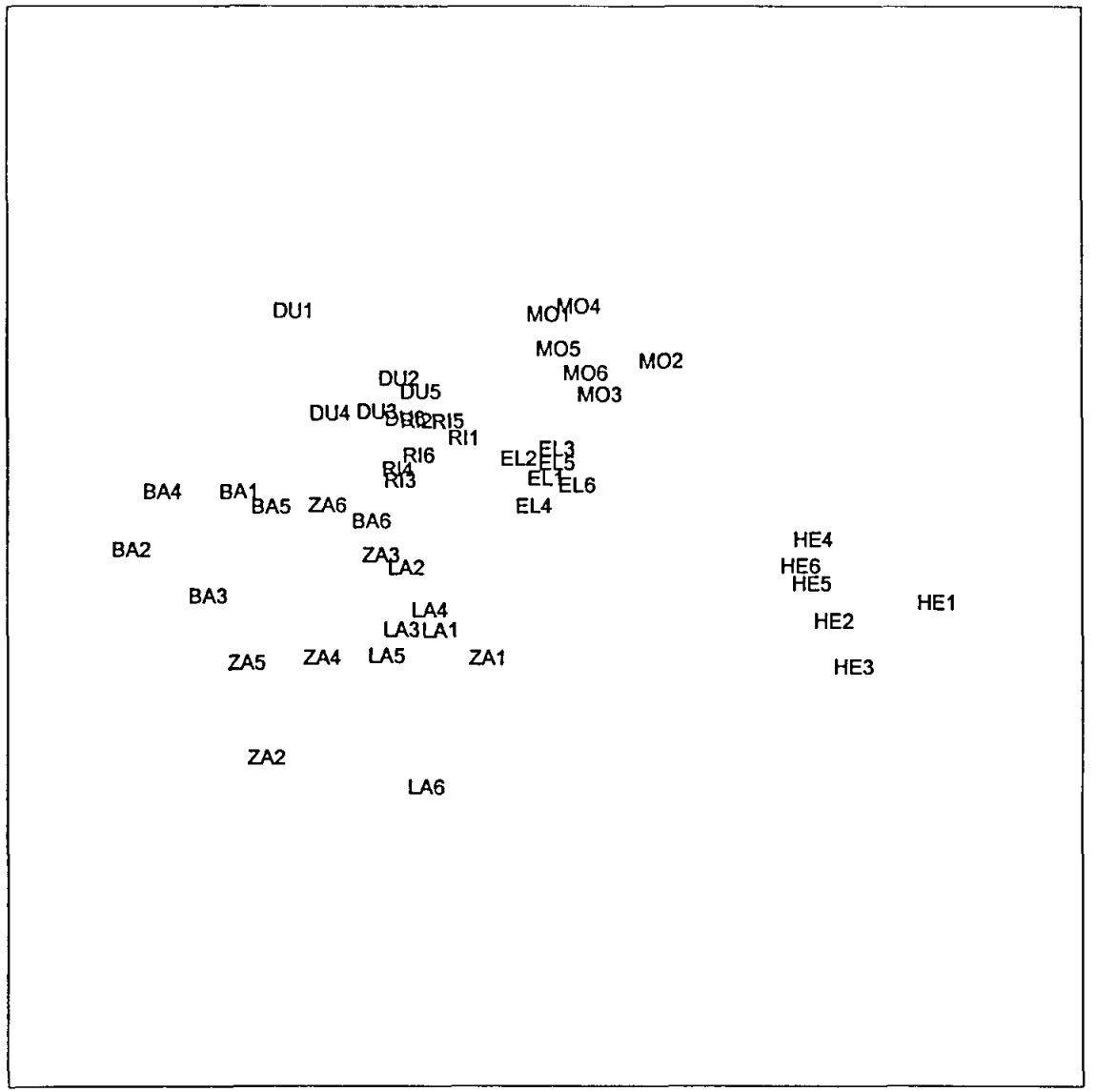


Figure 8.6 **Ordination, using non-metric MDS, of all 48 benthic macroinvertebrate samples collected in the March 1994 pilot study. The plot is based on the same similarity matrix as Figure 8.5. Samples represent family-level abundances, per 0.1 m^2 of riffle. Stress = 0.14.**

among all six samples of 67%. The Hex River again differed in terms of BMI composition from the other rivers, splitting off at a very low similarity of 42%. However, the assemblage composition of the Hex was fairly consistent among samples (69% similarity).

The ordination (Figure 8.6) tended to support the results shown in the dendrogram, although the stress value (0.14) indicated that although the plot is useful, it should only be used in conjunction with the classification results (Clarke and Warwick 1994). The Hex River remained distinctly separate from the other rivers in the ordination. The Molenaars and Elands groupings were retained, while that of Langrivier appeared less distinct than in the dendrogram. The low degree of separation of Riviersonderend and Du Toits samples indicated strong similarities between the sets of samples from these two rivers. Variability was highest among the Langrivier, Bakkerskloof, and Zachariashoek sites (Figure 8.6), as also indicated in the dendrogram (Figure 8.5).

Number of taxa and relative abundances of benthic macroinvertebrates

The numbers of taxa per sample typically ranged from a minimum of 12 (Bakkerskloof) to a maximum of 28 (Molenaars) (Appendix 8.1), for all rivers except the Hex. At the latter site, numbers of taxa per sample were low, with a maximum of only 11 families or higher taxa (Appendix 8.1).

Comparison of the total numbers of individuals per sample (Appendix 8.1) revealed that the three smallest rivers, Langrivier, Bakkerskloof, and Zachariashoek, exhibited lower abundances than the remaining moderately-sized rivers (Table 8.3). The Molenaars showed the highest abundances of all rivers, particularly for the families Chironomidae, Baetidae and Simuliidae (Appendix 8.1).

8.3.2 Physical habitat and discharge

In terms of general physical habitat, all eight sites provided suitable types of wetted habitat, with cobble/boulder riffles and high hydraulic diversity. However, surface areas available for sampling differed markedly among sites of various sizes (Tables 8.2 and 8.3).

Comparisons of calculated wetted widths for the eight pilot sites (Table 8.3) revealed that the Molenaars is widest, followed closely by the Hex River. In contrast, the Langrivier, Bakkerskloof, and Zachariashoek rivers are very narrow at the sites (1.3-2.8 m), reducing the area of wetted habitat available for repeated large scale BMI sampling.

Similar, relatively large, dry-season baseflows were recorded for the larger Hex, Molenaars and Elands rivers. Interestingly, the instantaneous discharge of the Elands was greater, though marginally so, than that of the Molenaars River downstream. Natural baseflows in the three smallest rivers were similarly very low, from 0.001-0.006 $m^3 s^{-1}$ (Table 8.3). Although flow diversion would have been simpler at the smaller sites, measurement of very low baseflows would have proved difficult.

Table 8.3 Instantaneous discharges and wetted channel widths for the eight pilot sites, for the March 1994 sampling trip. *Although a higher discharge was recorded for the Elands than the Molenaars River, this is atypical as average daily discharges are higher in the latter river.

PILOT SITE	DATE	INSTANTANEOUS DISCHARGE ($m^3 s^{-1}$)	WETTED CHANNEL WIDTH (m)
Bakkerskloof Stream	26/03/94	0.005	1.3
Du Toits River	08/03/94	0.266	7.4
Elands River	11/03/94	*0.682	11.6
Hex River	15/03/94	0.635	15.8
Langrivier	22/03/94	0.001	2.8
Molenaars River	10/03/94	*0.651	17.2
Riviersonderend River	24/03/94	0.491	7.7
Zachariashoek River	17/03/94	0.006	1.3

8.3.3 Water chemistry

The results of the chemical analyses are presented in Table 8.4. For most variables, especially conductivity (and TDS), major ions and total alkalinity, the Hex River showed considerably higher values than the other seven rivers. The Elands exhibited the highest phosphate concentrations of all eight sites, and along with the Hex and Molenaars sites, had high dry-season nitrate levels. These sites were the most anthropogenically altered ones, albeit still representing fairly natural conditions (Table 8.2). The seven rivers, excluding the Hex, thus grouped closely in terms of temperature and all chemical variables except nutrients. From a solely physicochemical perspective, these seven sites were sufficiently similar that they could be used for the low flow study.

8.4 CONCLUSIONS

Summarising the above sets of results, from biological and chemical perspectives, it was concluded that the Hex River is unsuitable for the low flow field experiment, as it is too dissimilar from the other rivers. This is probably a result of its geographic location, which results in a combination of fynbos and karroid vegetation, and a characteristic biota and chemistry. It is also the most anthropogenically impacted site. The most suitable and internally similar grouping of rivers is that of the Molenaars, Du Toits, Elands and Riviersonderend rivers, all tributaries contributing flow to the Breede River system. Within this group, the Elands and Molenaars formed a subgroup, as did the other two rivers.

In terms of river zonation, biological, chemical and physical differences between the mountain-stream and foothill zones did not preclude the use of sites representing different zones (Table 8.2), because biologically (at least at the family level) they were very similar. It was sufficient that the four sites were located on the upper reaches of rivers and had relatively low levels of anthropogenic disturbance. The elevated nutrient levels at two of the sites were, however, noted. Also, it was unfortunate that flow gauging was not ongoing at two of the sites.

Table 8.4 Summary of water chemistry conditions at the eight pilot sites, March 1994. SUB1 – subreach 1; SUB2 – subreach 2.

	SITE AND SUBREACHES											
	DU TOITS			MOLENAARS			ELANDS			HEX		
VARIABLE	SUB1	SUB2	MEAN	SUB1	SUB2	MEAN	SUB1	SUB2	MEAN	SUB1	SUB2	MEAN
sampling date	08/03/94	08/03/94	08/03/94	10/03/94	10/03/94	10/03/94	11/03/94	11/03/94	11/03/94	15/03/94	15/03/94	15/03/94
minimum temperature (°C)	21.0	21.0	21.0	21.0	21.0	21.0	19.0	19.0	19.0	22.0	22.0	22.0
maximum temperature (°C)	27.0	27.0	27.0	21.0	21.0	21.0	24.0	24.0	24.0	29.0	29.0	29.0
instantaneous temperature (°C)	22.6	20.9	21.8	20.9	20.3	20.6	22.9	19.0	21.0	25.1	23.8	24.5
pH (pH units)	5.8	5.6	5.7	6.7	7.0	6.8	5.8	5.7	5.7	7.1	7.1	7.1
conductivity (mS m^{-1})	3.36	3.21	3.29	2.81	2.83	2.82	2.57	2.61	2.59	13.45	14.05	13.75
oxygen (mg ℓ^{-1})	8.2	9.2	8.7	8.4	8.7	8.6	7.8	7.9	7.9	8.2	7.8	8.0
oxygen (% saturation)	94.9	103.4	99.1	94.4	96.8	95.6	91.1	85.3	88.2	99.5	92.9	96.2
TSS (mg ℓ^{-1})	0.00010	0.00005	0.00008	0.00120	0.00090	0.00105	0.00050	0.00010	0.00030	0.00090	0.00080	0.00085
TDS (mg ℓ^{-1})	21.11	21.77	21.44	29.32	26.22	27.77	25.73	22.03	23.88	90.52	90.40	90.46
sodium Na^+ (mmol ℓ^{-1})	0.198	0.203	0.201	0.169	0.175	0.172	0.150	0.168	0.159	0.547	0.547	0.547
potassium K^+ (mmol ℓ^{-1})	0.008	0.005	0.006	0.018	0.021	0.019	0.010	0.012	0.011	0.054	0.054	0.054
magnesium Mg^{2+} (mmol ℓ^{-1})	0.046	0.049	0.048	0.036	0.036	0.036	0.037	0.037	0.037	0.301	0.305	0.303
calcium Ca^{2+} (mmol ℓ^{-1})	0.015	0.018	0.017	0.040	0.039	0.039	0.027	0.021	0.024	0.320	0.313	0.317
total alkalinity HCO_3^- (meq ℓ^{-1})	0.060	0.070	0.065	0.090	0.100	0.095	0.070	0.070	0.070	0.210	0.280	0.245
chloride Cl^- (mmol ℓ^{-1})	0.249	0.263	0.256	0.157	0.155	0.156	0.168	0.169	0.169	0.690	0.691	0.690
sulphate SO_4^{2-} (mmol ℓ^{-1})	0.019	0.018	0.019	0.016	0.016	0.016	0.014	0.014	0.014	0.464	0.477	0.471
phosphate PO_4-P (µmol ℓ^{-1})	0.105	0.080	0.093	0.900	0.945	0.923	2.210	2.270	2.240	0.275	0.200	0.238
nitrate NO_3-N (µmol ℓ^{-1})	0.435	0.500	0.468	15.450	16.320	15.885	24.285	22.720	23.503	36.585	36.925	36.755
nitrite NO_2-N (µmol ℓ^{-1})	0.085	0.040	0.063	0.200	0.220	0.210	0.180	0.200	0.190	0.820	0.860	0.840
ammonium NH_4-N (mmol ℓ^{-1})*	n.d.	0.000	n.d.	n.d.	n.d.	n.d.	0.000	0.000	0.000	0.000	0.000	0.000
silicon SiO_2-Si (µmol ℓ^{-1})	1.060	0.670	0.865	6.100	6.850	6.475	1.670	1.590	1.630	91.000	48.750	69.875
turbidity (visual assessment)	clear	clear	clear	clear	clear	clear	clear	clear	clear	clear	clear	clear

*n.d. - not detectable; limit of detection for NH_4-N = 0.1 mg ℓ^{-1} or 0.007 mol ℓ^{-1}

Table 8.4 Continued.

	SITE AND SUBREACHES											
	ZACHARIASHOEK			LANGRIVIER			RIVIERSONDEREND			BAKKERSKLOOF		
VARIABLE	SUB1	SUB2	MEAN	SUB1	SUB2	MEAN	SUB1	SUB2	MEAN	SUB1	SUB2	MEAN
sampling date	17/03/94	17/03/94	17/03/94	22/03/94	22/03/94	22/03/94	24/03/94	24/03/94	24/03/94	26/03/94	26/03/94	26/03/94
minimum temperature (°C)	19.0	19.0	19.0	16.7	16.7	16.7	19.0	19.0	19.0	19.0	19.0	19.0
maximum temperature (°C)	22.0	22.0	22.0	17.0	17.0	17.0	21.0	21.0	21.0	21.0	21.0	21.0
instantaneous temperature (°C)	21.3	19.5	20.4	16.4	16.3	16.4	20.6	19.1	19.9	19.8	18.7	19.3
pH (pH units)	5.0	5.0	5.0	6.0	6.2	6.1	4.5	4.6	4.5	4.5	4.7	4.6
conductivity (mS m^{-1})	4.68	4.48	4.58	2.96	2.92	2.94	2.87	3.07	2.97	3.47	3.17	3.32
oxygen (mg ℓ^{-1})	8.3	7.7	8.0	8.5	7.2	7.9	8.9	8.7	8.8	7.4	5.8	6.6
oxygen (% saturation)	94.2	83.4	88.8	87.2	73.8	80.5	98.4	93.4	95.9	81.6	62.0	71.8
TSS (mg ℓ^{-1})	0.00005	0.00060	0.00033	0.00010	0.00005	0.00008	0.00090	0.00005	0.00048	0.00080	0.00120	0.00100
TDS (mg ℓ^{-1})	27.86	34.26	31.06	24.33	24.63	24.48	20.28	24.33	22.31	23.64	26.67	25.16
sodium Na^+ (mmol ℓ^{-1})	0.293	0.266	0.280	0.203	0.204	0.204	0.158	0.165	0.161	0.231	0.232	0.232
potassium K^+ (mmol ℓ^{-1})	0.004	0.005	0.005	0.011	0.011	0.011	0.003	0.007	0.005	0.005	0.006	0.006
magnesium Mg^{2+} (mmol ℓ^{-1})	0.070	0.058	0.064	0.038	0.039	0.038	0.041	0.042	0.041	0.051	0.051	0.051
calcium Ca^{2+} (mmol ℓ^{-1})	0.014	0.010	0.012	0.021	0.021	0.021	0.010	0.009	0.009	0.014	0.015	0.014
total alkalinity HCO_3^- (meq ℓ^{-1})	0.060	0.060	0.060	0.110	0.100	0.105	0.070	0.060	0.065	0.050	0.060	0.055
chloride Cl^- (mmol ℓ^{-1})	0.393	0.400	0.396	0.199	0.194	0.196	0.207	0.196	0.201	0.298	0.297	0.297
sulphate SO_4^{2-} (mmol ℓ^{-1})	0.019	0.019	0.019	0.013	0.012	0.013	0.015	0.014	0.015	0.010	0.009	0.009
phosphate PO_4-P (µmol ℓ^{-1})	0.050	0.020	0.035	0.150	0.165	0.158	0.665	0.670	0.668	0.125	0.140	0.133
nitrate NO_3-N (µmol ℓ^{-1})	4.290	4.480	4.385	1.460	1.360	1.410	0.180	0.090	0.135	0.195	0.335	0.265
nitrite NO_2-N (µmol ℓ^{-1})	0.030	0.080	0.055	0.110	0.165	0.138	0.285	0.235	0.260	0.165	0.165	0.165
ammonium NH_4-N (mmol ℓ^{-1})*	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
silicon SiO_2-Si (µmol ℓ^{-1})	0.720	2.080	1.400	5.365	5.420	5.393	0.420	0.630	0.525	1.560	2.080	1.820
turbidity (visual assessment)	clear	clear	clear	clear	clear	clear	clear	clear	clear	clear	clear	clear

*n.d. - not detectable; limit of detection for NH_4-N = 0.1 mg ℓ^{-1} or 0.007 mol ℓ^{-1}

Hence, on the basis of the results of the pilot study, the Du Toits, Elands, Molenaars and Riviersonderend rivers were selected as the sites for the low flow, field experiment (Chapters 9-11).

9. THE LOCATION, EXPERIMENTAL DESIGN AND PROCEDURES FOR IMPLEMENTATION OF THE LOW-FLOW STUDY

Written by R. Tharme

9.1 INTRODUCTION TO THE STUDY AREA
9.1.1 Topography
9.1.2 Climate and hydrology
9.1.3 Geology and geomorphology
9.1.4 Vegetation
9.1.5 Water chemistry
9.1.6 Riverine biota

9.2 GENERAL DESCRIPTION OF THE STUDY RIVERS AND STUDY SITES
9.2.1 Elands River and study site
9.2.2 Molenaars River and study site
9.2.3 Du Toits River and study site
9.2.4 Riviersonderend River and study site

9.3 HYDROLOGICAL CHARACTERISATION OF THE STUDY SITES
9.3.1 Availability of historical flow data and methods of analysis
9.3.2 General features of the hydrological regime

9.4 EXPERIMENTAL DESIGN AND PROCEDURES FOR FIELD IMPLEMENTATION
9.4.1 Experimental design
9.4.2 The structure of the temporary flow diversion weirs
9.4.3 The nature of the experimental flow reductions

9.5 METHODS OF DATA COLLECTION AND ANALYSIS
9.5.1 Methods of data collection and analysis for physical biotopes and discharge
9.5.2 Methods of data collection and analysis for water chemistry
9.5.3 Methods of data collection and analysis for benthic macroinvertebrates
9.5.4 Methods of data collection and analysis for physical microhabitat

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The study area and specific sites for the low-flow field experiment are introduced in this Chapter, in Sections 9.1 and 9.2, respectively. Section 9.3 provides a general background to the hydrological character of each of the study rivers, on the basis of the historical flow record for each study site. The design of the low-flow study and the steps for its implementation as a field experiment are described in Section 9.4. Methods of data collection and analysis form the final section of the Chapter. In this last section, some of the data that are referred to, were collected in addition to the data discussed in this report, as part of a doctoral thesis (hereafter referred to as Tharme in prep.).

9.1 INTRODUCTION TO THE STUDY AREA

The study area is located within the southwestern Cape, and the general features of this part of the Western Cape Province of South Africa are briefly described below. The study area encompasses all four catchments of the rivers identified as study sites in Chapter 8, namely the catchments of the Elands, Molenaars, Du Toits and Riviersonderend Rivers (Figure 9.1).

9.1.1 Topography

Topographically, most of the southwestern Cape is less than 500 m in altitude (Davies and Day 1998), and the lower river valleys are flat. The source areas and upper catchments of the major river systems, however, occur in mountainous terrain at altitudes up to about 2000 m a.m.s.l. (Figure 9.1; Everett and Quibell 1995). The broad terrain patterns, ranging from plains to mountains, correspond with recognised geomorphic provinces (Section 9.1.3), and comprise Coast Forelands and the Cape Fold Region (ENPAT97 Broad Terrain Patterns Map, DEAT 1997).

9.1.2 Climate and hydrology

Although South Africa is recognised as a semi-arid country, the southwestern Cape has a Mediterranean-type climate with cold, wet winters, and moderate to high precipitation (Davies and Day 1998). The study area falls within the winter rainfall region identified by Dyer (1976, cited in Davies *et al.* 1993), with roughly 80% of the precipitation falling between the months of April and September (DWAF 1994). Within the core study area (that is, the area including the catchments upstream of the four study sites), the mountains influence precipitation considerably. Consequently, the Mean annual precipitation (MAP) is high and spatially variable, and can reach a maximum of *c.* 2600 mm a^{-1} in the high lying western part of the core study area, while values in the southeastern coastal part are *c.* 600 mm a^{-1} (Everett and Quibell 1995; DWAF 1994b, 1997b). Winter rainfall results from a combination of orographic effects and cold fronts approaching from the southwest. Summer is the dry season, typically hot and windy. August minimum land temperatures range from 6-17 °C, with cooler temperatures in the mountainous areas (Dent *et al.* 1989). During the dry season, precipitation is usually in the order of only 126-250 mm a^{-1}, with isolated areas receiving 62.5-125 mm a^{-1} (Davies and Day 1998). Inter-annual variability of rainfall is fairly low, tending to range from 15-20% (Davies and Day 1998). The overall range of MAP for the southwestern Cape is from 200-1500 mm a^{-1} (WR90 map, updated by IWQS in 1998; Dent *et al.* 1989). Mean annual evaporation (MAE) (WR90 map, updated by IWQS in 1998; Midgeley *et al.* 1994) is less variable than MAP in the core study area, and ranges from typical values of about 1100 mm a^{-1} to a maximum of 1700 mm a^{-1} (DWAF 1994b, 1997b). There is a pronounced seasonal variation in evaporation in the core study area, with high summer values of 230-250 mm a^{-1}, and low winter values of 40-50 mm a^{-1} (DWAF 1997b).

The study area falls within one of the five limnological regions of southern Africa identified by Allanson *et al.* (1990) on the basis of geomorphological, geochemical and climatological features, namely the temperate, acid waters of the Cape Fold Region (see below). It is also part of Harrison's (1959) "Cape system" hydrobiological region. On the basis of regional hydrology, the Berg River and southwestern Cape regions identified by King *et al.* (1979) together contribute 4.0% to the total MAR of South Africa, while

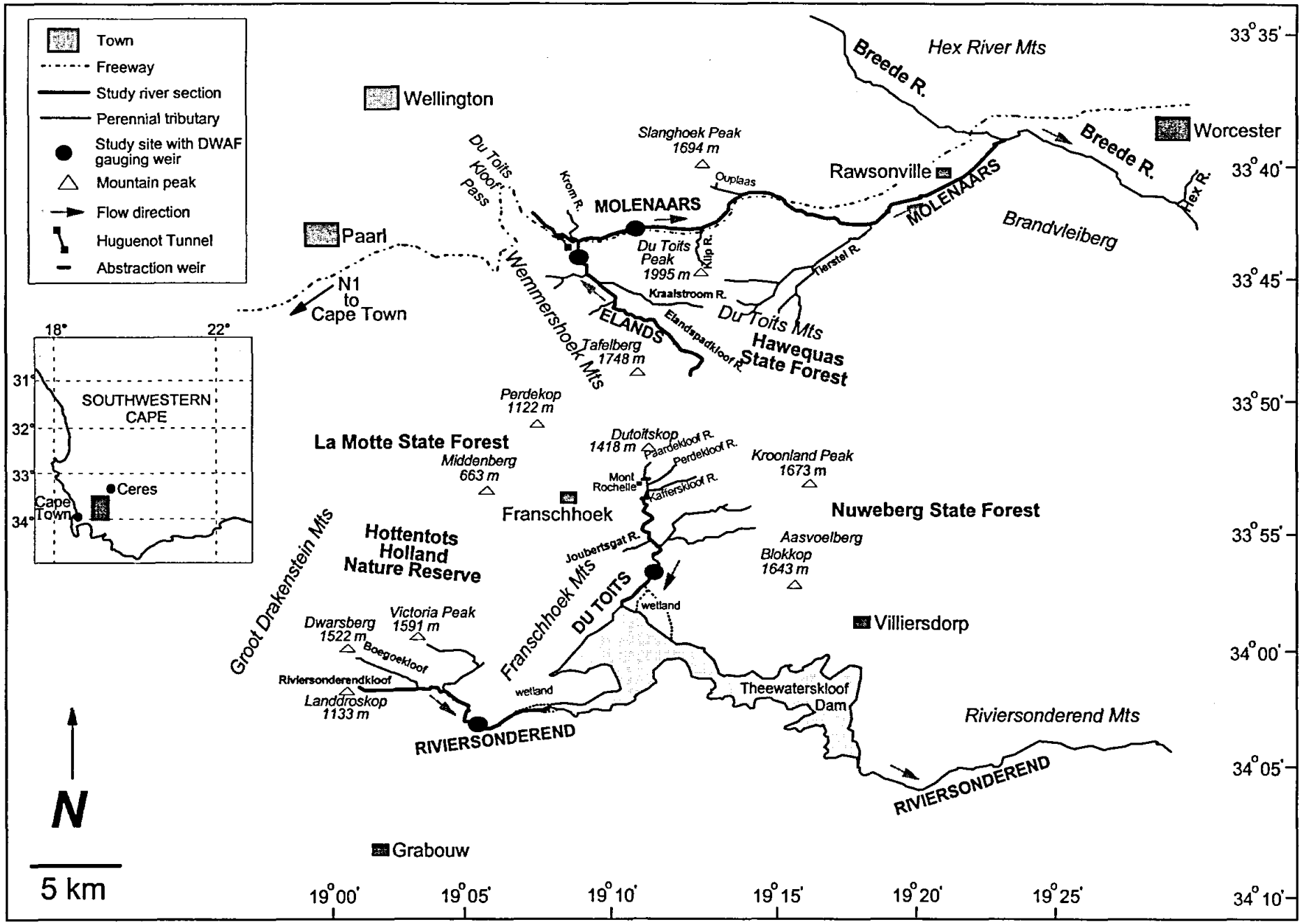


Figure 9.1 **Study area indicating the locations of the study sites on the Elands, Molenaars, Du Toits and Riviersonderend Rivers, southwestern Cape (modified from 1: 250 000 topographical sheets 3318 Cape Town 5th ed. and 3319 Worcester 3rd ed.).**

the Breede River region contributes a further 4.2%. The MAR ranges from about 40 mm to as high as 1250 mm (Midgeley *et al.* 1994).

Southern African rivers, including those of the southwestern Cape, are recognised as having a relatively high degree of flow variability and a low predictability of flow by international standards (Gordon *et al.* 1992; Davies *et al.* 1993). In a hydrological classification of South African rivers based on near-natural, long-term daily flow data, drainage region H which represents the Breede River catchment and includes all four study sites, has a calculated rainfall: runoff ratio of 20.2, with a coefficient of variation (CV, dimensionless) of precipitation of 0.20-0.25 and a CV of runoff of 0.26-0.72 (King and Tharme 1994).

On the basis of an analysis of seasonal flow patterns, the study area is included within the geographically distinct winter peak flow region of South Africa, defined in King and Tharme (1994). The 'winter moderate' group of gauging rivers within the region, which includes all four study rivers, typically exhibits highest monthly volumes during July and August.

Further information on the basic hydrological character of the study rivers is given in Section 9.3.

9.1.3 Geology and geomorphology

The study area largely falls within the Cape Folded Belt Geomorphic Province, which is bordered either side by the Southern Coast Geomorphic Province (IWQS map, updated in 1997; King 1963). Geologically, the southwestern Cape is dominated by the sedimentary strata of the Cape Supergroup, namely the Table Mountain Group or series, TMS), Bokkeveld Group and Witteberg Group, in order of decreasing age (Lambrechts 1979; Moon and Dardis 1992). These strata overly the older Cape Granite Suite (550 Ma). There are also localised areas of Quaternary sediments (Moon and Dardis 1992). The mountains of the study area typically comprise resistant, well-leached TMS sandstones and quartzites (500-320 Ma) (Lambrechts 1979) and some younger sediments of the Karoo Supergroup (265-190 Ma; Davies and Day 1998). Erosivity is predominantly low to medium, with very few areas of high potential erosion (ENPAT 97 map, DEAT 1997).

Although the fluvial geomorphology of the southwestern Cape appears poorly described in the literature, the rivers tend to be short and fairly steep (King *et al.* 1979) with few extensive, long systems. However, all the geomorphological zones identified for South African rivers are usually present (Noble and Hemens 1978).

9.1.4 Vegetation

The study area falls within the Fynbos bioregion (Brown *et al.* 1996). The catchments of the river systems naturally support fynbos, which contributes more than 80% of the species in the Cape Floral Kingdom (Cowling and Richardson 1995). Fynbos comprises fire-adapted, sclerophyllous shrubs and heathlands, and is exceptionally diverse in species, with many endemics. The vegetation comprises four main growth forms, proteoids (family Proteaceae), ericoids (including the dominant family, Ericaceae), restioids (family Restionaceae) and geophytes. Fynbos is uniquely characterised by restioids, and is confined to the nutrient-poor soils of the southwestern and southern Cape (Cowling and Richardson 1995). In the low-lying areas of

the southwestern Cape, outside the focus of the study area, there are some patches of thicket biome (Low and Rebelo 1996).

The steep, upper catchments of rivers in the study area are typically vegetated with mountain fynbos and are relatively undeveloped in terms of land use. However, infestation by alien vegetation is evident in these areas. Middle and lower catchments tend to be increasingly influenced by land uses such as agriculture, mainly in the form of irrigated vineyards and orchards, and afforestation.

Riparian vegetation within the study area predominantly comprises indigenous fynbos, but also includes elements of indigenous Afromontane forest, notably trees such as *Podocarpus*, *Olea* and *Rapanea* (Moll and Scott 1981; Cowling and Richardson 1995; HKS 1988). Dominant alien riparian tree species include *Acacia longifolia* (long-leaved wattle), *A. mearnsii* (black wattle) and *Sesbania punicea.*

Palmiet, *Prionium serratum*, is a common instream and marginal plant species of low-order southwestern Cape rivers and mountain streams (Davies *et al.* 1993). The macrophyte, *Isolepis digitata*, formerly known as *Scirpus digitatus*, also occurs in some of the rivers within the study area. Both of these dominant plants are strongly linked to low pH, particularly in upper catchments (Davies *et al.* 1993). There are no known alien macrophytes in the upper reaches of the study rivers, although a moss species might be present (Dr C. Boucher, Botany Department, University of Stellenbosch pers. comm.). However, some alien species are likely to occur in other parts of the Breede system.

9.1.5 Water chemistry

The rivers within the southwestern Cape are characteristically oligotrophic, as they drain well-leached soils associated with TMS sandstones and quartzites (King *et al.* 1979). The pH also tends to be below neutrality and may be as low as 4.3, due to the nature of the soils and to high concentrations of humic (weak organic) acids released from the surrounding fynbos (Davies *et al.* 1993). The naturally low buffering capacity of rivers in the region is a result of pH being determined largely by this vegetation type (Dallas and Day 1993). Salts of marine origin at lower altitudes variously influence the rivers, with a concomitant increase in buffering capacity to neutral pH (Davies *et al.* 1993). In terms of ionic dominance, the major ions of rivers in the study area tend to be Na^+ and $C\ell^-$ (King *et al.* 1992; Day and King 1995). The rivers of the study area have typically clear waters of low turbidity (and low TSS), and most sediment transportation occurs during the wet season, associated with flood events. Rivers in the region form two main groups, according to Harrison and Agnew (1962). The first group are strongly acidic (pH 5.0-5.9), with unbuffered waters, low TDS and are frequently peat-stained. The second group of rivers exhibits almost colourless waters, and a pH between 6.0 and 6.9.

An assessment by Dallas *et al.* (in press) for the southern and western coast water quality management region (which includes the study area), of the background concentrations of several system variables for least impacted river sites, showed the following trends. Median values for mountain-stream zones for TDS, conductivity, TSS and pH are: 26.3 mg ℓ^{-1}; 3.0 mS m^{-1}; 0.66 mg ℓ^{-1}; and 5.5 pH units, respectively. Similarly, median values for foothills are identified as the following, respectively: 32.0 mg ℓ^{-1}; 3.1 mS m^{-1}; 0.78 mg ℓ^{-1}; and 6.0 pH units. Corresponding concentration ranges are given in Dallas *et al.* (in press). Similar data have yet to be calculated for other chemical variables, such as nutrients.

The specific water chemistry of each site is described in Section 9.2, with particular reference to the concentrations of these different variables during the dry season.

9.1.6 Riverine biota

Aquatic invertebrates

Aquatic invertebrate communities of the southwestern Cape include a high number of species endemic to the typically acidic, humically-stained rivers of the region (Davies *et al.* 1993). Indeed, Harrison and Agnew (1962) described the Cape endemic fauna as being limited to the acidic streams and rivers of the western and southern Cape. The stony-bed communities, especially of low-order streams and rivers, are dominated by insects, both the aquatic larval stages of terrestrial species (e.g. Trichoptera) and aquatic adults of insects (e.g. Coleoptera). Indeed, insects typically constitute some 99% by number of the entire invertebrate assemblage of upper rivers (for example, King 1981). In terms of taxonomic orders of insects, Diptera tend to dominate, with Chironomidae and Simuliidae normally always present (Davies *et al.* 1993). These taxa are followed by Ephemeroptera, then Plecoptera, Coleoptera, Trichoptera, Odonata and Megaloptera. Representatives of non-insect invertebrate taxa, such as crustaceans, flatworms and oligochaetes are usually present in far lower numbers. River zonation into mountain stream, foothill and lower zones is typically reflected in the aquatic invertebrate communities inhabiting them. The mountain-stream zone tends to be inhabited by a slow-growing, insect-dominated community that is not seasonally differentiated, except by developmental stage (King 1981). Foothill communities typically show distinct seasonal changes in species composition and abundance, however, with a roughly four-month summer and eight-month winter community (King 1981). Non-insect taxa tend to increase in dominance in the summer community of lower rivers (King 1981).

Site-specific information on aquatic invertebrate assemblages is detailed in Section 9.2, and with specific reference to flow in Chapter 10.

Fish

The Cape component of the temperate fish fauna comprises only 15 species, which in the southwestern Cape are restricted to the rivers of the Cape Fold Mountains (Skelton 1993). Endemicity is highest in the southwestern Cape and the distinct fauna is dominated by the family Cyprinidae (Skelton 1993). It also includes the majority of threatened indigenous species in South Africa and several species with increasingly limited distributions. The diversity and abundance of fish species in the region is low relative to the rest of the country, with typically only a few indigenous species in a single river. (Skelton 1993). Several alien and translocated species occur in the region, such as trout, and smallmouth and largemouth bass (Skelton 1993).

Site-specific historical information on fish communities is provided in Section 9.2.

9.2 GENERAL DESCRIPTION OF THE STUDY RIVERS AND STUDY SITES

This section provides an overview of the ecological character and present condition of each of the four study rivers and the study sites located on their upper reaches. This information allows comparisons to be made

among the sites in terms of various physical, chemical and biological characteristics. It also provides an indication of the types and degrees of anthropogenic disturbance presently experienced by each river, particularly at the study site. Such information is of potential relevance in understanding any of the responses of various ecosystem components to natural and experimental low flows (Chapters 10 and 11).

9.2.1 Elands River and study site

The location of the study site on the Elands River, some 60 km northeast of Cape Town (1: 50 000 topographical sheet 3319CA Bain's Kloof, 2nd ed. 1979, partially modified from aerial photography 1987) is indicated in Figure 9.1. Site co-ordinates are provided in Table 8.1 and a brief description of the site is given in Table 8.2. Plate 9.1 (see Section 9.4) illustrates the natural character of the site.

Much of the information summarised below is derived from the pilot survey of the Elands River (Chapter 8). Other background information is taken from Ractliffe and Brown (1994) and Brown *et al.* (1995).

Catchment characteristics

The total catchment area of the Elands River, upstream of the study site, is the second largest of the four study sites at *c.* 61.0 km^2 (Table 8.1). The river forms part of the Molenaars system, and hence greater Breede system, which enters the sea at Cape Infanta on the southeast coast. Information on river runoff is provided in Section 9.3.2, where the hydrology of the study site is detailed. The MAP for the Elands subcatchment, which represents a portion of WR90 quaternary catchment H10J, is estimated at 2207 mm (DWAF 1997b). The figure for MAE (S-Pan) is similar to that calculated by NSI (1997b) for the Molenaars site, at 1266 mm.

The Elands (sometimes referred to as the Elandspad River) drains in a northerly direction, bounded on both sides by steep mountains. It receives flow contributions from numerous perennial and seasonal streams draining ranges including the Winterberg, Wemmershoekberge, Du Toitsberge and Tafelberg, with peaks at altitudes ranging from about 1995 m (Du Toits Peak) to 842 m. The main tributary of the Elands River is the mountainous Kraalstroom River, which has is source at about 1820 m, in a high-altitude area with peaks of 1870 m, 1846 m and 1776 m. The third order Elands mainstream is the main tributary of the Molenaars River (Section 9.2.2), feeding into it near the eastern portal of the Huguenot Tunnel (Figure 9.1).

The Elands catchment is largely undeveloped with respect to land use, and there is no urban development. Catchment vegetation appears to be entirely indigenous mountain fynbos in the vicinity of the study site, where it grows on the steep-sided valley slopes. Anthropogenic disturbance is low at the site, although it is the second most disturbed of the four study sites overall (Table 8.2). There are no areas of commercial afforestation or irrigation agriculture in the catchment (DWAF 1997b). Much of the catchment falls within an area protected by CNC and a hiking trail runs alongside the river at the study site. The area has further recreational value for flyfishing. A trout hatchery (and associated farm) is located on the Kraalstroom River, however, which is some 4 km upstream of the Elands study site. The hatchery has a significant detrimental impact on river water quality and the aquatic invertebrate fauna for a considerable distance downstream. Although major negative effects on the ecosystem do not extend into the study site, due to recovery (Brown *et al.* 1995), there is evidence of marked nutrient enrichment at the site (see section on water chemistry). Two DWAF hydrological gauging weirs are located on the Elands River within the

downstream end of the study site, H1H017, which is no longer operational, and a new weir, H1H033 (Table 8.1; Figure 9.1 and Section 9.3.1). Bridges and retaining walls associated with the Huguenot Tunnel are located in the river reach just downstream of the site. Water is diverted from H1H033 to cool the air conditioning system for the Huguenot Tunnel, and is reintroduced into the river further downstream. Presently, there is no other known abstraction of water in the vicinity of the site, although there is upstream abstraction from the trout farm.

Synopsis of background ecological information on the Elands River system

River classification, conservation importance and status

The Eland is a perennial river, and its overall conservation status can be considered high. Ractliffe and Brown (1994) calculated that the Elands in the vicinity of the site had an abiotic conservation status of 83.4% (Class 2), that is, it is largely natural from an abiotic perspective. The calculated biotic conservation status was lower, however, at 69.8% (Class 3). This represents an overall conservation status of 76.6% (Class 3), indicating that the section of river is moderately modified. Although there has been a loss and change of natural habitat and biota, basic ecosystem functioning appears predominantly unchanged.

River geomorphology

In terms of geomorphological zonation at the study site, the river represents the lower extent of the mountain-stream zone and, in general character, appears transitional to a foothill zone (Brown *et al.* 1996; Table 8.2; pers. obs.). The river channel is typically single, fairly narrow and well defined at the site. The bed gradient is moderately steep. The substratum is heterogeneous, dominated by small boulders and large to small cobbles, but with patches of gravel and several bedrock outcrops. Sand is uncommon. Physical biotope heterogeneity and associated hydraulic diversity are high. Dominant biotopes include pools, riffles, runs, transitional biotopes and a few, large backwaters (see Section 9.5.1 for definitions).

Riparian and instream vegetation

The riparian canopy is largely open and comprises entirely indigenous mountain fynbos scrub and small trees (Table 8.2). No alien species are apparent at the site. Instream vegetation is limited to occasional patches of *Isolepis* and *Prionium serratum*.

Although there are no historical data on algal species composition or densities for the river, the presence of primarily green algae has been noted in summer and winter (Ractliffe and Brown 1994; pers. obs.).

Water chemistry

The data obtained during the pilot study for the present project were used to gain an understanding of natural background water chemistry for the Elands River at the study site, during the dry season (March 1994; Table 8.4). Seasonal, long-term water chemistry data are available for weir H1H017 at the study site, although data for nutrients are considered of low reliability (Ractliffe and Brown 1994). Although these data were not analysed for the current report, a summary of the basic trends is provided in DWAF (1991b, cited in Ractliffe and Brown 1994). Historical ranges in chemical variables and temperature, especially for summer conditions, corresponded well with the present-day pilot data discussed below. Further information on natural water chemistry conditions in relation to changes in dry-season baseflows is given in Section 10.3.

In March 1994, daily water temperature ranged from 19.0-24.0 °C at the study site, and was typically about 21 °C. The water was acidic, at an average pH of 5.7, which is just above the median background pH value of 5.5 obtained for least-impacted mountain-stream zones in the southwestern cape (Dallas *et al.* in press). Average total alkalinity (TA) was 0.070 mEq ℓ^{-1}. Conductivity was very low at below 3 mS m^{-1}, and TDS values were correspondingly low at an average of 23.88 mg ℓ^{-1}. Both these figures approximate the median values for least-impacted mountain-stream zones in the region (Dallas *et al.* in press). The concentrations of major cations and anions were all very low, with sodium and chloride characteristically dominant, at 0.159 mmol ℓ^{-1} and 0.169 mmol ℓ^{-1}, respectively. Oxygen levels were moderately high, at 7.9 mg ℓ^{-1} (88.2 % saturation), indicating aerobic waters. Levels of TSS were negligibly low, corresponding with the high water clarity at the site and falling below the median values obtained for least-impacted mountain-stream zones in the region (Dallas *et al.* in press). Nitrate concentrations were relatively high, at an average of 23.503 µmol ℓ^{-1}, indicating mild nutrient enrichment at the site. This enrichment and some impairment of water quality previously have been attributed to the establishment of a trout farm upstream on a tributary of the Elands (Ractliffe and Brown 1994). Nitrites and phosphates were also quite high at 0.190 µmol ℓ^{-1} and 2.240 µmol ℓ^{-1}, respectively. Ammonium was recorded as negligible (below detection limits). The summer concentration of silicon was 1.630 µmol ℓ^{-1}. There was little deposition of inorganic sediment at the site. The surfaces of the substratum were virtually free of algal growth, and only occasional patches of algae were evident during the dry season (pers. obs.).

Results from rapid bioassessment using the South African Scoring System (SASS Version 4; Chutter 1995; Dallas *et al.* 1994) support the finding of high water quality and very good general health of the river at this site (see below).

Riverine biota

Aquatic macroinvertebrates:

The pilot study summer data, at the taxonomic level of family and above, and only for riffles (Appendix 8.1), indicated a total of 30 taxa in the reach of the river represented by the study site. The assemblage was insect-dominated, with nematodes, oligochaetes and planarians as the only non-insect taxa. Of the insect groups, sensitive species indicative of least impacted upper rivers included the following: Helodidae; Athericidae; Leptophlebiidae, Ephemerellidae; Notonemouridae; several cased trichopterans, including Barbarochthonidae and leptocerids; and Corydalidae. Relative abundances of taxa are given in Appendix 8.1.

SASS scores previously recorded for the Elands compare favourably with, and are within the upper values for, SASS scores reported from relatively undisturbed rivers elsewhere in the southwestern Cape (Tharme and Brown 1994; Ractliffe and Brown 1994). Summer SASS results for the Elands site, recorded in Tharme and Brown (1994) and Ractliffe and Brown (1994), were the following, respectively: SASS scores of 232 and 199; a total number of taxa of 30 and 23; and average scores per taxon (ASPTs) of 7.7 and 8.6.

Fish:

The fish recorded in the Molenaars system are listed in Section 9.2.2 Two of the alien species are known to occur in the Elands, namely *Oncorhynchus mykiss* (pers. obs.) and *Micropterus dolomieu* (Ractliffe and Brown 1994).

Existing and proposed water-resource developments

Existing water-resource developments

Presently, the Elands catchment is largely undeveloped in terms of its water resources, and there are no major impoundments on the mainstream or any of the tributaries. According to NSI (DWAF 1997b) data, there are no farm dams in the catchment.

Proposed water-resource developments

A site on the Elands River, near the eastern portal of the Huguenot Tunnel and just upstream of the confluence with the Molenaars River, has been investigated as part of the Western Cape System Analysis (WCSA). There is a proposal to construct a diversion weir, the Klein Drakenstein Diversion, to divert water to the greater Cape Town area (DWAF 1994b).

9.2.2 Molenaars River and study site

The location of the study site on the Molenaars River (1: 50 000 topographical sheet 3319CA Bain's Kloof, 2nd ed. 1979, partially modified from aerial photography 1987) is indicated in Figure 9.1. Site co-ordinates are provided in Table 8.1, and a brief description of the site is given in Table 8.2. The catchment area extends into the area described by 1: 50 000 topographical sheet 3319CC Franschhoek (2nd ed. 1979, partially modified from aerial photography 1987). Plate 9.3 (see Section 9.4) illustrates some of the natural features of the site.

Much of the information summarised below is derived from the pilot assessment of the Molenaars River (Chapter 8). Other background information is taken from Ractliffe and Brown (1994) and Brown *et al.* (1995). A detailed description of the catchment of the lower Du Toit's Kloof Valley is given in HKS (1988).

Catchment characteristics

The catchment area of the Molenaars River upstream of the study site is the largest of the four study sites at about 109.85 km^2 (Table 8.1). It comprises a portion of WR90 quaternary catchment H10J (DWAF 1997b). The river drains from west to east, forming a major tributary of the Breede system. MAP in the part of the catchment immediately upstream of DWAF gauging weir H1H018 is estimated by NSI (DWAF 1997b) as 1851 mm a^{-1}, with an overall MAP for H1H018 and H1H017 (Section 9.2.1) of 2047 mm a^{-1}. The MAE has been calculated as 1266 mm a^{-1} (DWAF 1997b). Runoff at the site is documented in Section 9.3.2 where the hydrological regime is described.

The source of the Molenaars is at the top of the Du Toit's Kloof Pass, in the Klein Drakenstein Mountains, at an altitude above 1000 m a.m.s.l. The river is also fed by perennial and seasonal streams and seeps arising on the southern slopes of the Witteberg. Further drainage contributions to the river are from the northeastern slopes of the Klein Drakensteinberge, the northern DuToitsberge and the southeastern slopes of the Slanghoekberge. The Molenaars flows eastwards, confined to the north and south by mountain ranges with peak heights of 1995 m (Du Toits Peak), 1736 m (The Pinnacles), 1694 m (Slanghoek Peak) and 1384 m (Krom River Dome) to altitudes just above 800 m. The main tributary of the Molenaars is the Elands River (Section 9.2.1), which joins it just east of the Huguenot Tunnel. At the intersection of the Witteberg and Slanghoekberge, at Ouplaas, an alluvial fan, through which the Tierkloof River flows, enters the river

over a distance of about 0.7 km. Once the river leaves the Du Toit's Kloof Valley, it is joined by the Tierstel and Holsloot Rivers, before its confluence with the Breede River at Worcester.

The catchment is vegetated primarily with indigenous fynbos, notably shrubland and woodland plant communities. Less than 25% of it is located within the Hawequas Mountain Catchment Area, where it is largely protected from anthropogenic disturbance. The remaining catchment area is government- or privately-owned land. Anthropogenic disturbance is moderate at the site (Table 8.2), making this site the most disturbed of all four sites. There is no significant urban development in the catchment. The N1 national road, and associated Huguenot Tunnel and bridges, represent the dominant human influence in the area, as the road runs directly alongside the river. There is no irrigation, afforestation or water resource development (see below) in the upper catchment (land use data as at 1993; DWAF 1997b). However, an area of alien riparian vegetation, of a density greater than 25%, occurs downstream of the confluence of the Elands River (DWAF 1997b). During the study, large scale road and bridge construction, the latter crossing the river, was underway to widen the N1, both upstream and downstream of the study site (pers. obs.; Ractliffe and Brown 1994). In addition, an alien vegetation clearance programme by CNC had commenced, upstream of the study site (pers. obs.; Ractliffe and Brown 1994). Two trout hatcheries, De Poort Trout Farm and Devon Trout Farm, are located on the mainstream (Brown *et al.* 1995). The former is located on the mountain-stream zone of the Molenaars, upstream of the study site (the other trout farm is located on the Elands tributary; Section 9.2.1). A hotel, private residences and fruit orchards are located along the river reaches upstream of the study site. Protea farming also takes place in the Du Toit's Kloof Valley, while the lower sections of the river flow through winelands. The area has recreational value, and is used for flyfishing, whitewater kayaking, hiking and rock-climbing (pers. obs.). A DWAF hydrological gauging station, H1H018, is located on the Molenaars River just upstream of the confluence of the Klip River, partway along the Du Toit's Kloof Valley (Figure 9.1; Table 8.1).

Synopsis of background ecological information on the Molenaars River system

River classification, conservation importance and status

The Molenaars is a perennial river. O'Keeffe (1985) has described it as exhibiting substantial alteration of the riverine ecosystem, such as invasion by alien biota and locally severe water pollution. The Molenaars is recognised as having a high conservation importance, while some of its tributaries, notably the Tierkloof Stream, are rated of exceptionally high importance (Ractliffe and Brown 1994). It is a designated Special Standards River (Amendments to the Water Act 1984), which implies a higher level of protection from polluting effluents. Moreover, it has an unusually long foothill zone characterised by an exceptionally high number of riverine wetlands and seeps (Boucher 1988, cited in Ractliffe and Brown 1994). The latter are scarce nowadays in the southwestern Cape. The foothill zone also exhibits an exceptional biodiversity in terms of its aquatic macroinvertebrate fauna (see below).

The overall conservation status of the Molenaars River is considered fairly high, according to Ractliffe and Brown (1994). However, the abiotic conservation status of the section of river including the site was calculated as only 62.4% (Class 3) and the biotic conservation status as 64.6% (Class 3). This means that this section of river is moderately modified. Although there has been a loss and change of natural habitat and biota, basic ecosystem functioning appears predominantly unchanged.

River geomorphology

A longitudinal profile of the river from its source to the end of the Du Toit's Kloof Valley is given in Ractliffe and Brown (1994). The river exhibits a steep gradient from its source for some 5 km, before the Krom and Elands tributaries join it. The river gradient then flattens out for the approximately 14 km of river course through the Du Toit's Kloof Valley, where the site is located. The confluence of the Elands with the mainstream constitutes a geomorphological segment boundary, upstream of which the Molenaars River exhibits physical features typical of a mountain stream (Wadeson 1994, cited in Ractliffe and Brown 1994).

Downstream of its confluence with the Elands River, the Molenaars is a fourth-order river, typical of a southwestern Cape foothill river *sensu* Harrison (1965). From the Elands confluence to the end of the Du Toit's Kloof Valley, the Molenaars River is divided into four distinct geomorphological reaches (Wadeson 1994, cited in Ractliffe and Brown 1994). The study site is located within one of these reaches, within the foothill zone. This entire section of the river, especially from the Huguenot Tunnel to just downstream of the Klip River (Figure 9.1), is relatively broad, with gentle slopes underlain by Cape granite (Carter 1988, cited in Ractliffe and Brown 1994). Further downstream, the valley is deeply incised and narrow, and is characterised by steep, sandstone slopes (Carter 1988, cited in Ractliffe and Brown 1994).

The river channel at the study site is typically single, with occasional multiple channel sections (as defined in Gordon *et al.* 1992). The bed gradient is moderately gentle at about 0.013 (Wadeson 1995). The channel is fairly wide and well defined. The substratum is heterogeneous, dominated by deep, multiple layers of boulders and cobbles, but also with patches of gravel and sand. Several bedrock outcrops occur at the site and there are occasional midchannel cobble bars, some with small amounts of vegetation. Physical biotope heterogeneity and hydraulic diversity are high, and biotopes include pools, riffles, runs, transitional biotope patches and backwaters (see Section 9.5.1 for definitions).

Riparian and instream vegetation

Most tributaries contributing flow in the mountain-stream zone of the Molenaars are close-canopied (Ractliffe and Brown 1994), while others like the Krom and Elands have open canopies. Longitudinal stands of wetland vegetation are associated with the extensive network of seepage lines on the mountain slopes, through which water is additionally contributed to the mainstream (Ractliffe and Brown 1994).

The riparian canopy is largely open at the study site and comprises mostly indigenous mountain fynbos (Table 8.2). Species include Restionaceae, Cyperaceae and small trees like *Salix mucronata*. Alien species have heavily invaded the length of the riparian zone upstream of and at the site, in particular *Acacia mearnsii* (black wattle) and *A. longifolia* (Boucher 1988, cited in Ractliffe and Brown 1994). Marginal and instream stands of indigenous *Prionium serratum* occur at the site.

Water chemistry

The data obtained during the pilot study for the present project were used to gain an understanding of natural background water chemistry for the Molenaars site, during the dry season (March 1994; Table 8.4). Seasonal, long-term water chemistry data have been recorded at DWAF gauging weir, H1H018. Although these data were not examined in detail for the present report, they are summarised in DWAF (1991c, cited in Ractliffe and Brown 1994) and are compared with chemistry data collected by Ractliffe and Brown (1994).

Some basic trends are presented below, in comparison with the 1994 pilot data (Chapter 8). Examination of DWAF data revealed clear seasonal differences in water quality and temperature. DWAF water quality data and additional data collected by Ractliffe and Brown (1994) in summer for sites upstream and immediately downstream of the present study site, compared well with the 1994 data.

The daily water temperature was about 21.0 °C at the time of sampling in March 1994. Winter temperatures are considerably lower. The water was mildly acidic to neutral, at a pH of 6.8. This value exceeded the median background pH of 6.0 obtained for least-impacted foothill zones in the region, but fell within the overall regional range of 4.0-7.2 units (Dallas *et al.* in press). Lower pH values occur in the river in winter, due to increased leaching of organic acids from fynbos (Ractliffe and Brown 1994). Average TA was 0.095 mEq ℓ^{-1}. Conductivity was low at about 3 mS m^{-1}, and TDS values were correspondingly low at an average of 27.77 mg ℓ^{-1}. Both figures are just less than the background median values of 3.1 mS m^{-1} and 32.0 mg ℓ^{-1}, respectively, for least-impacted foothill zones (Dallas *et al.* in press). Conductivities are similar in summer and winter, while TDS concentrations tend to be lower in winter (Ractliffe and Brown 1994). The concentrations of major cations and anions were all low, with sodium and chloride dominant, at 0.172 mmol ℓ^{-1} and 0.156 mmol ℓ^{-1}, respectively. Oxygen levels were very high, at 8.6 mg ℓ^{-1} (95.6% saturation), indicating highly aerobic waters. Ractliffe and Brown (1994) could not detect marked seasonal trends in oxygen concentrations, with high levels recorded year-round. TSS levels were extremely low in March 1994, at 0.001 mg ℓ^{-1}, falling well below the regional background median TSS concentration for least-impacted foothills (Dallas *et al.* in press). The TSS levels correspond with the high water clarity at the site in summer. Ractliffe and Brown (1994) recorded summer TSS ranges within the foothill zone of 0.70-3.05 mg ℓ^{-1}. Turbidity levels typically increase markedly as a result of floods in winter. Summer nitrate concentrations were relatively high, at an average of 15.885 µmol ℓ^{-1}, indicating mild nutrient enrichment at the site (Dallas and Day 1993). Ractliffe and Brown (1994) recorded similar summer values and clearly lower winter maxima. Examination of DWAF data also revealed summer peaks in nutrient concentrations, which have been principally attributed to the effects of upstream trout farming activities (Ractliffe and Brown 1994). March 1994 nitrite and phosphate concentrations were fairly low at 0.210 µmol ℓ^{-1} and 0.923 µmol ℓ^{-1}, respectively. Winter nitrite and phosphate levels are typically lower than summer values (Ractliffe and Brown 1994). Ammonium levels were below detection limits for the pilot study, but a summer range from 1.8-9.1 µmol ℓ^{-1} for the foothill zone was given in Ractliffe and Brown (1994). The summer concentration of silicon was high at 6.475 µmol ℓ^{-1}.

There was some deposition of silt at the site, but the surfaces of the substratum were virtually free of algal growth at the time of sampling in March 1994. Some patches of algae were evident, however, during the dry season (pers. obs.). Although freshwater algal species have not been recorded from the site, species records are available for reaches upstream and downstream of it, and high algal densities, both in summer and winter, have been observed along the foothill zone (Ractliffe and Brown 1994). This provides evidence of biological response to increased availability of nutrients. A further indication that the river is mildly enriched at times is provided by the growth of heterotrophic slimes in slower-flowing areas (Ractliffe and Brown 1994; pers. obs.). Gray (1985) notes that such growths are a response by riverine bacteria and fungal communities to increased availability of dissolved organic carbon.

Further information on natural water chemistry conditions, specifically in relation to baseflow fluctuations, is provided in Section 10.3. From the pilot study results, DWAF records, data from Ractliffe and Brown (1994) and SASS results (see below), it is clear that the river is in good condition in terms of its water quality, although mild nutrient enrichment exists and has been occurring for more than two decades.

Riverine biota

Aquatic macroinvertebrates:

A total of roughly 129 taxa, at the species level and above, has been historically recorded along the Molenaars River (Ractliffe and Brown 1994) and 33 summer invertebrate taxa, at the family level and above, were recorded from the study site during the pilot assessment (Appendix 8.1). Abundances of taxa are given in Appendix 8.1.

On the basis of the various families and species recorded for the river, a characteristic southwestern Cape, foothill aquatic macroinvertebrate community inhabits the river at the site. The assemblage is insect-dominated and particularly diverse, with nematodes, oligochaetes and planarians as the main non-insect taxa. Of the insects, sensitive taxa indicative of least impacted upper rivers included the following: Helodidae; Athericidae; Leptophlebiidae, Ephemerellidae; Notonemouridae; Leptoceridae; Pyralidae and Corydalidae. The invertebrate communities also included a number of previously undescribed species of considerable scientific interest, such as a new, rare Gondwanaland relict trichopteran of the Tribe Leucotrichiini. A provisional species list for various sites within the foothill zone of the river is provided in Ractliffe and Brown (1994). Invertebrate abundances are variable and may reach exceptionally high densities of up to 120 000 individuals m^{-2} (Ractliffe and Brown 1994; this study). Primary consumer productivity has been rated far in excess of rates expected for oligotrophic systems in the southwestern Cape (Ractliffe and Brown 1994). Population explosions of opportunist species are known to occur in summer, probably as a response to warm summer temperatures, low flows, nutrient enrichment and high algal production (Ractliffe and Brown 1994; this study).

SASS scores previously recorded from the river compare favourably with, and are within the upper values for, SASS scores reported from relatively undisturbed rivers elsewhere in the southwestern Cape (Ractliffe and Brown 1994). SASS scores for the foothill zone in summer ranged from 160-196, number of taxa from 20-23, and ASPTs from 7.6-8.7; winter scores were lower. These results indicate that the river at the study site is in very good health and has water of high quality.

Fish:

Eleven fish species are known to inhabit the Molenaars River system (Ractliffe and Brown 1994). Three common, indigenous fish species have been recorded from the Molenaars River, namely: *Galaxias zebratus* (Cape galaxias), *Sandelia capensis* (Cape kurper) and *Anguilla mossambica* (longfin eel) (Ractliffe and Brown 1994). Two Red Data species, *Pseudobarbus burchelli* (Burchell's redfin) and *Barbus andrewi* (witvis) are known to inhabit the Elands River and the mainstream Molenaars (at Ouplaas), respectively. Alien species are known to include the widespread *Oncorhynchus mykiss* (rainbow trout) and *Micropterus dolomieu* (smallmouth bass). A further four alien species have been recorded from the river downstream of the valley, *Lepomis macrochirus* (bluegill sunfish), the tilapias *Oreochromis mossambicus* (Mozambique tilapia) and *Tilapia sparrmanii* (banded tilapia), and *Micropterus salmoides* (largemouth bass).

Existing and proposed water-resource developments

Existing water-resource developments

Presently, the Molenaars catchment is largely undeveloped in terms of its water resources, and there are no major impoundments on the mainstream or any of the tributaries.

Proposed water-resource developments

Several sites on the river have been investigated as possible water storage areas, within the WCSA (DWAF 1994b, 1997b):

- An abstraction weir, the Upper-Molenaars River Diversion, on the Molenaars mainstream, immediately upstream of the confluence of the Elands River.
- A dam of two possible capacities on the mainstream at Ouplaas (Figure 9.1).
- An abstraction weir on the Molenaars mainstream, immediately downstream of the confluence of the Elands and Molenaars rivers.

9.2.3 Du Toits River and study site

The location of the study site on the Du Toits River (or Du Toit's River) (1: 50 000 topographical map 3319CC Franschhoek, second edition 1977, partially revised from aerial photography 1987) is indicated in Figure 9.1. The longitude and latitude of the site are given in Table 8.1, while a brief comparison of the site with other sites is provided in Table 8.2. Natural characteristics of the river at the site are shown in Plate 9.5 (see Section 9.4).

Most of the information below is derived from Tharme and Brown (1994), which is the only detailed assessment of the riverine ecosystem to date, the pilot survey (Chapter 8) and DWAF (1997b).

Catchment characteristics

The Du Toits site falls within WR90 quaternary subcatchment H60B (DWAF 1997b), for which the total subcatchment area is the second smallest of the four study sites at 46.02 km^2 (Table 8.1). The subcatchment receives a MAP in the order of 1445 mm a^{-1}, with a corresponding MAE (S-Pan) of 1288 mm a^{-1} (DWAF 1997b). Runoff figures are given in Section 9.3.2, along with a description of the hydrological regime at the study site.

The Du Toits River forms part of the Breede system, and is essentially a tributary of the Riviersonderend River (Section 9.2.4), but is truncated by Theewaterskloof reservoir. The river has its source in the north, in the Middagkransberg and Franschhoek Mountains, at an altitude of about 1500 m. It is fed by several seasonal mountain streams, and the Paardekloof, Perdekloof and Kafferskloof Rivers are the main first-order perennial tributaries forming the headwaters (Figure 9.1). Downstream of the confluence of these tributaries, the mainstream Du Toits forms a short, second order river which flows for *c.* 13.4 km, from the confluence of the Kafferskloof River with the mainstream, through the steep-sided, narrow Franschhoek Pass. The arterial road linking Franschhoek and Villiersdorp runs alongside the river, through the Pass. The Du Toits passes under a road bridge, before dividing into multiple channels that flow through a large palmiet-dominated wetland and into Theewaterskloof Dam (Figure 9.1). The wetland is a recognised Natural Heritage Site (Tharme and Brown 1994). In its short middle reaches, the Du Toits receives perennial flow contributions from the Joubertsgat River and several other perennial tributaries (Figure 9.1).

The largely mountainous upper catchment, in the Franschhoek Mountains, is a designated Mountain Catchment Area (MCA) (Mr Steyn, Franschhoek Municipality pers. comm.). Much of the area is located within the 1759 ha Mont Rochelle Nature Reserve. The subcatchment is largely undeveloped, and land use is primarily limited to recreation as much of the subcatchment falls within protected conservation areas. Land use estimates reported here are as at 1993 (DWAF 1997b). Several scattered patches of alien vegetation, including *Pinus* species, occur in the MCA, with the area of alien vegetation with a density greater than 25% estimated as 3.5 km^2 (DWAF 1997b). The majority of the area is covered with indigenous mountain fynbos, and there are no afforested areas (commercial plantations) and no irrigation agriculture. The latter land-use activities do, however, occur in adjacent subcatchments (DWAF 1997b). The middle and lower subcatchment are primarily located within the Hottentots Holland Nature Reserve, and a section of the Boland Hiking Trail runs close to the river for a short distance. There is a Youth Camp alongside the river in its upper reaches, and a Camping Site immediately upstream of the lower-reach wetland (Figure 9.1). A single DWAF hydrological gauging weir, H6H007, is situated within the study site, on the reach of river flowing through Purgatory Outspan, upstream of the wetland (Figure 9.1). The weir is located immediately downstream of the confluence of an unnamed perennial tributary that arises in the Baviaans Kerk 2 region, with the mainstream (Figure 9.1). The closest town to the study site is Franschhoek (Figure 9.1), situated on the southwestern side of the Franschhoek Mountains, so there is no urban development in the catchment.

Synopsis of background ecological information on the Du Toits River system

River classification, conservation importance and status

The Du Toits is a perennial river. Although there is currently no designated conservation importance for the river, it is likely to be of high importance as much of the river system and its catchment is located within conservation areas, is relatively undisturbed by anthropogenic impacts and is, therefore, in a largely natural condition (Tharme and Brown 1994).

Tharme and Brown (1994) estimated the abiotic conservation status of the upper Du Toits River as Class 2, indicating that the functioning of the riverine ecosystem is essentially still natural, although a small change in natural habitats and biota may have occurred. Existing water abstraction was given as the main reason for not allocating the river a Class 1 abiotic conservation status. The biotic present conservation status for the upper river was calculated as extremely high, a rating of Class 1. That is, the reach has 100% of its potential value, and is natural and unmodified. Overall, the river has a conservation status of Class 2.

River geomorphology

The upper river has a high gradient and exhibits physical features characteristic of the mountain-stream zone of a southwestern Cape river. These include a narrow, single channel dominated by shallow riffle and short cascade sections interspersed with deep bedrock-bottomed pools. Further downstream, in the vicinity of the DWAF gauging weir and study site, the river transforms into a foothill river (Table 8.2), with a gentler gradient and shallow, cobble-dominated bed (pers. obs.). The substratum comprises a maximum of two layers of fairly angular to rounded cobbles, above bedrock (this study).

Riparian and instream vegetation

The Du Toits River has a largely open riparian canopy of indigenous mountain fynbos along much of its length and at the site, although the canopy tends to be closed along the headwater streams.

There are no visible signs of removal of indigenous riparian vegetation or encroachment of exotic species at the study site. Although no formal survey has been made of the distribution and composition of the riparian vegetation, some of the dominant species identified in Tharme and Brown (1994) from the reaches upstream of the study site are the following: *Metrosideros angustifolia* (lance-leaved myrtle); *Podalyria calyptrata; Euryops* sp.; *Brabejum stellatifolium* (Cape wild almond); *Erica caffra; Myrica serrata; Isolepis*; and *Prionium serratum*. The area of Purgatory Outspan, downstream of the study site, is the only remaining locality of three *Erica* species, viz. *Erica purgatoriensis*, *E. chrysocodon* and *E. hibertia*. It also includes a large *P. serratum* wetland that obtains its water from the Du Toits River (Tharme and Brown 1994).

Water chemistry

The data obtained during the pilot study for the present project were used to gain an understanding of natural background water chemistry for the study site during the months of the dry season (March 1994; Table 8.4). There is no summarised literature on historical winter and summer water chemistry patterns, although there are historical DWAF chemistry records for the study site (DWAF 1990b).

The instantaneous, summer water temperature was about 22 °C, with a minimum daylight temperature of 21.0 °C and a maximum as high as 27.0 °C (March 1994). The water was acidic, at a pH of 5.7. This value falls between the median pH values of 5.5 and 6.0 obtained for least-impacted mountain-stream and foothill zones in the region (Dallas *et al.* in press). Average TA was low at 0.065 mEq ℓ^{-1}. Conductivity was very low at about 3 mS m^{-1}, and TDS values were correspondingly low at an average of 21.44 mg ℓ^{-1}. Both figures are just less than the background median values of both least-impacted mountain-stream and foothill zones of the same region (Dallas *et al.* in press). The concentrations of major cations and anions were low, with sodium and chloride constituting the dominant ions, at 0.201 mmol ℓ^{-1} and 0.256 mmol ℓ^{-1}, respectively. Oxygen levels were near saturation, indicating highly aerobic waters for instream biota. TSS levels were negligibly low, corresponding with high water clarity during summer. The figures are well below the regional background median TSS concentration for least-impacted mountain stream and foothill zones (Dallas *et al.* in press). Nutrient levels, specifically nitrates, nitrites and phosphates, were low, at 0.468 µmol ℓ^{-1}, 0.063 µmol ℓ^{-1} and 0.093 µmol ℓ^{-1}, respectively. Ammonium levels were below detection limits. Such nutrient concentrations are typical of least impacted upper rivers (Dallas and Day 1993). The summer concentration of silicon was about 0.865 µmol ℓ^{-1}.

SASS results support the above data, indicating that the river is in excellent condition (Tharme and Brown 1994).

Riverine biota

Aquatic macroinvertebrates:

The pilot study summer data, at the taxonomic level of family and above (Appendix 8.1), showed a total of 29 taxa in the reach of the river represented by the study site. The assemblage is insect-dominated, with nematodes and oligochaetes as the main non-insect taxa. Of the insect groups, sensitive species indicative

of least-impacted upper rivers included the following: Helodidae; Athericidae; Leptophlebiidae, Ephemerellidae; Notonemouridae; several cased trichopterans, including Leptoceridae; and Corydalidae. Abundances of taxa are given in Appendix 8.1. There are no known exotic aquatic macroinvertebrates in the Du Toits River (Tharme and Brown 1994).

A once-off SASS assessment of the headwaters and mountain-stream zones of the Du Toits, in March 1994, indicated a range of 16-20 macroinvertebrate families and higher order taxa; there is no species list for the river (Tharme and Brown 1994). SASS Scores ranged from 171-133, and ASPTs from 8.3-8.6. The SASS results are well within the ranges considered typical of near-pristine southwestern Cape mountain-stream and foothill zones (Tharme and Brown 1994). Consequently, the river can be classed as being in excellent biological condition.

Fish:
According to Tharme and Brown (1994), the hardy, common and widespread southwestern Cape freshwater fish species, *Galaxias zebratus* (Cape galaxias, Family Galaxiidae; Skelton 1993) has been recorded from the Du Toits River (CNC pers. comm.). Although not recorded from the river, the Berg River redfin, *Pseudobarbus burgi* (Family Cyprinidae; Skelton 1993), possibly inhabits it (CNC pers. comm.). *Pseudobarbus burgi* is a Red Data species, accorded "endangered" status (Skelton 1987). There are currently no known alien fish species in the Du Toits River (Tharme and Brown 1994).

Existing and proposed water-resource developments on the Du Toits River system

Existing water-resource developments
There is no water storage in farm dams in the Du Toits catchment (DWAF 1997). Although there are also no major impoundments on the Du Toits River or any of its tributaries, Franschhoek Municipality presently abstracts water from two different locations on the river system, principally for domestic use. Both of these abstraction points are located some distance upstream of the study site. The first established abstraction point is a gravity-fed diversion weir on the Paardekloof River (Figure 9.1). Franschhoek Municipality has a year-round unlimited water allocation from this tributary, and specific records of volumes of water abstracted are not kept (Tharme and Brown 1994). The second abstraction weir is located on the mainstream approximately 100 m downstream of the confluence of the Kafferskloof River (Tharme and Brown 1994). Run-of-river abstraction from the second site occurs during the summer months, from November until March or April. Abstraction is by pumping from a natural bedrock pool, where pool capacity has been increased by a small, *c.* 0.5 m wide, low brick weir across the downstream end of the pool. Franschhoek Municipality is currently entitled to pump a maximum of 667.2 kℓ day^{-1}, all year round from the river at the site. However, no detailed records of abstracted water volumes are available, and the absence of a daily gauge means that, on any one day, more than the permitted daily allocation could be abstracted from the river (Tharme and Brown 1994). In March 1994, Tharme and Brown (1994) recorded an instantaneous abstraction of a full 67% of flow in the river.

As the Du Toits River flows into Theewaterskloof Dam (Figure 9.1), it is a contributor to the Theewaterskloof Scheme (Tharme and Brown 1994).

Proposed water-resource developments

Although in June 1993, Franschhoek Municipality applied to the Cape Water Court for permission to increase the existing daily water allocation to an unlimited one at the abstraction site on the mainstream, the application was denied (Tharme and Brown 1994). No other developments appear imminent (DWAF 1997b).

9.2.4 Riviersonderend River and study site

The Riviersonderend study site (1: 50 000 topographical sheet 3419 AA Grabouw, 2nd ed. 1979, partially modified from aerial photography) is located just upstream of the top end of Theewaterskloof reservoir, as indicated in Figure 9.1. Site co-ordinates are provided in Table 8.1. A brief comparison of the site with other sites is made in Table 8.2. The natural character of the site is illustrated in Plate 9.7 (see Section 9.4).

Most of the information below on the river system is derived from a situation assessment of the entire Riviersonderend River system by Ractliffe *et al.* (1996) and the pilot survey (Chapter 8).

Catchment characteristics

The total catchment area of the Riviersonderend River upstream of the study site is represented by WR90 quaternary subcatchment H60A (DWAF 1997b). At 38.57 km^2, it has the smallest area of the four study catchments (Table 8.1). (The total Riviersonderend Basin has an area of 494 km^2). The MAP of the subcatchment is 2314 mm a^{-1}, with a corresponding MAE (S-Pan) of 1175 mm a^{-1} (DWAF 1997b). The virgin and present-day MARs at the study site are given in Section 9.3.2, where the hydrology is described. The Riviersonderend River forms part of the Breede River system. The linkages of several of its tributaries with the mainstream, for instance the Du Toits River, are interrupted by Theewaterskloof reservoir (Section 9.2.3).

The river rises in the western Groot Drakenstein Mountains near Landdroskop, at an altitude of about 1590 m a.m.s.l. It flows in a southeasterly direction through Riviersonderendkloof for approximately 15 km, before it enters Theewaterskloof reservoir, at 300 m a.m.s.l. It is joined by two main perennial tributaries in its upper reaches, Boegoekloof tributary and an unnamed tributary arising near Victoria Peak (Figure 9.1). Downstream of the impoundment, the Riviersonderend flows eastwards through the Donkerhoekberge, which are situated at the southwestern limit of the Riviersonderend Mountains. The Riviersonderend Mountain range extends in a broad east-west band. It receives considerable, though unpredictable, orographic, summer precipitation from onshore southeasterly winds along the south coast. However, most rainfall is received in winter, from northwesterly frontal systems. The river is fed by runoff from the southern slopes of the Riviersonderend Mountains for some 90 km, before flowing north-east through the range at "Die Poort", near Rheenen. It flows eastwards for a further 15 km, before joining the Breede River west of Swellendam.

The natural vegetation of the Riviersonderend catchment is determined in part by the underlying geology (Ractliffe *et al.* 1996). The upper catchment, and areas close to the Riviersonderend Mountains, are covered with mesic and dry fynbos. In the vicinity of Theewaterskloof Dam and in the wide river valley downstream of it, there is some renosterveld on areas of Bokkeveld shales. However, these areas have been extensively cleared for agriculture and little natural vegetation remains.

The upper catchment, above Theewaterskloof Dam, is largely unimpacted by human disturbance and falls largely within the Hottentots Holland Nature Reserve. There is a small amount of afforestation with commercial pine plantations (land use as at 1993; DWAF 1997b). This activity, and associated bulldozing, is clearly apparent in some smaller tributaries (Ractliffe *et al.* 1996). However, such impacts are not directly in evidence at the study site, with only occasional, scattered *Pinus* present on the mountain slopes flanking it (pers. obs.). There are no areas within the MCA with a density of alien vegetation greater than 25%, and no area of the upper catchment under irrigation (DWAF 1997b). Physical impacts at the site are the closed DWAF gauging weir, H6H008, and a hiking trail (part of the Boland Hiking Trail). A wooden bridge is located just downstream of the gauging weir, and a dirt road ends near it. There is no significant urban development in the catchment.

Agriculture is the major land use in the catchment downstream of Theewaterskloof Dam, and has resulted in significant detrimental changes to the river system, especially in conjunction with flow regulation by the dam (Ractliffe *et al.* 1996).

Synopsis of background ecological information on the Riviersonderend River system

River classification, conservation importance and status

The Riviersonderend is classed as a naturally perennial river, with distinct seasonality in flow patterns (see Section 9.3). The study site is in the mountain-stream zone. Ractliffe *et al.* (1996) assigned a high conservation importance to the Riviersonderend River.

A conservation status assessment of the geomorphological reach that includes the study site was performed by Ractliffe *et al.* (1996). The reach was attributed a high overall conservation status of 96%, with an abiotic conservation status of 96% and a biotic conservation status of 95%. These scores place the site in a Class 2 conservation-status class, indicating that the functioning of the riverine ecosystem is essentially still natural, although a small change in natural habitats and biota might have taken place (Ractliffe *et al.* 1996).

River geomorphology

The source and upper reaches of the river, including the study site, cut through predominantly TMS sandstones and quartzites of the Cape Fold Mountains (Lambrechts 1979). The Riviersonderend Mountains comprise a similar geology, while patches of Bokkeveld Shales occur along the middle and lower reaches of the river course from Theewaterskloof Dam to the Breede River (Lambrechts 1979). A coarse-scale macro-reach analysis revealed that four of a total of 12 geomorphological reaches identified for the Riviersonderend, occur upstream of the impoundment (Ractliffe *et al.* 1996). The average gradient for the upper river is 32 m km^{-1}. The mountain stream (Reach 4), of altitude 480-320 m and length 5 km, includes the present study site. Geologically, in this reach the river passes through shale (often micaceous), with thin bands of sandstone, tillite, grit and conglomerate (Ractliffe *et al.* 1996). Channel substratum at the site comprises predominantly boulders and cobbles, with several bedrock outcrops instream and along the banks. The major biotopes are riffles, runs and large deep pools, with some backwaters and upstream cascades (see Section 9.5.1 and King and Tharme 1994, for definitions).

Riparian and instream vegetation

Riparian vegetation at the site is virtually entirely natural, comprising the typical elements of indigenous mountain fynbos, including shrubs such as *Brabejum stellatifolium* and *Metrosideros angustifolia*.

However, occasional young pine trees have invaded the riparian belt (pers. obs.). *Prionium serratum* occurs in small areas along the river margins at the site, increasing to high densities at the upstream end of Theewaterskloof reservoir. Instream *Isolepis* is highly abundant on the tops of boulders and bedrock.

Water chemistry

Historical winter and summer water chemistry data are available from records taken at the DWAF gauging weir, H6H008. Ractliffe *et al.* (1996) analysed these data from 1977 to 1992. The data obtained during the pilot study for the present project were used, in addition, to gain an understanding of natural concentrations during the dry season (March 1994; Table 8.4). The following background water chemistry for the study site was obtained from these data sets. It pertains to summer conditions, unless otherwise specified.

The instantaneous water temperature (March 1994) was about 20.0 °C (minimum of 19.0 °C, maximum of 21.0 °C). The water was highly acidic, with a pH of 4.5. According to historical data, the median pH is below 4.7 in both summer and winter. These figures are well below the median background pH value of 5.5 given for least-impacted mountain-stream zones (Dallas *et al.* in press). Median pH values tend to be lower in the winter, consistent with greater leaching of acidic polyphenols from the catchment fynbos (Midgeley and Schafer 1992). An average TA of 0.065 mEq ℓ^{-1} at the time of the pilot study closely matches the historical median value. Conductivity was very low at about 3 mS m^{-1}, and TDS was similarly low at 22.31 mg ℓ^{-1}. Both figures approximate the background median values calculated by Dallas *et al.* (in press) for least-impacted mountain-stream zones. The pilot survey values compare well with the historical data, where both summer and winter median conductivities are well below 5 mS m^{-1}. The concentrations of major cations and anions were low, and fall within the range of values recorded in the historical record. Sodium and chloride were the dominant ions, at 0.161 mmol ℓ^{-1} and 0.201 mmol ℓ^{-1}, respectively. According to the historical data, median concentrations of chloride, sodium and magnesium are marginally higher in summer than winter, although seasonal differences are not marked. For the other major ions, summer and winter concentrations are similar. Oxygen levels in March 1994 were high at about 8.8 mg ℓ^{-1} (an average saturation of 96%), indicating a suitably high level of aeration for instream biota. TSS levels were negligible at the site, and corresponded with high water clarity. They are well below the regional background median TSS concentration for least-impacted mountain stream zones (Dallas *et al.* in press). Nutrient levels, specifically nitrates, nitrites and phosphates, were very low, at 0.135 µmol ℓ^{-1}, 0.260 µmol ℓ^{-1} and 0.668 µmol ℓ^{-1}, respectively. Ammonium levels were below detection limits. Such nutrient concentrations are typical of unimpacted upper rivers (Dallas and Day 1993). The concentration of silicon was recorded as 0.525 µmol ℓ^{-1}. Although there appears to be occasional mild nutrient enrichment of the river at the site on the basis of historical data, such enrichment was not apparent during the present study.

Riverine biota

Aquatic macroinvertebrates:

The pilot study summer data, at the taxonomic level of family and above (Appendix 8.1), showed a total of 29 taxa in the reach of the river represented by the study site. The assemblage is insect-dominated, with non-insect taxa being planarians and oligochaetes. Of the insect groups, sensitive species indicative of least impacted upper rivers included the following: Helodidae; Athericidae; Leptophlebiidae, Ephemerellidae; Notonemouridae; several trichopterans, including Leptoceridae; Pyralidae; and Corydalidae. *Potamonautes perlatus* (Cape river crab) occurs at the site (Ractliffe *et al.* 1996). It is a common, hardy and omnivorous

inhabitant of Western Cape rivers, occurring in rivers with both natural and regulated flow regimes. Ractliffe *et al.* (1996) provides information on macroinvertebrates from various sites along the section of the Riviersonderend River downstream of Theewaterskloof Dam.

Fish:
Five indigenous fish species have been recorded from the river downstream of Theewaterskloof Dam, namely: *Pseudobarbus burchelli* (Burchell's redfin); *Barbus andrewi* (Breede witvis); *Sandelia capensis* (Cape kurper); *Galaxias zebratus* (Cape galaxias); *Anguilla mossambica* (longfin eel) (Ractliffe *et al.* 1996). The first two species are rare and vulnerable, respectively (Skelton 1987; Skelton 1993). It is not certain how many of these species, or if any additional species, occur in the mountain-stream zone above the dam. Similarly, seven alien species have been recorded from Theewaterskloof Dam and the downstream river: *Oreochromis mossambicus* (Mozambique tilapia); *Tilapia sparrmanii* (Banded tilapia); *Micropterus dolomieu* (smallmouth bass); *Micropterus salmoides* (largemouth bass); *Lepomis macrochirus* (bluegill sunfish); *Tinca tinca* (tench); and *Cyprinus carpio* (carp). It is not known whether or not any of these species occur in the upper river.

Other fauna:
Aonyx capensis (Cape clawless otter) is likely to occur along the upper river as it has been recorded from more impacted reaches downstream of the dam (Ractliffe *et al.* 1996). Ractliffe *et al.* (1996) indicated that four non-threatened, non-endemic riverine and six wetland amphibian species occur in the Riviersonderend downstream of the dam. It is not known whether or not these species are present at the upstream study site.

Existing and proposed water-resource developments

Existing water-resource developments
There are no existing water-resource developments, including farm dams, above or at the study site (DWAF 1997b). However, the downstream reaches of the Riviersonderend are regulated by a major water-resource development in the form of Theewaterskloof Dam, with a storage capacity of 482 x 10^6 m^3 (Ractliffe *et al.* 1996). The dam was completed and operational in 1980, but only reached full supply capacity in 1993. It acts as a source reservoir for the Riviersonderend-Berg-Eerste River Government Water Scheme, an interbasin water transfer project that supplies water to greater Cape Town, Stellenbosch and the upper Berg River catchment (Ractliffe *et al.* 1996). The dam has a substantial effect on the flow regime of the downstream river. A full description of the dam and its various ecological effects on the downstream river is provided in Ractliffe *et al.* (1996).

Proposed water-resource developments
Presently. there are no proposed water-resource developments for the river upstream of Theewaterskloof Dam (DWAF 1997b).

9.3 HYDROLOGICAL CHARACTERISATION OF THE STUDY SITES

The following section provides a brief description of the general features of the natural hydrological regime at each study site. Knowledge of the hydrological regimes of the four rivers is deemed essential for this study, as it enabled the results of the low flow experiment to be assessed relative to the long-term, natural

flow disturbance history of the river. Additional information on the dry-season baseflow patterns and natural, low flow characteristics of each site is provided in Section 10.2. Further hydrological analyses are beyond the scope of this report and are presented elsewhere (Tharme in prep.).

9.3.1 Availability of historical flow data and methods of analysis

Availability of historical hydrological data

Table 9.1 summarises the availability and quality of the historical daily flow records presently worked-up from raw data (by DWAF) for each of the four sites, and should be used in conjunction with the relevant sections of Table 8.1.

Elands River

Hydrological data from two DWAF gauging weirs at the site were used for the study, the now- closed weir H1H017 and the new weir H1H033 (Table 8.1).

The accuracy of DWAF gauging weir, H1H017, is rated as moderate (Table 8.1). Discharge data for this weir are considered adequate for flows at less than 1.1 m above the weir crest (Mr P. Rademyer, Hydrology Division, DWAF, Pretoria pers. comm.), a stage level that is equivalent to 17.0 $m^3 s^{-1}$. Hence, low flow readings for this weir can be considered of acceptable accuracy. Winter flows are unreliable, often exceeding the rating curve, as flows regularly spilled over the left bank, to the left of the gauge tower and weir. The precise stage corresponding with this event is not known (Mr P. Rademyer pers. comm.). The record for the period 10/02/76-05/10/76 is considered unsuitable for use, as the natural rating cross-section was altered during activities related to the Huguenot Tunnel. Data for the period 06/79-11/84 need to be treated with caution as the weir pool was regularly filled with washed-in river stones during this period, affecting high approach velocities, and the inlet pipe was buried under stones.

As a result of the above problems with discharge measurement at H1H017, DWAF gauging weir H1H033 was constructed upstream of the old weir. H1H033 is rated as "good" in terms of accuracy (Mr P. Rademyer pers. comm.). Cross-correlation of H1H033 with H1H017 for the 14 month period of overlap in data recording, from 04/1991-06/1992 indicated a good correlation of the recorded flow data (Mr P. Rademyer pers. comm.). However, the time period was too short to undertake a comprehensive correlation between the two weirs.

For the present study, all the daily flow data for H1H017 were used until the day for which the first record was available for H1H033. The data from H1H033 were used from that date onwards, for the entire period of overlap in record. As the probable low reliability of the record from 10/02/76 to 05/10/76 (Mr P. Rademyer pers. comm.) was not known about prior to data analysis, these data were included in the present study.

Twenty-nine years of hydrological data, including months with a total of 321 missing days, were used in the hydrological analyses for this site (Table 9.1).

Table 9.1 Summary of the Department of Water Affairs and Forestry historical, average daily discharge data, computed to date from raw gauging weir records at the four study sites, for use in the hydrological analyses.

SITE	DWAF GAUGING WEIR NO.(S)	START AND END DATES OF USABLE RECORD	NO. OF YEARS OF RECORD	NO. OF DAYS OF MISSING DATA	NO. OF DAYS OF WEIR EXCEEDANCES
Elands	H1H017 / H1H033	11/03/1969 - 26/02/1997	29	321	11
Molenaars	H1H018	27/02/1969 - 31/01/1997	29	249 (includes 43 days of experimental impact phase)	8
Du Toits	H6H007	01/10/1964 - 31/08/1992	29	353	536
Riviersonderend	H6H008	01/10/1964 - 30/06/1992	29	208	349

Molenaars River

The accuracy of DWAF gauging weir, H1H018, is given as moderate (Table 8.1). However, the velocity of approach for the weir may be higher than originally estimated by a factor of some 20% (Mr F. Mouski, DWAF, Western Cape region pers. comm.). This has obvious implications for the degree of accuracy of the weir, particularly at high flows.

For the Molenaars site, the original historical daily discharge record from the weir included the period for which flows were experimentally reduced below natural levels, for the period 20/02/95-03/04/95. Discharges for these 43 days were excised from the record and labelled as missing data, prior to any hydrological analyses (Table 9.1).

In total, 29 years of hydrological data were used in the hydrological analyses for the Molenaars site (Table 9.1). The record included a total of 249 missing days of record.

Du Toits River

DWAF gauging weir, H6H007, located at the Du Toits site, has an extremely low DWAF accuracy rating, with an expected error of up to 50% (Table 8.1). Hence, the hydrological data for this site should be treated with caution. However, the majority of the lack of accuracy appears to be attributable to flood events, where peak magnitudes often could not be measured during overtopping (Table 9.1). The weir was closed in 1992, so there are no records of discharge for the duration of the low-flow study (Tables 8.1 and 9.1).

In total, 29 years of hydrological data were used in the hydrological analyses for this site, excluding 353 days of missing data. (Table 9.1).

Riviersonderend River

DWAF gauging weir, H6H008, at the Riviersonderend study site is rated as highly accurate by DWAF, with a low error margin (Table 8.1). However, weir exceedances occur frequently, in almost every winter month, albeit of short durations (Table 9.1). The weir was closed in 1992, so there are no records of discharge for the duration of the low-flow study (Tables 8.1 and 9.1).

In total, 29 years of hydrological data, including months with a total of 208 missing values, were used in the hydrological analyses (Table 9.1). Overtopping occurred frequently in the record, reducing the accuracy of flood estimates (Table 9.1).

Methods of analysis of historical flow records

None of the rivers are highly regulated upstream of the sites, and the historical flow records represent near-natural flow conditions (pers. obs.). This had been verified prior to the pilot survey by previous testing of the data from the four sites for non-homogeneity (King and Tharme 1994). Only pre-screened rivers were considered for the present study (Table 7.1). Therefore, no naturalisation of the present-day flow records was required.

The DWAF historical, average daily flow data for each site were checked for errors, missing data values and periods of weir overtopping (Table 9.1). The records were not patched, due to constraints of time and available expertise. All the worked-up data available for each site were used for each type of analysis, not only the coincident periods of record among them. For days of overtopping, the recorded discharge had to be used, with consideration of the fact that the higher flows were likely to have been underestimated.

The DWAF data were converted from the two kinds of DWAF formats to a standard NSI format, using NSI programs. This enabled the data to be used in Microsoft Excel Version 7.0 and in various NSI software programs (unpubl. software, NSI, Cape Town). The NSI program, Table, was used to generate basic summary statistics, while the Series program was used to generate flow duration curves and various flow distribution series. Summary data were automatically rounded to two decimal places by the NSI programs, although the DWAF data are recorded to three places. In some instances, it was necessary to include the third decimal place for very low flows (see below). For instance, in terms of the flow record for the Du Toits River, for January 1988, the calculated minimum average daily discharge was 0.003 $m^3 s^{-1}$ as opposed to a rounded value of 0.00 $m^3 s^{-1}$. Monthly flow data were calculated directly from the daily time series of flows.

The following hydrological indices were calculated from various analyses of the **average daily discharge** data:

- Summary statistics for each month of the year, both for each individual year of record and over the entire historical record. For each month and each instance, the following indices were computed:
    - Minimum daily discharge ($m^3 s^{-1}$).
    - Maximum daily discharge ($m^3 s^{-1}$).
    - Mean daily discharge ($m^3 s^{-1}$).
    - Modal daily discharge ($m^3 s^{-1}$).
    - Median daily discharge ($m^3 s^{-1}$).
    - Seven-day consecutive low flow ($m^3 s^{-1}$) - the average daily discharge over a period of seven consecutive days, and equal to the lowest of all total discharges calculated for the various series of 7 days within a month (the 6 days from any preceding month are used in a month when identifying the lowest flow series of 7 days).
    - Monthly flow volume (million cubic metres, Mm^3).

- Similarly, summary statistics for each month of the year, for all years of record in combination. For each month, the following indices were computed:
    - MinMin daily discharge - the overall minimum, average daily discharge, calculated from the minimum values for each year of record.
    - MaxMax daily discharge - the maximum of all annual maxima, as average daily discharge, over the entire flow record.
    - MeanMean daily discharge - the average of all the average daily discharges calculated for the full record.
    - Median 7-d low - the median discharge of all the 7-day low flow discharges, over the entire record.
- One-day flow duration curves (FDCs), derived for each month of the year – these were used to calculate various percentage exceedance values, specifically the 5%, 25%, 50% (median), 75% and 95% exceedances. These percentiles are also referred to as Q_5, Q_{25}, etc. Other percentage exceedances were measured directly from plotted 1-day FDCs. Techniques for the generation of FDCs from average daily discharge data are described in Gordon *et al.* (1992).
- One-day FDCs based on all average daily discharge data combined for all months of each year, for the entire record. Plot y-axis scales were expanded, so as to ascertain percentiles for low flows.
- Ranked 7-d low flows according to the years of record. These data were not used for the present report (Tharme unpubl. data).
- Ranked maximum average daily discharges. These data were not used for the present report (Tharme unpubl. data).
- Ranked maximum average daily discharges, for each month, over the entire period of record. These data were not used for the present report (Tharme unpubl. data).
- Recurrence intervals for the 1:2, 1:5, 1:10 and 1:20 year floods. These data were not used for the present report (Tharme unpubl. data).

It was not possible to use Aquapack software (Gordon *et al.* 1992) during the present study to further analyse the hydrological data, as all of the programs require complete patching of the record (Prof. B. Finlayson, Melbourne University, Australia pers. comm.). Analyses of flow variability, flow predictability (Colwell 1974), low flow spell analysis, *inter alia*, are presented elsewhere (Tharme in prep.).

9.3.2 General features of the hydrological regime

The general features of the flow regime at each study site are briefly discussed in this section. Specific low flow characteristics of the hydrological regime are discussed in Chapter 10 (Section 10.2), however, with particular reference to the results of the low-flow study.

Mean annual runoff

The most up to date MARs and runoff percentages available for all four sites have been calculated by NSI as part of the WCSA (Table 9.2; DWAF 1994b, 1997b). It should be noted that the naturalised figures, representing estimated near-natural or virgin flow conditions, reflect patching (infilling of missing flow values) and simulated extension of the length of the original observed DWAF flow record, by NSI. Both the original observed DWAF data and the same data set with patched-in missing values are also given in Table 9.2.

Table 9.2 Observed original and patched Mean Annual Runoff (MAR), estimated naturalised MAR (Mm3) and naturalised runoff coefficients for the four study sites (modified from DWAF 1994b, 1997b). The periods of record for which the various MARs were calculated are indicated in parentheses, where available. See Section 9.3.2 for an explanation of terms.

SITE	OBSERVED MAR FROM ORIGINAL DWAF DATA (x 10^6 m^3)	OBSERVED MAR FROM PATCHED DWAF DATA (x 10^6 m^3)	NATURALISED PATCHED EXTENDED MAR (x 10^6 m^3)	NATURALISED RUNOFF COEFFICIENT (%)
Elands	c. 87.03 (-)	- (-)	85.4 (S.D. = 31.43) (1928-1988)	63
Molenaars	134.50 (1969-1993)	136.70 (1969-1993)	135.53 (1927-1993)	60
Du Toits	41.33 (1964-1991)	42.19 (1964-1991)	42.01 (1927-1993)	63
Riviersonderend	60.74 (1964-1991)	61.93 (1964-1991)	60.85 (1927-1993)	68

Runoff coefficients, indicating the estimated percentage of rainfall that reaches the river as runoff under natural conditions, are similar for all four sites at 60% and above (Table 9.2). The uppermost site in terms of geomorphological zonation, the mountain stream Riviersonderend, has the highest runoff coefficient, as expected based on rainfall and other local factors (Section 9.2.4).

For the Elands site, the naturalised MAR of 85.4 Mm^3 can be assumed to be the same as the present-day MAR as there are no major offstream demands in the catchment upstream of the site (Table 9.2). The naturalised MAR for the Molenaars, of 135.53 Mm^3 is the greatest of all four sites, reflecting the differences in catchment area among sites (Tables 9.2 and 8.1). The use of water by alien vegetation along the Molenaars upstream of the site has been calculated by NSI as 0.64 Mm^3 for the period 1969-1993, but naturalised and current MARs can be assumed to be highly similar (DWAF 1997b). The naturalised and present-day MARs at the Riviersonderend site, have been assumed to be identical by NSI, at about 60.85 Mm^3 (DWAF 1997b), as there is no regulation of the hydrological regime upstream of the site. The lowest naturalised MAR applies to the Du Toits River, at 42.01 Mm^3. Although a water demand for alien vegetation has been estimated as 0.37 Mm^3 by NSI (Table 9.2; DWAF 1997b), and there is some year-round abstraction upstream of the site (Section 9.2.3; Tharme and Brown 1994), current and naturalised MARs can be assumed to be highly similar.

Distribution of monthly flow volumes

A comparison of the near-natural, annual hydrological flow regime at the four sites, on the basis of monthly flow volumes over the whole record (Figure 9.2), shows that all four rivers exhibit a distinctly seasonal flow pattern, with peak flows during the wet season, from June to August. Transitional flow months are May, when a dramatic increase in flow volume occurs with the climatic change to Autumn conditions, and September to October, when there is a sharp decline in flow volume with the onset of Spring. The upper Du Toits and Riviersonderend Rivers exhibit the most gradual increase to and decline from peak winter flow

volumes of all four sites, while the other two sites peak sharply in flow volume in June/July (Figure 9.2). At all sites, the dry season characteristically extends from about November to April, with February being the month of lowest flow volume. For all sites, except the Riviersonderend, January, February and March are the three consecutive months of lowest flow. For the Riviersonderend site, the average flow volume for December was marginally lower than that of March.

The Molenaars River has the highest average total volumes of flow in all months of the year, while the Du Toits River consistently has the lowest flow volumes (Figure 9.2). Although the catchment area of the Riviersonderend site is marginally smaller than that of the Du Toits, the winter flow volumes are markedly higher in the former case (Figure 9.2). This difference can be attributed to the considerably greater runoff generated by the Riviersonderend catchment (Table 9.2).

Although the differences among the sites in flow volumes during the winter months are considerable and generally reflect the different sizes of the rivers, all four sites have similarly low volumes of flow during the dry season (Figure 9.2). Hence, much of the flow volume of the larger rivers is generated by winter flood events.

Seasonal distribution of average daily discharges

Examination of the seasonal distribution of average daily discharges for each site, using box-whisker plots derived from percentage exceedance values from the historical flow data, showed the following patterns for the four sites (Figures 9.3-9.6).

Elands River

For the Elands site (Figure 9.3), November to March represent the typical extent of the dry season, with the majority of baseflows in each month being similar at less than 1.0 $m^3 s^{-1}$. The range of discharges is least variable in February, when all discharges are below this figure, and the median baseflow is about 0.3 $m^3 s^{-1}$. During the other months of the dry season, higher baseflows (and freshes) representing the 25th and lower FDC percentiles occur, though infrequently. The lower dry-season baseflows, as represented by the 95th percentile (Figure 9.3) are highly similar in magnitude, in the order of 0.2 $m^3 s^{-1}$. Hence, the lowest flows experienced at the site, over much of the dry season, tend to be the same, irrespective of the actual month. This indicates that the dry-season baseflow regime is a fairly constant one. At the 95% exceedance level, baseflows at the end of the dry season (April) are generally lower than flows at the outset (November/December). It would, thus, appear that a late winter or protracted dry season is more common, in terms of low flow magnitude, than an early winter or short dry season. At higher flow percentiles, like the 5th percentile, indicative of summer freshes and early wet season floods (Figure 9.3), this trend is no longer apparent. For instance, modal, mean and 5th percentile flows in April markedly exceed the equivalent discharges in November and December. Typically, the range of different magnitude discharges gradually increases from April, through May, to the wet season proper. Although the largest median flow events occur in July, the timing of the highest magnitude floods of the winter is typically June, with 5th percentile discharges just below 28.0 $m^3 s^{-1}$. September and October are transitional months in terms of the overall distribution of flows.

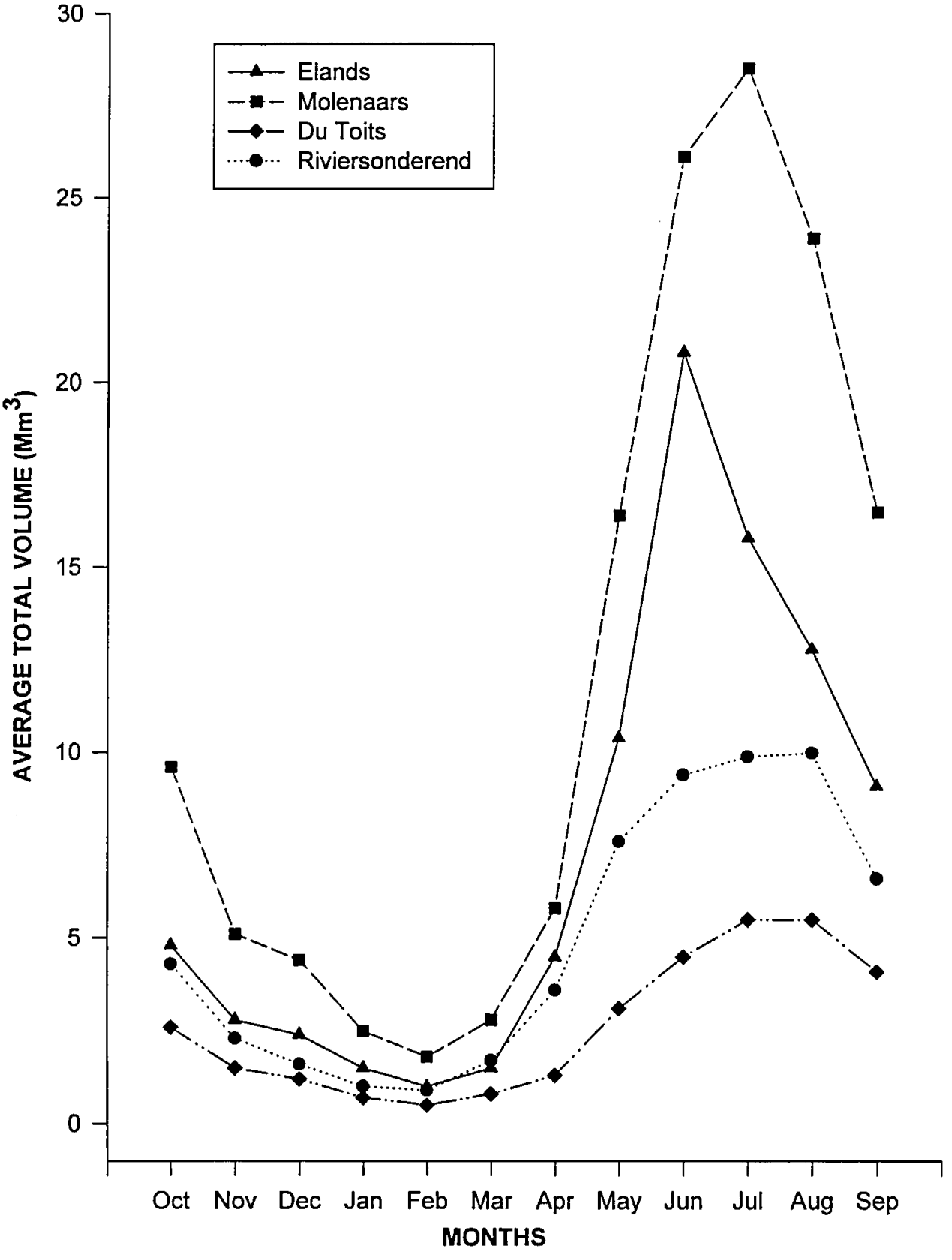


Figure 9.2 **Characteristic seasonal flow patterns at the four study sites. Flows are expressed as volumes in Million m^3 (Mm^3), and are not transformed to eliminate the scale effects due to differences in river size.**

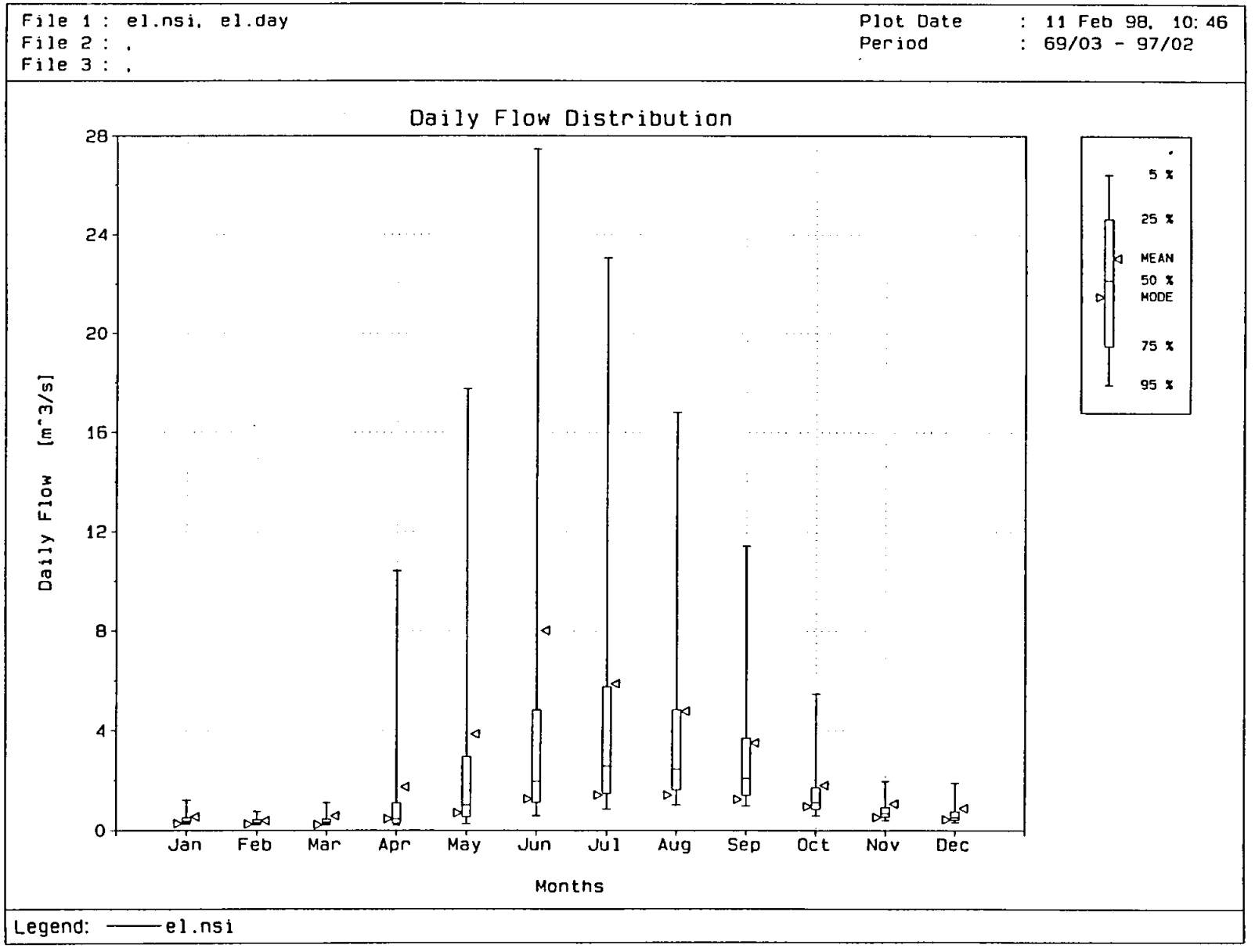


Figure 9.3 **Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Elands site, for the period 11/03/1969 to 26/02/1997. The mean, mode, and 5%, 25%, 50% (median), 75% and 95% daily exceedance flows are indicated.**

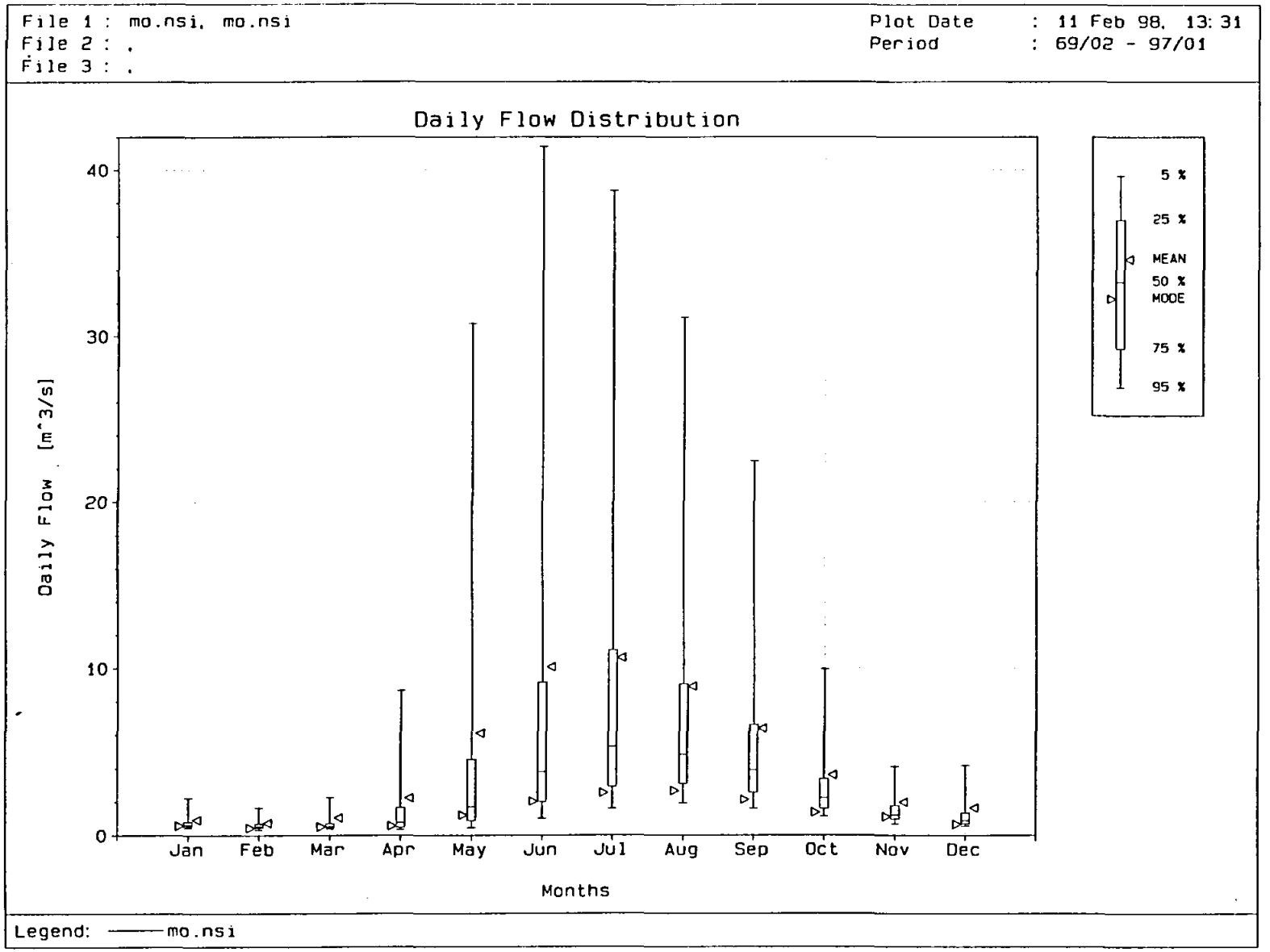


Figure 9.4 **Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Molenaars site, for the period 27/02/1969 to 31/01/1997. The mean, mode, and 5%, 25%, 50% (median), 75% and 95% daily exceedance flows are indicated.**

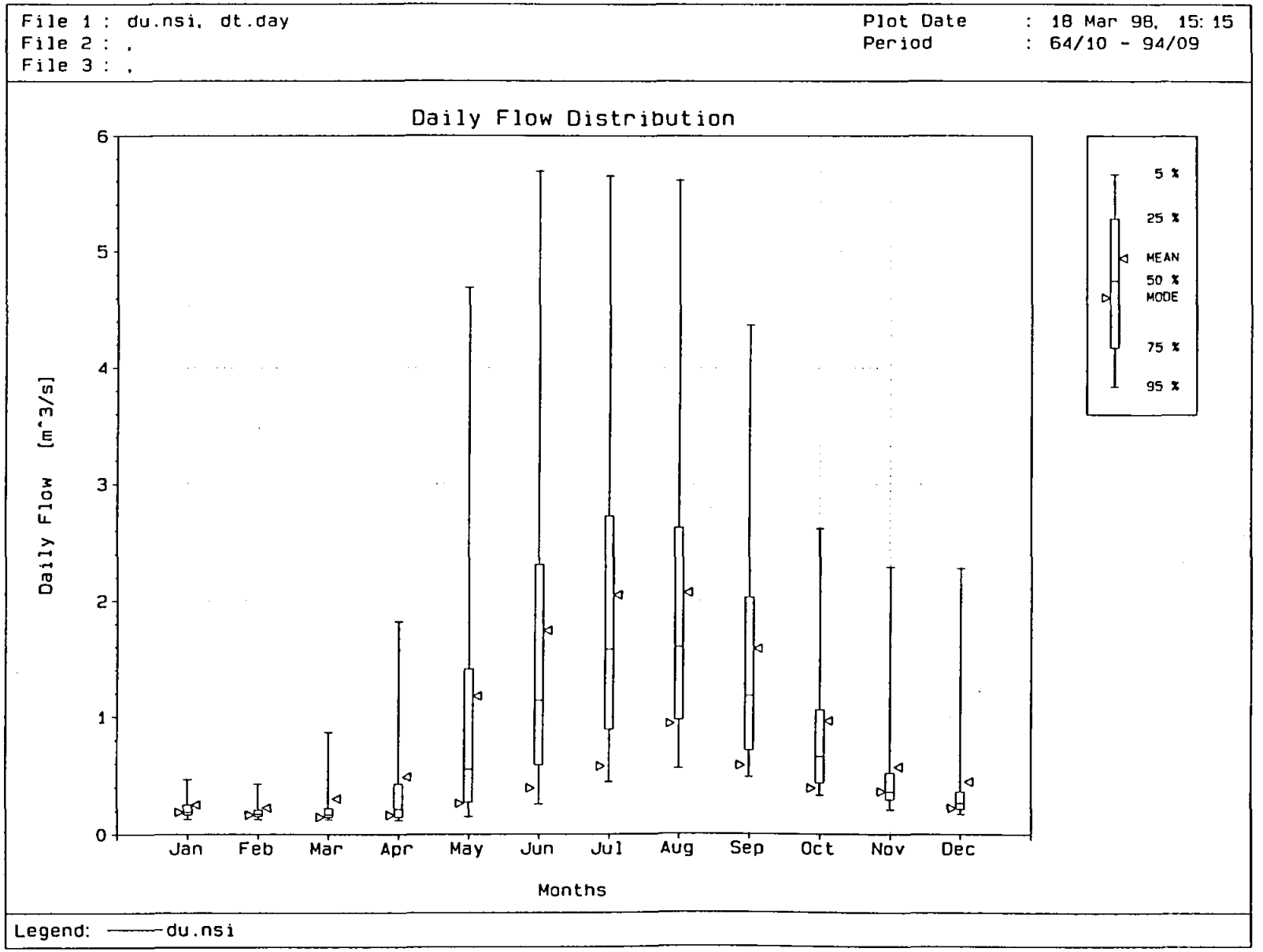


Figure 9.5 **Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Du Toits site, for the period 01/10/1964 to 31/08/1992. The mean, mode, and 5%, 25%, 50% (median), 75% and 95% daily exceedance flows are indicated.**

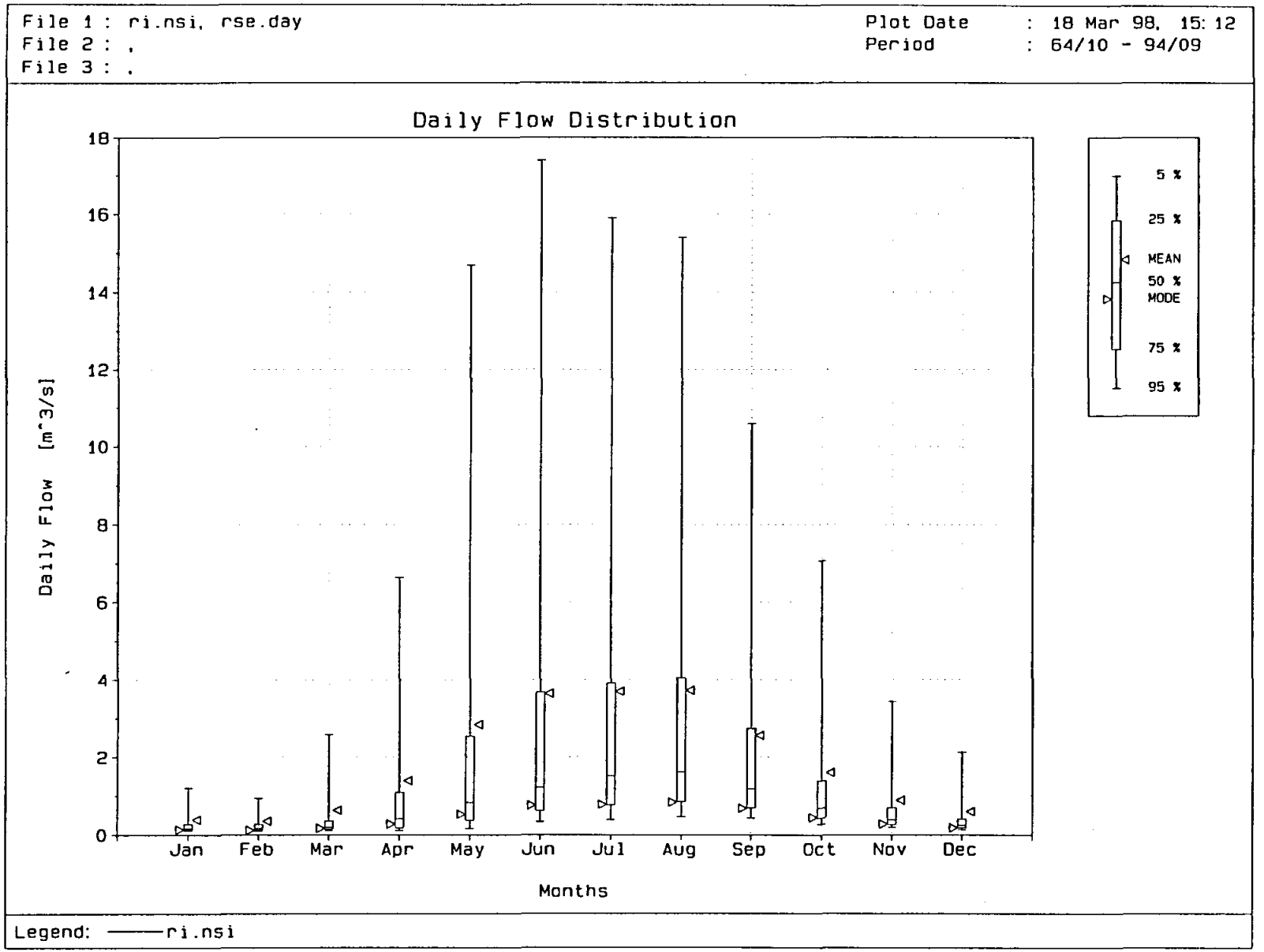


Figure 9.6 **Monthly distribution of average daily discharges ($m^3 s^{-1}$) at the Riviersonderend site, for the period 01/10/1964 to 30/06/1992. The mean, mode, and 5%, 25%, 50% (median), 75% and 95% daily exceedance flows are indicated.**

Molenaars River

The distribution of discharges for the Molenaars site (Figure 9.4), indicates that the months of November to March represent the dry season proper, with baseflows typically below 1.0 $m^3 s^{-1}$. February has the lowest overall range of magnitudes of discharge, and is the month of lowest flow. The 95% exceedance flows for this month are only marginally lower than those of March, at about 0.3 $m^3 s^{-1}$. Moreover, median baseflows for February and March are similar, at *c.* 0.5 $m^3 s^{-1}$. From November to April, a very low range of discharges represents the 75%-95% exceedance flows, and the lowest flows on record are fairly similar in all months. In April, however, the upper range of discharges, represented by the 25th and lower FDC percentiles, increases markedly above the preceding months, due to the influence of fresh events and small floods. Maximum and average discharges in November/December are lower than those in April/May, as the latter months include higher flows associated with the onset of winter rains. At the 95% exceedance level (i.e. for very low flows), baseflows at the end of the dry season, extending to April and May, are lower than flows in early summer (November/December). At higher flow percentiles, like the 5th percentile, indicative of summer freshes and early wet season floods (Figure 9.4), this trend is no longer apparent. Mean and median discharges peak in midwinter, July. However, the widest range in discharge magnitude occurs in June, with a maximum, at the 5% exceedance value, above 40.0 $m^3 s^{-1}$.

Du Toits River

For the Du Toits site (Figure 9.5) the dry season is not very clearly defined, with a significantly wide range of flow magnitudes still occurring in the record for November and December. Indeed, the average and median daily discharges for November, and the median discharge in December, are greater than in April. The peak dry-season months are January to March, with low median flows of less than 0.2 $m^3 s^{-1}$. The lower end of the range of flows are of similar magnitude in all three months. Although the median discharge in March is characteristically lower than that in February, the upper range of flows occurring in the river in March is considerably broader. The narrowest range of flows occurs in February. January and February represent the dry-season months with the most stable baseflow regime. Typically, the range of different magnitude discharges increases dramatically from April to May. The winter period is characterised by three months of similar peak flows, June to August. The upper range of discharges is only marginally higher for June than for the other two months. Over the entire year, the highest magnitude flows are relatively small, generally below 6.0 $m^3 s^{-1}$. This, in part, reflects the fairly small catchment area generating runoff for this river. However, the high number of days of weir overtopping at this site and low accuracy of recording (Tables 9.1 and 8.1, respectively) indicate that there was probably also an underestimation of flood magnitudes.

Riviersonderend River

For the Riviersonderend (Figure 9.6), November to March represent the typical extent of the dry season on the basis of median discharges, with peak dry-season flows below 0.2 $m^3 s^{-1}$. The lowest range of discharges occurs in February, when even the Q_5 is less than 1.0 $m^3 s^{-1}$. Thus, this month is also the most consistent and constant with respect to baseflow. During the other months of the dry season, higher discharges are represented by the 25th and lower percentiles, indicating the increasing influence of freshes on the flow regime closer to both autumn and spring. April exhibits a marked influence of freshes, while some relatively high flow events already occur in May. Thus, the Riviersonderend exhibits a fairly early and rapid response in hydrological regime to winter flow conditions. Of the three peak wet season months, June

to August, the median discharge in August is highest though average flows are similar in all the months. The greatest range of discharges, on the basis of 5th-95th percentiles, occurs in June.

9.4 EXPERIMENTAL DESIGN AND PROCEDURES FOR FIELD IMPLEMENTATION

9.4.1 Experimental design

The low-flow field experiment was structured as a spatially replicated extension of the spatial, unreplicated Before-After, Control-Impact (BACI) design advocated by Stewart-Oaten *et al.* (1986) and Underwood (1993). The conceptual nature of the BACI design is detailed in these papers and briefly outlined in Figures 9.7A-D. Overall, this type of experimental design was aimed, as far as possible, at avoiding criticisms of pseudoreplication (see Hurlbert 1984, for a full explanation). Pseudoreplication may be defined as "the use of inferential statistics to test for treatment effects with data from experiments where either treatments are not replicated (though samples may be) or replicates are not statistically independent" (Hurlbert 1984). It can be avoided, for instance, by using a control treatment(s) to account for the temporal change exhibited by biological systems.

In terms of the experimental design, there were replicated locations (two per site, where locations were equivalent to the subreaches at each study site defined in Section 8.1) at multiple sites (four sites, one on each of the study rivers). At each of the sites, the more upstream of the two locations was not subjected to any unnatural reduction in flow and acted as an experimental control. The more downstream locations were the impact or experimental locations, in that they had various proportions of flow diverted from them (see below, for the specific proportions of flow abstracted at each site). While flow reductions occurred at three sites, the fourth site acted as the overall control for the experiment by having no flow diversion. Thus, at the control site, both the control and experimental locations reflected conditions of natural flow. The experiment was designed to focus on the effects of both natural and experimental, dry-season baseflows on the physical biotopes, water chemistry and BMIs of the rivers at the sites. The entire control site, and control locations at the three experimental sites, thus reflected the natural ecological condition of the river during the dry season, while the three impact locations at the experimental sites were subjected to unnaturally low flows.

Figures 9.8A-D schematically illustrate the experimental layout adopted at each of the four study sites. At each site, a 60-m long control location, representing the full width of the active channel, was delineated upstream of a predetermined flow diversion point, while a 60-m long impact location was located no more than *c.* 80 m downstream of the control location. The locations were located close together so that they could be considered to be within the same reach, and are thus likely to be hydrologically, geomorphologically and biologically similar (as established by the results of the pilot study; see Chapter 8). They were also situated in close proximity (within a maximum of 200 m) to a DWAF hydrological gauging weir located in the study reach, upstream and/or downstream of it (see Figures 9.8A-D), so that no tributary inflows or other significant discharge contributions to the mainstream occurred between the cross-sections within each location and the weir. This ensured that the historical flow record and hydrological data collected during the low-flow experiment, reflected the hydrological regime at each site. Although it was deemed preferable to use sites at which flow gauging was ongoing, this was only feasible for two sites, the Elands and Molenaars sites (see Table 8.1 and Section 9.3).

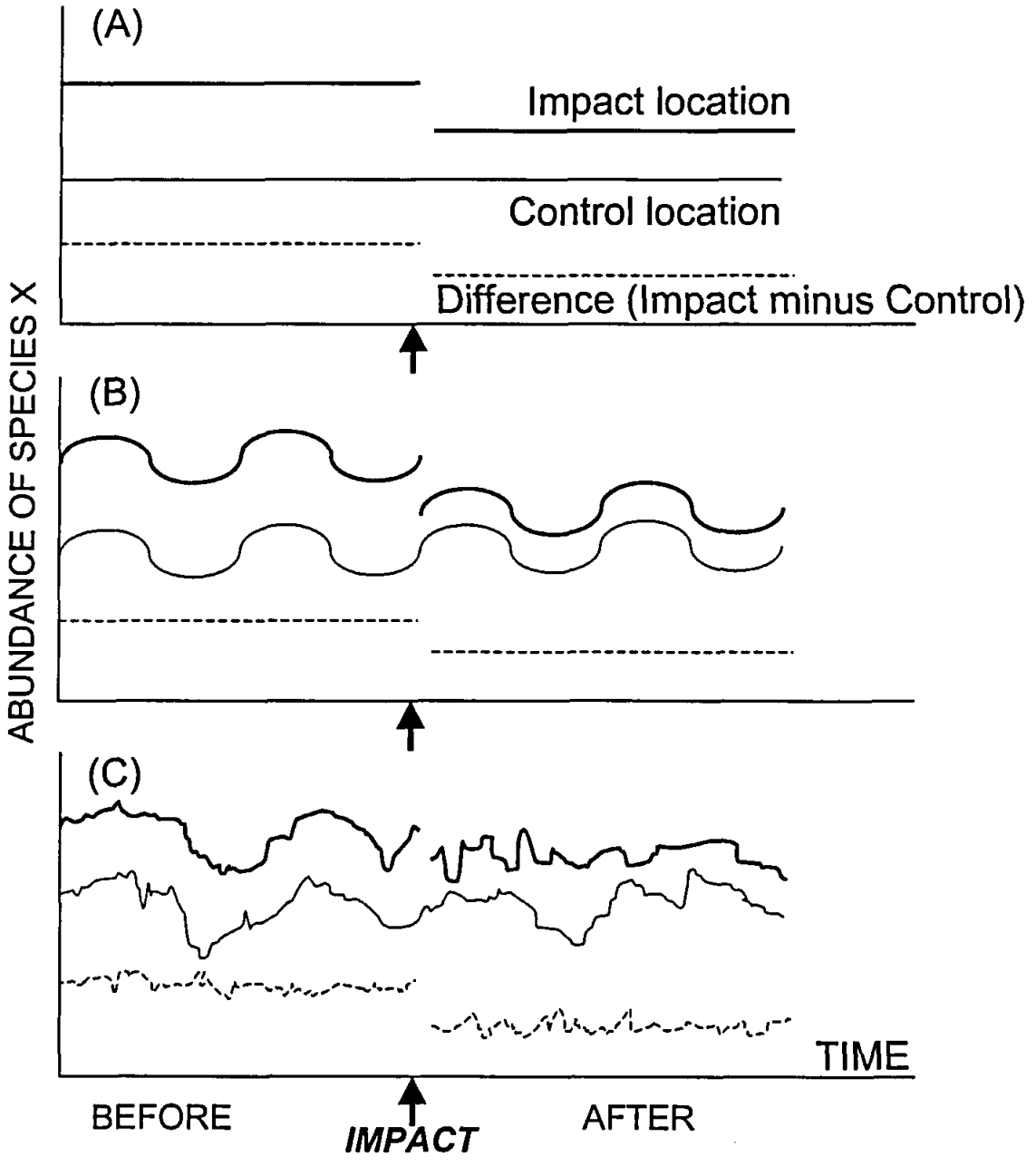


Figure 9.7 **Hypothetical abundances of "Species X" at the impact and control locations, and the difference of the abundances, as functions of time, in three versions of assessment of an impact (e.g. experimental flow reduction). (A) Simplistically, abundance is constant at each location (though possibly different for different locations) except for a decrease in abundance at the impact location at the start of the impact. (B) More plausibly, the abundances fluctuate (e.g. seasonally), but the difference in abundance (Impact minus Control) remains constant, except at the start of the impact. (C) However, realistically, the abundances fluctuate partly in synchrony and partly separately, due to random influences (e.g. local movement of individuals, mortalities, drift and weather changes). Some of these influences might have the same effect at both locations, thus not affecting the difference between them, but others affect the two locations differently, and thus affect the difference. As a result, both the abundances and (perhaps to a lesser extent) the difference vary. The difference of relevance for assessment of the impact is that between the *mean* heights of the dashed curves in the Before and After periods. The fluctuations in the dashed lines within each period show that the difference between locations is constantly changing, but an additional difference can still be distinguished between locations, not present in the Before period, that is present in the After period. The differences between locations within both periods need to be characterised, so that the added difference due to the impact can be distinguished (modified from Stewart-Oaten *et al.* 1986).**

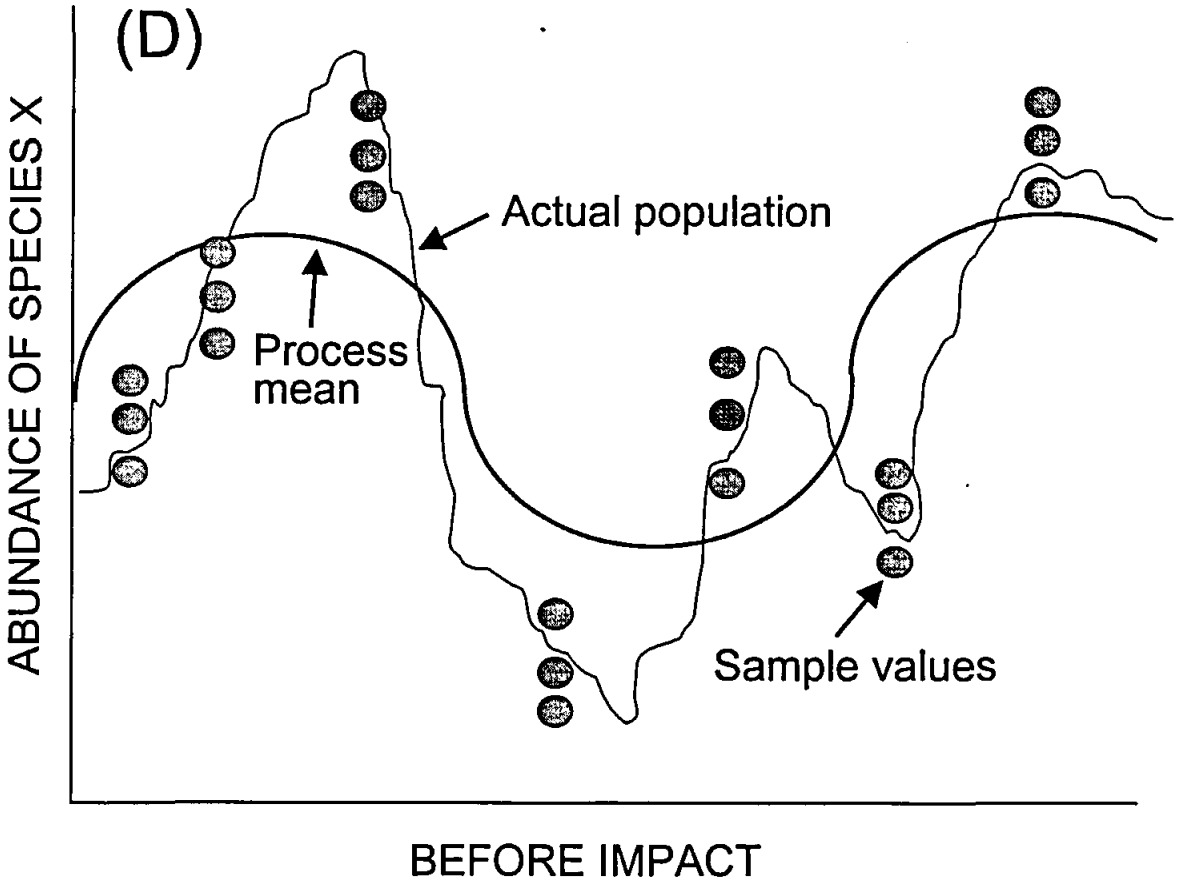


Figure 9.7 **(D) The three functions to be distinguished in the characterisation of the differences between the Control and Impact locations illustrated in Figure 9.7A-C: the smooth curve represents the true mean abundance of the population. The aim of the assessment is to determine if this mean abundance at the Impact location has been altered by the Impact (e.g. flow reduction). The thin, irregular line represents the path traced out by the actual population, which represents a single realisation of the population-producing process. This population will differ from the mean as a result of a variety of chance factors (see Figure 9.7A-C). Finally, the closed circles represent estimates, based on replicate samples, made of the actual population by the investigator. If unbiased, these estimates are centered on the actual population (modified from Stewart-Oaten *et al.* 1986).**

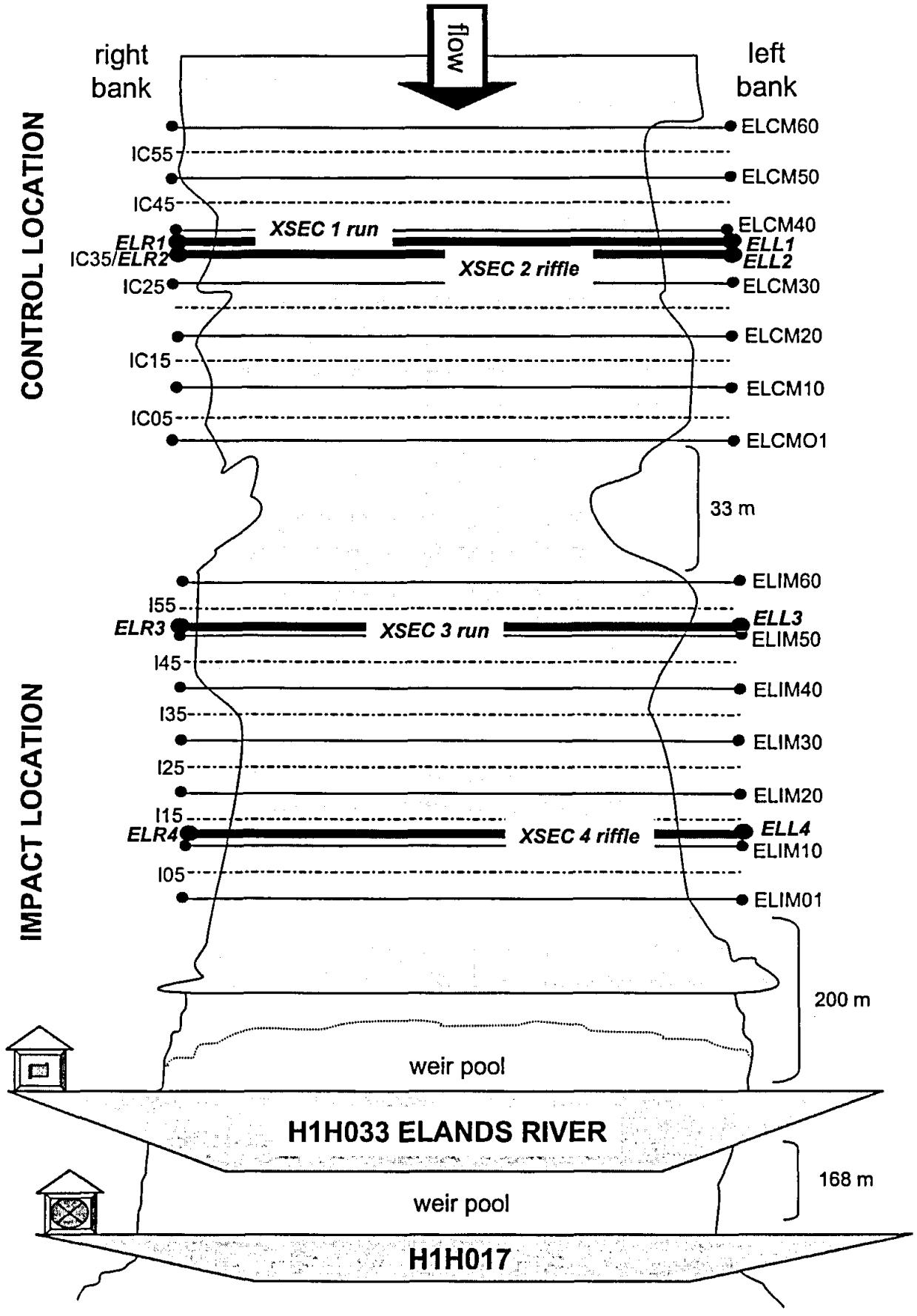


Figure 9.8A Schematic of the Elands study site, showing the 60 m control and impact locations. As this was chosen as the control site, it had no diversion weir. The positions of all transects delimiting the locations, and the four cross-sections (XSEC1-4) are indicated. Gauging weirs H1H033 and H1H017 (non-operational) are shown. Further explanation of site layout is provided in Sections 9.4.1 and 9.5.1, and Appendix 9.1.

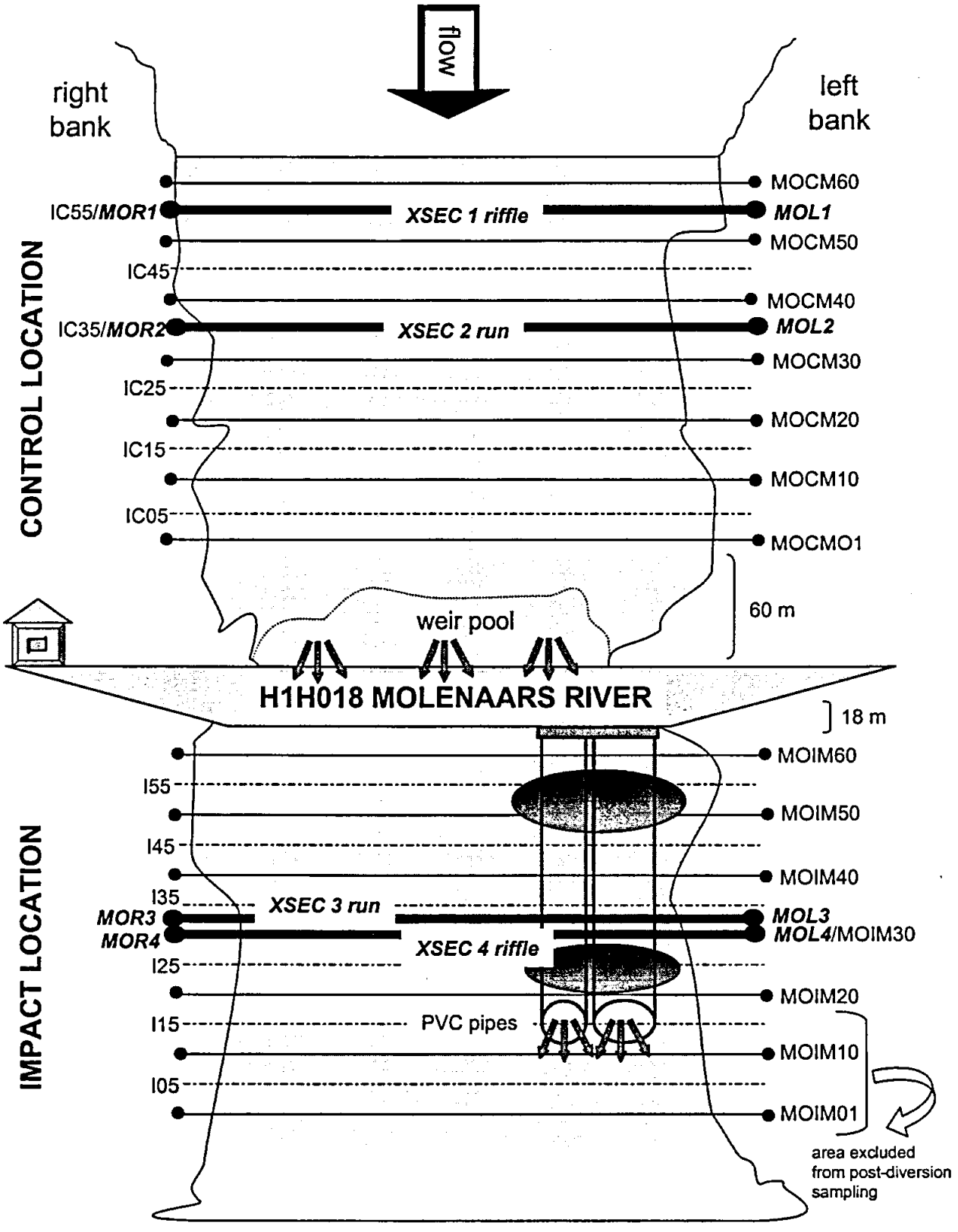


Figure 9.8B Schematic of the Molenaars study site, showing the 60 m control and impact locations. Diversion of about 36% of the natural baseflow took place at the existing gauging weir, H1H018. Water was reintroduced to the site at the downstream end of the impact location. Shaded ellipses represent boulders used as props for the two offtake pipes. The positions of all transects delimiting the locations, and the four cross-sections (XSEC1-4) are indicated. Further explanation of site layout is provided in Sections 9.4.1 and 9.5.1, and Appendix 9.1.

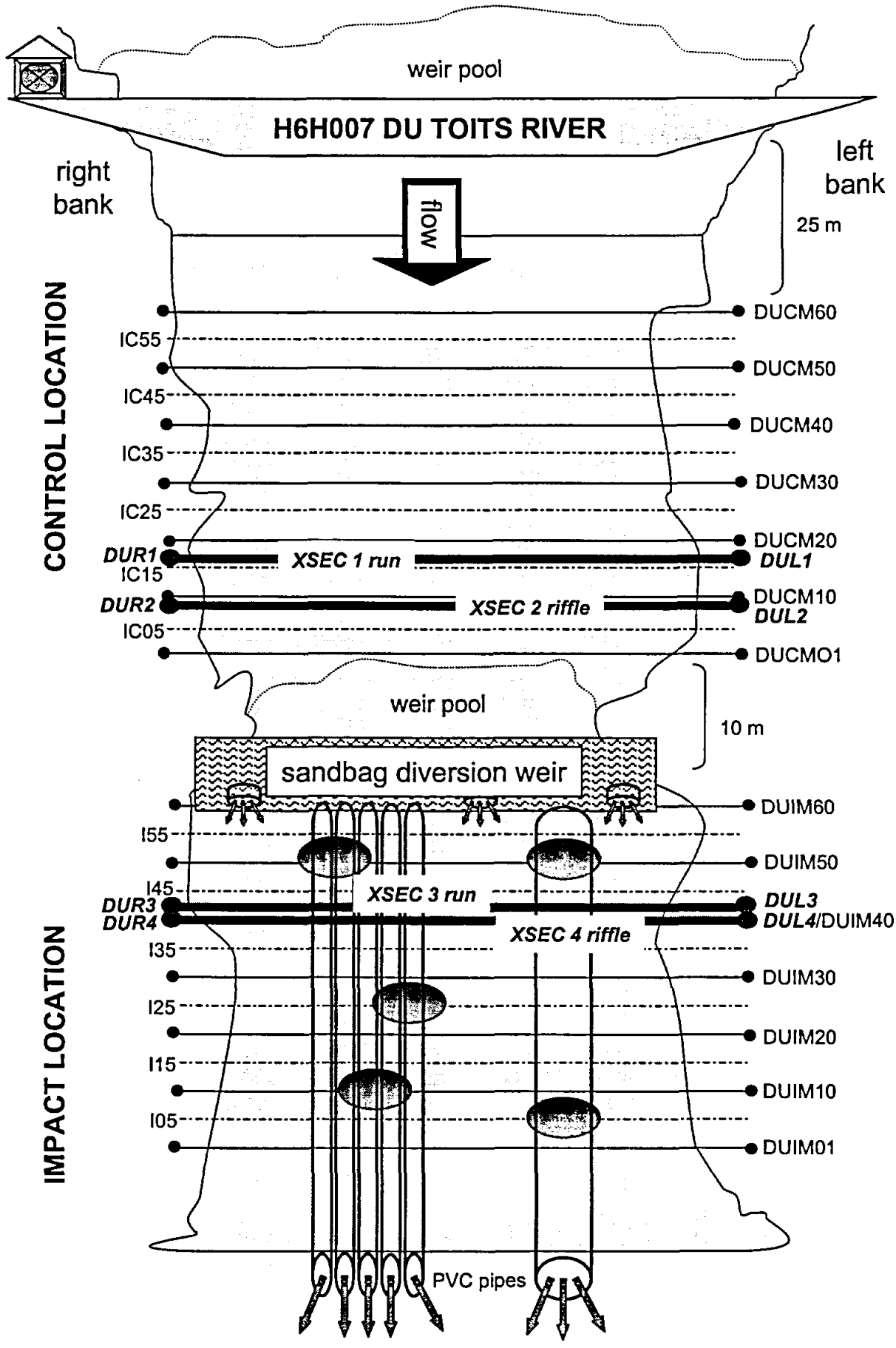


Figure 9.8C Schematic of the Du Toits study site, showing the 60 m control and impact locations. About 86% of the natural baseflow was diverted at a weir at the upstream end of the impact location. Water was reintroduced to the site below the impact location. Shaded ellipses represent boulders used as props for the two sets of offtake pipes. The positions of all transects delimiting the locations, and the four cross-sections (XSEC1-4) are indicated. Gauging weir H6H007 (non-operational) is shown. Further explanation of site layout is provided in Sections 9.4.1 and 9.5.1, and Appendix 9.1.

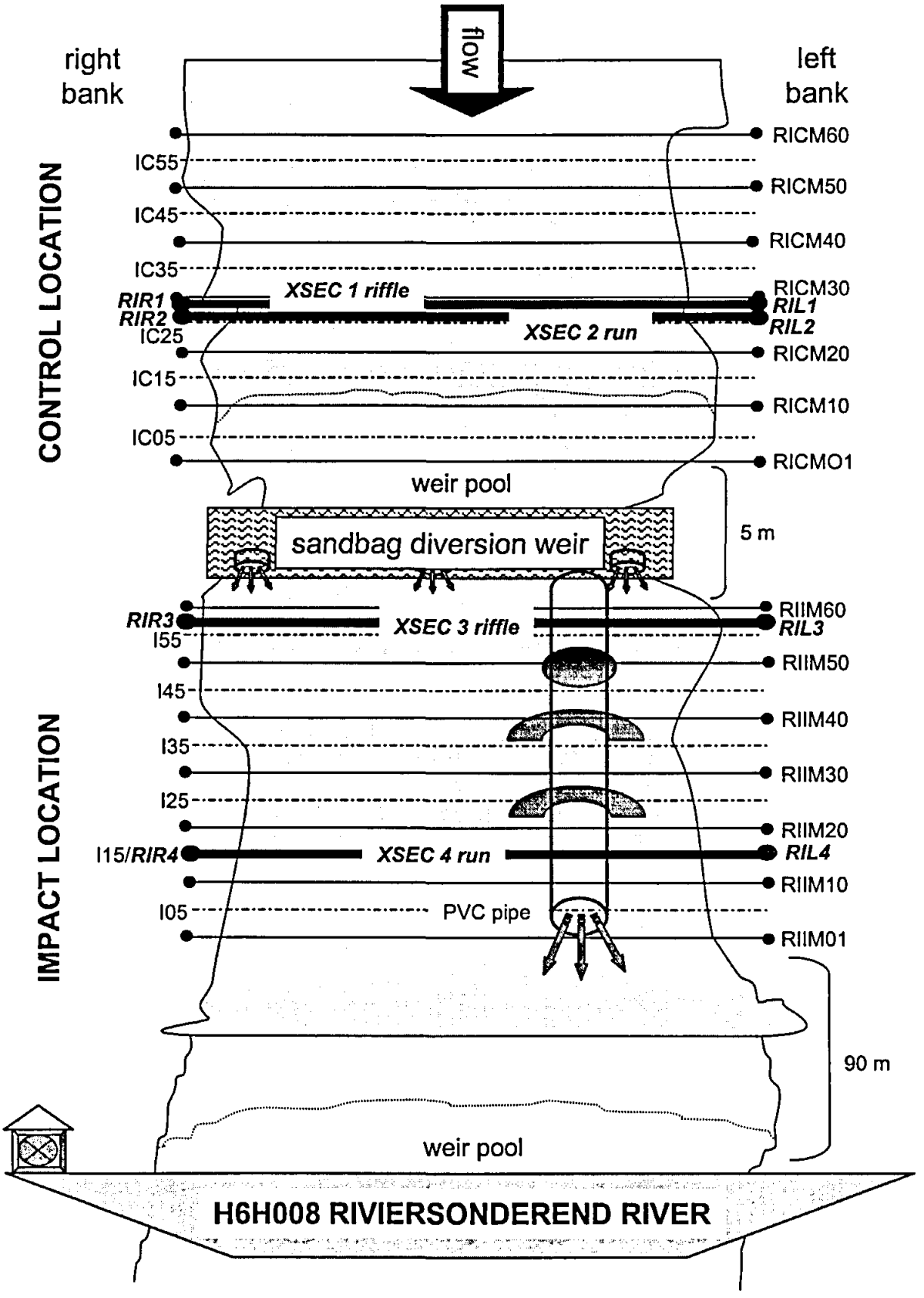


Figure 9.8D Schematic of the Riviersonderend study site, showing the 60 m control and impact locations. About 86% of the natural baseflow was diverted at a weir just upstream of the impact location. Water was reintroduced to the site below the impact location. Shaded ellipses and semi-circles represent boulders and wooden props for the single offtake pipe. The positions of all transects delimiting the locations, and the four cross-sections (XSEC1-4) are indicated. Gauging weir H6H008 (non-operational) is shown. Further explanation of site layout is provided in Sections 9.4.1 and 9.5.1, and Appendix 9.1.

The total length of river identified for each location, for each site, was within 10 to 14 times the average width of each river. This ensured that each location included a series of at least two riffle-run sequences (Bovee and Milhous 1978). Riffles and runs were identified as the dominant biotopes (see Section 9.5.1). Pools were also included in the study as they were recognised as potential refugia for BMIs. Of the three biotopes sampled for the low-flow study, riffles were considered most essential, as they tend to be the elevated areas in river reaches where flow ceases first (King and Tharme 1994).

Downstream controls, to account for the potential confounding error of consistent positioning the control locations at all four sites upstream of the experimental locations, were not used at any of the sites. However, the two locations at the Elands site acted as adequate controls to account for this problem. Although additional controls could have been put in place, it was felt that the pilot study had indicated sufficiently high inter-reach similarity on the basis of BMI and water chemistry samples collected in two locations (subreaches), to make them unnecessary (Chapter 8). Moreover, it was felt that controls downstream of the impacted locations would likely have been subject to the effects of invertebrate drift and changes in chemistry associated with any flow-related impacts experienced upstream. Lastly, data collection, should such additional controls have been included, would have been too intensive within the constraints of the study.

9.4.2 The structure of the temporary flow diversion weirs

The temporary flow diversion weirs employed in the experiment were designed in consultation with several hydraulics experts, including Prof. A. Rooseboom, Civil Engineering, University of Stellenbosch, and Mr W. Rowlston, then at Hydraulic Studies Subdirectorate, DWAF, Pretoria. Members of the Hydrology Division, DWAF, Worcester, were responsible for the erection and dismantling of the weirs.

The general approach adopted was the diversion of water from an existing gauging weir or temporary sandbag weir through a series of segmented PVC pipes, from immediately downstream of the control location through the impact location (Figures 9.8A-D). A limited number of pipes were hired by DWAF rather than purchased, as the costs of new materials were prohibitive. The pipes were located within the main river channel along the length of the impact location. Pipe segments were linked, so as to include in-channel bedrock outcrops and boulders as props to elevate pipes above the water surface (Figures 9.8B-D); additional wooden props were used for the Riviersonderend site (Figure 9.8D). The lengths of pipe ensured that the diverted water was re-introduced immediately downstream of the impact location, directly from the pipe openings (see below).

The form of the flow diversion structure was necessarily site-specific, as illustrated in Figures 9.8A-D and Plates 9.4, 9.6 and 9.8. Plate 9.2 illustrates the general characteristics of the Elands impact location, where no flow was diverted. Theoretically, a flow diversion structure should have been erected at the Elands site, which acted as a control for the presence of the diversion weirs and through which the total discharge of the river passed unimpeded (Figure 9.8A). However, the logistical difficulties, time and costs associated with the construction of the diversions rendered a control diversion impractical (Mr M. Acker, DWAF Hydrology Division, Worcester pers. comm.).

Originally, at the Du Toits and Riviersonderend sites, it was intended to use the existing gauging weirs as the points at which flow was diverted, for practical reasons. However, this was not possible at either site,

for the reasons given in Section 8.1. Therefore, at the upstream ends of the Riviersonderend and Du Toits experimental locations, a sandbag weir was constructed. The channel bottom was first sealed off across the channel with plastic sheeting, on top of which the sandbags were stacked. Offtake pipes were set within the wall of sandbags, and continued along the bottom of the river channel (Figures 9.8C and D; Plates 9.6 and 9.8). Short sections of pipe in the sandbag walls ensured that the non-diverted flow was evenly distributed within the main channel immediately downstream of each weir. At the Du Toits site (Figure 9.8C; Plate 9.6), five small diameter pipes and one large diameter pipe were used. At the Riviersonderend site (Figure 9.8C; Plate 9.8), a single, large pipe was used.

As a result of the large baseflow magnitude of the Molenaars River (Table 8.3), and as the reaches upstream and downstream of the gauging weir fulfilled all the main requirements for the experimental locations (Table 7.1), it was possible to use the existing gauging weir as the offtake point (Figure 9.8B; Plate 9.4). Two large pipes were fitted to the existing weir scour gate using a specially welded plate. Permission was granted by DWAF for the experiment to take place at the gauging weir, as the reduction in flow resulted in an artificial modification to the flow record during the impact phase of the experiment. The ongoing flow record at the site enabled a check to be kept of the proportion of flow being diverted during construction. For this site, an insufficient length of pipe was available to divert the water over the entire length of the impact location, so only *c.* 45 m of river length were impacted by the reduction in flow (Figure 9.8B).

In all cases, the construction of the diversion weirs followed a structured set of procedures aimed at minimising, as far as possible, the physical disturbance to the river channel, as this might have had detrimental effects on instream biota and water chemistry. Establishment of each weir was done within a period of at most three days, and dismantling took place within a matter of hours. Sand for the sandbags used for the Du Toits and Riviersonderend weirs was clean river sand, excavated on site from the river margins downstream of the sites. It was returned to the excavation areas when the weirs were dismantled. Construction work on the weirs was done from the banks or instream bedrock/boulders, and walking through the wetted river bed was avoided. The diversion pipes within the wetted channel were also kept out of the water (see above).

9.4.3 The nature of the experimental flow reductions

A graded series of reductions in discharge (the experimental treatments) was applied to the four sites, where each reduction represented a proportion of the natural instantaneous baseflow (see Section 10.2). The assignment of a percentage reduction in flow to a particular site was random and approximate (Table 9.3). Exact percentage reductions in discharge were not feasible, due to the difficulties associated with construction of the temporary diversion weirs, but the numbers and diameters of pipes of each of the three diversions were adjusted until an approximate reduction in the series 0%, 30%, 80% and 85% of natural baseflow was achieved.

Calculations of actual proportions of discharge, based on averages of all discharge cross-sections taken after diversion, revealed that roughly 35% of the baseflow had been taken off from the Molenaars River, 80% from the Du Toits, and 82% from the Riviersonderend (Table 9.3). The Elands River was assigned a 0% reduction in dry-season baseflow, as the control site (Table 9.3).



Plate 9.1 The control location of the Elands study site, situated upstream of the impact location and gauging weir H1H033, showing the natural character of the river.

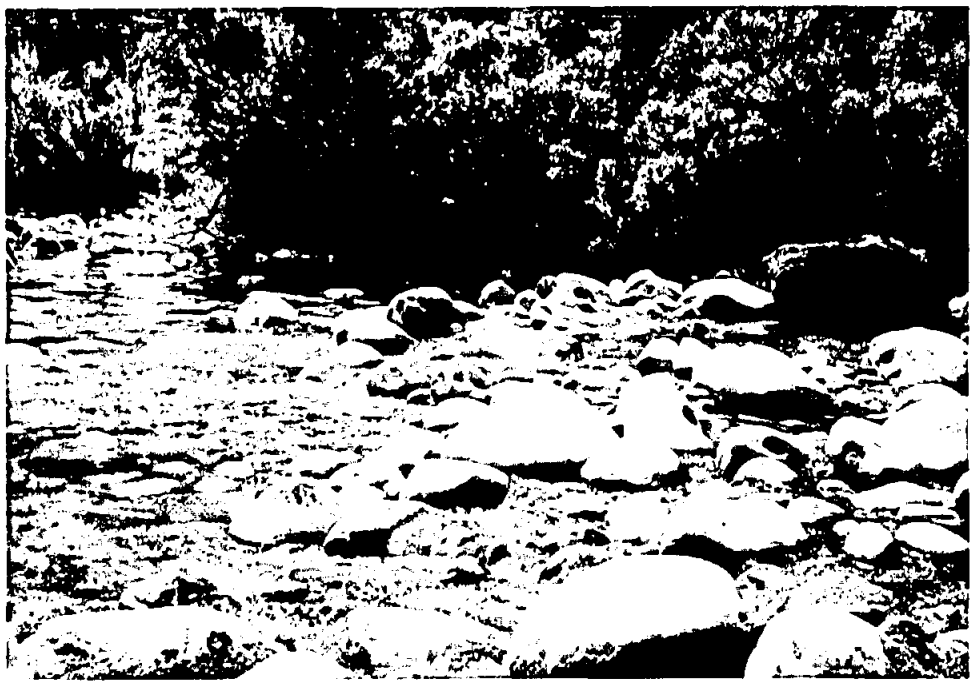


Plate 9.2 The impact location of the Elands study site, situated immediately downstream of the control location and within the same reach. Gauging weir H1H033 is located downstream of site. There was no experimental reduction in dry-season baseflow in the impact location, as the Elands represented the control site for the low-flow study.



Plate 9.3 **The control location of the Molenaars study site, upstream of gauging weir H1H018, showing the natural character of the river. Several sample markers are depicted.**

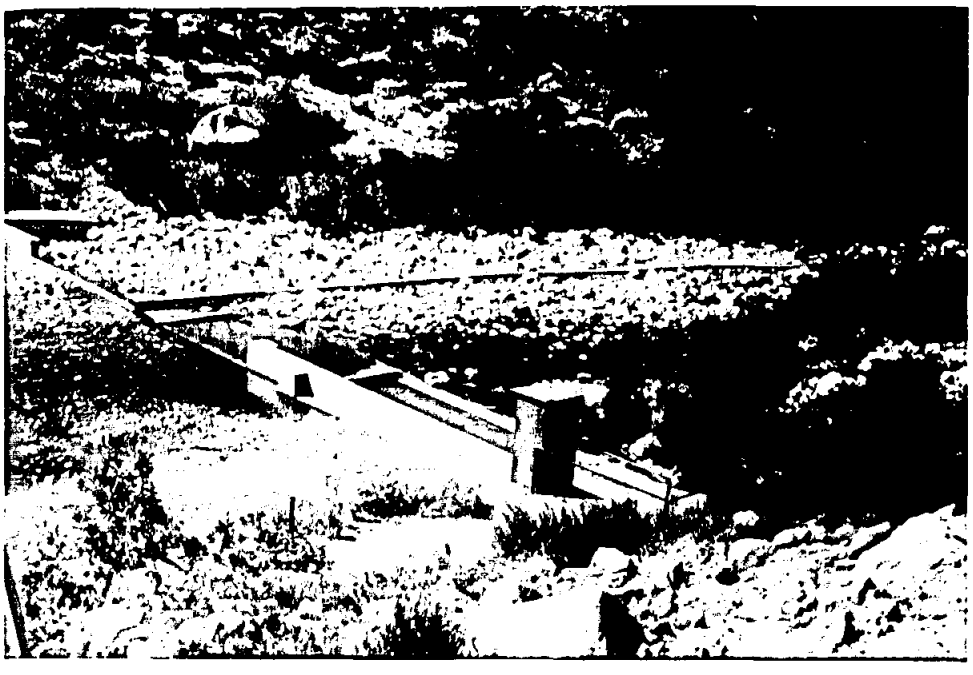


Plate 9.4 **The impact location of the Molenaars study site, situated immediately downstream of gauging weir H1H018. Diversion of approximately 36% of the natural dry-season baseflow of the river took place at this weir, as illustrated.**



Plate 9.5 **The control location of the Du Toits study site, situated just downstream of gauging weir H6H007, showing the natural character of the river. The position of a run cross-section is indicated by the survey tape.**

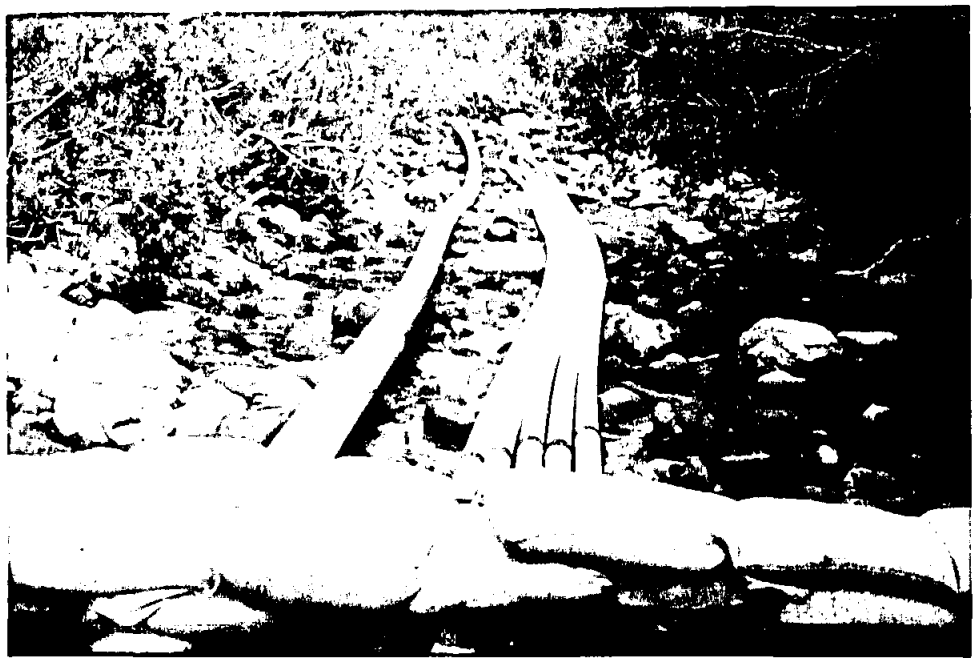


Plate 9.6 **The impact location of the Du Toits study site, situated immediately downstream of the control location. Diversion of approximately 86% of the natural dry-season baseflow of the river took place at a temporary diversion weir constructed at the upstream end of the impact location, as illustrated.**



Plate 9.7 The control location of the Riviersonderend study site, situated upstream of the impact location and gauging weir H6H008. The natural character of the river at the site is depicted. A Department of Water Affairs and Forestry surveyor is present in the foreground.



Plate 9.8 The impact location of the Riviersonderend study site, upstream of gauging weir H6H008. Diversion of approximately 86% of the natural dry-season baseflow of the river took place at a temporary diversion weir constructed immediately upstream of the location, as illustrated.

Table 9.3 The timing and duration of the pre-impact, impact and post-impact phases of the low-flow experiment. Durations are calculated on the basis of first and last sampling dates for the pre- and post-impact phases for the three experimental sites and for all phases for the control site (Elands). For the impact phase, at the experimental sites, durations are based additionally on the dates of completion and dismantling of the temporary flow diversion weirs.

SITE	PRE-IMPACT PHASE		IMPACT PHASE			POST-IMPACT PHASE	
	TIMING	DURATION (days)	TIMING	DURATION (days)	Q REDUCTION (%)	TIMING	DURATION (days)
ELANDS							
Control location	14/12/94-18/02/95	67	19/02/95-09/04/95	50	0	10/04/95-17/05/95	38
Impact location	13/12/94-26/02/95	76	27/02/95-19/04/95	52	0 (control)	20/04/95-17/05/95	28
MOLENAARS							
Control location	18/12/94-16/02/95	61	17/02/95-10/04/95	53	0	11/04/95-16/05/95	36
Impact location	18/12/94-19/02/95	64	20/02/95-03/04/95	43	35	04/04/95-16/05/95	43
DU TOITS							
Control location	05/12/94-14/02/95	72	15/02/95-08/04/95	53	0	09/04/95-12/05/95	34
Impact location	05/12/94-14/02/95	72	15/02/95-03/04/95	48	80	04/04/95-12/05/95	39
RIVIERSONDEREND					0		
Control location	03/12/94-10/02/95	70	11/02/95-07/04/95	56	82	08/04/95-11/05/95	34
Impact location	03/12/94-09/02/95	69	10/02/95-03/04/95	53		04/04/95-11/05/95	38

Care was taken to ensure that the flow of water into the offtake pipes was maximised and unobstructed, so that the percentage reductions in flow remained roughly consistent with natural upstream increases and decreases in baseflow, at each impact location over time. This also reduced the possibility of overtopping of the diversion weirs during occasional summer rainfall events.

Independent temporal replication, within the BACI design (Section 9.4.1), was achieved by sampling both locations at each site twice before the flow diversions were operational, twice during the phase of flow reduction, and twice after the diversions were removed and the natural flow regimes were reinstated (Tables 9.3 and 9.4). The time sequence of sampling was dependent both on the expected time scale of response, on natural increases and decreases in flow over the summer period, and on pragmatic considerations, such as the time required to collect the requisite number of samples from each location. The duration of the total impact (flow reduction) phase of the experiment was, therefore, highly dependent on the time required to sample each of the locations at each site, for both of the sampling trips undertaken during that phase of the experiment. However, sampling trips were staggered as best as possible, to enable fairly similar durations between sampling events at each location and site, for the purposes of direct comparison of results (see Table 9.3 for the durations of each experimental phase and Table 9.4 for sampling dates).

The low-flow field experiment was designed to represent a full dry season, and extended from 3 December 1994 to 17 May 1995, a total duration of 166 days. The dry season in the southwestern Cape is typically represented by the months of November to March, with the two months either side constituting transitional months. Theoretically, the experiment should have spanned the months of October to March to cover the entire dry season. However, requisite permission for the experiment from DWAF and a necessary test run of all field procedures on the Zachariashoek River in November 1994 (Tharme unpubl. data), delayed the onset of the fieldwork until early December. Fortuitously, late rains during 1994 resulted in higher flows than under average conditions during October and November 1994. Additionally, the onset of autumn in 1995 was later than usual, with the typical increases in baseflow occurring late in May. Indeed, the field experiment was completed before the first major winter rains commenced.

9.5 METHODS OF DATA COLLECTION AND ANALYSIS

The following section details the methods employed for both the collection of the different types of data, and the techniques of data analysis. Section 9.5.1 details the methods of data collection and analysis for the assessment of biotopes and for calculations of discharge. Sections 9.5.2 and 9.5.3 describe the methods employed for water chemistry and BMIs, respectively.

The dates of the two pre-impact, two impact and two post-impact field trips, and the numbers of samples or data sets collected in each instance, are provided for all four sites and the two locations at each of them in Table 9.4. All except one of the six sampling trips conducted from December to May was completed within a single month. In the case of the third trip, all sampling for the Du Toits and Riviersonderend Rivers and BMI collections for the other two sites were completed within February. However, biotope and cross-section data collection for the Molenaars and Elands sites extended into early March (Table 9.4), due to logistical delays experienced by DWAF.

Table 9.4 **Summary of the types of data and number of samples collected, and sampling dates, for all six sampling trips during the low-flow study. Similar sets of data were collected for each of the two locations at each of the four sites, as indicated. Data that were analysed for this report (Section 9.5) are shaded. BMIs – benthic macroinvertebrates.**

EXPERIMENTAL PHASE	ELANDS SITE			
TRIP NO.	CONTROL LOCATION		IMPACT LOCATION	
TYPE OF DATA	Sampling date(s)	No. of samples	Sampling date(s)	No. of samples
PRE-IMPACT PHASE				
TRIP 1 - DECEMBER				
BMIs	23/12/94; 27-28/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	21/12/94; 28/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	27/12/94	1 full set	27/12/94	1 full set
Biotope maps	14/12/94	1 set for 60 m	13/12/94	1 set for 60 m
Cross-sections	14-15/12/94	1 riffle; 1 run	13/12/94	1 riffle; 1 run
TRIP 2 - JANUARY				
BMIs	19/01/95	• 6 riffle • 6 run • 6 pool	19/01/95; 24/01/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	24/01/95	1 full set	24/01/95	1 full set
Biotope maps	12/01/95	1 set for 60 m	12/01/95	1 set for 60 m
Cross-sections	12/01/95	1 riffle; 1 run	12/01/95	1 riffle; 1 run
IMPACT PHASE				
TRIP 3 - FEBRUARY/EARLY MARCH				
BMIs	19/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	27/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	28/02/95	1 full set	28/02/95	1 full set
Biotope maps	02/03/95	1 set for 60 m	03/03/95	1 set for 60 m
Cross-sections	02/03/95	1 riffle; 1 run	02-03/03/95	1 riffle; 1 run
TRIP 4 - MARCH				
BMIs	16-17/03/95	• 6 riffle • 6 run • 6 pool	17/03/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	30/03/95	1 full set	30/03/95	1 full set
Biotope maps	27/03/95	1 set for 60 m	27/03/95	1 set for 60 m
Cross-sections	27/03/95	1 riffle; 1 run	27/03/95	1 riffle; 1 run
POST-IMPACT PHASE				
TRIP 5 - APRIL				
BMIs	10/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	24/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	10/04/95	1 full set	24/04/95	1 full set
Biotope maps	20/04/95	1 set for 60 m	20/04/95	1 set for 60 m
Cross-sections	20/04/95	1 riffle; 1 run	20/04/95	1 riffle; 1 run
TRIP 6 - MAY				
BMIs	07/05/95	• 6 riffle • 6 run • 6 pool	10/05/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	01/05/95	1 full set	01/05/95	1 full set
Biotope maps	17/05/95	1 set for 60 m	17/05/95	1 set for 60 m
Cross-sections	17/05/95	1 riffle; 1 run	17/05/95	1 riffle; 1 run

Table 9.4 Continued.

EXPERIMENTAL PHASE	MOLENAARS SITE			
TRIP NO.	CONTROL LOCATION		IMPACT LOCATION	
TYPE OF DATA	Sampling date(s)	No. of samples	Sampling date(s)	No. of samples
PRE-IMPACT PHASE				
TRIP 1 - DECEMBER				
BMIs	18/12/94; 26-27/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	24-26/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	18/12/94	1 full set	18/12/94	1 full set
Biotope maps	20/12/94	1 set for 60 m	19/12/94	1 set for 60 m
Cross-sections	20/12/94	1 riffle; 1 run	19/12/94	1 riffle; 1 run
TRIP 2 - JANUARY				
BMIs	22-23/01/95	• 6 riffle • 6 run • 6 pool	21-22/01/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat		As per BMIs		As per BMIs
Water chemistry	23/01/95	1 full set	23/01/95	1 full set
Biotope maps	17/01/95	1 set for 60 m	17/01/95	1 set for 60 m
Cross-sections	17/01/95	1 riffle; 1 run	17/01/95	1 riffle; 1 run
IMPACT PHASE				
TRIP 3 - FEBRUARY/EARLY MARCH				
BMIs	17/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	25/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat		As per BMIs		As per BMIs
Water chemistry	17/02/95	1 full set	24/02/95	1 full set
Biotope maps	07/03/95	1 set for 60 m	07/03/95	1 set for 60 m
Cross-sections	07/03/95	1 riffle; 1 run	07/03/95	1 riffle; 1 run
TRIP 4 - MARCH				
BMIs	15/03/95	• 6 riffle • 6 run • 6 pool	22/03/95; 24/03/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat		As per BMIs		As per BMIs
Water chemistry	29/03/95	1 full set	29/03/95	1 full set
Biotope maps	28/03/95	1 set for 60 m	28/03/95	1 set for 60 m
Cross-sections	28/03/95	1 riffle; 1 run	28/03/95	1 riffle; 1 run
POST-IMPACT PHASE				
TRIP 5 - APRIL				
BMIs	11/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	13/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat		As per BMIs		As per BMIs
Water chemistry	11/04/95	1 full set	13/04/95	1 full set
Biotope maps	19/04/95	1 set for 60 m	19/04/95	1 set for 60 m
Cross-sections	19/04/95	1 riffle; 1 run	19/04/95	1 riffle; 1 run
TRIP 6 - MAY				
BMIs	09/05/95; 14/05/95	• 6 riffle • 6 run • 6 pool	09/05/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	01/05/95	1 full set	01/05/95	1 full set
Biotope maps	16/05/95	1 set for 60 m	16/05/95	1 set for 60 m
Cross-sections	16/05/95	1 riffle; 1 run	16/05/95	1 riffle; 1 run

Table 9.4 Continued.

EXPERIMENTAL PHASE	DU TOITS SITE			
TRIP NO.	CONTROL LOCATION		IMPACT LOCATION	
TYPE OF DATA	Sampling date(s)	No. of samples	Sampling date(s)	No. of samples
PRE-IMPACT PHASE				
TRIP 1 - DECEMBER				
BMIs	18/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	10/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMI	As per BMI	As per BMI	As per BMI
Water chemistry	10/12/94	1 full set	10/12/94	1 full set
Biotope maps	05-06/12/94	1 set for 60 m	05/12/94	1 set for 60 m
Cross-sections	06/12/94	1 riffle; 1 run	06/12/94	1 riffle; 1 run
TRIP 2 - JANUARY				
BMIs	16/01/95; 18/01/95	• 6 riffle • 6 run • 6 pool	15-16/01/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMI	As per BMI	As per BMI	As per BMI
Water chemistry	18/01/95	1 full set	18/01/95	1 full set
Biotope maps	11/01/95	1 set for 60 m	11/01/95	1 set for 60 m
Cross-sections	11/01/95	1 riffle; 1 run	11/01/95	1 riffle; 1 run
IMPACT PHASE				
TRIP 3 - FEBRUARY				
BMIs	15-16/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	20-21/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMI	As per BMI	As per BMI	As per BMI
Water chemistry	16/02/95	1 full set	20/02/95	1 full set
Biotope maps	23-24/02/95	1 set for 60 m	23/02/95	1 set for 60 m
Cross-sections	24/02/95	1 riffle; 1 run	23/02/95	1 riffle; 1 run
TRIP 4 - MARCH				
BMIs	14/03/95	• 6 riffle • 6 run • 6 pool	13/03/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMI	As per BMI	As per BMI	As per BMI
Water chemistry	12/03/95	1 full set	12/03/95	1 full set
Biotope maps	21/03/95	1 set for 60 m	21/03/95	1 set for 60 m
Cross-sections	21/03/95	1 riffle; 1 run	21/03/95	1 riffle; 1 run
POST-IMPACT PHASE				
TRIP 5 - APRIL				
BMIs	09/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	14/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMI	As per BMI	As per BMI	As per BMI
Water chemistry	09/04/95	1 full set	14/04/95	1 full set
Biotope maps	17/04/95	1 set for 60 m	17/04/95	1 set for 60 m
Cross-sections	17/04/95	1 riffle; 1 run	17/04/95	1 riffle; 1 run
TRIP 6 - MAY				
BMIs	08/05/95; 12/05/95	• 6 riffle • 6 run • 6 pool	03/05/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMI	As per BMI	As per BMI	As per BMI
Water chemistry	03/05/95	1 full set	03/05/95	1 full set
Biotope maps	12/05/95	1 set for 60 m	12/05/95	1 set for 60 m
Cross-sections	12/05/95	1 riffle; 1 run	12/05/95	1 riffle; 1 run

Table 9.4 Continued.

EXPERIMENTAL PHASE	RIVIERSONDEREND SITE			
TRIP NO.	CONTROL LOCATION		IMPACT LOCATION	
TYPE OF DATA	Sampling date(s)	No. of samples	Sampling date(s)	No. of samples
PRE-IMPACT PHASE				
TRIP 1 - DECEMBER				
BMIs	04/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	03-04/12/94; 10/12/94	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat	As per BMI	As per BMI	As per BMI	As per BMI
Water chemistry	03/12/94	1 full set	03/12/94	1 full set
Biotope maps	12/12/94	1 set for 60 m	07/12/94	1 set for 60 m
Cross-sections	12/12/94	1 riffle; 1 run	07/12/94	1 riffle; 1 run
TRIP 2 - JANUARY				
BMIs	09/01/95	• 6 riffle • 6 run • 6 pool	13/01/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat		As per BMI		As per BMI
Water chemistry	13/01/95	1 full set	13/01/95	1 full set
Biotope maps	10/01/95	1 set for 60 m	10/01/95	1 set for 60 m
Cross-sections	10/01/95	1 riffle; 1 run	10/01/95	1 riffle; 1 run
IMPACT PHASE				
TRIP 3 - FEBRUARY				
BMIs	11-12/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	14-15/02/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat		As per BMI		As per BMI
Water chemistry	15/02/95	1 full set	15/02/95	1 full set
Biotope maps	22/02/95	1 set for 60 m	22/02/95	1 set for 60 m
Cross-sections	22/02/95	1 riffle; 1 run	22/02/95	1 riffle; 1 run
TRIP 4 - MARCH				
BMIs	10/03/95	• 6 riffle • 6 run • 6 pool	10-11/03/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat		As per BMI		As per BMI
Water chemistry	12/03/95	1 full set	12/03/95	1 full set
Biotope maps	20/03/95	1 set for 60 m	20/03/95	1 set for 60 m
Cross-sections	20/03/95	1 riffle; 1 run	20/03/95	1 riffle; 1 run
POST-IMPACT PHASE				
TRIP 5 - APRIL				
BMIs	08/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool	22/04/95	• 3 riffle • 3 riffle • 3 run • 3 run • 6 pool
Physical microhabitat		As per BMI		As per BMI
Water chemistry	08/04/95	1 full set	22/04/95	1 full set
Biotope maps	18/04/95	1 set for 60 m	18/04/95	1 set for 60 m
Cross-sections	18/04/95	1 riffle; 1 run	18/04/95	1 riffle; 1 run
TRIP 6 - MAY				
BMIs	04/05/95	• 6 riffle • 6 run • 6 pool	04-05/05/95; 13/05/95	• 6 riffle • 6 run • 6 pool
Physical microhabitat	As per BMIs	As per BMIs	As per BMIs	As per BMIs
Water chemistry	04/05/95	1 full set	04/05/95	1 full set
Biotope maps	11/05/95	1 set for 60 m	11/05/95	1 set for 60 m
Cross-sections	11/05/95	1 riffle; 1 run	11/05/95	1 riffle; 1 run

9.5.1 Methods of data collection and analysis for physical biotopes and discharge

Methods of data collection for biotope maps

Biotope map structure

During the first field trip, in December 1994, both the control and impact locations at each of the four sites were subdivided according to a series of transects, to produce a baseline reference grid from which biotope maps could be produced (Figures 9.8A-D). The layout of the grids also enabled referencing of the approximate positions at which BMI and water chemistry samples were taken.

For each location, a 60 m reach was marked out using a 100 m survey tape. At each 10 m interval, a survey peg was set in concrete on one river bank, where possible, or in a few instances a paint mark was used to demarcate a transect beacon. It was not possible to concrete beacons in place on both banks due to time constraints. The 10 m beacons were labelled according to a standard set of codes devised for surveying purposes (see Appendix 9.1). At 5 m intervals, survey pegs were hammered into either the left or right river bank, but were not put in concrete. The 5 m pegs were coded as intermediate pegs (I) and numbered according to distance along the subreach representing the location (Appendix 9.1). The first transect was located at a distance of 1 m from the start of the tape, and transects were numbered from 1 m to 60 m, moving in an upstream direction, for both locations at each site.

The transect-based biotope maps were created by surveying and then mapped by freehand, according to the procedures outlined below. A total of eight surveyed biotope maps and eight complementary freehand maps were produced in December (see below; Table 9.4). For the remaining five trips, minor amounts of surveying were done (see below), and a further 40 freehand maps were drawn (Table 9.4). The dates of biotope mapping trips are given in Table 9.4. The maps are presented in Tharme (in prep.).

Surveying equipment

Surveying of the biotope maps (and cross-sections, see below) was done by members of DWAF Surveying Division, Worcester, and this author, using three main types of surveying instrument: a Leica TC2002K Total Station, with an accuracy of 0.5 seconds, 1 ppm (i.e. accurate to 1 mm for every 1 km); a Nikon DT10ALG, with an accuracy of 3 s, 3 and 5 ppm (i.e. accurate to 3 mm for every 5 km); and a Pentax PSC, with an accuracy of 10 s, 5 and 7 ppm (i.e. accurate to 5 mm for every 7 km). Circular type prisms, accuracy 5 mm and zero correction 3 mm, mounted on a ranging rod, were used with all three surveying instruments. Data recording was done using a Psion Electronic Organiser II, Model LZ, with recordings booked by the surveyor using the program Organisurv.

Biotope mapping procedures

Biotope mapping of the diversity and surface areas of biotopes was undertaken in two complementary ways. Firstly, during the first field trip in December 1994, a detailed biotope map of each location, at each site, was surveyed, so as to provide a template for use during subsequent field trips. As it was only possible to survey the biotopes in each location at a fairly coarse degree of resolution, and only on the first field trip (due to cost and time constraints), a more detailed biotope map was drawn freehand, making use of the same grid of transects for reference points. The freehand map included finer-scale details such as the precise location of

the wetted channel edge, positions of exposed boulders and locations of smaller biotopes. Freehand mapping took place at the same time as the surveying of the biotope map and cross-sections.

On subsequent field trips, for the surveyed maps, only the waters' edges (WEs) and water surface elevations (WSEs) at 10 m intervals were surveyed in, while dimensions and locations of biotopes were mapped freehand, using the map from the previous field trip for reference.

The survey team comprised a surveyor and two persons holding ranging rods with prisms. One surveyor surveyed in the major points delimiting the structure of each biotope map, while the author surveyed in the biotopes, using the same instrument and on the same day. Communication of a system of standardised survey codes developed to facilitate the surveying procedure (Appendix 9.1), was effected using a system of two-way radios.

For the biotope mapping exercise, for each location, the left and right WEs of the channel (left and right banks defined facing in a downstream direction) were surveyed, at every 5 m along the transect grid. In addition, at every 10 m transect, three representative WSEs were taken at varied intervals across the wetted channel width. All of the beacons defining the transects of each biotope map were also surveyed. The main biotopes, defined visually and hence subjectively, on the basis of flow type and substratum composition (see below), were surveyed as patches, by taking spot shots (surveying measurements) that defined the outer boundaries of each patch. The spatial extent of large bed elements, such as bedrock slabs and large boulders were surveyed, while spot shots were taken at the centres of smaller boulders, the dimensions of which were then measured using a survey tape. The various survey codes used to identify physical habitat features, including biotopes, are provided in Appendix 9.1.

The survey data recorded were the following: the site and location name; date; an automatically generated number for each spot shot or specific code for a beacon (Appendix 9.1); the biotope code assigned to each spot shot (Appendix 9.1); and X and Y (in combination, representing horizontal distance) and Z (representing elevation) survey co-ordinates from a common datum, for each surveyed point.

The biotope map surveyed for the control location at a site was linked to the map similarly produced for the impact location, by means of benchmarks. The established sets of benchmarks were used on all subsequent trips as points of reference. The benchmarks were particularly important as the survey co-ordinates were not linked by DWAF surveyors to real land elevations (m) within the national grid system.

Identification and classification of biotopes

A biotope, in its broadest sense, may be defined as an homogeneous environment that satisfies the habitat requirements of a biotic community (King and Tharme 1994). It thereby recognises the integration of those physical, chemical and biological features that define the habitats of the individual species within the community. Wadeson (1995) defines a hydraulic biotope as the habitat assemblage with a characteristic range of temporarily variable hydraulic and substratum characteristics that can be associated with morphological units (i.e. the basic structures comprising the channel morphology, such as rapids and pools).

Hydraulic biotopes, also referred to as physical biotopes (Padmore 1997), as the focus is on physical features, were visually assessed in the field, primarily on the basis of the definitions presented in King and

Tharme (1994) (see below). However, cognisance was also taken of the fact that definitions of physical biotopes were under continual revision during the time of the present study (see Wadeson 1995; Rowntree 1996; Padmore 1997). It should be noted that all biotopes are spatially and temporally variable, particularly in relation to discharge magnitude and channel geomorphology, and may be hydraulically 'transitional' in nature. **Transitional biotopes** were defined by this author as areas that were difficult to visually categorise in the field without hydraulic measurements, as they tended to fall somewhere between well-defined types of biotopes. For example, riffle-run and pool-run transitions were sometimes observed. Biotopes that spanned the entire channel width, or represented most of it, were referred to as **transect biotopes,** according to terminology in Padmore (1997) (see Figure 9.9). In many instances, these transect biotopes exhibited smaller areas along them that represented other types of biotope. Such biotopes, within a transect biotope, were referred to as **cell biotopes** (Padmore 1997; see Figure 9.9). On the basis of more recent work outside this study, transect biotopes might be more appropriately classified as morphological units, and the cell biotopes would then represent single biotopes which might or might not contain different flow types and substrata. Work is underway in a current WRC project to characterise hydraulically and biologically such flow type-substratum combinations (Dr J.M. King and Ms D.M. Schael, FRU, University of Cape Town pers. comm.).

Riffles:

Riffles are high points of coarse sediment deposition in the long profile of the undulating bed, characterised by a low depth to substratum ratio. They exhibit shooting flow with a steep, broken water surface (visible flow turbulence). The substratum is predominantly cobble and boulders, with little deposition of fines. There is generally a noticeable change in bed slope from riffle head to foot. Riffles migrate longitudinally with changes in flow. They can transform into a rapid at higher flows and a run at lower discharges.

Runs:

Runs are characterised by tranquil, smooth flow with velocity sufficient to cause some surface disruption (but not a turbulent, clearly broken water surface). They have variable substrata, often small or large cobbles and/or boulders. Runs exhibit a high depth to substratum ratio, and water depth is variable. They are typically located as areas of transition between pools or rapids, and riffles. There is no apparent change in river bed slope along the length of the run. At very low discharges, runs may transform to 'trickle runs' or pools, and at high flows to rapids. 'Trickle runs' are patches of slow, shallow flow, where the flow is between, and not submerging, most substratum elements.

Pools:

These biotopes are characterised as shallow to deep, slow-flowing (low velocity) areas with little or no disruption of the water surface (except possibly at a pool head or foot). Pools are low points within the long profile of the undulating bed. The accumulation of fine sediment is often promoted by low velocities, which are sometimes in association with deep water. Pools have variable substrata, ranging from bedrock to sand. They typically occur between riffle units, either in midchannel or along the channel margins. Pools may become entirely isolated from the main channel at very low discharges, as standing bodies of water with zero velocity, due to a loss of connectivity of wetted area. At high flows, scouring action and eddy effects can occur and pools may transform to pool/run transitions or runs.

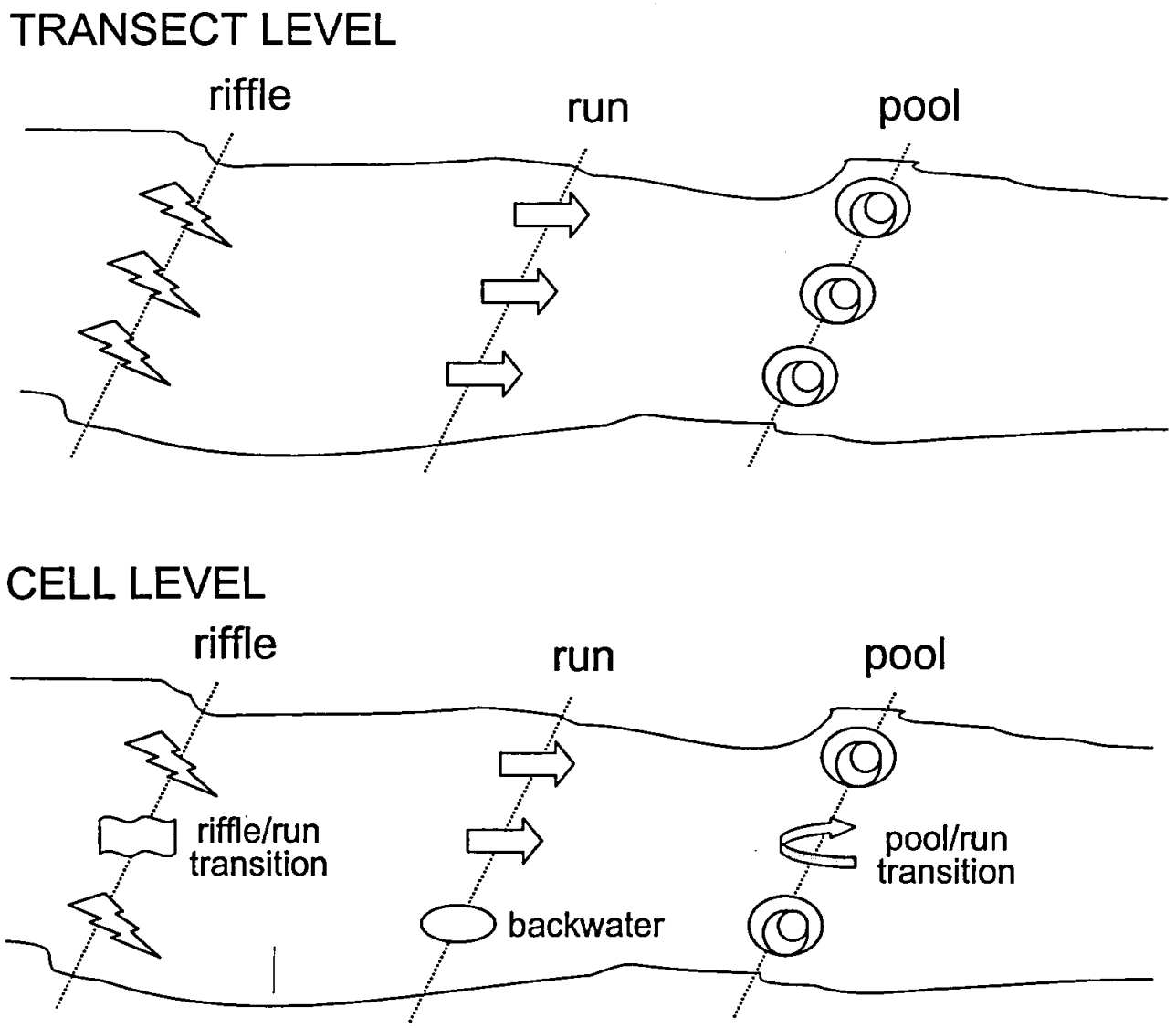


Figure 9.9 **Schematic illustrating the identification of physical biotopes at the transect and cell levels within a river reach. 'Transect biotopes' represent the dominant biotope type across the full channel width. Within the wetted width, other smaller, different biotopes might be present, termed 'cell biotopes' (modified from Padmore 1997).**

Backwaters:
Backwaters are defined as 'hydraulically detached' alcoves that tend to occur along the river margin, often resulting from the isolation of small side, flood channels when discharge decreases. There is typically no through-flow of water, and water tends to enter and leave using the same route. Depth is variable, while velocity tends to be low, often zero. Backwaters exhibit a wide range of substratum types, though they are generally dominated by cobbles/gravel and areas of deposition of fine material, such as sand, silt and organic detritus. They can transform to runs at high flows. This biotope is considered by some authors to be a type of pool.

Methods of data analysis for biotope maps

The survey data were downloaded by DWAF survey personnel, and basic surveying calculations were done on a PC using the program Survey Assistant. Preliminary plots of maps and the positions of cross-sections were created using Model Maker and Ultimate Map Maker (CAD) programs.

Composite biotope maps incorporating both surveyed and freehand data are being digitised using Geographical Information System programs, to calculate the reach-scale proportions of biotopes at different discharges (Tharme in prep.). These analyses are beyond the scope of this report.

Methods of data collection for cross-sections

For each of the six sampling trips (Table 9.4), two cross-sections were established in each location, for each site, using both tape and surveying techniques (see below). One cross-section was located across a riffle and the other across a run, as indicated in Figure 9.8A-D. This represented 16 cross-sections per sampling trip and an overall total of 96 cross-sections. Where possible, cross-section locations were established so as to coincide with transects delimiting the biotope map, to minimise surveying effort.

The cross-sections were used to provide the following information:

- Discharges, and stage-discharge relationships for the two, main 'transect biotopes' (*sensu* Padmore 1997). That is, the biotopes represented by the full width of each cross-section, namely riffles and runs.
- Changes in hydraulic variables across major biotopes specifically water depth and velocity, with natural and unnatural increases and decreases in discharge.
- 'Cell biotope' (*sensu* Padmore 1997; the various smaller-scale biotopes across each cross-section) dynamics, with natural and experimental changes in discharge. For this purpose, the proportions of different biotopes were calculated from recorded data on the distance (m) each biotope represented across each cross-section. Data from the two cross-sections in each location were then combined.
- Information on substratum composition.

Cross-section headstakes were located as far from the wetted channel as possible, while still enabling adequate visibility for surveying. They were secured in concrete and labelled according to specific survey codes, which included the site, bank and cross-section number (Appendix 9.1). For example, codes DUR1 and DUL1 indicate the cross-section headstakes for cross-section 1 on the right and left banks of the Du Toits River, respectively (Figure 9.8C). Cross-sections 1 and 2 were located in the control location and cross-sections 3 and 4 in the impact location, with numbering from upstream to downstream (Figures 9.8A-D). Headstakes were cross-linked to enable their re-establishment, if necessary at a later stage.

It was attempted to ensure that runs were suitable for developing step-backwater calculations for the purposes of hydraulic simulation modelling. All the survey data were recorded such that they can be used in hydraulic simulations within PHABSIM II, IFIM (Milhous *et al.* 1989), and the point hydraulic data can be incorporated into the same data sets as those collected with the biological samples. Furthermore, each cross-section was placed in the middle of a biotope or at its upstream end, because riffle areas particularly (as hydraulic controls) tend to migrate hydraulically longitudinally with increases or decreases in discharge.

A rating was made by the author of cross-sections of potentially high or low accuracy for measurements of discharge and WSEs.

The following data were surveyed-in (see above for the surveying equipment used) or recorded at each cross-section:

- The dominant biotope of each cross-section ('transect biotope').
- Fixed-point photographs of the cross-section at the recorded discharge.
- Horizontal survey tape distances from the cross-section headstake to each surveyed point.
- Generally, five WSEs across the channel. These were taken by touching the base of the surveying rod on the water surface. WSEs were fairly evenly spaced and taken at points where flow was least turbulent, to ensure the most accurate measurements possible.
- Locations of the WEs on the right and left banks (facing downstream).
- Typically 20-30 spotshots of bed elevation across each cross-section for the wetted channel. Spotshots were generally at set increments across the survey tape, and were scaled according to river width. However, spotshots were close together near the WEs of the channel both instream and on the bank, as this is the area where the most obvious changes in wetted area with discharge were expected to occur. Additional spotshots were also taken where distinct changes in hydraulic characteristics were apparent.
- Water depth (to the nearest 0.01 m), measured at each point using a top-setting wading rod.
- Average velocity (velocity at 0.6 of the water depth from the surface) and near-bottom velocity (velocity recorded directly above the substratum), measured at each point using a calibrated Scientific Instruments Price AA current meter fitted with a standard or mini bucket wheel. In shallow areas of extreme turbulence and rapid flow, three replicate readings were taken. In most instances, relatively slow flow, shallow depths, and flow between large bed elements necessitated the use of the mini bucket wheel. Measurement of velocities at very low flows was problematic. As a result of the limitations of the equipment for velocity measurement, where water depth was ≤ 0.06 m, near-bed velocity was assumed equivalent to average velocity.
- Visual estimates of proportions of substratum particles of various sizes (β axis) using the standard Wentworth grade scale (as given in King and Tharme 1994).
- Visual categorisation of 'cell biotope' type at each surveyed point on the wetted cross-section.
- Large, instream substratum elements recorded both in terms of their dimensions and the distance they occupied along each cross-section.
- Locations of marginal and instream vegetation.
- Proportions of silt, organic/inorganic detritus and algae covering the surface of the substratum.
- Proportions of instream and overhead cover.

- The presence of large substratum elements located upstream, sufficiently close to the cross-section that they are probably exerting an influence on downstream hydraulics.

Calculations of discharge

Calculations of flow proportions representing the flow reductions at the diversion weirs

Prior to initiation of any sampling in the experimental impact phase, at least two discharge cross-sections were established at suitable places upstream and downstream of the temporary diversion weirs at the Du Toits and Riviersonderend sites, immediately following their construction. Cross-sections were placed in channel sections where the most accurate discharge measurements could be obtained, given the low flows. The calculated discharges (Equations 9.1 and 9.2) were used to establish whether or not the weirs were able to divert the approximate, pre-determined proportions of flow identified for each site (Section 9.4.3). For the Molenaars diversion, the continuous data record at the DWAF gauging weir was used for the same purpose and it showed that roughly the correct proportion of flow reduction was taking place. In the case of the Du Toits site, a greater proportion of flow than required was being diverted according to the calculated discharges, so one diversion pipe was eliminated. However, as a result of experimental error in discharge measurement, the resultant proportion of discharge diverted from the Du Toits impact location was actually highly similar to the experimental flow in the Riviersonderend impact location (at about 86%), rather than at the expected lower value of 80%.

Discharges at riffle and run cross-sections

Discharges were calculated for each of the 96 experimental cross-sections (Table 9.4). For the calculations, the total discharge for each cross-section was calculated using the partial section concept (Bovee and Milhous 1978). Each partial section or hydraulic cell across the cross-section was defined by the distance between the vertical at each point measurement of depth and average velocity, extending half the measured tape distance (or a specified distance) to the verticals on either side. The discharge through each partial section (q_1) is calculated using the following equation:

$$q_1 = a_1 \times v_1 = w_1 \times d_1 \times v_1 \qquad \textbf{Equation 9.1}$$

Where:

q_1 = the discharge through a cell

a_1 = the area of the cell

v_1 = mean column velocity (0.6-depth from surface), measured at the vertical

d_1 = mean depth of the cell, measured at the vertical

w_1 = width of the cell = the sum of half the distance, or a specified distance, to each of the two adjacent verticals.

The total discharge through the cross-section (Q) is then calculated as the sum of all the individual cell discharges:

$$Q = \Sigma q_i \qquad \textbf{Equation 9.2}$$

Discharge calculated in the above manner is instantaneous, that is, it represents the discharge in the river at the recorded time of day. This is in contrast to an average daily discharge (e.g. derived from the continuous record at a gauging weir) that represents the average flow over a 24 h period.

For all instances where discharges were required for analyses, only the discharges calculated from cross-sections of runs were used, rather than averages of riffle and run discharges. This was done for three main reasons:

- Experience in hydraulic measurement at low flows (this author) has shown that it is considerably more difficult to obtain accurate measurements of depths and velocities (and WSEs) in riffles than in runs. This is supported by literature that indicates that the hydraulics of riffles are more complex than runs, particularly as bed element roughness is typically very high (King and Tharme 1994; Wadeson 1995; Padmore 1997).
- Visual assessment of the degree of variability in discharge measurements from riffles compared with those from runs (using the data from this study) showed that estimates for runs were less variable than estimates for riffles. Riffles also tended to cause overestimates of discharge, although the trend was not consistent (see also Padmore 1997). Moreover, particularly with riffles, it was evident that measurements of discharge were increasingly less accurate as flow decreased.
- Riffle cross-section 3 of the Riviersonderend site (Figure 9.8D) was shown to produce highly imprecise estimates of discharge. Comparison of calculated discharges before and after the known baseflow reduction of *c.* 86%, for both sampling trips during the impact phase of the experiment, showed that the run discharge for the impact phase reflected the amount of water diverted, but that the riffle discharge was consistently an order of magnitude less precise (0.101 $m^3 s^{-1}$ as compared with 0.017 $m^3 s^{-1}$).

9.5.2 Methods of data collection and analysis for water chemistry

Sets of water chemistry data were collected in both the control and impact locations at each of the four study sites, for each of the six main sampling trips (Table 9.4). A total of 48 sets of physical and chemical data were collected. As a result of financial constraints, it was not possible to install chemistry data loggers at each location and site, so only instantaneous measurement of chemical variables was possible, with its inherent limitations. The time of day at which water chemistry sampling took place was recorded. As the main emphasis of the work was to identify any large scale changes in variables, water chemistry data were recorded from flowing sections of the main channel, even during the period of unnaturally low flows. In the latter instance, data were also recorded in isolated and/or non-flowing (standing) areas of the channel (both impact and control locations) to gain an understanding of possible extremes in the ranges of variables. This second set of measurements was only possible for system variables that could be directly recorded in the field.

Field-based methods of collection of physical and chemical data

Temperature

Instantaneous temperature was measured in the field using a Syland Simplair Model F5SP oxygen meter, accurate to ± 0.1 °C, during specific trips to collect water chemistry data. Readings were cross-checked with measurements obtained using a standard mercury thermometer, accurate to ± 0.5 °C. Measurements were made in the three main biotopes, namely riffles, runs and pools, and additional measurements were made in standing water where it occurred during the impact and recovery phases of the experiment (such areas were not encountered during the pre-impact phase).

Longer-term, diel ranges in temperature in each location were recorded using minimum/maximum thermometers. The thermometers were consistently located in the same place in each location and in the same biotope for all sites, over the entire duration of the field experiment. This was done to minimise within-location variability as far as possible. Readings of maximum, instantaneous and minimum temperature were taken whenever each location was visited.

pH

pH was measured on site with a Crison Portable 506 field pH meter, accurate to 0.01 pH units. The pH was recorded in flowing water within the main channel for all sampling trips, and was also recorded for patches of standing water where these occurred.

Conductivity

Conductivity was measured on site with a Crison Conductimeter Portable 523 field meter, with built-in temperature compensation of 25 °C. The meter is accurate to 0.1 μS cm^{-1} (accuracy > ± 0.3%; reproducibility > 0.2%). Conductivity values were recorded as μS cm-1 and then converted to mS m^{-1}.

Dissolved oxygen

Instantaneous measures of DO were obtained on site, using a portable Syland Simplair Model F5SP oxygen meter with inbuilt compensation for altitude.

Turbidity

As project constraints prohibited the purchase of a turbidity meter, a relative index of turbidity was obtained by visual estimation.

Nutrients, ions, total dissolved solids and total suspended solids

Water for laboratory analyses of nutrients, cations and anions was pre-filtered through an 80 μm-mesh sieve to remove large particulate matter, and then filtered through pre-combusted and pre-weighed Whatman GF/F glass microfibre filters, of pore size 0.7 mm.

A measured volume of water, approximately one litre, was similarly filtered for laboratory analysis of total dissolved solids (TDS), and the filter paper was used for analysis of total suspended solids (TSS).

All filtered water, except that for analysis of ammonium, was bottled in polythene containers that had been pre-cleaned in 5% phosphate-free ExtranR solution, and rinsed in deionised and then double-distilled water.

Samples for analysis of ammonium were stored in HCℓ-washed glass vials, within polythene containers. All water samples were kept frozen while in the field, and in the laboratory until the time of analysis. Analyses took place within a maximum of one month of field collection to minimise contamination through long-term storage.

Laboratory-based analyses of water chemistry samples

Anions

Concentrations of sulphate ($SO4^{2-}$), chloride ($C\ell^-$), and fluoride (F^-) were measured using the techniques of high-performance ion exchange chromatography as described in Haddad and Jackson (1990). The Dionex Ion Chromatograph of the Department of Geology and Geochemistry, University of Cape Town was used, equipped with a conductivity detector and an automated computer-linked output facility. Detection is by conductivity with chemical eluent suppression. The anions were separated on an HPIC-AS4A anion exchange separator column, with a carbonate/bicarbonate buffer eluent. Standard solutions were made up, and frozen until required. Calibrations were based on peak height, and results expressed in ppm (mg ℓ^{-1}), with an estimated accuracy greater than 0.005 ppm. Results were then converted to mmol ℓ^{-1}. The percentage error on repeat runs with the same standard is estimated at approximately 10%, and is possibly greater for very low concentrations (< 0.5 ppm). F^- levels were mostly below detection limits and the variable was therefore excluded from further analyses.

Total alkalinity

Total alkalinity was measured by titration of samples with 0.005M HCl (methyl orange indicator), according to the method prescribed by Golterman *et al.* (1978). Standardisation was against NaOH, titrated with 0.005M oxalic acid (phenolphthalein indicator). Three replicate titrations were performed for each water sample. The precision of the method is estimated at 2% at TA = 1 mmol ℓ^{-1}, and 2-10% at TA = 1-0.1 mmol ℓ^{-1} (Golterman *et al.* 1978). Results were obtained as mg ℓ^{-1} $CaCO^3$, HCO_3^- (bicarbonate) and expressed as mEq ℓ^{-1}. Phenolphthalein alkalinity was not encountered.

Cations

Concentrations of potassium (K^+), sodium (Na^+), calcium (Ca^{2+}), magnesium (Mg^{2+}) were measured in a similar manner to the anions by ion exchange chromatography, but using an HPIC-AS4A cation exchange separator column and the appropriate eluent. The standards used were previously frozen. Calibrations were based on peak area, and results expressed in ppm. The percentage error on repeat runs with the same standard is estimated at approximately 10%, and is possibly greater for very low concentrations (< 0.5 ppm).

Total suspended solids

Each GF/F filter of known pre-filtration dry mass, through which a known volume of water had been filtered in the field, was dried at 60 °C for 48 h and then reweighed. The filter was weighed using a Mettler AE 100 laboratory balance with a readability and reproducibility of 0.1 mg. The difference in dry mass pre- and post-filtration in relation to the volume of water filtered constituted a measure of TSS in mg ℓ^{-1}, after a correction factor was applied to account for losses in mass as a result of field handling procedures. The organic fraction of TSS was calculated as a percentage for the sample filters, using the difference in mass before and after combustion of the dried filter at 450 °C for 4 h.

Unfortunately, a departmental fire resulted in the loss of many of the TSS filter papers, precluding the use of the data in this report.

Total dissolved solids

A known volume of water, filtered in the field, was evaporated at 60 °C from preweighed lightweight glass beakers. Once all the water had evaporated, the beakers were reweighed, using a Sartorius precision laboratory balance accurate to 1 mg, and the difference in mass represented the concentration of TDS, expressed in mg ℓ^{-1}.

Nutrients

Nitrates (NO3-N), nitrites (NO2-N), and phosphates (PO4-P) were analysed using a TECHNICON AutoAnalyser II (AA), Seaweed Research Unit, Botany Department, University of Cape Town. The principles of the methods employed are outlined in Mostert (1983), while the methods had been modified by the CSIR as described in Windt (1993) to improve detection limits. Results were expressed as either mg ℓ^{-1} or mmol ℓ^{-1} of the nutrient atom, i.e. NO3-N, NO2-N and PO4-P, and were converted by the author to µmol ℓ^{-1}. For all three variables the detection limit was 1 µg ℓ^{-1} (Windt 1993).

It was not possible to analyse ammonium (NH4-N) samples on the AA, due to poor levels of detection at the time of this study. Consequently, ammonium was analysed in a similar manner to the anions by ion exchange chromatography, but using an HPIC-AS4A cation exchange separator column and the appropriate eluent. Ammonium standards were made up freshly, on a daily basis, while all other standards used were previously frozen. Calibrations were based on peak area, and results expressed in ppm. Percentage errors on repeat runs are similar to those for analysis of anions. Ammonium values were below the detection limit of 0.2 ppm (0.2 mg ℓ^{-1}) in all instances.

Reactive silicon

Reactive silicon (SiO_2-Si) was determined using the AA (see above). The method employed is a standard CSIR procedure, and is outlined in Windt (1993). Results were expressed as mg ℓ^{-1} and were converted to µmol ℓ^{-1}.

Metals

Water samples were not analysed for any metals, due to project constraints.

Assessment of the quality of results for certain chemical variables

A cation-anion balance method, as recommended in Day (1990), was used to check that the sum total concentration of anions matched that of cations. In three instances where the difference in the total anions versus cations was greater than 0.1 mmol ℓ^{-1}, a spare water sample was used for reruns of both anions and cations. An additional three samples were also rerun as controls. The balance method was then successfully reapplied.

The balance between major ions and TDS was also checked, using the method recommended in Day (1990).

Methods of analysis of water chemistry data

Data on all measured chemical variables were plotted according to the actual days on which the data were collected, for visual interpretation of trends. For the variables where data were collected both in a flowing section of the mainstream and in isolated, standing pools, error bars denoting upper and/or lower limits were included where the difference between the two measured values was ≥ 10%. Ten percent was an arbitrarily designated threshold. For temperature, data were recorded as falling into one of four categories according to time of day, as follows: (1) 06h00-09h00; (2) 09h01-12h00; (3) 12h01-15h00; (4) 15h01-18h00. This information was not used for the present report. Statistical analyses of observed differences in chemical variables with changes in flow are reported on elsewhere (Tharme in prep.).

9.5.3 Methods of data collection and analysis for benthic macroinvertebrates

Methods of data collection

For all locations, and for all six sampling trips, six BMI samples were collected from each of three biotopes, namely riffles, runs and pools (Table 9.4). These data constituted the main BMI data set. An overall total of 864 samples was thus collected for the low-flow experiment, with each sample comprising a top, bottom and underneath part (a total of 2592 sample parts; see below). Of these samples, a total of 144, comprising a total of 432 individual sample parts, were analysed for the present report, as indicated in Table 9.4. The BMI data for this report were collected during the December, February and April sampling trips and only samples from riffles and runs were examined.

Methods of collection of benthic macroinvertebrates for the main data set

Benthic macroinvertebrate samples were taken from individual stones, where stones ranged in size from large gravel to medium cobble (about 64-256 mm); the majority of samples were collected from small cobbles (range 64-128 mm, according to the standard Wentworth grade scale; King and Tharme 1994). For each stone, the BMI sample was divided into three separate parts, as illustrated in Figure 9.10, namely: (1) Top of stone (T); (2) Bottom of stone (B); (3) Substratum underneath the stone (U). The stone top was routinely sampled first, by scrubbing the stone surface and collecting the BMIs using either a small or large 80 µm-mesh, sampling net with a terminal collecting bottle (designed by this author) held directly over and downstream of the stone, or a box sampler. Physical microhabitat data were then collected (see section 9.5.4). A medium-sized box sampler was then put in place around each stone, which was carefully lifted and its bottom surface area scrubbed. BMIs were collected in the 80 µm-mesh, collecting bottle of the box sampler. The substratum underlying each stone was sampled immediately after the stone bottom, using either the hand held sampling net or a modified, small box sampler (designed by this author). The surface of the substratum that had been directly in contact with each stone was scrubbed, providing a semi-quantitative sample (Figure 9.10). In contrast, top and bottom stone samples were fully quantitative, according to surface area (Figure 9.10; see below). For each individual stone, the most appropriate combination of sampling devices was decided upon before sampling. The choice of sampling device depended on the position and size of each stone. Net mesh size was standard with all samplers at 80 µm.

The BMIs from parts T, B and U of each sample were stored in separate collecting bottles, and were preserved in a mixture of 8% buffered formalin diluted to 4% using river water.

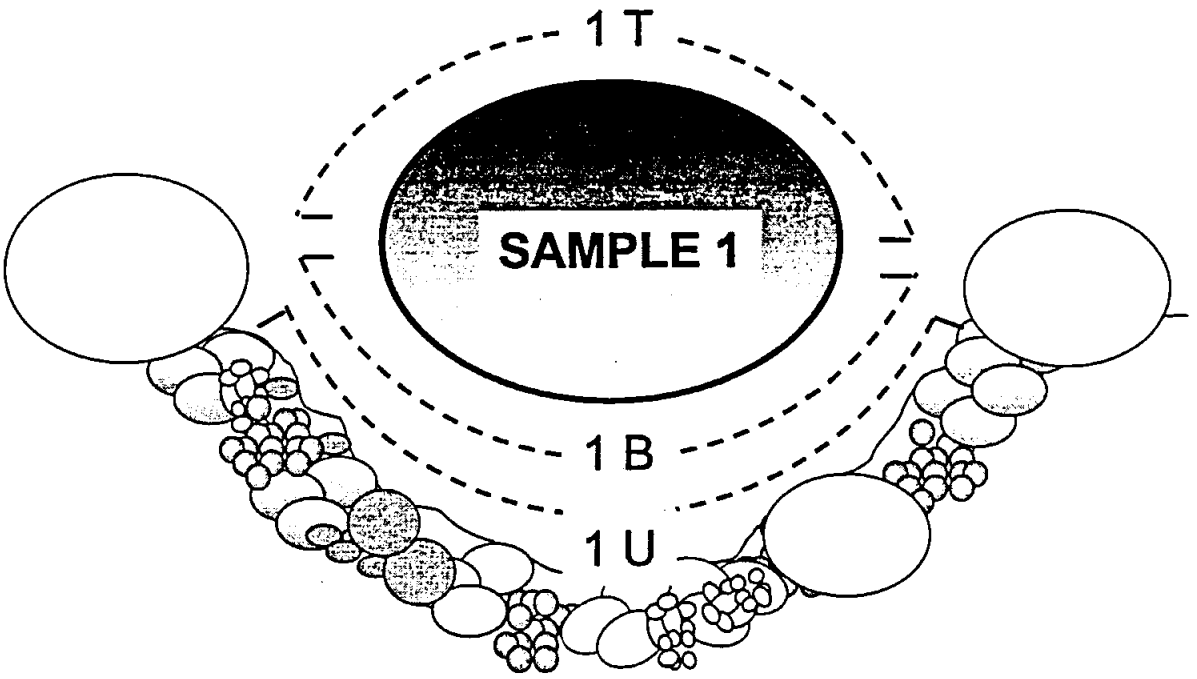


Figure 9.10 Schematic illustrating the division of each sample stone into three separate sample parts, from which benthic macroinvertebrates were collected. Sample 1 is divided into: 1T - top stone surface; 1B - bottom stone surface; 1U – the surface of the substratum underlying the stone.

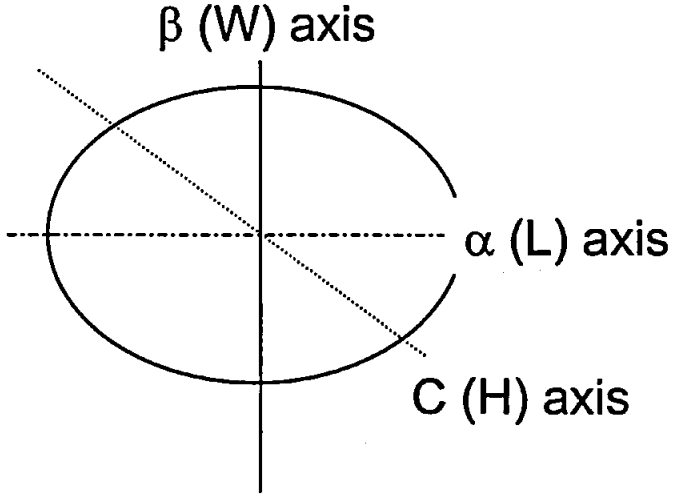


Figure 9.11 Schematic illustrating the three axes measured (in mm) in the field for each sample stone in order to calculate surface area (Section 9.5.3): L – length; W – width; H - height. The width or β axis was used to determine stone size according to the standard Wentworth substratum grade scale.

On return to the laboratory, all fixed BMI samples were transferred from 4% formalin to 70% ethanol for long-term preservation. For each sample part, the BMIs were picked from the organic detritus (see below for assesment of percentage organics). The set of samples used in this report (see Table 9.4) was not size-fractioned, although subsequent samples were divided according to the following mesh sizes: 950 μm; 500 μm; 250 μm; 125 μm and 80 μm (Tharme in prep.). BMIs were identified to the taxonomic level of family, or higher taxon, and counted, using a Nikon stereo dissecting microscope. Subsampling was undertaken only for sample parts with very high abundances. In such instances, a macrosort was performed, whereby large individuals clearly visible with the naked eye, were picked from the sample part for the duration of an hour. The remaining sample was divided into four, using a numbered 12-compartment subsampler (designed by this author) from which the animals of three compartments were sorted using a microscope (a 25% microsort). The identification of subsampler compartments was based on a system of selection of random numbers.

Although certain groups of taxa, such as the Chironomidae, have been further identified to genus- or species-level (Tharme in prep.), these species-level data are not presented in this report. For individuals of each taxon identified to family or higher level, and for each sample part, a unique catalogue number was assigned, according to the recommendations of the Curator of Freshwater Invertebrates, Albany Museum, Grahamstown (Dr F. de Moor pers. comm.).

For the purposes of quantitative comparisons of animal abundances among samples (and parts), numbers of individuals of each taxon were multiplied up to the minimum common unit surface area of 0.1 m^2. Multiplication factors were based on calculations of the top and bottom surface areas of the sampled stones, and of the surface area of substratum directly underlying each stone (Figure 9.10), using measurements of the α, β and c axes (Figure 9.11).

The results of three methods of calculation of stone surface area were compared in order to select the most suitable available method. The first method of calculating surface area, based simply on the multiplication of the L and W axes (Figure 9.11) consistently produced the lowest surface area estimates, while the use of an equation providing a surface area of spherical approximations (see Graham *et al.* 1988), provided the second lowest estimates. Graham *et al.* (1988) proposed a third method as the most accurate one of a number they reviewed, including spherical approximation. Their method (see below) consistently produced slightly higher values than the other two methods when tested for application in this study, although the results of the spherical approximation method were highly similar. Although the equations recommended by Graham *et al.* (1988) were used for the present study, it should be noted that the specific slopes of the regression lines of the equation for surface area might differ for stones from different environments. For this report, Graham *et al.'s* (1988) coefficient from a survey of river-worn stones was adopted without modification.

The surface area of each whole stone was calculated using one of the two forms of Graham *et al.*'s (1988) equation, which utilises measurements of the three main stone axes (Equations 9.3 and 9.4; Figure 9.11). Measurements of stone dimensions were made as part of the collection of physical microhabitat data (Section 9.5.4). The mean percentage error of estimation of the method was calculated by Graham *et al.* (1988) as 3.9%, S.D. = 3.1%.

Whole stone surface area = 1.15 x {(L x W) + (W x H) + (H x L)} **Equation 9.3**

The surface area of the top and bottom parts of each sample stone were then calculated on the assumption that the two parts were always equal in area, for the purposes of standardisation (Figure 9.10):

For the purposes of simplification, the surface area of part U of the sample (Figure 9.10) was assumed to be of the same size as the top (or bottom) part of the stone, as only the upper surface of the underlying substratum directly covered by each sample stone was sampled for invertebrates.

Surface area of part T/B/U = 0.5 x Whole stone surface area **Equation 9.4**

However, the BMI data from part U of each sample should be regarded as only semi-quantitative in that the accurate surface area would be partially dependent on substratum composition and particle size distribution. Although these data were collected, they were not analysed for the purposes of this report (Section 9.5.4). Moreover, quantification of the surface area of this sample part would be complex.

For each part of each BMI sample, an assessment of the percentage of organic material was made once all BMIs had been removed. All of the organic material, ranging in size from fine organic detritus to leaves, was combined and placed in muffle boats and dried in an oven at 60 °C to constant weight. The samples were pre-weighed, combusted in a muffle furnace at 450 °C for 5 hours to burn off all organics and then re-weighed to calculate the total percentage of organic material. The results of this analysis are reported on elsewhere (Tharme in prep.).

Methods of collection of benthic macroinvertebrates from wetted channel margins

A further set of three kinds of BMI samples, termed 'gravel', 'partially exposed' and 'damp fully exposed', was collected twice during the flow reduction phase, from wetted edges of the channel. The samples were only collected from the impact location, with the Elands samples representing control samples. The numbers of samples of each type and dates of sampling are summarised in Appendix 9.2. A total of 72 samples were collected, of which 48 constituted a total of 144 parts (T, B, and U; as per Figure 9.10).

The objective of the sampling was to ascertain which BMI taxa, if any, did not need to move or were stranded with shrinkage of the wetted channel and were thus able to withstand various degrees of drying. Natural biotic composition was reflected by the Elands data, while the Molenaars, Du Toits and Riviersonderend data indicated the BMIs at channel margins that had shrunk from the natural extent as a result of unnaturally low baseflow conditions.

The set of 24 'gravel' samples (Appendix 9.2) was taken from damp gravel next to wetted edges of biotopes. In each instance, 25 gravel particles were carefully rinsed above an 80 µm-mesh sieve. Each sample was checked for living BMIs for a total time of *c.* 15 minutes. The live individuals were counted and assessed in terms of their physical condition. These individuals were stored in 4% formalin in a separate vial from the remaining organic material, which included any dead specimens. The following additional

information was collected: a grid reference for each sample; range of gravel size and d_{50} (median) particle size; and a description of the sampled area and adjacent biotope(s).

The 24 'partially exposed' BMI samples were collected from substratum of sizes large gravel to small cobble (32-128 mm) (Appendix 9.2). The stones were in water, but the tops were completely exposed and dry. As for the main BMI samples, each stone was divided into a top (T), bottom (B) and gravel underlying the stone (U). Each surface was scrubbed with a toothbrush and the BMIs were collected in an 80 μm-mesh sieve. As for the gravel samples, living BMIs were stored separately from the rest of the sample.

The set of 24 'fully exposed' BMI samples was collected from stones which were completely dry, but resting on a damp substratum (Appendix 9.2). Stones of similar sizes to the partially exposed ones were sampled. As for the main BMI samples, each stone was divided into a top (T), bottom (B) and gravel underlying the stone (U). Each surface was scrubbed with a toothbrush and the BMIs were collected in an 80 μm-mesh sieve. As above, any living BMIs were stored separately from the rest of the sample.

For both the 'partially exposed' and 'damp fully exposed' samples, the following additional data were collected: a grid reference for each sample; stone α, β and c axes (mm); proportions of substratum types, range of particle sizes, and d_{50} values, for the U component; stone shape, roughness, compactness, percentage embeddedness; percentages instream and overhead cover; the percentage of side cover provided by adjacent stones; distances to upstream water, downstream water, and the channel edge; and a description of the sampled area and adjacent biotope(s). For 'partially exposed' stones, the proportion of the stone that was immersed in water was also calculated.

None of these samples was analysed for this report, but the data are presented elsewhere (Tharme in prep.).

Methods of analysis of benthic macroinvertebrate data

The BMI family and higher taxon data used in the analyses are listed in Tharme (in prep.).

A group of software programs specifically developed for multivariate and non-parametric, statistical analyses of multispecies data and associated environmental variables, PRIMER Version 3.1a (User Guide to PRIMER 1993), was the principal means by which patterns in the BMI data were explored. The theory and principles on which PRIMER is based are discussed in Field *et al.* (1982), Clarke and Warwick (1994), and outlined in King and Tharme (1994). Specific data requirements and functions of each of the Primer programs are outlined in User Guide to PRIMER (1993).

Normal or q-type analysis was used for the classification and ordination of the macroinvertebrate data (Field *et al.* 1982), whereby the samples were ultimately arranged into groups which each had a similar biotic composition. The stages of q-type analysis are summarised in King and Tharme (1994). In its general form, the BMI data consisted of a matrix of *n* samples by *s* taxa. Various combinations of the data were analysed ranging from assessments at the level of the two locations per site, for all sites and trips, through to detailed assessments of the possible impacts of flow reduction on the distribution of BMIs in the two biotopes on tops, bottoms and underneaths of stones.

In all cases, the BMI data were analysed at the level of presence/absence data and abundances standardised to 0.1 m^2 unit area. Abundances were root-root transformed (Equation 9.5) to downweight the importance of very abundant taxa so that the less dominant, and even rare ones, played some role in determining similarity levels between samples (Clarke and Warwick 1994). The similarity coefficient in this form of transformation is invariant to a scale change (Field *et al.* 1982).

Root-root transformation:

$$Yij = \sqrt{\sqrt{Xij}} = Xij^{1/4} \qquad \textbf{Equation 9.5}$$

Where X_{ij} = raw data score of the *i*th species in the *j*th sample; Y_{ij} = corresponding transformed score

Classification

Classification, to produce an hierarchical agglomerative clustering of the samples/taxa was performed using the Bray-Curtis similarity matrix (Equations 9.6 and 9.7) generated by the PRIMER program, CLUSTER. The results of each classification procedure were represented as a dendrogram created using the program DENPLOT, with group-average linking of the Bray-Curtis similarities (Clarke and Warwick 1994). Group-average sorting joins two groups of samples together at the average level of similarity between all members of one group and all members of the other and so considers natural variability between samples.

Bray-Curtis dissimilarity measure:

$$\delta_{jk} = \frac{\sum_{i=1}^{s} |Y_{ij} - Y_{ik}|}{\sum_{i=1}^{s} (Y_{ij} + Y_{ik})} \qquad \textbf{Equation 9.6}$$

Where Y_{ij} = score for the *i*th species in the *j*th sample; Y_{ik} = score for the *i*th species in the *k*th sample; δ_{jk} = dissimilarity between the *j*th and *k*th samples summed over all *s* species. δ_{jk} ranges from 0 (identical scores for all species) to 1 (no species in common) and is the complement of the similarity S_{jk}

Bray-Curtis similarity measure:

$$S_{jk} = 1 - \delta_{jk} \qquad \textbf{Equation 9.7}$$

Although dendrograms have the advantage of clustering samples into discrete groups, the cut-off levels for the groupings are arbitrary. There are also several disadvantages to this approach which make it advisable to employ an additional method of presentation of the group relationships (Field *et al.* 1982). Firstly, once a sample has been placed in a group its identity is lost. Dendrograms show only inter-group relationships and the sequence of samples in a dendrogram is arbitrary, with two adjacent samples in a group not necessarily

being the most similar. Moreover, dendrograms tend to over-emphasize discontinuities and may, therefore, force a continuum or graded series into discrete classes.

Ordination

As recommended in Field *et al.* (1982) and Clarke and Warwick (1994), clustering analysis was used in conjunction with ordination analysis, as superimposition of the clusters obtained from dendrograms at various levels of similarity on corresponding ordination plots enabled inter-group relationships to be re-assessed on a continuous scale. Furthermore, agreement between the two forms of representation of the data lends support to the consistency and adequacy of both methods in describing assemblage relationships.

The non-metric, multidimensional scaling method of ordination (non-metric MDS) was performed, using the programs MDS and CONPLOT (User Guide to PRIMER 1993). The principles of non-metric MDS are described in Shepard (1962, cited in Clarke and Warwick 1994) and Kruskal (1964, cited in Clarke and Warwick 1994).

Briefly, in the context of sample analysis, the purpose of non-metric MDS is to construct an ordination of the n sites, in a specified number of dimensions, by interpreting some function of the dissimilarity measure between each pair of sites as Euclidean distance. It is an iterative procedure where a starting map of the n sites is constructed (in an arbitrary fashion or by an ordination technique) in the required number of dimensions, and the configuration is perturbed in a direction which decreases the stress (Equation 9.8) to an acceptable minimum (see below). The interpoint distances of the configuration are then regressed on the corresponding dissimilarities using a general monotonic transformation to distance. The MDS is non-metric in that the regression is non-parametric, hereby making allowance for the typically non-linear relation of dissimilarities to distance for biological q-type analysis (Field *et al.* 1982). Similar samples will be located close together and highly dissimilar ones far apart in the resultant ordination.

In the MDS program, a scatter plot or Shepard diagram is constructed, of distance against dissimilarity, for all pairs of values in the plot. The goodness-of-fit of the regression is then indicated by the stress value, which totals the scatter around the regression (Equation 9.8). Stress provides an indication of the distortion involved in compressing the data to a small number of dimensions (usually 2-d for ease of interpretation); low stress indicates that the sample relationships are well represented by a plot of the sites in the specified dimensionality (Field *et al.* 1982). As a general rule-of-thumb, Field *et al.* (1982) state that a stress value < 0.05 provides an excellent representation of the data, and < 0.1 gives a good ordination with misleading interpretation being highly unlikely. At higher stress values, increasingly less reliance should be placed on the plot, and it may be necessary to use higher-dimensional ordinations to improve the representation.

Stress formula:

$$\text{Stress } 1 = \frac{\sum_{j}^{n}\sum_{k>j}^{n}\left(d_{jk} - \hat{d}_{jk}\right)^2}{\sum_{j}^{n}\sum_{j>k}^{n} d_{jk}^{\,2}} \qquad \textbf{Equation 9.8}$$

Where n = number of stations; d_{jk}: $k > j$; $j = 1, ...n$ = interpoint distances between jth and kth stations of the configuration (ordination plot), which are then regressed on the corresponding dissimilarities (δ_{jk}); $\hat{d}_{jk}$ = distance estimated from the regression, corresponding to dissimilarity, δ_{jk}

Diversity indices

The PRIMER program, DIVERSE, was used to calculate several diversity indices for the BMI samples, of which the results using the common Shannon-Weiner diversity index (Equation 9.9) are presented in this report.

Two different aspects of community structure contribute to the concept of community diversity (Clarke and Warwick 1994):

- Species richness, which is a measure of the total number of species.
- Equitability or evenness, which expresses how uniformly the numbers or individuals are distributed among the different taxa.

The Shannon-Weiner index was used because it incorporates both the species richness and equitability components of community diversity (Clarke and Warwick 1994).

Shannon-Weiner diversity index (H'):

$$H' = -\sum_i p_i (\ell n\, p_i) \qquad \textbf{Equation 9.9}$$

Where p_i = proportion of the total count arising from the ith species

9.5.4 Methods of data collection and analysis for physical microhabitat

For each instream stone from which a BMI sample was taken, representative physical microhabitat data were also recorded. A total of 864 sets of physical microhabitat data were collected, of which 144 sets correspond with the BMI data reported on here (Table 9.4). Apart from stone dimensions, these data were not used in the current report (Tharme in prep.).

The following types of data were collected for each sample (Tharme unpubl. data):

- Grid reference, based on the biotope maps.
- Depth from the water surface to the top of the stone (measured as described in section on discharge).
- Mean water column velocity (at 0.6 x depth) and near-bottom velocity (immediately above the top of the stone) were recorded (measured as described in section on discharge). The degree of turbulence and any effects of hydraulic refuge were noted.
- Dimensions of each sampled stone, recorded by measurement of the α, β (diameter, Wentworth grade scale) and c axes of each stone using a flexible tape (Figure 9.11). Slide calipers generally could not be used due to the large particle sizes sampled. These data were used for the calculations of surface area (Section 9.5.3).
- Stone roughness and compactness were rated on a scale of low, moderate or high, and the percentage embeddedness in fines was recorded.

- For the substratum underlying each stone, visual estimates of the percentages of the sample comprising various particle sizes were made. For each size category, the β axis of the minimum, maximum and approximate d_{50} particle size were measured.
- Instream and overhead cover, visually estimated as percentages.
- Percentage of silt and detritus and leaf litter under and surrounding each stone.
- Percentage of algae associated with each stone.
- The degree of biotope isolation from the main channel, specifically in the case of marginal pools, was estimated as a percentage, and the adjacent biotope patches on all sides were noted.

10. PRELIMINARY RESULTS OF THE LOW-FLOW STUDY

Written by R. Tharme

10.1 INTRODUCTION

10.2 HYDROLOGICAL EFFECTS AND LINKS WITH THE LONG-TERM HISTORICAL FLOW RECORD

10.2.1 Natural changes in dry-season baseflow and the hydrological effects of unnatural flow reductions

10.2.2 Linking natural and unnatural dry-season discharge patterns with hydrological indices generated from the historical flow record

10.3 EFFECTS OF NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS ON RIVER PHYSICAL AND CHEMICAL VARIABLES

10.3.1 Introduction

10.3.2 Effects of natural and unnatural dry-season baseflows on temperature

10.3.3 Effects of natural and unnatural dry-season baseflows on pH and total alkalinity

10.3.4 Effects of natural and unnatural dry-season baseflows on conductivity, total dissolved solids and total suspended solids

10.3.4 Effects of natural and unnatural dry-season baseflows on conductivity, total dissolved solids and total suspended solids

10.3.5 Effects of natural and unnatural dry-season baseflows on dissolved oxygen

10.3.6 Effects of natural and unnatural dry-season baseflows on major ions

10.3.7 Effects of natural and unnatural dry-season baseflows on nutrients and silicon

10.4 EFFECTS OF NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS ON PHYSICAL BIOTOPE CHARACTER AND DYNAMICS

10.4.1 Effects of natural and unnatural dry-season baseflows on the hydraulic character of physical biotopes, across riffles and runs

10.4.2 Effects of natural and unnatural dry-season baseflows on the dynamics of physical biotopes

10.5 RESPONSES OF BENTHIC MACROINVERTEBRATE ASSEMBLAGES TO NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS

10.5.1 Changes in the number of benthic macroinvertebrate taxa in response to natural and unnatural dry-season baseflows

10.5.2 Changes in total benthic macroinvertebrate abundance in response to natural and unnatural dry-season baseflows

10.5.3 Changes in taxon dominance in benthic macroinvertebrate assemblages in response to natural and unnatural dry-season baseflows

10.5.4 Changes in the diversity of benthic macroinvertebrate assemblages in response to natural and unnatural dry-season baseflows

10.5.5 Effects of natural and unnatural dry-season baseflows on the overall composition of benthic macroinvertebrate assemblages

10.1 INTRODUCTION

The following sections describe some preliminary results of the experimental reduction of dry-season baseflow in the four study rivers. Further exploration of the topic is beyond the scope of this report and forms part of a doctoral thesis hereafter referred to as Tharme (in prep.).

The first section deals with direct hydrological changes associated with the experimental manipulation of flow and their relationship with the long-term hydrological disturbance history of the river (Section 10.2). Sections describing effects of both natural and unnatural changes in flow on water chemistry and temperature (Section 10.3), physical or hydraulic habitat (Section 10.4) and the composition of BMI assemblages (Section 10.5) follow this. A summary discussion of these various results is presented in Chapter 11.

10.2 HYDROLOGICAL EFFECTS AND LINKS WITH THE LONG-TERM HISTORICAL FLOW RECORD

10.2.1 Natural changes in dry-season baseflow and the hydrological effects of unnatural flow reductions

Natural changes in discharge throughout the dry season, and unnatural changes associated with the experimental reductions in flow, are illustrated in Figure 10.1. Differences in discharge at the control locations of the experimental sites on the Molenaars, Du Toits and Riviersonderend rivers, and for the whole control site on the Elands River, provide an indication of the natural variability in discharge within the same reach, during a single dry season. Discharges measured at the impact locations indicate the reductions in flow achieved by experimental diversion.

Problems with the measurement of low flows

The discharge results for the almost six month experiment, which lasted from December to May (Section 9.4.3) revealed several problems with the measurement of low flow at all four sites. These problems are briefly discussed, before a discussion of the results, as they have important implications for the trends presented in this section and for low-flow assessment in general. The results obtained for the Elands site are used as examples. Similar experimental errors were encountered at the other sites.

As the Elands was the control site, discharges at the control and impact locations, and for riffles and runs, should have been highly similar for each month. However, experimental error attributed to the physical measurement of discharge at low flows, resulted in high variability both between riffle and run measurements and average discharges for the control and impact locations. The S.D. associated with the average of discharges calculated for this site for each month, ranged overall from 0.054 $m^3 s^{-1}$ to as high as 0.146 $m^3 s^{-1}$. For the averages of discharges measured for the control location for the six sampling trips, a 0.001-0.138 $m^3 s^{-1}$ range in S.D. was obtained. The lowest S.D. in the range corresponded with an average discharge of 0.220 $m^3 s^{-1}$ in early March, and the highest S.D. was associated with a May average of 0.391 $m^3 s^{-1}$. This range in S.D. compared with a wide range of 0.04-0.194 $m^3 s^{-1}$ for the impact location. The lowest S.D. corresponded with an average discharge of 0.290 $m^3 s^{-1}$ in March, and the highest one related to a May average discharge of 0.474 $m^3 s^{-1}$. The range of variability was lower for averages of run

discharge measurements throughout the study, than for averages of riffle data, at S.D.s of 0.028-0.066 $m^3 s^{-1}$ and 0.003-0.106 $m^3 s^{-1}$, respectively. For runs, the lowest S.D. corresponded with an average discharge of 0.301 $m^3 s^{-1}$ in January, and the highest S.D. was associated with an April average discharge of 0.237 $m^3 s^{-1}$. The lowest S.D. calculated for riffle data corresponded with an December average discharge of 0.487 $m^3 s^{-1}$, and the highest S.D. pertained to a January average discharge of 0.350 $m^3 s^{-1}$. This trend was expected, as flow is smoother in runs, due to the greater depth to particle size ratio.

The variability in discharge measurements resulted in differences in the ranking of months from lowest to highest discharge, according to which sets of data were examined. This shows that the experimental error of measurement at low flow often exceeded small-scale, natural changes in dry-season baseflow.

For the purpose of consistency, and for the reasons discussed in Section 9.5.1 and above, the patterns presented in Figure 10.1 pertain only to run discharge data. Although riffle discharge data were excluded from the assessment, any major discrepancies between discharges for cross-sections or locations are noted where relevant. The results presented below represent instantaneous discharges that were usually measured within a few hours of one another (Section 9.5.1), but might thus differ slightly from corresponding average daily discharges. The latter variable derives from the averaging of all recorded instantaneous discharges for a full 24 h period. A single set of discharge measurements was made at each site, for each of six sampling trips (Table 9.4).

Natural flow patterns at the Elands site

Fluctuations in natural instantaneous discharge from early summer (December) to the onset of autumn (May) were fairly low for the Elands, from a low of 0.190 $m^3 s^{-1}$ (from a run, April) to a maximum of 0.361 $m^3 s^{-1}$ (December), indicating that the river has stable dry-season baseflows (Figure 10.1). The recorded discharge range represents almost a doubling of baseflow within the experimental period (a factor of 1.9). During the experimental period, flow was similarly low in the months of January to April. It was only marginally lower than all other months in March (impact location) and April (control location), with percentage reductions in magnitude relative to antecedent (early March) baseflows of 5% and 6%, respectively. Discharges were thus similar from the beginning of summer through to the onset of autumn. Increases in baseflow magnitude in May, with the transition to autumn, were in the order of 19% for the impact location and 54% for the control location. Comparison of the instantaneous discharges with average daily discharges from the historical flow record (Section 9.3.2) revealed differences between the two in the months of lowest and highest discharge. From the hydrological record, the month of the year of lowest flow is characteristically February, both in terms of average daily discharge and flow volume (Figures 9.2 and 9.3). Furthermore, highest flows over the six month study period typically occur in May (Figure 9.3).

As the control site, there was no unnatural reduction in baseflow at the Elands impact location during the months of February and March. However, from Figure 10.1, discharges in this location are consistently marginally higher than discharges in the control location. This pattern can be, in part, attributed to experimental error. Moreover, a small amount of surface flow was observed entering the impact location from the banks during most site visits. It is likely, therefore, to be largely a function of a small contribution from surficial and groundwater sources.

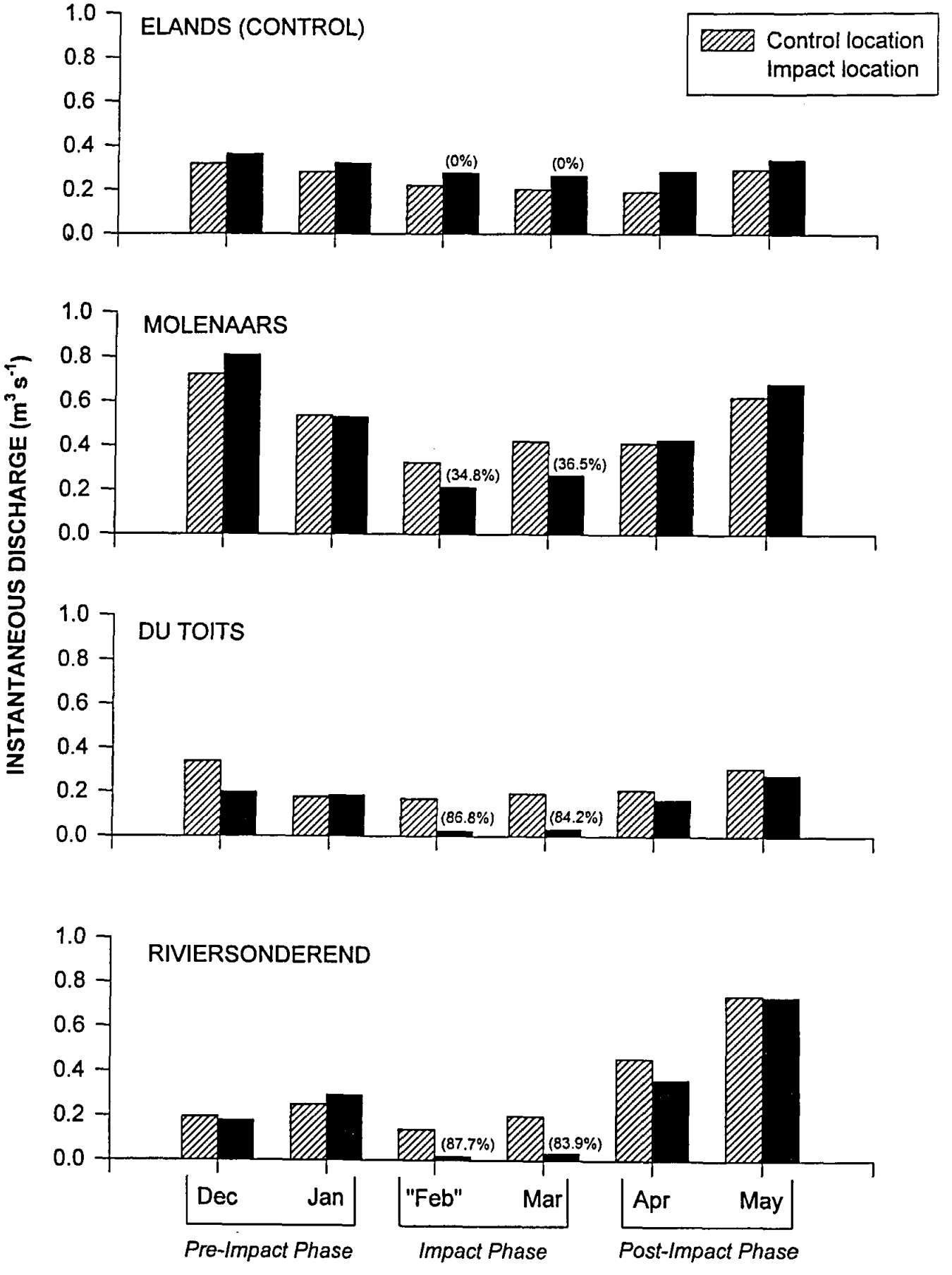


Figure 10.1 **Instantaneous discharges ($m^3 s^{-1}$) for all three phases of the low-flow study, at the four sites. Percentage flow reductions at the impact locations, during the impact phase, are indicated in parentheses. Natural discharges are represented by the values for the control location and the discharges for the impact location both pre-and post-impact. For the third fieldtrip, "Feb" discharge values for the Molenaars and Elands sites were recorded in early March, while all sampling was completed in February for the other two sites.**

The degree to which the natural and unnatural flow patterns differed from established longer-term hydrological trends is explored further in Section 10.2.2.

Natural and unnatural flow patterns at the Molenaars site

Fluctuations in natural instantaneous baseflow from December to May were moderately low for the Molenaars site, ranging from 0.322 $m^3 s^{-1}$ in early March (and hence probably also February) to 0.721 $m^3 s^{-1}$ in December (a factor of 2.2). Natural baseflows were lowest in early March and only slightly higher in April (Figure 10.1). The former observation is in accordance with the long-term flow pattern for the site (Section 9.3.2, Figures 9.2 and 9.4). However, on the basis of historical data, May baseflows typically greatly exceed April flows, which in turn, usually exceed all baseflows from December to the end of March. The observed discharges therefore point to a naturally dry summer of protracted duration during the study period.

On average, on the basis of instantaneous discharges calculated during the low-flow study, 36% of the total dry-season baseflow was abstracted from the river upstream of the impact location, during the impact phase (February to March). This left some 64% of natural baseflow in the river. The overall average proportional reduction in discharge, taking into consideration discharges measured during actual construction of the flow diversion structure, was estimated as 35% (Table 9.3). An increase in baseflow magnitude of *c.* 61% occurred in the impact location with the reinstatement of natural flows from March to April, during the post-impact phase. This was in contrast with a natural slight decrease in magnitude of 2% in the control location (Figure 10.1).

The pattern of baseflows at the flow-impacted location remained unchanged from the natural one (see above), with lowest baseflows being experienced in early March (0.210 $m^3 s^{-1}$) and therefore, probably in February, and greatest discharges occurring in December.

Further comparison of the natural and unnatural flow patterns obtained during the present study with longer-term hydrological trends is made in Section 10.2.2.

Natural and unnatural flow patterns at the Du Toits site

Natural instantaneous baseflows were similarly low in the Du Toits river from January right through to April, indicating fairly constant dry-season baseflows. December and May discharges only marginally exceeded peak summer values (Figure 10.1). The lowest baseflow recorded was 0.167 $m^3 s^{-1}$, in February, and the highest baseflow of 0.338 $m^3 s^{-1}$ occurred in December (representing an increase factor of 2.0). An assessment of average total flow volumes and average daily discharges (Figures 9.2 and 9.5, respectively) showed a similar pattern, with a gradual decrease and increase in flow at the ends of winter and summer, respectively. Moreover, the month of lowest flow, on the basis of the long-term hydrological record, corresponded with the study data. However, from the historical record, May average flows typically exceed those for December, due to the inclusion of flood events.

About 85.5% of the total dry-season baseflow was abstracted from the river upstream of the impact location, leaving only about 14% of natural baseflow in the river. The overall average proportional reduction in discharge, taking into consideration discharges measured during actual construction of the flow diversion structure, was estimated as 80% (Table 9.3). An increase of *c.* 440% occurred in the impact location with

the reinstatement of natural flows in April, in comparison with a natural increase in baseflow magnitude in the control location of only 8% (Figure 10.1).

The pattern of baseflow magnitudes at the impact location remained unchanged from the natural condition, with the lowest baseflows being experienced in February (0.022 $m^3 s^{-1}$).

Further comparison of the natural and unnatural flow patterns obtained during the present study with longer-term hydrological trends is made in Section 10.2.2.

Natural and unnatural flow patterns at the Riviersonderend site

Natural baseflows at the Riviersonderend site were variable, ranging from a minimum of 0.138 $m^3 s^{-1}$ in February to 0.741 $m^3 s^{-1}$ in May (Figure 10.1). This range represented an increase factor, from minimum to maximum, of 5.4. January instantaneous baseflow was higher than that of the neighbouring months, possibly as a result of a dry-season fresh event, while February was the month of lowest flow. Comparison with the historical flow record (Figures 9.2 and 9.6) showed that February baseflow is typically the lowest of the dry season. However, the general trend differed somewhat from that of the present study, with a progressive, consistent decrease in baseflows from early spring through summer. Slight, gradual increases in flow characteristically occur from March to April, and there is a marked increase in total flow volume and average daily discharge in May. The latter pattern was reflected in the data collected in this study.

Marginally more flow was diverted from the river than at the Du Toits site, at *c.* 85.8% of the total dry-season baseflow. Approximately 14% of the natural baseflow thus passed through the impact location. The overall average proportional reduction in discharge, taking into consideration discharges measured during actual construction of the flow diversion structure, was estimated as 82% (Table 9.3). An increase of about 1028% occurred with the reinstatement of natural flows in the impact location, in April, in comparison with a natural baseflow increase in the control location of only 130% (Figure 10.1).

The pattern of baseflows in the impact location remained unchanged from the natural condition, with the lowest baseflows being experienced in February (0.017 $m^3 s^{-1}$).

Natural and unnatural discharges recorded during the present study are compared with a series of hydrological indices based on the historical flow record in Section 10.2.2.

Comparisons among sites

For all sites, lowest, natural instantaneous baseflows tended to occur in February/early March. In all instances, except the Riviersonderend, highest instantaneous discharges were recorded in December. At the Riviersonderend site, elevated May baseflows reflected the earlier transition to winter baseflow conditions at this site. This more rapid response is probably a result of seasonally earlier or greater rainfall in the mountainous upper catchment. The site is the most upstream one and in a mountain-stream zone. It also showed the most variable baseflow pattern, with an increase in baseflow from December to January, and a subsequent decrease to naturally very low flows in February. Three of the sites showed a magnitude range of baseflows throughout the experimental period in the order of a factor of two, while the Riviersonderend site exhibited a far greater degree of change in baseflow. The natural May baseflow in the Riviersonderend

River was greater than the highest instantaneous discharge recorded for the Molenaars, even though the latter is a far larger river at the site. However, as the discharges are instantaneous, higher flow events at the Molenaars and other sites could have occurred among sampling dates. Discharge magnitudes for days other than sampling dates are only known for the two sites for which hydrological gauging is ongoing (Table 8.1).

Although evaluation of the above data was beyond the scope of this report, it is evident from the Molenaars gauged data that the unnaturally low baseflows remained roughly constant for each of the two months of the impact phase. At the Du Toits and Riviersonderend sites, impacted baseflows were also likely to have been near-constant at the instantaneous discharges recorded for February and March, due to the design of the diversion weirs (Section 9.4.2). In terms of experimental flow reduction, the most flow was diverted at the Riviersonderend, followed by the Du Toits and then the Molenaars River.

10.2.2 Linking natural and unnatural dry-season discharge patterns with hydrological indices generated from the historical flow record

A first attempt was made to link the changes in instantaneous discharge observed throughout the dry season and early autumn in the field, with the long-term historical flow data for each site, by means of assessment of the general pattern of the hydrological regime and of various low-flow hydrological indices. The purpose of this link is twofold:

- Firstly, it enables the experimental, unnatural reductions in baseflow to be contextualised in terms of the long-term natural flow disturbance history of each river.
- Secondly, it provides preliminary indications of which of the commonly computed hydrological indices have particular merit in assessments of the impacts of unnaturally low, dry-season baseflows on the riverine ecosystems. Ultimately, these various low flow indices will be linked with any significant physical and chemical, biotope-level or biotic responses to natural and unnatural dry-season baseflows (as indicated in Sections 10.3 to 10.5, respectively).

The long-term patterns of daily discharges used for comparison with the flow pattern observed during the low-flow study are described in Section 9.3. The flow indices selected for the same types of comparison are the following: MeanMean discharge; MedianMedian discharge; Median 7 day-low flow discharge; MinMin discharge, and a series of percentiles from FDCs. Explanations of these indices are provided in Section 9.3. The maximum daily discharge for each month, over the historical record (MaxMax), far exceeded the magnitudes of the other indices calculated, for all four sites. This index was, therefore, was not considered further. Links with additional low flow indices, specifically indices that incorporate aspects of flow variability and predictability, form part of further work by Tharme (in prep.).

The following sections describe some links between the observed natural and unnatural baseflows and low-flow indices, for each of the four sites.

Elands site

The general annual pattern of daily flows for the Elands site is described in Section 9.3 and illustrated in Figure 9.3. Figure 10.2 illustrates the magnitudes of natural instantaneous discharges measured at the control and impact locations during the low-flow study, and the magnitudes of various hydrological indices, calculated using the methods outlined in Section 9.3. In addition to these indices, a series of flow percentiles was calculated from the long-term record, from 1-day FDCs for the study months (Table 10.1). The

corresponding monthly average daily discharges and modal discharges are presented for comparative purposes.

On the basis of the historical flow record for the Elands River, the various low flow indices given in Figure 10.2 tend to form a fairly well-defined envelope of natural baseflows from above 0.1 $m^3 s^{-1}$ to around 0.5 $m^3 s^{-1}$. However, the minimum discharges on record are clearly lower than the median 7-day low flows. The various indices group quite closely for the months of December to March, but start to diverge in April and more so in May. This indicates the incorporation of freshes (as defined in Section 2.2) into the median (i.e. MedianMedian) discharges, particularly. The graph illustrating the long-term trend in average discharges is similar to the other indices in magnitudes of indices only in the peak dry season (February; Figure 10.2). As expected, of the indices depicted in Figure 10.2, average discharge is the index most influenced by the occurrence of dry-season fresh events, especially towards the beginning and end of the dry season. Of the percentage exceedance values calculated for the Elands (Table 10.1), the 75th and 95th percentiles are of particular interest, as they represent the lower end of the spectrum of flows. The 95% exceedance values are highly similar for all months of the study period at about 0.20-0.25 $m^3 s^{-1}$, except in December, when the figure is somewhat higher. They are not markedly lower than the 75% exceedances, except in December and to a greater extent in May (Table 10.1).

An assessment of the 1-day annual FDC generated for the site, showed that an overall percentage exceedance of 80% (or Q_{80}) corresponds with a discharge in the order of 0.4 $m^3 s^{-1}$, irrespective of flow timing (such as month of occurrence). The Q_{90} corresponds with a discharge of about 0.3 $m^3 s^{-1}$.

Natural instantaneous baseflows for the months of December to May (Figure 10.2) fell below the longer-term median discharge established for each month, but above the long-term minimum discharges. The natural baseflows approximated the medians of the 7-day low flow periods over the whole record, as recorded for each month. This was most apparent when the average of the discharges at the control and impact location was compared with the (median) 7-day low flow values. The natural instantaneous discharges were slightly higher than the corresponding 7-day low flow discharges for January and early March, and generally slightly lower than the 75th percentiles (Table 10.1). Comparison of the discharges represented by the 95th percentiles, on a month-by-month basis (Table 10.1), with the natural baseflows during the study months, showed that natural dry-season flows were the same or only slightly higher than the Q_{95} values in all months. Comparison of the instantaneous discharges with the Q_{80} and Q_{90} values from the 1-day annual FDC (see above), also showed that baseflows at the time of the low-flow study were generally in the order of the 90-95th percentile range.

The above comparisons of the natural baseflow regime at the time of the low-flow study with the long-term flow pattern, revealed that the dry season was probably protracted during the experiment and that flow magnitudes were quite low. This is, in part, based on the established assumption that Q_{95} values represent low flows. Recorded natural discharges closely approximated the median 7-day low flows and 90-95% exceedances for each month. Although small fresh events probably occurred outside the times of sampling, in response to localised storm events (pers. obs.), the general trend appears to be one of a drier than average 1994/1995 dry season. This would, however, require confirmation, using relationships between flow indices and wet, dry and intermediate years (Tharme in prep.).

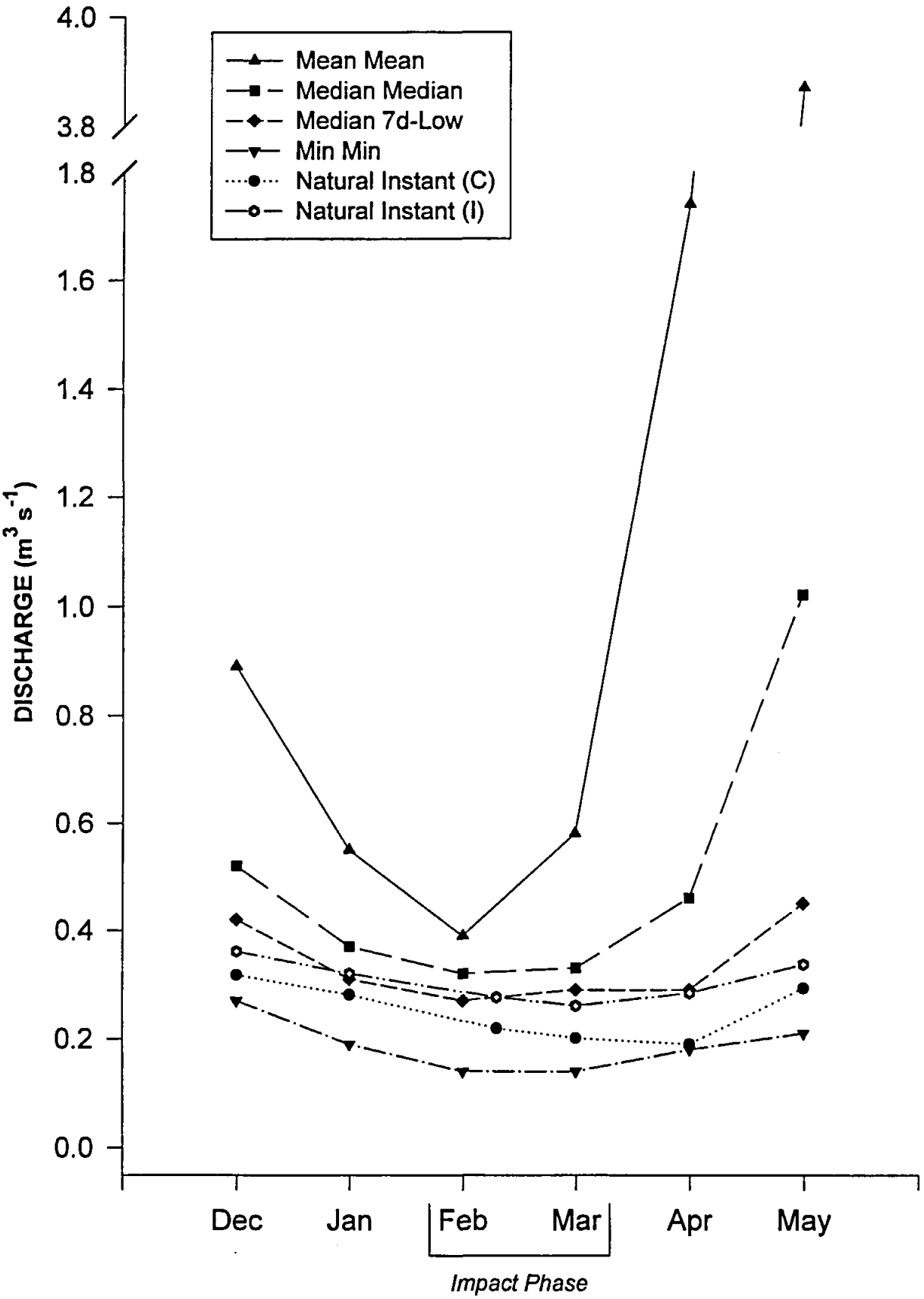


Figure 10.2 **Comparison of natural instantaneous discharges (m^3 s^{-1}) for the Elands control (Natural Instant (C)) and impact (Natural Instant (I)) locations with various hydrological indices, for the duration of the low-flow study. The derivations of the indices are explained in Section 9.3. For the third (February) field trip, discharges were recorded in early March.**

Table 10.1 **Mean and modal natural average daily discharges, and average daily discharges corresponding with 5%, 25%, 50% (median), 75%, and 95% exceedance values derived from daily flow duration curves. Data were calculated for the entire period of historical record, for the months corresponding with the duration of the low-flow study. Units are $m^3 s^{-1}$.**

SITE	DEC	JAN	FEB	MAR	APR	MAY
ELANDS						
Mean	0.885	0.545	0.391	0.578	1.743	3.866
Mode	0.869	0.710	0.303	0.550	1.104	1.659
5%	1.910	1.211	0.755	1.116	10.420	17.760
25%	0.763	0.507	0.431	0.452	1.100	2.964
50%	0.521	0.374	0.316	0.327	0.461	1.018
75%	0.414	0.301	0.257	0.255	0.284	0.538
95%	0.317	0.251	0.209	0.209	0.209	0.263
MOLENAARS						
Mean	1.626	0.920	0.726	1.043	2.226	6.107
Mode	1.064	0.974	0.482	1.113	1.682	2.597
5%	4.168	2.221	1.638	2.246	8.664	30.770
25%	1.348	0.805	0.685	0.698	1.660	4.542
50%	0.890	0.623	0.517	0.517	0.757	1.710
75%	0.706	0.527	0.442	0.458	0.490	0.889
95%	0.561	0.438	0.338	0.368	0.358	0.452
DU TOITS						
Mean	0.449	0.252	0.225	0.304	0.487	1.175
Mode	0.272	0.129	0.209	0.249	0.244	0.271
5%	2.280	0.468	0.428	0.866	1.820	4.690
25%	0.364	0.252	0.206	0.219	0.428	1.410
50%	0.266	0.190	0.171	0.167	0.214	0.556
75%	0.215	0.167	0.154	0.145	0.148	0.275
95%	0.170	0.129	0.127	0.127	0.120	0.151
RIVIERSONDEREND						
Mean	0.606	0.388	0.350	0.632	1.392	2.842
Mode	0.453	0.576	0.341	0.500	0.578	0.818
5%	2.130	1.210	0.937	2.580	6.630	14.700
25%	0.409	0.274	0.267	0.355	1.090	2.540
50%	0.251	0.172	0.174	0.195	0.419	0.836
75%	0.182	0.139	0.129	0.148	0.182	0.371
95%	0.135	0.110	0.097	0.106	0.116	0.165

Molenaars site

The general annual pattern of daily flows is described in Section 9.3 and illustrated in Figure 9.3. Figure 10.3 illustrates the relative magnitudes of natural and unnatural, instantaneous discharges during the low-flow study. The natural instantaneous discharge is represented by the discharge recorded at the control location at this site (and similarly for the other two experimental sites). The instantaneous discharges, in the impact location, represented natural magnitudes, but only for the pre- and post-impact phases of the study. During February and March, instantaneous discharges at this location reflected the manipulated 36% reduction in flow (Figure 10.1). Figure 10.3 also shows the extent to which the measured discharges corresponded with various hydrological indices. A series of flow percentiles calculated from the long-term record for the Molenaars site is given in Table 10.1.

Of the various hydrological indices generated for the Molenaars, the absolute minimum discharges on record (MinMin) represent a variable range of discharges, from effectively zero (0.001 $m^3 s^{-1}$) for December through to similarly high values of 0.330 $m^3 s^{-1}$ and 0.340 $m^3 s^{-1}$ in January and May, respectively (Figure 10.3). The accuracy of a near zero December discharge for this large, perennial river is, however, deemed questionable. A similarly low, probably anomalous discharge of 0.004 $m^3 s^{-1}$ was recorded for November of the same year, 1988. The February minimum is the lowest of all months except December. The minimum discharges on record are markedly lower than the (median) 7-day low flows. Seven-day low flows and median (MedianMedian) discharges follow a similar pattern, with lowest values for the peak dry-season months of February and March. Median discharges are marginally higher in all cases. However, the median values increase more rapidly for April and May, due to significant high flow contributions. Average (MeanMean) discharges far exceed the values for other indices in all months, although the pattern is similar to that of all other indices except the minima. Lowest average discharges occur in February. This index is most influenced by the occurrence of higher flow events, especially at the beginning and end of summer.

Of the percentage exceedance values calculated from the long-term record for the Molenaars (Table 10.1), the 75th and 95th percentiles are most relevant to the present study, as they represent the lower range of flows. Q_{75} values are all about 0.4-0.5 $m^3 s^{-1}$ for January to April, and are markedly elevated for the months either side. Ninety-fifth percentiles follow a similar trend, though at somewhat lower values.

Assessment of the 1-day annual FDC, showed that an 80% exceedance corresponds with a discharge of about 0.63 $m^3 s^{-1}$, and a 90% exceedance with a discharge of *c.* 0.49 $m^3 s^{-1}$.

Comparison of the measured, natural instantaneous baseflows for the months of December to May with the plotted hydrological indices (Figure 10.3), revealed the following trends. Firstly, during the study, natural baseflows were at lower than or similar magnitudes to the median 7-day low flows for all months of the year of the study, except December when they were a little higher. For the driest month, February, the observed, natural discharge was midway in magnitude between the 7-day low flow and the absolute minimum on record for the month (Figure 10.3). Natural flows during the study diverged most from the absolute minima for the month of December, and to a lesser degree in May.

Natural instantaneous baseflows approximated the 75% exceedances for December and January, and were between the 75th and 95th percentile figures for March to May (Table 10.1 and Figure 10.3). Flows

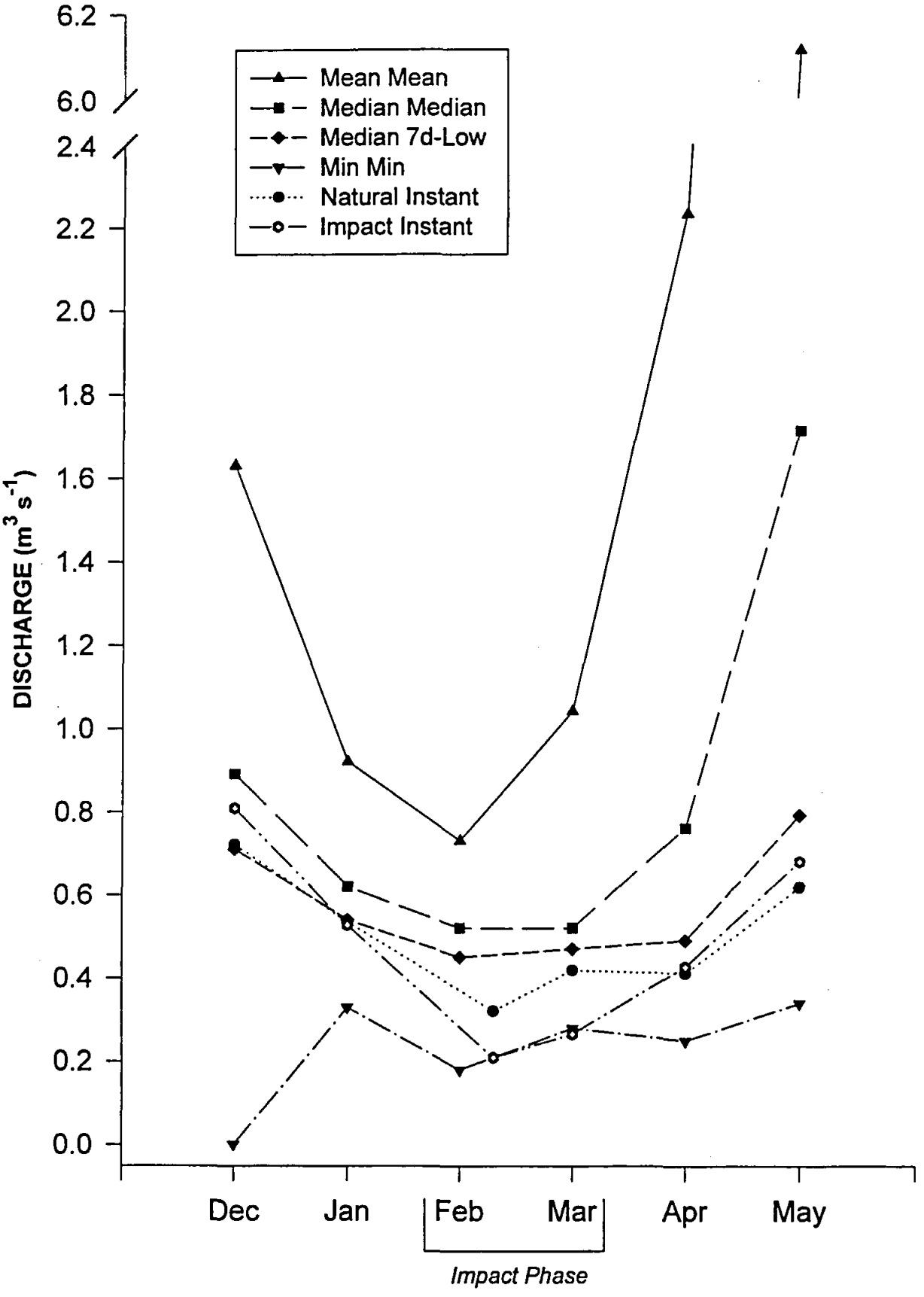


Figure 10.3 **Comparison of natural instantaneous discharges ($m^3 s^{-1}$) for the Molenaars control (Natural Instant) and impact (Impact Instant) locations with various hydrological indices, for the duration of the low-flow study. The derivations of the indices are explained in Section 9.3. For the third (February) field trip, discharges were recorded in early March.**

recorded in February at the site corresponded closely with the Q_{95} for that month, and were below discharges corresponding to the Q_{90} on a 1-day annual FDC. These comparisons of the natural baseflow regime at the time of the low-flow study with the long-term flow pattern, revealed that the dry season was probably protracted during the time of the experiment and that flow magnitudes were also lower than average. This assumption of a dry summer requires confirmation (Tharme in prep.).

The experimental reduction of flow during the impact phase of the study in February/March resulted in unnaturally low, instantaneous discharges in the order of 0.2 $m^3 s^{-1}$, that were at markedly lower magnitudes than the natural 95% exceedance flows (Table 10.1). Moreover, the observed flows corresponded with the absolute minima on record for the same time of year (Figure 10.3). Hence, the river and its biota were subjected to a flow disturbance that qualifies as extreme in magnitude and duration, relative to the long-term, natural disturbance history of the site.

Du Toits site

The general annual pattern of daily flows is described in Section 9.3 (see also Figure 9.5). Figure 10.4 illustrates the magnitudes of natural and unnatural, instantaneous discharges during the low-flow study, as well as the trends in various hydrological indices. Flow percentiles for this site are given in Table 10.1. The natural instantaneous discharge represents the discharge recorded at the control location at each site. The instantaneous discharges in the impact location represent natural magnitudes for the pre- and post-impact phases of the study. During February and March, however, instantaneous discharges at this location reflect the 86% experimental reduction in flow (Figure 10.1).

All of the hydrological indices except the average discharge (MeanMean) produced a flat graph, with similar values for most months. The minimum discharges (MinMin) are highly similar for all months except January, for which an absolute minimum of 0.003 $m^3 s^{-1}$ (for the year 1988) was recorded (Figure 10.4). Seven-day low flows and median (MedianMedian) discharges follow a similar pattern, in close proximity to each other, with consistently low values for the peak dry-season months of January to March. The median discharge diverges greatly from the 7-day low flow for May, due to high flow contributions. Average (MeanMean) discharge is lowest for February. It then increases sharply from March through April, to May, with the onset of higher flow events.

Discharges corresponding with the 75th percentile, range from a low of 0.145 $m^3 s^{-1}$ in March to a high of 0.275 $m^3 s^{-1}$ in May (Table 10.1). The 95% exceedance figures are consistently low, reaching a minimum of 0.120 $m^3 s^{-1}$ in April. It is evident from the different flow indices that February is not necessarily the month of lowest flow at this site. However, the higher discharges that correspond with lower percentage exceedances (25% and 5%) accord with Figure 9.5, in showing that there is little evidence of the influence of freshes in this month.

Assessment of the 1-day annual FDC showed that a 80% exceedance corresponds with a discharge of 0.19 $m^3 s^{-1}$ and a 90% exceedance with a discharge of 0.15 $m^3 s^{-1}$.

Comparison of the natural instantaneous baseflows for the months of December to May with the range of hydrological indices (Figure 10.4) firstly showed that the natural baseflows were similar in range to the median (MedianMedian) and (median) 7-day low flows during December. This was due to variability in

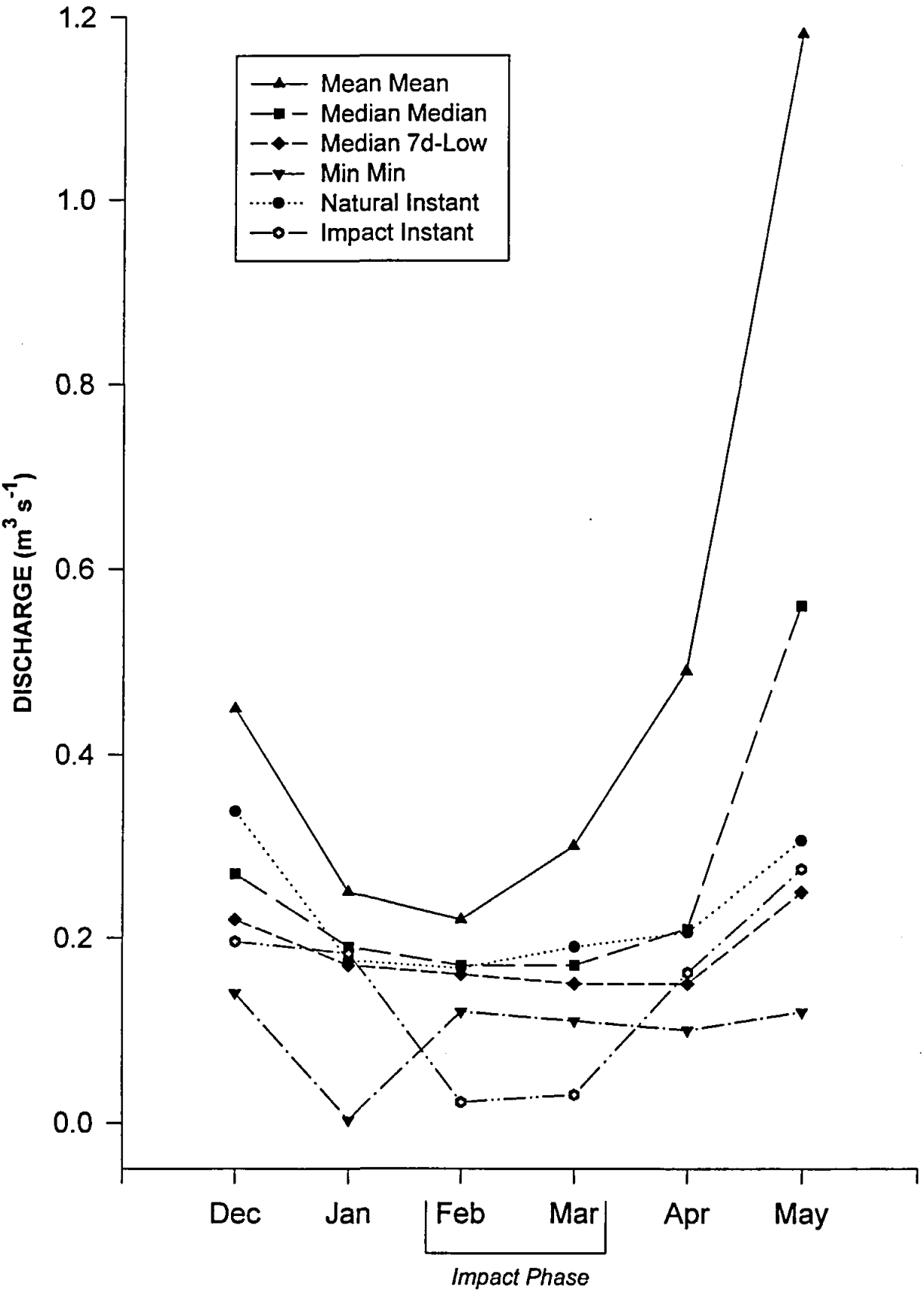


Figure 10.4 **Comparison of natural instantaneous discharges (m^3 s^{-1}) for the Du Toits control (Natural Instant) and impact (Impact Instant) locations with various hydrological indices, for the duration of the low-flow study. The derivations of the indices are explained in Section 9.3.**

discharge within the study reach. For January to April, natural baseflows approximated the median index, while they were closer to the 7-day low flow in May.

The natural instantaneous baseflows did not bear a consistent relationship with any single flow percentile over the December to May period (Table 10.1 and Figure 10.4). For instance, they fell between the 50-75% exceedance flow for January, but were close to the 50th percentile for April. None of the natural baseflows corresponded closely with the Q_{95} for a particular month. Thus, the trend of a drier summer than usual, observed for the Elands and Molenaars sites, is not apparent at this site.

The experimental reduction of flow during the impact phase of the study resulted in unnaturally low, instantaneous discharges that were significantly lower than both the natural 95% exceedance flows (Table 10.1) and the absolute minima on record, for February and March (Figure 10.4). Hence, the experimental reduction in dry-season baseflow represented an extreme event not experienced previously in the river, both in duration and magnitude.

Riviersonderend site

The general annual pattern of daily flows is described in Section 9.3 (see also Figure 9.6). Figure 10.5 illustrates the magnitudes of natural and unnatural, instantaneous discharges during the low-flow study, as well as trends in various hydrological indices. Flow percentiles are provided in Table 10.1. The natural instantaneous discharge represents the discharge recorded at the control location at each site. The instantaneous discharges measured in the impact location for the pre- and post-impact phases of the study represent natural magnitudes. During February and March impact months, instantaneous discharges at this location reflect the approximately 86% experimental reduction in flow (Figure 10.1).

The minimum discharges (MinMin) on record are highly similar for all months, reaching a low of 0.020 $m^3 s^{-1}$ in February (Figure 10.5). Median seven-day low flows are also highly similar, ranging only from 0.17-0.13 $m^3 s^{-1}$ in all months, except May. The May 7-day low flow, although a low flow index, reflects the characteristic occurrence of high flow events in this month (Figure 9.6). Median (MedianMedian) and average (MeanMean) discharges follow a fairly similar pattern, increasing sharply in April and again in May. However, median and mean discharges are lowest in January and February, respectively.

In terms of flow percentiles derived from the historical record (Table 10.1), 75th percentile flows are similar in all months, at less than 0.2 $m^3 s^{-1}$. The 95% exceedance figures are consistently low, reaching a minimum of 0.097 $m^3 s^{-1}$ in February and a maximum of 0.165 $m^3 s^{-1}$ in May. Assessment of the 1-day annual FDC calculated for the Riviersonderend, showed that an 80% exceedance corresponds with a discharge of about 0.20 $m^3 s^{-1}$ and a 90% exceedance with a discharge of 0.14 $m^3 s^{-1}$.

The natural instantaneous baseflows did not bear a consistent relationship with any particular hydrological index over the December to May period (Figure 10.5). Baseflows were similar to median (MedianMedian) discharges from March to May, approximated (median) 7-day low flows in December and February, and were greater than the long-term median for January.

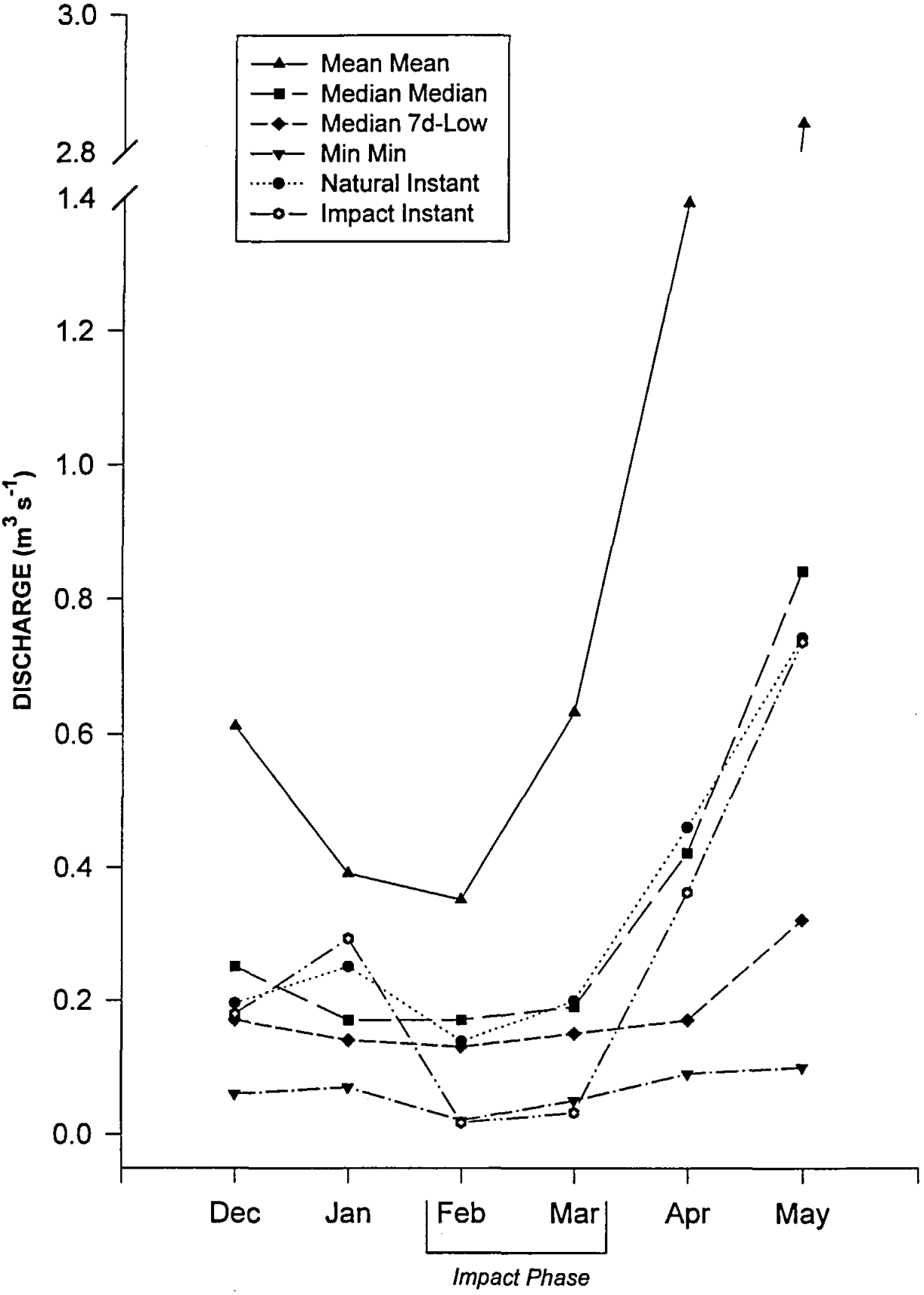


Figure 10.5 **Comparison of natural instantaneous discharges (m^3 s^{-1}) for the Riviersonderend control (Natural Instant) and impact (Impact Instant) locations with various hydrological indices, for the duration of the low-flow study. The derivations of the indices are explained in Section 9.3.**

The observed, natural flows also bore no consistent relationship with a specific flow percentile over the study period (Table 10.1 and Figure 10.5). For instance, they were greater than the flows represented by the median percentile for January and April, but just greater than the 75th percentile for December. None of the natural baseflows corresponded closely with the corresponding long-term 95% exceedance flow. Thus, the trend of a drier summer than usual, observed for the Elands and Molenaars sites, is not apparent at this site.

The experimental reduction of flow during the impact phase of the study resulted in unnaturally low, instantaneous discharges that were significantly lower than natural Q_{95} flows (Table 10.1). The measured discharges were also just below the absolute minima on record for the months of the impact phase (Figure 10.5). Hence, the experimental reduction in dry-season baseflow represented an extreme event, both in duration and magnitude, beyond what had been experienced before.

10.3 EFFECTS OF NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS ON RIVER PHYSICAL AND CHEMICAL VARIABLES

10.3.1 Introduction

This section describes the preliminary results of sampling physical and chemical variables, during each of the six field trips to each study site. The physical and chemical variables (hereafter simply referred to as water chemistry) recorded at each site, are listed in Section 9.5.2. Sampling dates are given in Table 9.4. All raw data, including outputs from the analyses by ion exchange chromatography, are housed by the author. The limitations inherent in instantaneous, single-event sampling of water temperature and chemistry should be borne in mind. These limitations include the absence of repeated sampling at short time intervals, during the flow reduction phase and immediately after the temporary weirs were removed, when poor quality water was flushed downstream. Too few records of the different physical and chemical variables may also result in unobserved changes in water quality with flow reduction.

The influence of natural changes in discharge over the dry season is indicated by the physicochemical data collected from the control locations at the four sites, and throughout the entire study for the control site on the Elands River. It is also reflected in the data for the impact location during the pre-impact phase of the low-flow study, at all sites.

Water chemistry conditions reflecting unnaturally low flows are represented by the data for the impact locations of all sites bar the control, for the impact phase of the experiment. Any recovery from impacted to natural water quality is shown by the data for these locations during the post-impact phase of the study.

The three phases of the study are delimited on the basis of the dates of diversion of flow and reinstatement of natural flow, as provided in Table 9.3. Due to logistical constraints, the durations for each phase differ somewhat for the control and impact locations (Tables 9.3), so delimitation is based primarily on durations for the impacted location in each instance.

10.3.2 Effects of natural and unnatural dry-season baseflows on temperature

Three measures of temperature were recorded during the study, for variable durations (numbers of days) between consecutive visits to each site: maximum, minimum and instantaneous temperature (largely as spot measures). Although the time of day was always noted, it is not incorporated here.

Elands site

Natural ranges in temperature for the control site, over the full dry season and into autumn, are presented in Figure 10.6, for both locations.

As expected, inter-reach variability in temperature was very low, as control and impact temperatures were similar for instantaneous, maximum and minimum temperatures. For all three measures of temperature, values progressively increased from December through to January/February, after which they gradually decreased to a low in April/May (Figure 10.6). The absolute difference in temperature (difference between MaxMax and MinMin) was 18.6 °C. During February to March (impact phase), the natural difference in temperature was roughly 11.9 °C (minimum of 14.0 °C, and maximum of 25.9 °C).

The maximum temperature records revealed the following trends (Figure 10.6). The absolute maximum temperature was high at 28.5 °C, in both locations, in January. This coincided with the onset of naturally low baseflow conditions in the river (Section 10.2). Lowest maxima were recorded in May, at 15.6 °C and 15.1 °C, in the control and impact locations, respectively. Instantaneous temperatures were slightly more variable than minimum and maximum figures, probably due to time of day, but followed a similar monthly pattern (Figure 10.6).

The absolute minimum temperatures recorded over the study were 9.9 °C (control location) and 10.3 °C (impact location), in May. They coincided with marked increases in baseflow (Section 10.2) and climatic changes associated with a natural, seasonal transition to autumn. Minima were more stable over the study than maxima or instantaneous temperatures, only decreasing markedly at the end of the dry season.

Molenaars site

Temperatures recorded during the low-flow study are provided in Figure 10.7.

Under natural conditions, inter-reach variability in temperature was low. For all three measures of temperature, there was a decrease in early December, after which figures remained elevated until the end of the dry season (about March) (Figure 10.7). The absolute difference in natural temperatures over the study period was 18.9 °C. During the impact phase, temperatures at the flow-impacted site ranged from a minimum of 15.9 °C to a maximum of 26.0 °C, representing a difference of 10.1 °C. The natural range of temperatures for the same time frame was 26.0-14.0 °C, a greater difference of 12.0 °C.

The absolute maximum temperature was similarly high at 28.9 °C and 29.0 °C, for the control and impact locations respectively, in January (i.e. prior to flow reduction). This coincided with the onset of naturally low baseflow conditions in the river (Section 10.2). Lowest maxima were recorded in May, at 16.0 °C and 16.2 °C, in the control and impact locations, respectively. Maximum temperatures were highly similar at

both locations throughout the flow impact and recovery phases, indicating that there was no apparent effect of unnaturally low baseflows on them (Figure 10.7).

Instantaneous temperatures were slightly more variable than the maximum figures, but followed a similar monthly pattern (Figure 10.7). Temperatures in the impacted location were consistently elevated above natural values. However, comparison of the pre-impact data showed a fairly high degree of variability and impact location temperatures were often slightly higher than control temperatures. Statistical analysis would be required to elucidate whether or not the instantaneous temperatures during and after unnatural flow reduction are significantly above natural values (Tharme in prep.).

The absolute minimum temperatures recorded over the study were 10.0 °C (control location) and 9.3 °C (impact location), both in May. They coincided with marked increases in baseflow (Section 10.2) and climatic changes associated with the natural transition to autumn. Minima tended to be more stable, over the months of December to March, than maxima or instantaneous temperatures, only decreasing significantly at the end of the dry season. Although minima at the impacted location were higher than the control values, this trend was initiated prior to the unnatural baseflow reduction. Hence, the difference is unlikely to be of significance in terms of the experiment.

Du Toits site

Variations in temperature for this site, over summer and early autumn, are presented in Figure 10.8.

Under natural conditions, inter-reach variability in temperature was low. For all three measures of temperature, there was an increase from December to January, after which figures remained fairly constant until the end of the dry season. A progressive decrease in temperature occurred consistently from the end of March to May (Figure 10.8). The absolute difference in natural temperatures over the study period was 16.7 °C. During the impact phase, temperatures at the impacted site ranged from a minimum of 15.5 °C to a maximum of 25.0 °C, representing a difference of 9.5 °C. The natural range of temperatures for the same period was 26.9-15.9 °C, a greater difference of 11.0 °C.

The absolute maximum temperature was similarly high at 27.2 °C and 27.0 °C, for the control and impact locations respectively, in January, prior to flow reduction. This coincided with the onset of naturally low baseflow conditions in the river (Section 10.2). The lowest maximum was recorded in May in the control location, at 17.0 °C. The lowest maximum temperature recorded for the impact location, 16.0 °C, occurred earlier, in April, during the recovery phase. Maximum temperatures were highly similar at both locations before the experimental reduction in discharge. However, maxima at the impacted location were consistently lower than natural, for both the impact and recovery periods (Figure 10.8). High similarity in temperature between locations occurred in May, with the onset of elevated baseflows. The possibility that the difference between temperatures at the two locations for these periods is significant, requires testing. It is possible that the difference, though visually apparent, is within a range reflecting natural variability.

Instantaneous temperatures in the impact location were similar or lower than control figures, throughout the study period (Figure 10.7). Statistical analysis would be required to detect any difference associated with the experimental reduction in discharge.

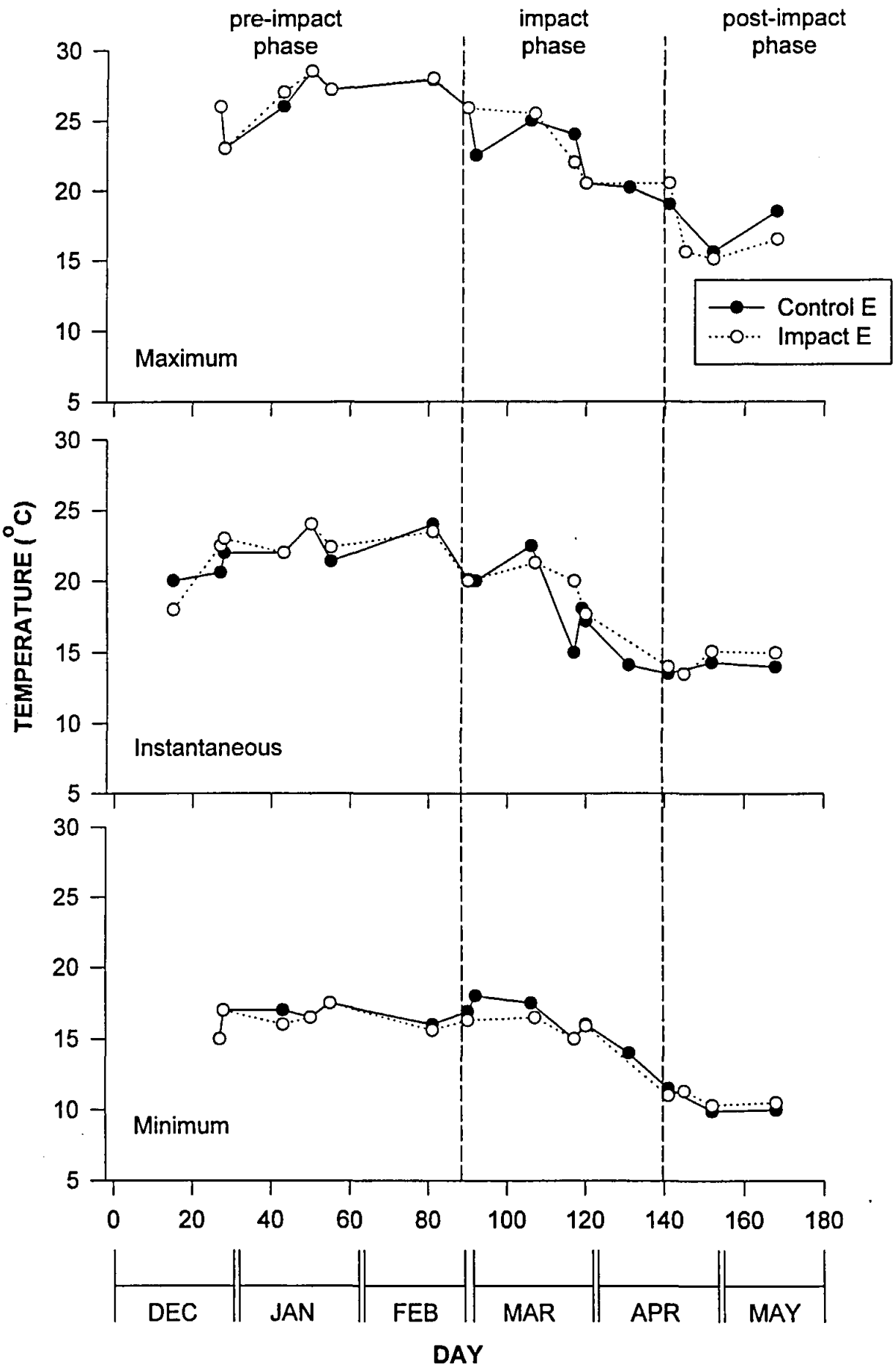


Figure 10.6 **Maximum, instantaneous and minimum temperatures (°C) recorded at the Elands (E) control and impact locations, during the three phases of the low-flow study.**

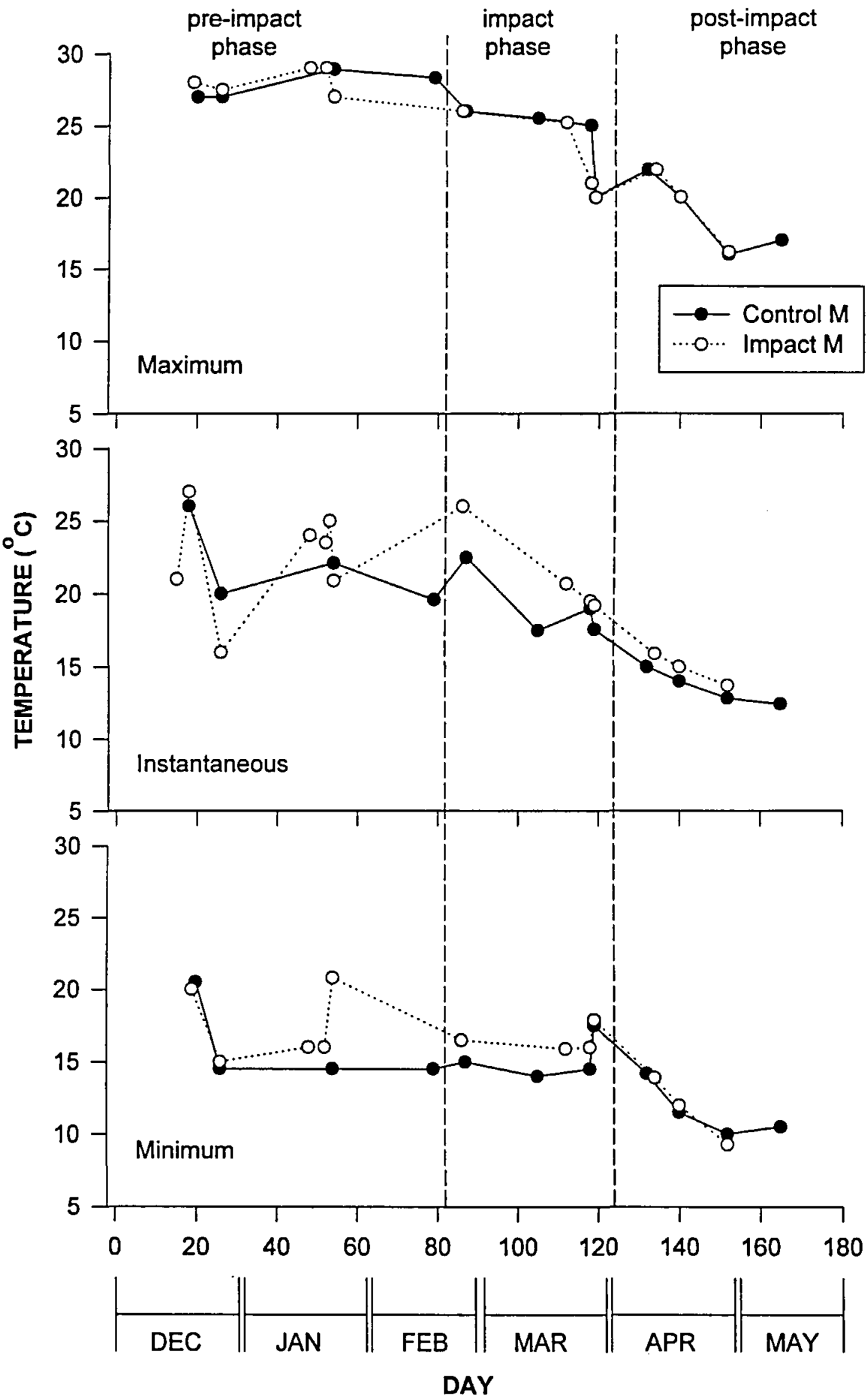


Figure 10.7 **Maximum, instantaneous and minimum temperatures (°C) recorded at the Molenaars (M) control and impact locations, during the three phases of the low-flow study.**

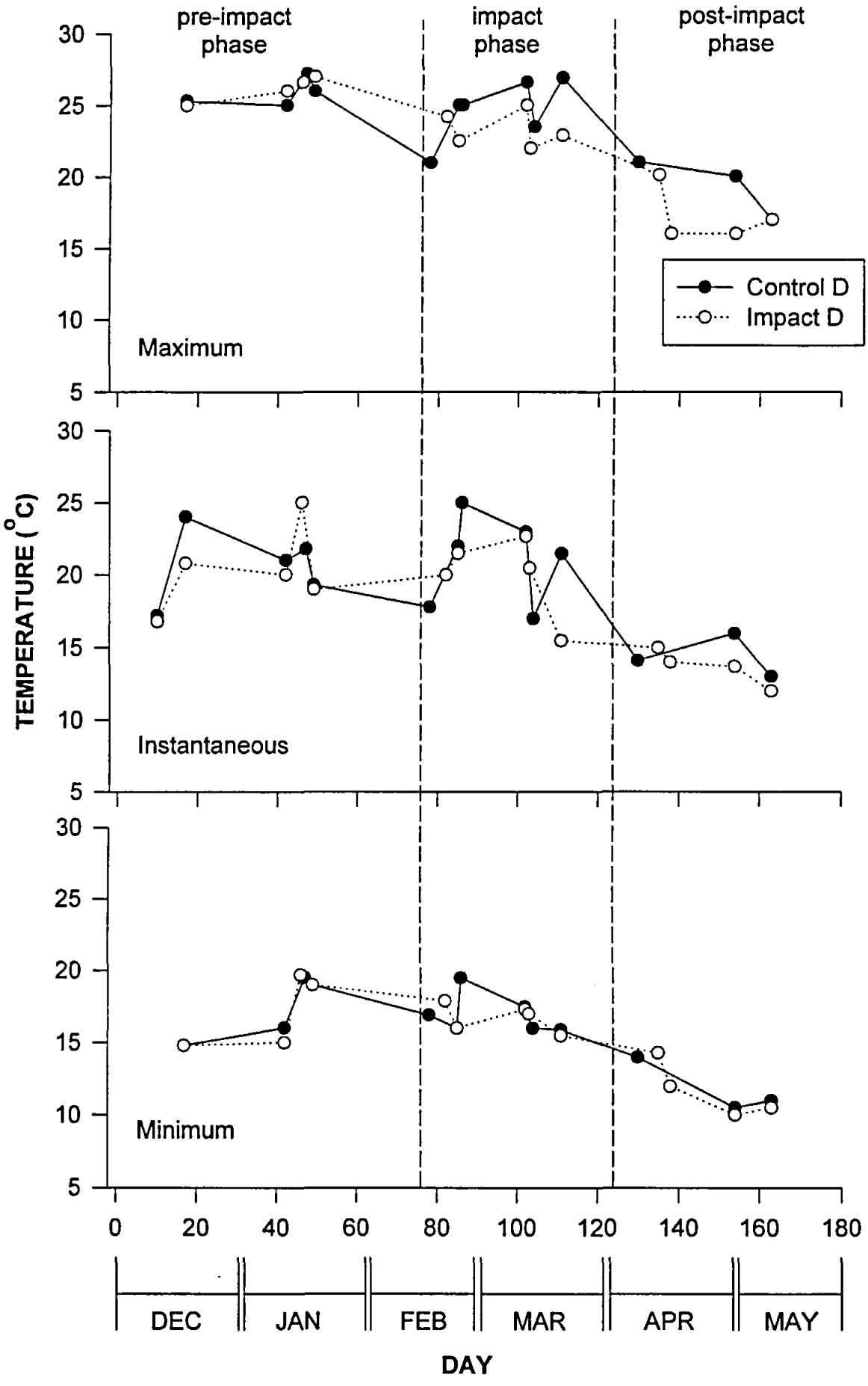


Figure 10.8 **Maximum, instantaneous and minimum temperatures (°C) recorded at the Du Toits (D) control and impact locations, during the three phases of the low-flow study.**

The absolute minimum temperatures recorded over the study were 10.5 °C (control location) and 10.0 °C (impact location), both in May, coincident with natural increases in baseflow (Section 10.2) and climatic changes associated with the transition to autumn. No differences between the minima at the control and impact locations could be detected for the impact or recovery phases (Figure 10.8).

Riviersonderend site

Variations in temperature for this site, over the dry season and early autumn, are illustrated in Figure 10.9.

Under natural conditions, inter-reach variability in temperature was moderately high, especially in terms of minima (Figure 10.9). For all three measures of temperature, there was an increase from December to January, after which temperatures remained fairly elevated until the end of the dry season. A fairly steep decline in temperature occurred with the occurrence of higher flows in May (Figure 10.9). The absolute difference in natural temperatures over the study period was 18.7 °C. During the impact phase, temperatures at the impacted site ranged from a minimum of 16.0 °C to a maximum of 27.7 °C, a difference of 11.7 °C. The natural range of temperatures for the same period was 27.5-17.0 °C, a marginally lesser difference of 10.5 °C.

The absolute maximum temperature was similarly high at 27.5 °C (in March) and 28.0 °C (in both January and February), for the control and impact locations, respectively. These high temperatures were reached during the peak months of the dry season (Section 10.2). The lowest maximum of 14.0 °C was recorded in April in the impact location. The lowest maximum temperature recorded for the control location was recorded as 19.5 °C, in December. However, this result is more likely a function of few observations for April/May, when temperatures would be expected to be lowest. Maximum temperatures were similar at both locations or marginally higher for the impact location during all phases of the study, and no marked impact of the 86% reduction in flow (Section 10.2) on maximum temperature was evident.

Instantaneous temperatures were generally variable, with temperatures in the impacted location marginally higher than natural. There was no marked indication of any effect of unnatural flow reduction (Figure 10.9).

The absolute minimum temperatures recorded over the study were 8.8 °C (control location) and 9.5 °C (impact location), both in May, coincident with natural increases in baseflow (Section 10.2) and climatic changes associated with the transition to autumn. No differences between the minima at the control and impact locations could be detected for the impact or recovery phases (Figure 10.8). The temperatures at the impact location prior to unnatural flow reduction were often well below those of the control location, which is a somewhat anomalous result. It indicates that natural variability in minimum temperatures is rather high for this mountain-stream site.

Comparison of temperatures at the four sites

Natural temperature ranges were highly similar among sites, as well as within the same reach, with maximum temperatures of about 27-29 °C and minima in the order of 9-10 °C. The natural, absolute differences in temperature (maximum to minimum) for the sites, for the entire six month period, were: 18.6 °C for the Elands River; highest at 18.9 °C, for the Molenaars River; lowest at 16.7 °C, for the Du Toits

River; and 18.7 °C for the Riviersonderend River. There was a slight tendency for the foothill reaches to have higher maxima and higher minima than the cooler mountain-stream and transitional zones. However, the difference was not marked during the peak dry season. Maximum water temperatures were very high during the middle of summer in the upper reaches. Minimum temperatures tended to be more stable than maxima.

Natural temperatures were fairly consistent over the peak dry-season months of January to March, and declined with the onset of autumn instream conditions. This decrease in temperature was not necessarily associated with a large increase in baseflow. The range in temperature during the impact phase was lowest for the Riviersonderend site, at 10.5 °C and highest at the Molenaars site, at 12.0 °C. Intermediate differences from maximum to minimum temperature were obtained for the Du Toits River (11.0 °C) and Elands River (11.9 °C).

With unnatural reduction in baseflows, there were generally no consistent or marked effects on temperature. The only exception was an apparent decrease in maximum temperature, with 86% reduction in discharge, at the Du Toits site. A moderately large reduction in discharge possibly resulted in an increase in instantaneous and minimum temperature during the impact phase. In addition, a slight decrease in instantaneous temperature was apparent for one of the sites (Du Toits) at which flow reduction was high. However, these kinds of trends require statistical validation (Tharme in prep.). The range between maximum and minimum temperature, recorded during the impact phase, did not follow any trend in relation to the magnitude of flow reduction. The difference in temperature was lowest for the Du Toits site, at 9.5 °C, greater for the Molenaars site (10.1 °C), and highest for the Riviersonderend River, at 11.7 °C. For only the former two sites, was the difference lower than under natural flow conditions.

10.3.3 Effects of natural and unnatural dry-season baseflows on pH and total alkalinity

The changes in pH and alkalinity, at each site and location, for the duration of the low-flow study, are illustrated in Figures 10.10 and 10.11, respectively.

Elands site

The natural range in mainstream pH, as indicated by both the control and impact locations for the Elands was fairly low, from 5.23 (late February) to 6.00 (January) for the control location, a difference of 0.8 pH units. For the impact location, the range extended from 4.60 (March) to 6.03 (January), a greater difference of 1.4 pH units. Natural variability in pH appears moderate, increasing during the late dry season and early autumn (Figure 10.10).

Total alkalinity was naturally highly variable throughout the study period, ranging overall from 0.036 mEq ℓ^{-1} to 0.121 mEq ℓ^{-1}, a difference of 0.085 mEq ℓ^{-1} (Figure 10.11). This variability rendered the differences in alkalinity observed at the impacted sites (Figure 10.11; see below) questionable in terms of their actual significance. Further analysis is required to ascertain whether or not changes in alkalinity at the other sites might be a function of unnaturally low flows (Tharme in prep.).

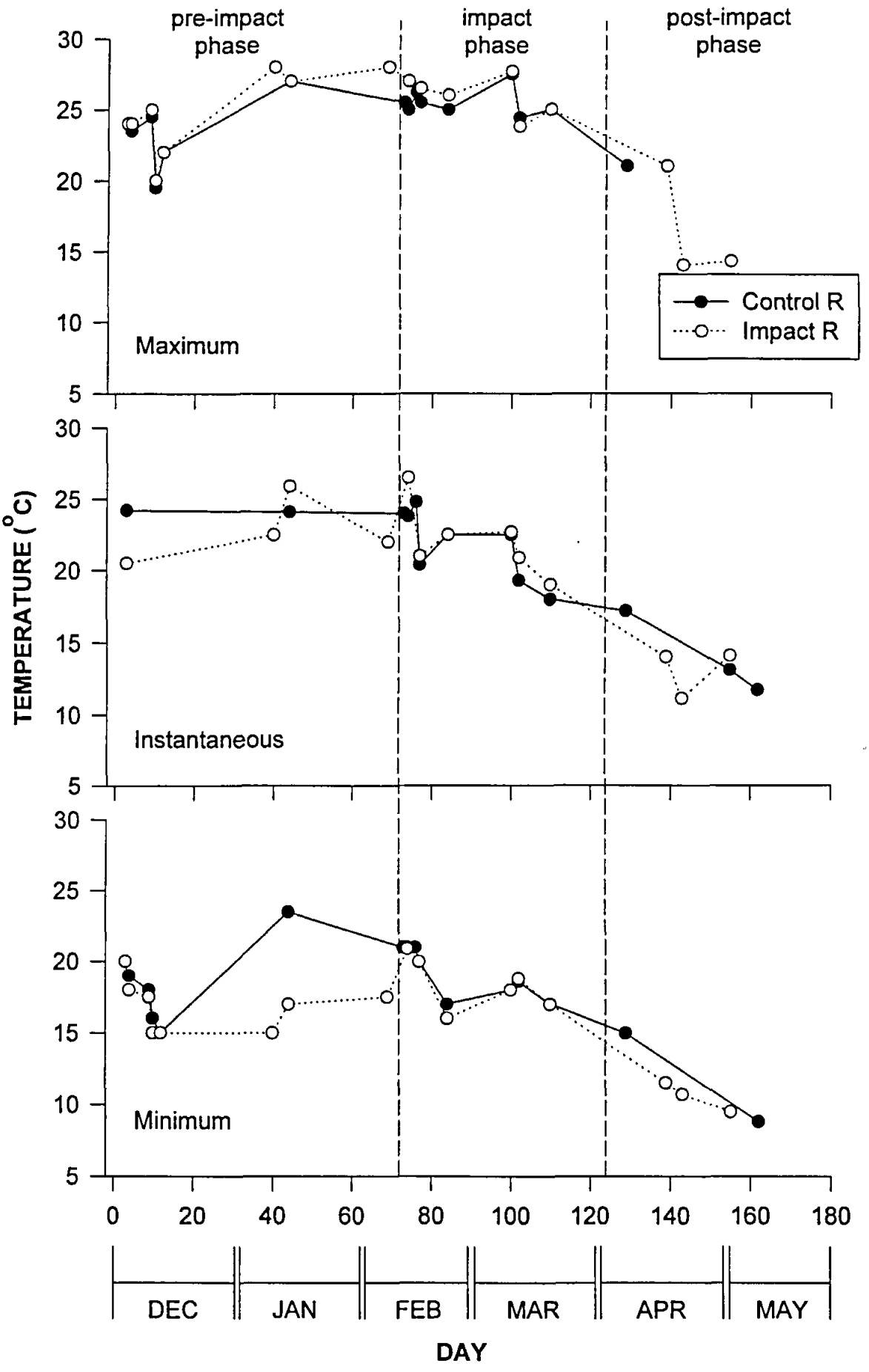


Figure 10.9 **Maximum, instantaneous and minimum temperatures (°C) recorded at the Riviersonderend (R) control and impact locations, during the three phases of the low-flow study.**

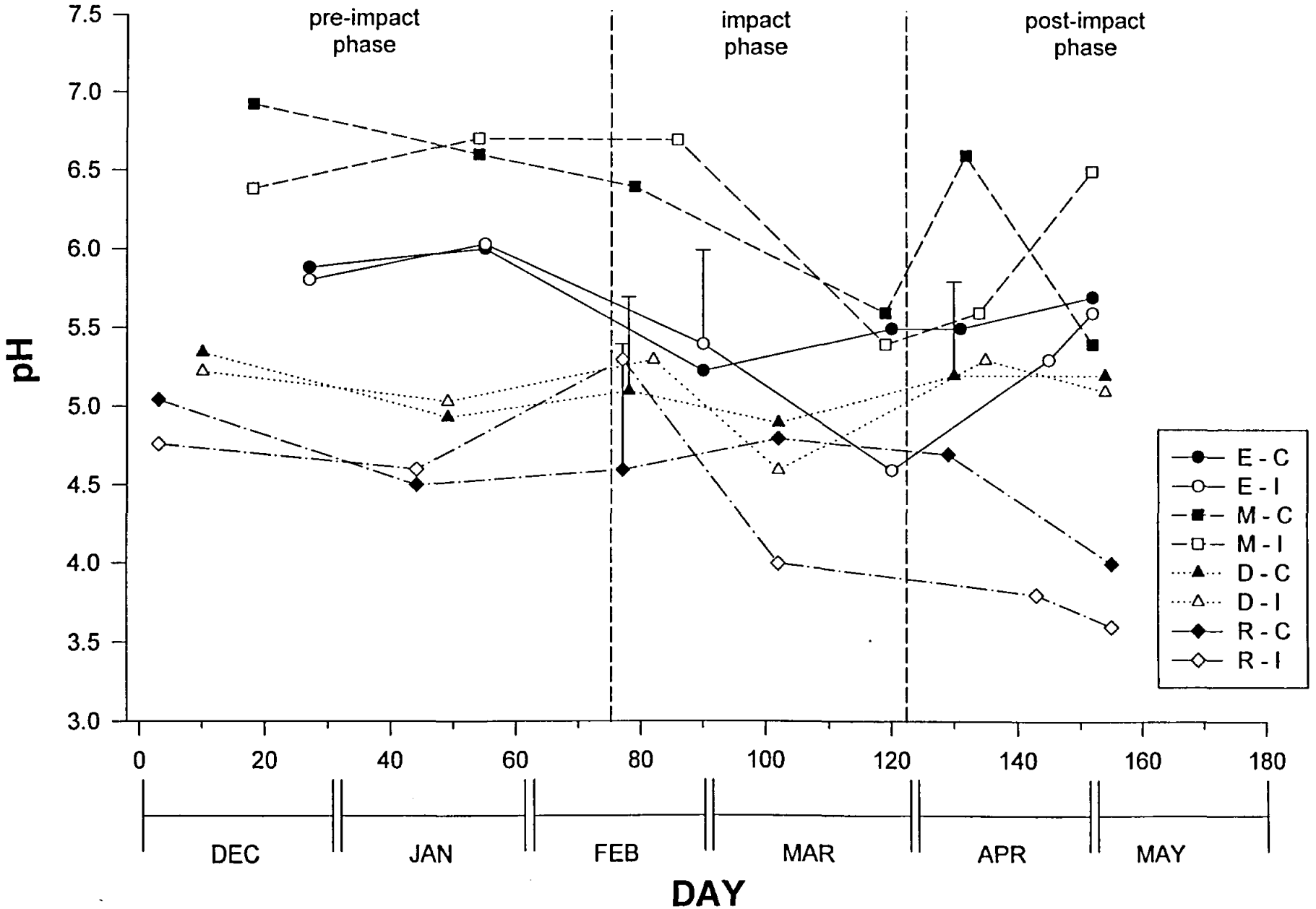


Figure 10.10 Changes in pH at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates). Upper error bars indicate pH values in standing pools, where pH was ≥ 10% of mainstream pH.

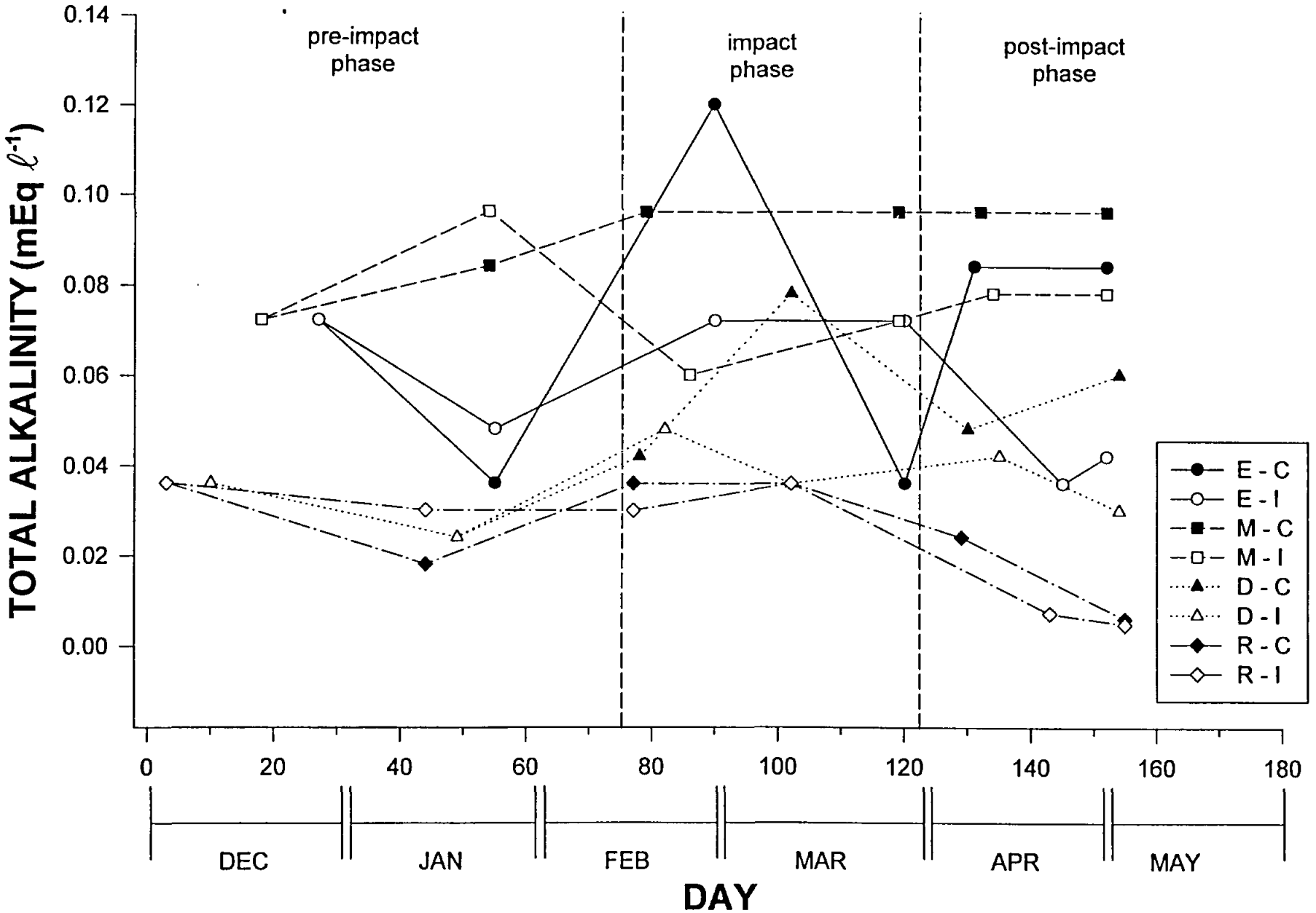


Figure 10.11 Changes in total alkalinity (mEq ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

Molenaars site

The pH values were generally similar in the control and impact locations, and among months, irrespective of unnatural flow reduction (Figure 10.10). However, there was some divergence between pH at the two locations early in the recovery phase when baseflows were still low (Section 10.2); April pH was lower in the impacted location.

The natural range in mainstream pH for the control location was moderate, from 5.40 (May)-6.92 (December), representing 1.5 pH units. For the impact location, the range extends from 5.40 (March)-6.70 (January and February), a lesser difference of 1.3 pH units.

Total alkalinities were noticeably lower with the unnatural reduction in baseflow than control values, during both the impact and recovery phases (Figure 10.11). However, they were within the natural range of variability recorded for the control site (Elands; see above). Control values were stable through summer and autumn, and ranged from 0.072-0.096 mEq ℓ^{-1}. Alkalinities at the impact location ranged from 0.060 mEq ℓ^{-1} (February)-0.096 mEq ℓ^{-1} (pre-impact, January).

Du Toits site

The pH values were very similar in the control and impact locations, and among months, irrespective of unnatural reductions in discharge (Figure 10.10). The natural range in mainstream pH for the control location was very low at 0.4 pH units (4.90 in March, to 5.34 in December). In the impact location, the range extended from 4.60 (March)-5.30 (February and April), a greater but similarly low difference of 0.7 pH units.

Total alkalinities were very similar in both locations before unnatural reduction in baseflow. Alkalinities were lower in the impacted location for the duration of the impact, and to a lesser extent after it was removed (Figure 10.11). However, they were within the natural range of variability recorded for the Elands (see above). Natural TAs ranged from a minimum of 0.024 mEq ℓ^{-1} in mid-summer (January) to a maximum of 0.078 mEq ℓ^{-1}, experienced in late summer (March). Lowest TAs at the impact location were the same as natural values, but reached a lower maximum of 0.048 mEq ℓ^{-1} (February). This represented a very low difference of 0.024 mEq ℓ^{-1} compared with a natural difference of 0.054 mEq ℓ^{-1}.

Riviersonderend site

The pH values were highly similar in the pre-impact phase, but diverged notably during the impact and recovery phases (Figure 10.10). In the impacted location, pH was initially higher than under natural conditions, and subsequently became increasingly acidic from March to May. The pH values approximated each other for the two locations, in May, with the flushing action of higher flows (Section 10.2). High acidities were probably the result of flushing out of high levels of humic acids associated with the observed desiccation and decomposition of plant matter, especially *Isolepis digitata*, at unnaturally low flows. Statistical analysis would be required to test whether or not the observed difference is significant (Tharme in prep.). During the recovery phase, flushing of the impacted section of river resulted in the river turning dark orange-brown in colour, probably as a result of leachates from decomposed vegetation. Such a change in

colouration was also observed in impacted isolated pools (pers. obs.). A natural increase in humic staining of the water also occurred in the control location with the onset of elevated baseflows in May, although to a far lesser extent.

The natural range in mainstream pH for the control location was fairly low at 1.0 pH units (4.00 in May, to 5.04 in December). In the impact location, the range was high, extending from only 3.60 (May) to 5.30 (February), a greater difference of 1.7 pH units.

Total alkalinities were very similar under conditions of natural and unnaturally low baseflows (Figure 10.11). Ranges in variability were similarly low in both locations, and minimum values were extremely low in May at 0.006 mEq ℓ^{-1} (control location) and 0.005 mEq ℓ^{-1} (impact location).

Comparison of pH and alkalinity at the four sites

Natural pH and total alkalinity values were greatest for the Molenaars site and then the Elands site, while the Riviersonderend site exhibited the lowest TAs and highly acidic waters. Values for these variables were slightly higher at the Du Toits site than for the Riviersonderend River.

With unnatural reductions in flow there were no marked trends in pH. Although the range in pH was almost double for both the highly flow-impacted locations of the Du Toits and Riviersonderend sites, it was similarly nearly double for the Elands River. However, the trend may still be noteworthy, as the latter site appears naturally more variable in pH than the other two sites (Tharme in prep.).

In isolated standing pools in the locations experiencing natural dry-season baseflows, pH values were generally higher than those of the mainstream, and only occasionally lower or the same (Tharme in prep.). In isolated pools within locations impacted by unnatural reduction in flow, there was no obvious trend (Tharme in prep.). Considering only differences in pH in pools that exceeded 10% of mainstream pH, only increases in pH were observed, and solely for non-impacted locations (Figure 10.10). Such differences were not greater than a single pH unit and typically occurred in the month of lowest flow. According to Dallas *et al.* (in press), diel changes of 1 pH unit have been found to occur naturally. There were no marked differences in pH between the mainstream and isolated pools for impacted locations (Figure 10.10).

Flow reduction to moderately and extremely low levels, appeared to result in a reduction in alkalinity, both during the period of flow reduction and immediately following it. However, this was not the case for the site with the lowest pH, and hence lowest buffering capacity (Dallas and Day 1993). This inconsistent pattern requires further examination (Tharme in prep.).

10.3.4 Effects of natural and unnatural dry-season baseflows on conductivity, total dissolved solids and total suspended solids

The changes in conductivity and TDS, at each site and location, for the duration of the low-flow study, are illustrated in Figures 10.12 and 10.13, respectively. TSS concentrations were extremely low in all instances where data were available (Section 9.5.2), and are therefore not discussed further here.

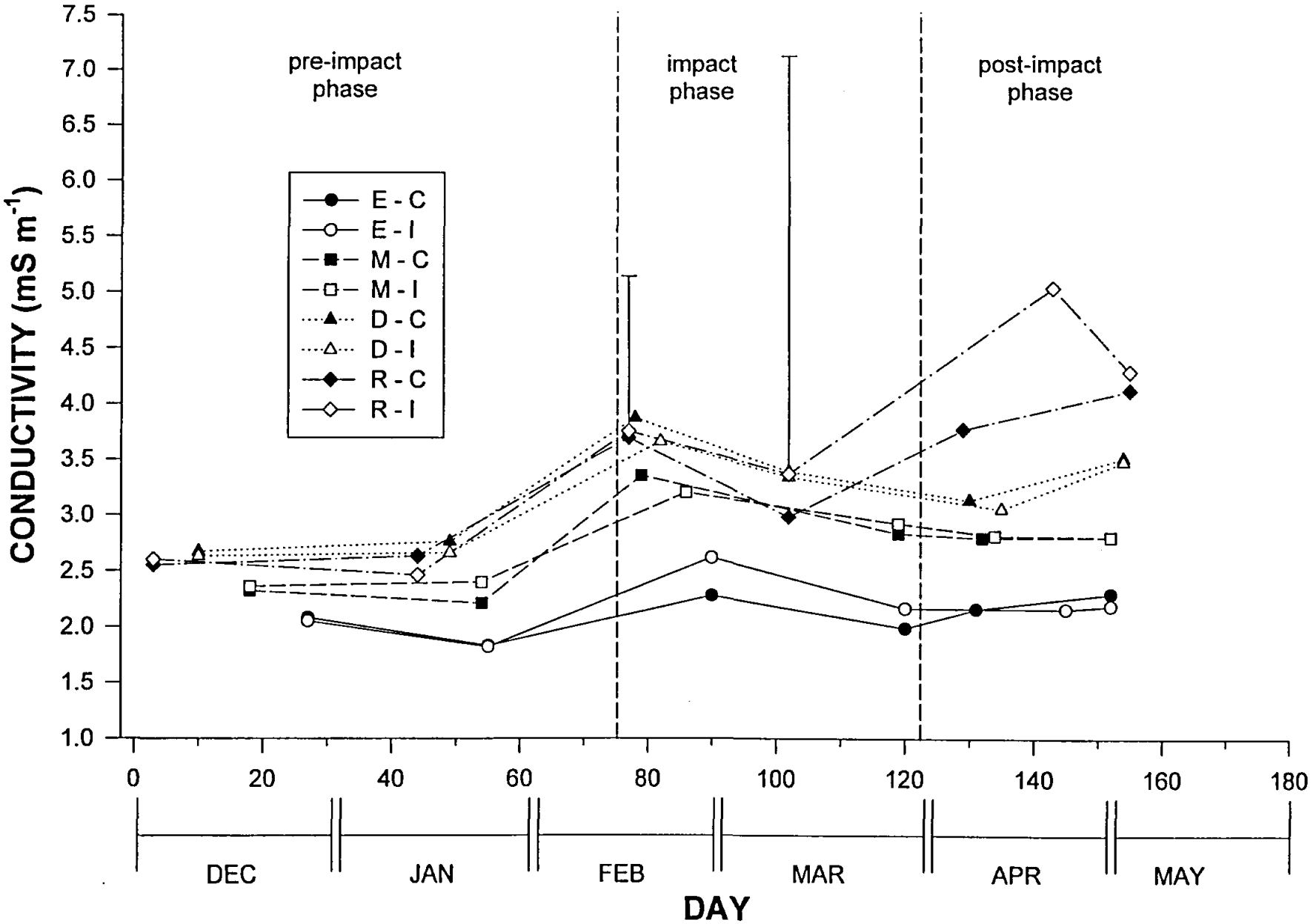


Figure 10.12 Changes in conductivity (mS m^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates). Upper error bars indicate conductivities in isolated standing pools, where values were ≥ 10% of mainstream conductivity.

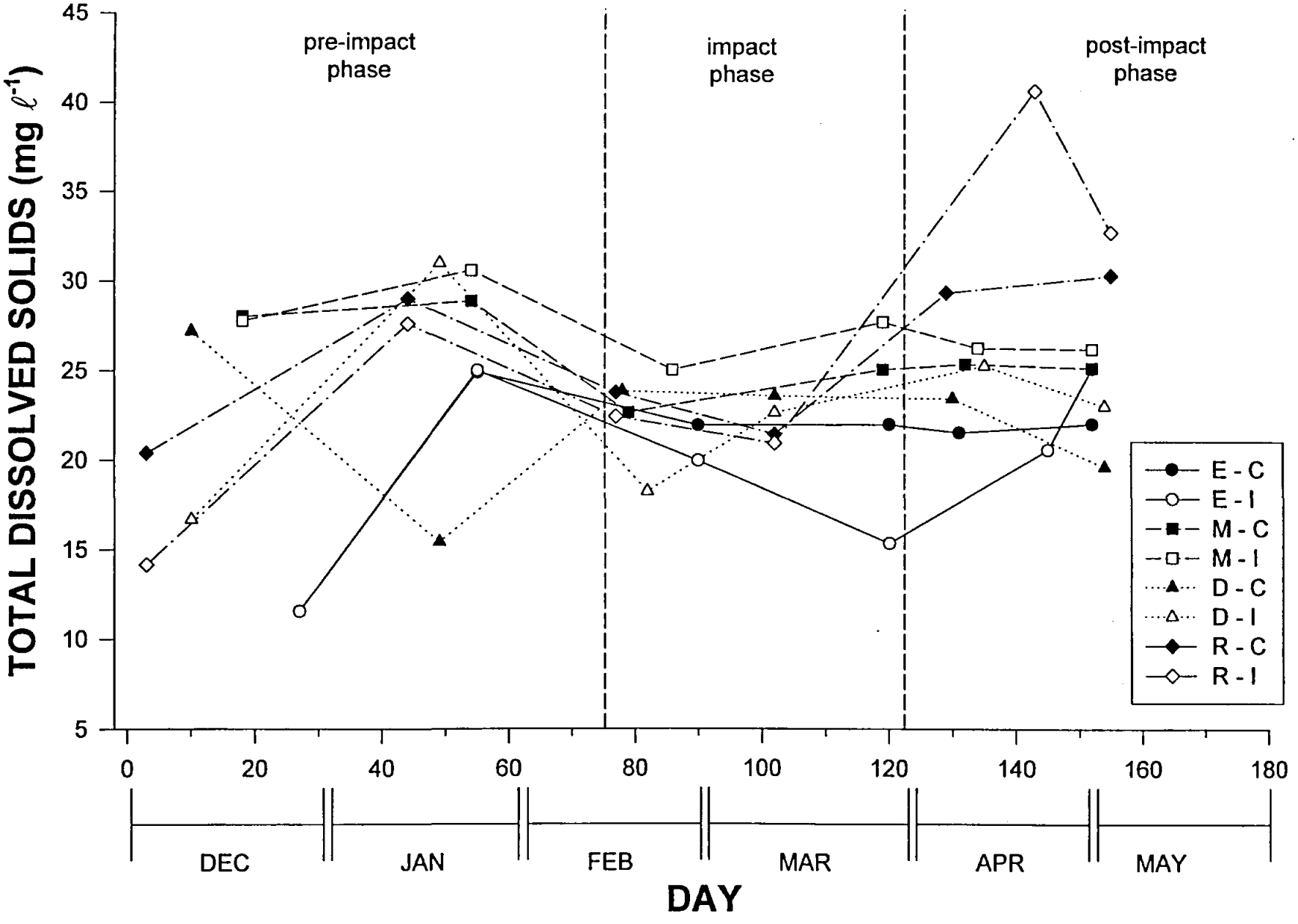


Figure 10.13 Changes in total dissolved solids (mg ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

Elands site

Figure 10.12 shows that natural conductivities were low, highly similar and stable throughout the dry season in the Elands River, ranging from 1.82 mS m^{-1} in January to 2.62 mS m^{-1} at the end of February (a difference of 0.8 mS m^{-1}). TDS values were more variable in the peak dry season than conductivities, and the timing of highest and lowest concentrations was not coincident. The overall range in TDS was from 11.54-25.13 mg ℓ^{-1} (Figure 10.13).

Molenaars site

Conductivities were highly similar, low and stable throughout the dry season. There did not appear to be any effect of unnatural reduction in flow (Figure 10.12). Control values ranged from 2.21 (January)-2.84 mS m^{-1} (March), while the maximum conductivity recorded at the impacted location was higher at 3.21 mS m^{-1} (February). TDS values were similarly consistent, and similar for the control and impact locations. The recorded minima were 22.65 mg ℓ^{-1} and 25.06 mg ℓ^{-1}, and maxima 28.87 mg ℓ^{-1} and 30.58 mg ℓ^{-1}, respectively (Figure 10.13).

Du Toits site

Conductivities were highly similar, low and stable throughout the dry season. There did not appear to be any effect of unnatural reduction in discharge (Figure 10.12). Minimum and maximum values were similarly highest for the control and impact locations in the month of lowest flow (February), at 3.88 mS m^{-1} and 3.67 mS m^{-1}, respectively. TDS values were more variable in range, but also did not show any obvious effects associated with unnaturally low baseflows (Figure 10.13). The maximum TDS was 30.97 mg ℓ^{-1} (impact location) and the minimum was 15.40 mg ℓ^{-1} (control location).

Riviersonderend site

Conductivities in the control and impact locations diverged during the impact phase and were markedly dissimilar during the recovery phase of the study, about 19 days after natural baseflows were reinstated (Figure 10.12). Natural conductivities were low and ranged from 2.55 mS m^{-1} (December) to 4.13 mS m^{-1} in May, a difference of 62%. In contrast, conductivities in the impacted location were elevated to a maximum mainstream concentration of 5.05 mS m^{-1} in April, a difference from minimum to maximum concentration of 105%. This was most likely a function of flushing of pools of high conductivity, once connectivity was re-established with the increase in baseflow. The most pronounced effects of the unnaturally low baseflows at this site occurred in areas of the bed where pools entirely lost connectivity with flowing areas. Conductivities in such areas were up to 112% above already impacted, mainstream values, reaching 7.15 mS m^{-1} in March (Figure 10.12).

TDS values were variable under natural and unnatural flow conditions. They only showed a distinct increase to a maximum of 40.70 mg ℓ^{-1}, coincident with an increase in conductivity, at the impact location, after natural flows were reinstated in April (Figure 10.13). No records of TDS were made for isolated pools. Recovery to natural conductivity and TDS values was rapid, and coincided with high flows in autumn.

Comparison of conductivity and total dissolved solids at the four sites

Natural conductivities and TDS concentrations were similarly low at all four sites, with the lowest values recorded for the Elands River. Typically, conductivities increased to a maximum in the peak dry season, at low baseflows, and subsequently decreased. TDS values did not follow an obvious pattern, and were similar at all sites.

The Riviersonderend was the only site that showed a clear increase in conductivity with unnatural baseflow reduction, both in the still-flowing sections of the mainstream and in isolated pools. However, TDS values generally did not reflect very low flows, except during flushing of the impacted location in April. Conductivities recorded from natural and impacted standing pools at the other sites did not show any consistent increases or decreases in conductivity greater than 10% of mainstream values. The impacts of flow reduction on the two chemical variables appeared short-lived.

10.3.5 Effects of natural and unnatural dry-season baseflows on dissolved oxygen

Dissolved oxygen (DO) naturally fluctuates diurnally (Dallas *et al.* 1994), so the following figures illustrate only the broad patterns in DO, as percentage saturation, observed at each site, for the three main biotopes from which BMIs were sampled. Trends in DO as mg ℓ^{-1}, were much the same as for percentage saturation.

Elands site

DO concentrations for the Elands site are shown in Figure 10.14.

Oxygen concentrations were variable within the same reach, but showed the same trends for both locations. Dissolved oxygen concentrations increased dramatically from pools, to runs to levels around 100% saturation in riffles. The absolute minimum natural DO value was recorded in a pool in February, at 38.6%. Variability in DO was lowest in riffles and greatest in pools.

Molenaars site

Oxygen concentrations were naturally variable, and increased dramatically from pools, to runs to levels around 100% saturation in riffles (Figure 10.15). During the period of 36% reduction of baseflow, DO levels in impacted riffles and runs decreased from natural levels. However, the degree to which such changes are significant, given the considerable natural variability in DO, needs to be assessed (Tharme in prep.). Lowest DO concentrations (34.9%) were recorded in an impacted pool, during the month of lowest flow, February. Variability in DO was characteristically lowest in riffles and greatest in pools.

Du Toits site

For the Du Toits site, DO concentrations followed a natural trend of increasing from pools, to runs to levels around 100% saturation in riffles (Figure 10.16). Variability between locations was low, and extreme flow reduction did not appear to have any effect on oxygen concentrations. DO concentrations reached a minimum of 40.0% under conditions of natural baseflow, early in the dry season.

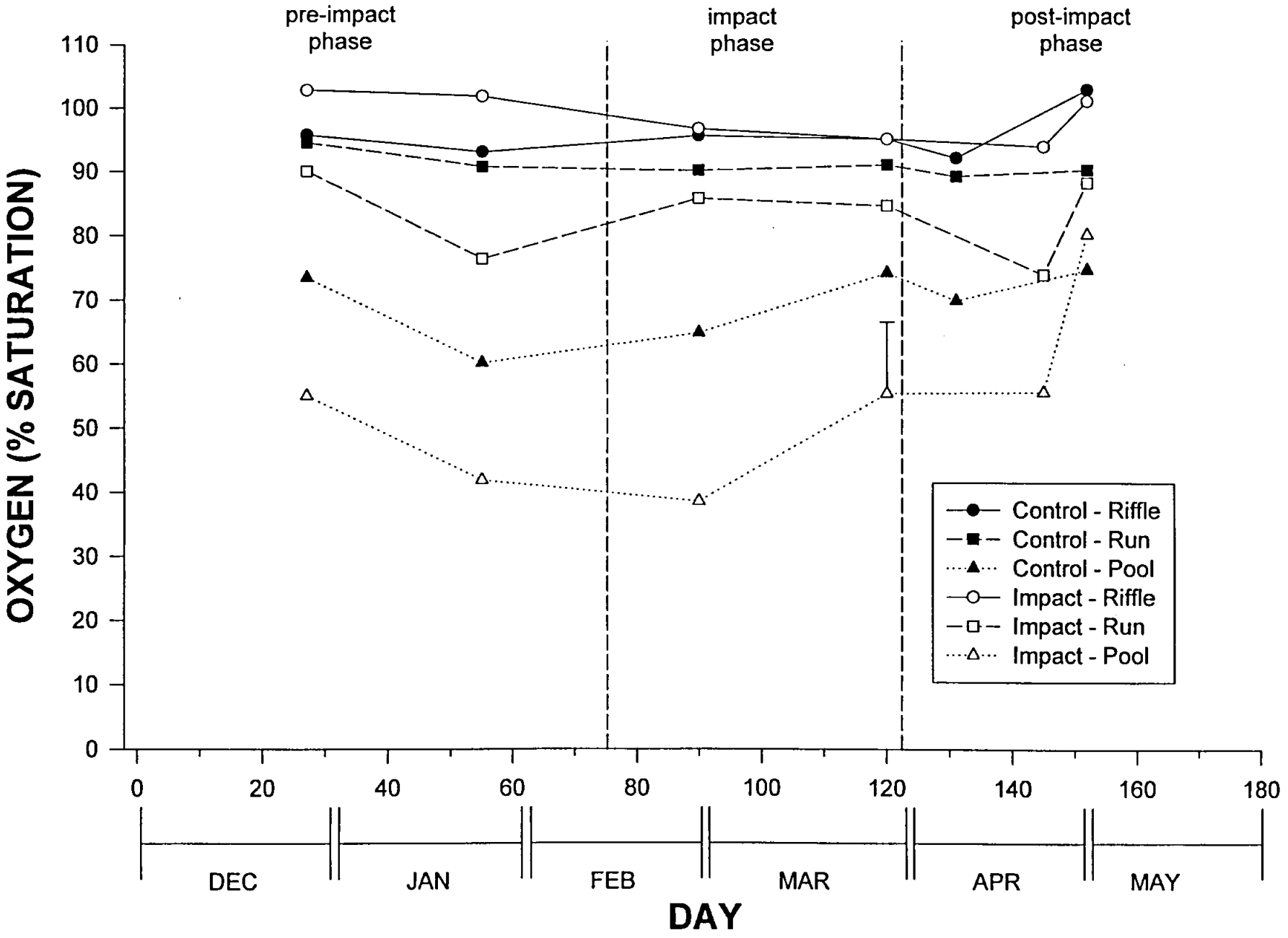


Figure 10.14 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Elands site, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates). The upper error bar indicates an oxygen concentration in an isolated standing pool that was ≥ 10% of the oxygen level in mainstream pools.

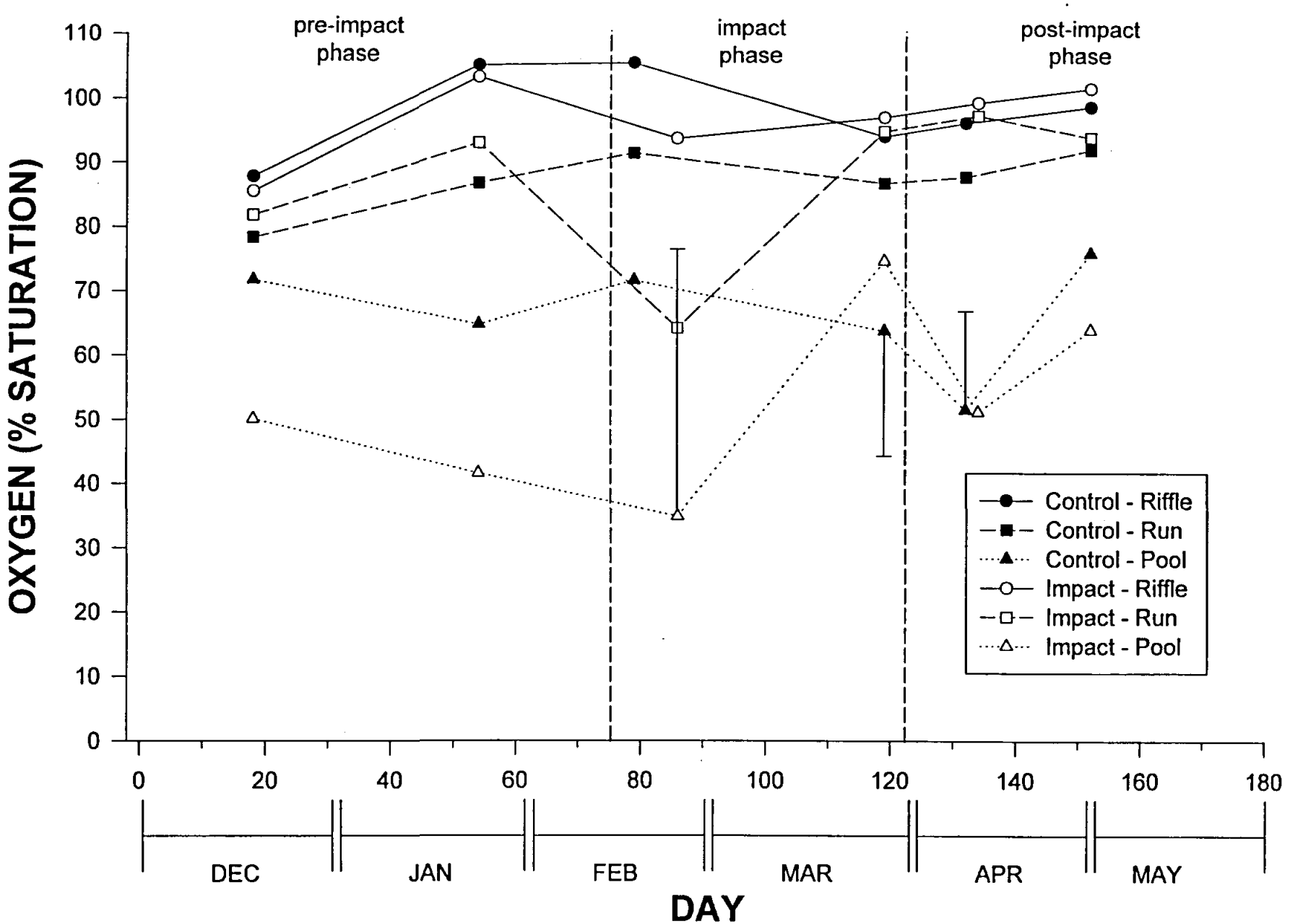


Figure 10.15 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Molenaars site, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates). Error bars indicate oxygen concentrations in isolated standing pools that were ≥ 10% of the oxygen level in mainstream pools.

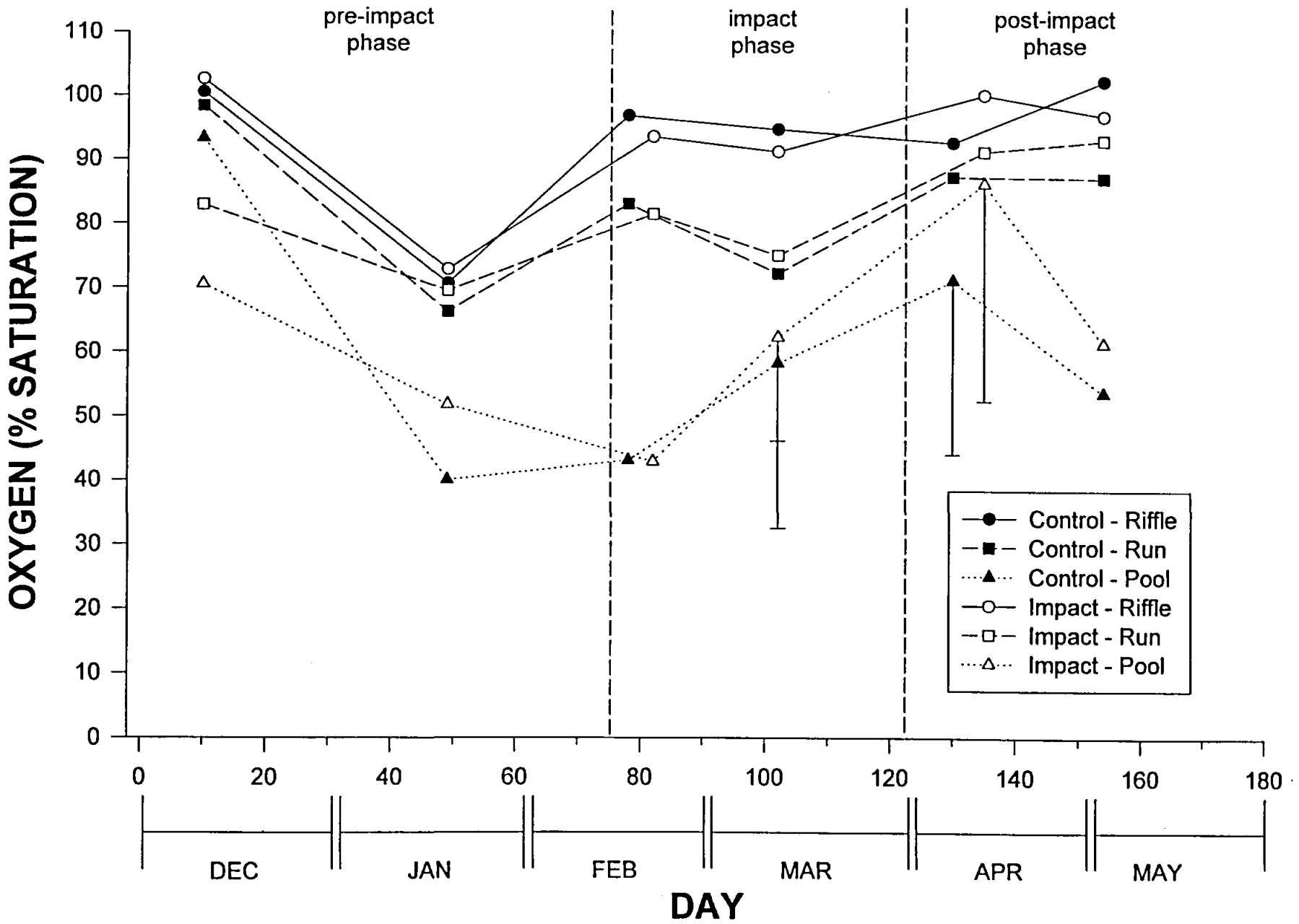


Figure 10.16 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Du Toits site, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates). Lower error bars indicate oxygen concentrations in isolated standing pools that were ≥ 10% of the oxygen level in mainstream pools.

Riviersonderend site

DO concentrations for this site followed a natural progression of increasing concentration from pools to runs, to levels of 100% saturation in riffles (Figure 10.17). Variability in DO was fairly high among months and locations, and there was no apparent effect of an 86% baseflow reduction. DO concentrations reached a minimum of 37.8% under natural, peak dry-season baseflows.

Comparison of dissolved oxygen concentrations at the four sites

DO concentrations for various biotopes were highly similar in range among sites. Riffles consistently had the highest oxygen levels, and pools exhibited the lowest levels. This trend indicates that increased discharge likely has a secondary influence on oxygen levels, by increasing flow velocity and mixing through turbulence. There was no apparent effect of extremely low flows on oxygen concentrations, even in isolated pools. However, a decrease from natural concentrations was observed in natural riffles and runs, with 36% reduction in baseflow. Although DO levels were similarly low in pools under conditions of natural and unnatural dry-season baseflows, protracted isolation of pools would likely result in progressively anaerobic conditions. Detection of such changes would require intensive, continuous monitoring of oxygen levels, which was beyond the scope of this study.

10.3.6 Effects of natural and unnatural dry-season baseflows on major ions

Changes in the concentrations of the major cations at each site and location, for the duration of the low-flow study, are illustrated in Figures 10.18-10.21, while anion concentrations are shown in Figures 10.22-10.23. Total alkalinities, as HCO_3^-, are discussed in Section 10.3.3.

From Figure 10.18, it is evident that concentrations of the dominant cation, sodium, were similar in range for all sites. The Elands River showed the greatest natural variability in concentration, with both the overall maximum concentration of 0.206 mmol ℓ^{-1} (February) and minimum concentration of 0.103 mmol ℓ^{-1} (January) occurring in the control location at this site. Sodium appeared to be a stable variable on a month-by-month basis, with no obvious relationship with natural baseflow magnitude. There was also no apparent, direct effect of unnaturally low baseflows on this variable.

Potassium concentration was most variable for the Elands, with a peak in the dry season (Figure 10.19). Unnatural baseflow reduction did not appear to alter K^+ levels from natural ones in the Molenaars (Figure 10.19). Although values for the impacted location were higher than natural for the Du Toits (February only) and Riviersonderend (Figure 10.19), they are not likely to be significantly different. The concentrations of K^+ for the Du Toits and Riviersonderend sites are naturally, roughly an order of magnitude lower (*c.* 0.002 mmol ℓ^{-1} in January, to 0.006 mmol ℓ^{-1} in May) than those of the Elands and Molenaars (*c.* 0.007 mmol ℓ^{-1} in February, to 0.017 mmol ℓ^{-1} in March) (Figure 10.19). The former two sites showed a gradual increase in K^+ from December to May, while values for the latter sites decreased after the peak dry season.

Calcium was the most variable of all the cations, though within a moderately restricted overall range for all four sites (*c.* 0.002 mmol ℓ^{-1} in December, to 0.040 mmol ℓ^{-1} in April) (Figure 10.20). For the Molenaars,

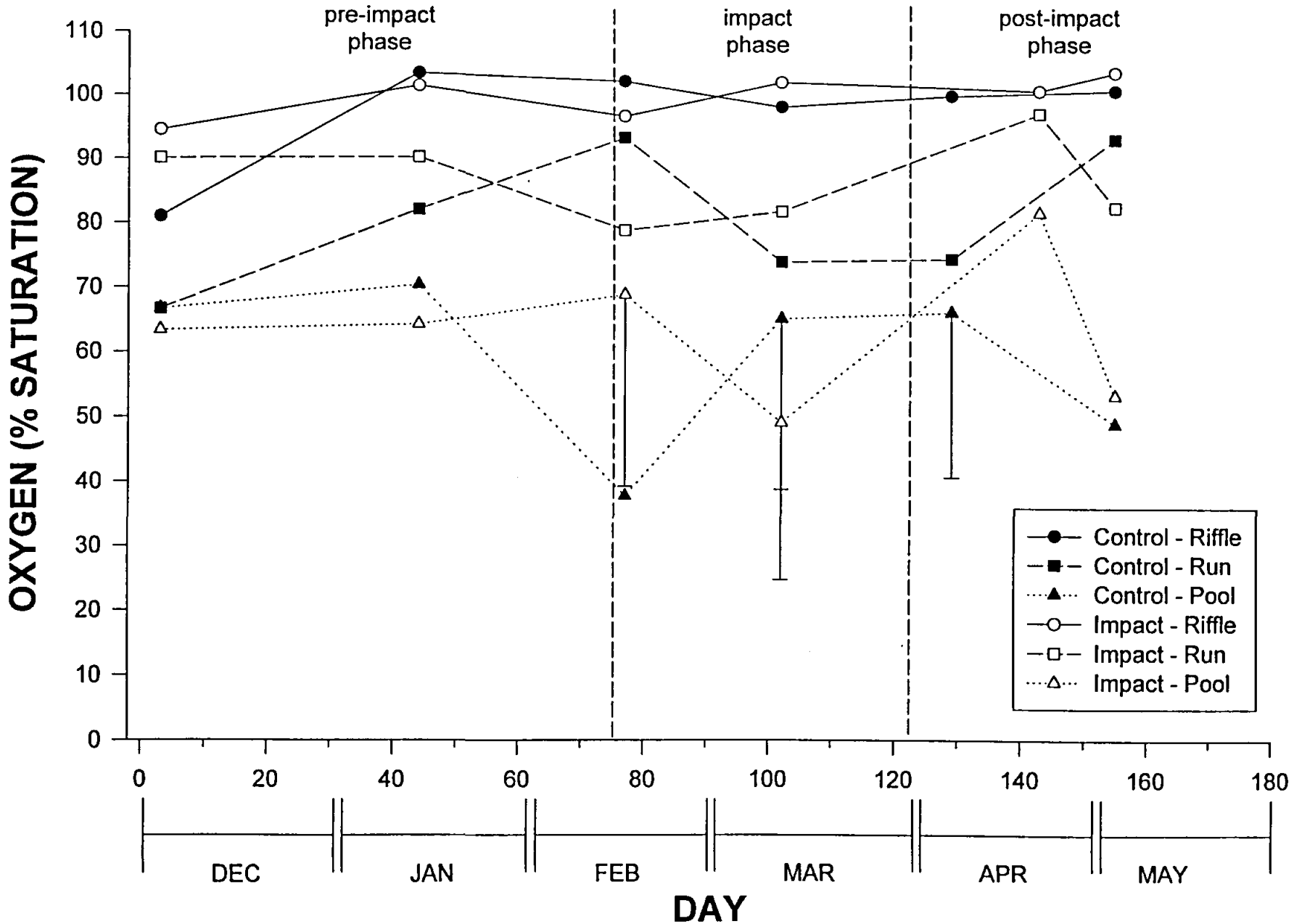


Figure 10.17 Changes in dissolved oxygen, as % saturation, for three major biotopes at the Riviersonderend site, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates). Lower error bars indicate oxygen concentrations in isolated standing pools that were ≥ 10% of the oxygen level in mainstream pools.

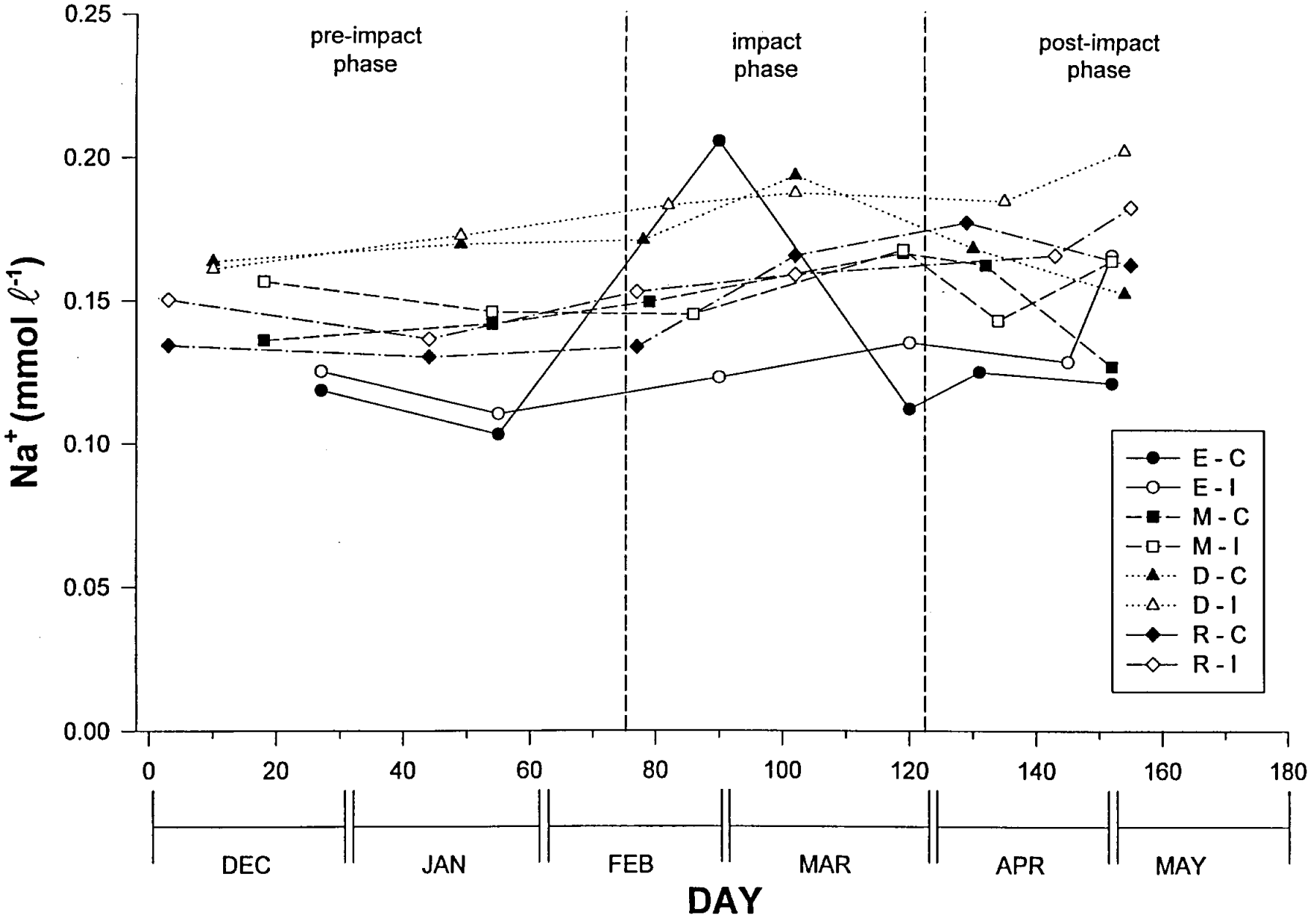


Figure 10.18 Changes in Na^+ (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

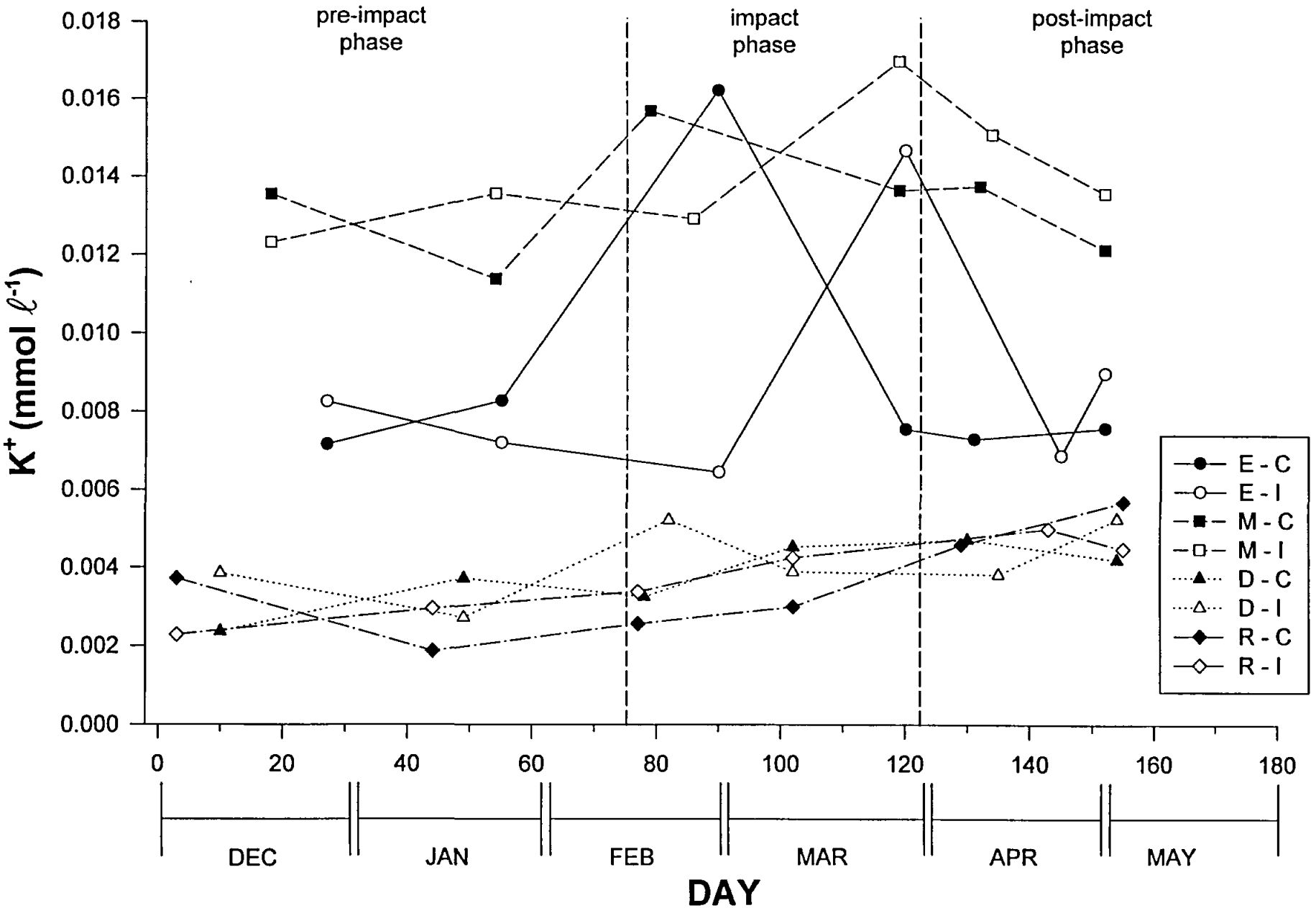


Figure 10.19 Changes in K^+ (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

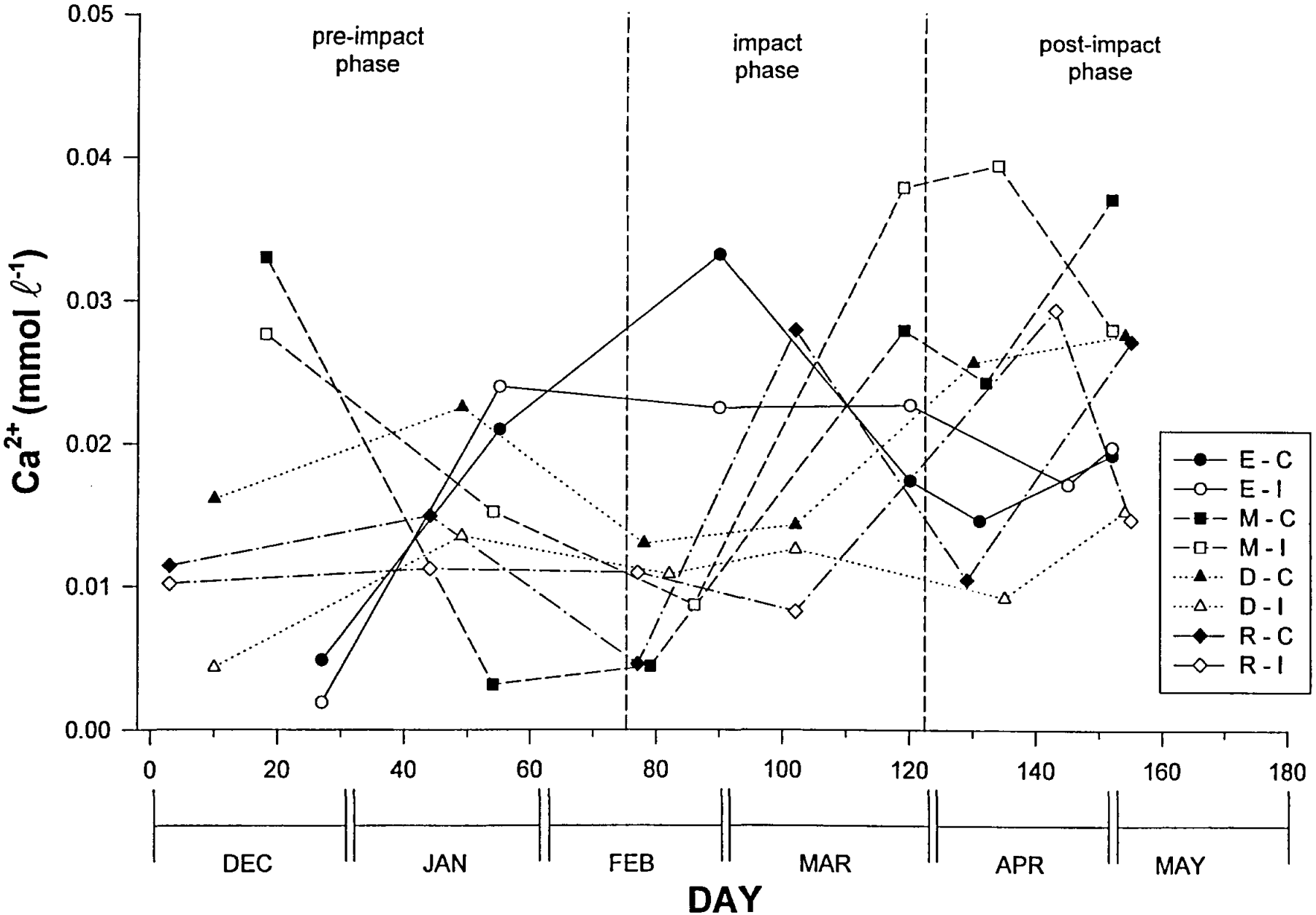


Figure 10.20 Changes in Ca^{2+} (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

concentrations during the impact and recovery phases were greater than control concentrations (Figure 10.20), but the difference is unlikely to be significant as the same trend was apparent prior to flow reduction. For the sites of greatest unnatural reduction in baseflow, the Du Toits and Riviersonderend, there were no obvious changes in calcium with flow reduction.

Magnesium concentrations were naturally similarly low in all four rivers (0.035-0.068 mmol ℓ^{-1}), and stable over the six-month period irrespective of natural baseflow magnitudes (Figure 10.21). Baseflow reduction below natural levels did not appear to alter Mg^{2+} in most instances (Figure 10.21). However, in the latter half of the impact phase (March), concentrations of Magnesium in the impacted location of the Riviersonderend were markedly lower than control values, at 0.035 mmol ℓ^{-1} and 0.056 mmol ℓ^{-1}, respectively. Later, in the recovery phase, the pattern was reversed (Figure 10.21).

From Figure 10.22, it is evident that the Riviersonderend and Du Toits sites grouped at naturally higher concentrations (a range of 0.161 mmol ℓ^{-1} in February, to 0.0.241 mmol ℓ^{-1} in March) of the dominant anion, chloride, than the Elands and Molenaars sites (a range of 0.065 mmol ℓ^{-1} in April, to 0.154 mmol ℓ^{-1} in January). This was opposite to the trend for potassium (Figure 10.19). For a percentage flow reduction of 36% (Molenaars) there did not appear to be a pronounced effect on Cl^- concentration (Figure 10.22). Looking at the general trend for the Du Toits (Figure 10.22), the markedly lower natural value in February is most likely anomalous, and there was no apparent effect of extreme low flows on Cl^- concentration. Chloride levels were similar in the two locations of the Riviersonderend during the pre-impact phase, but concentrations in the impact location were consistently higher throughout the other two experimental phases (Figure 10.22, suggesting a possible effect of large scale baseflow reduction. However, such changes are within the natural range of variability shown by the control site.

Natural sulphate concentrations were similar at all four sites, ranging from an overall minimum of 0.010 mmol ℓ^{-1} to a maximum of 0.040 mmol ℓ^{-1} (a difference of 0.030 mmol ℓ^{-1}) (Figure 10.23). The only evidence of a possible effect of extremely low baseflows on sulphate concentration was the elevated February value of 0.040 mmol ℓ^{-1} for the impacted location of the Riviersonderend, in comparison with the natural background value of 0.023 mmol ℓ^{-1} (Figure 10.23). The significance of the difference in concentration of 0.017 mmol ℓ^{-1} requires investigation (Tharme in prep.).

10.3.7 Effects of natural and unnatural dry-season baseflows on nutrients and silicon

Nutrient concentrations at each site and location, for the duration of the study, are illustrated in Figures 10.24-10.26. Ammonium is not included as concentrations were below the detection limits of 0.2 mg ℓ^{-1} in all instances. Silicon concentrations, as SiO_2-Si, are shown in Figure 10.27.

Nitrate concentrations were naturally extremely low in the Du Toits and Riviersonderend rivers, showing their oligotrophic character (Figure 10.24). Generally, only trace amounts were recorded, below the detection limit of 0.009 µmol ℓ^{-1}. The maximum nitrate level was only 2.585 µmol ℓ^{-1}, in the Du Toits control location, in May. In contrast, the more disturbed Elands and Molenaars sites (see Section 9.2.1 and 9.2.2), exhibited mild nutrient enrichment, with peak nitrate levels of 18.001 µmol ℓ^{-1} and 12.682 µmol ℓ^{-1}, respectively (Figure 10.24). Maxima for these two sites were not necessarily associated with the month of

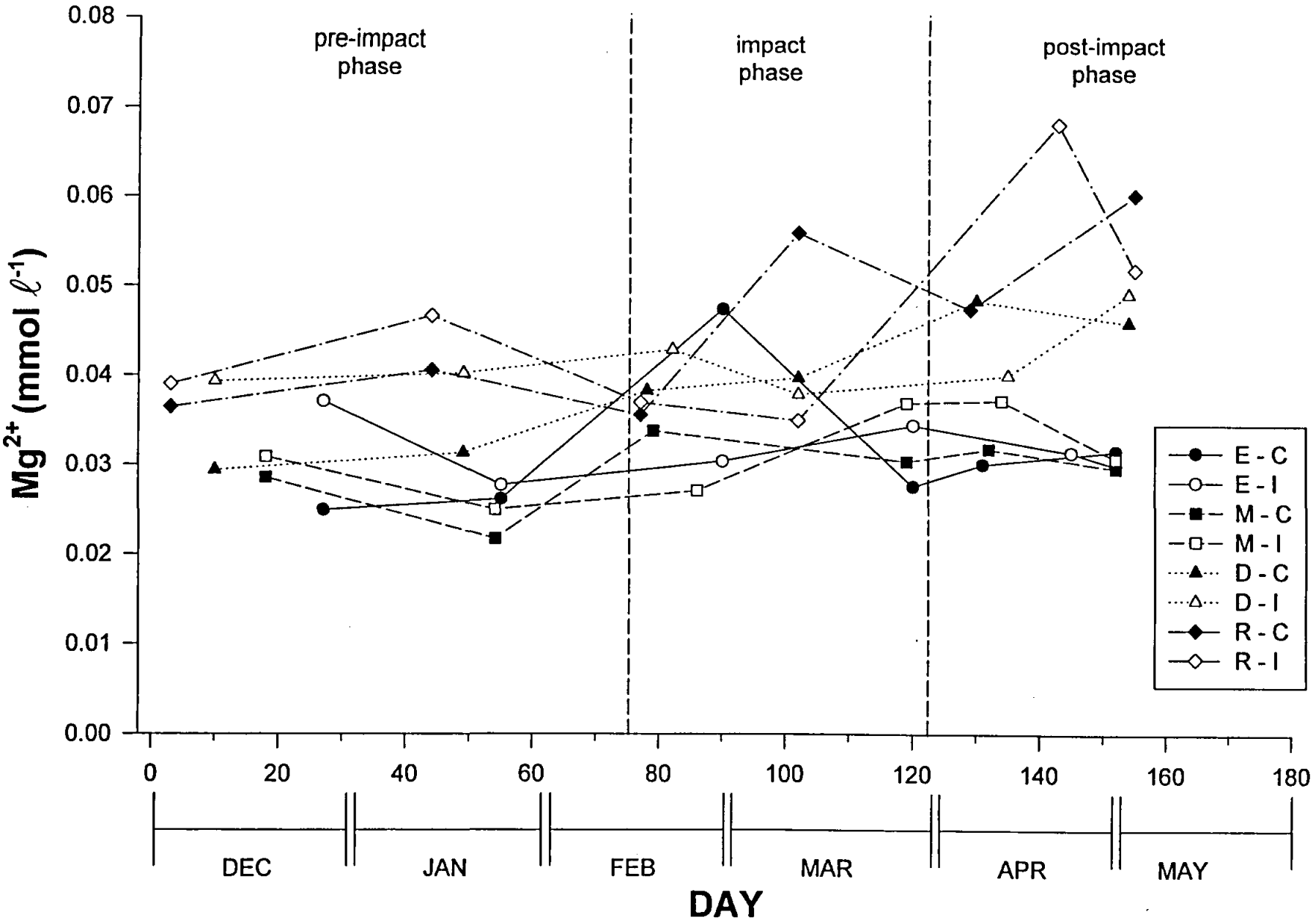


Figure 10.21 Changes in Mg^{2+} (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

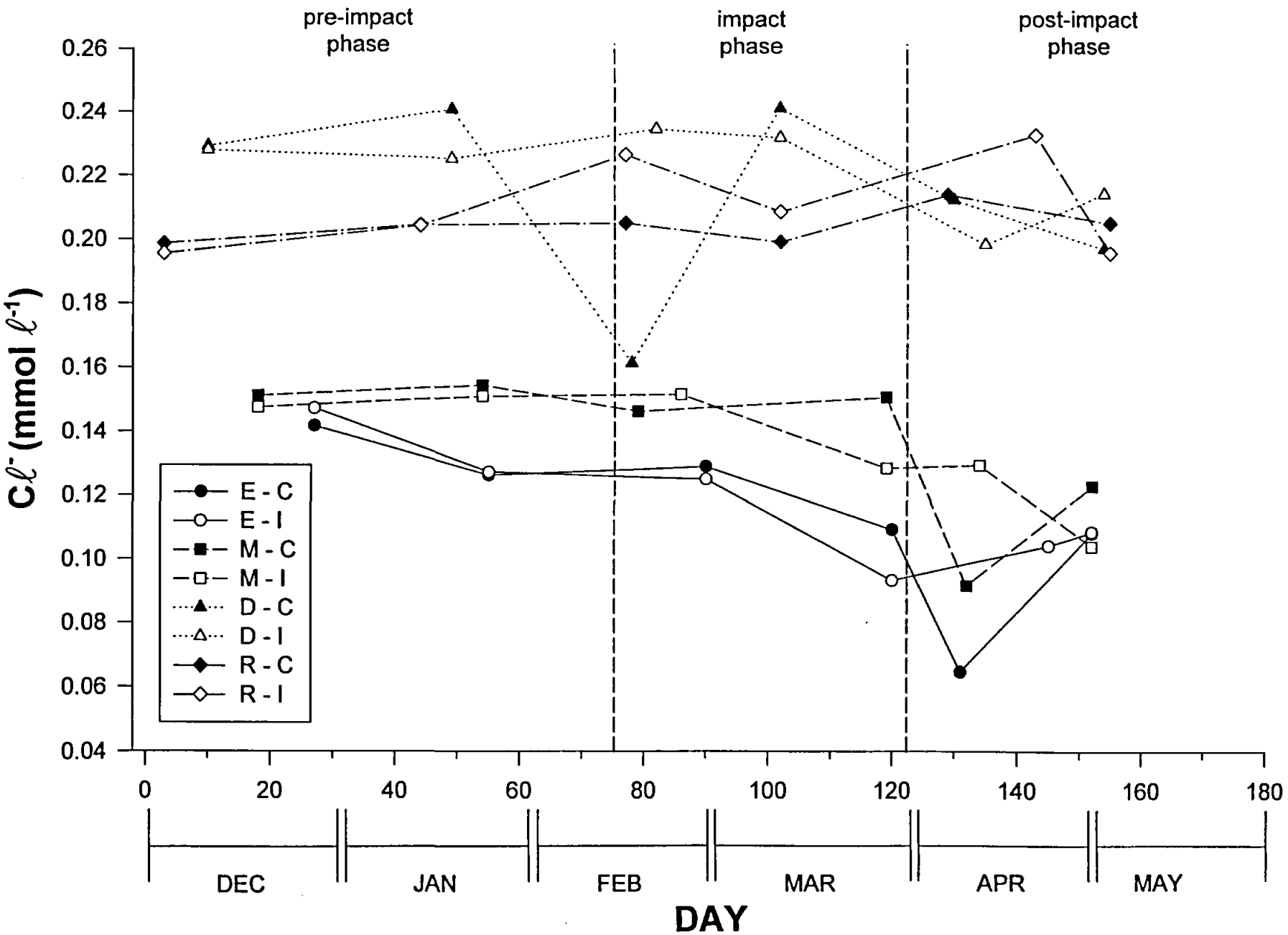


Figure 10.22 Changes in $C\ell^-$ (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

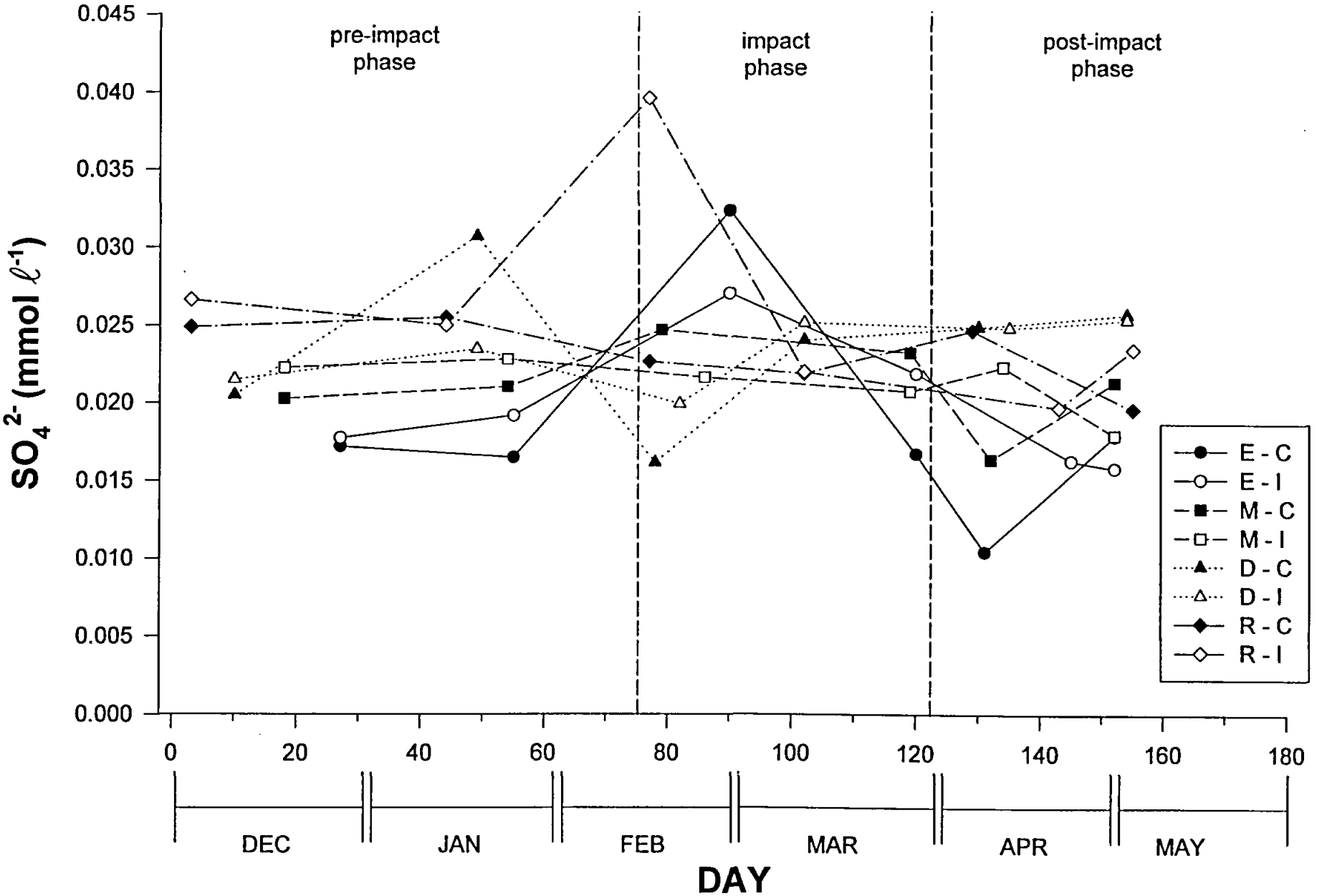


Figure 10.23 Changes in SO_4^{2-} (mmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

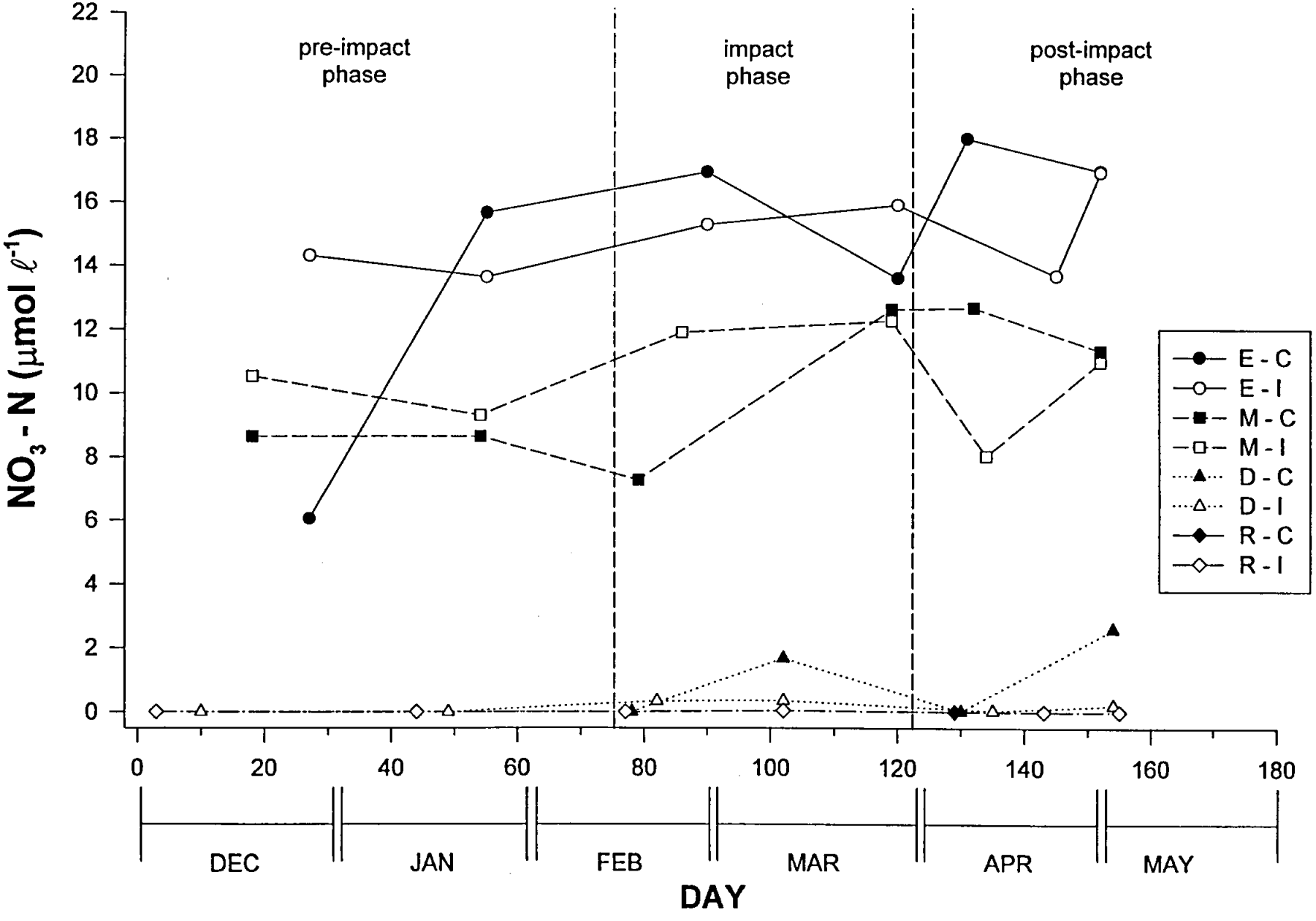


Figure 10.24 Changes in nitrate (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

lowest natural baseflow. There were no distinct changes in nitrates as a result of unnaturally low baseflows (Figure 10.24), this despite the high degree of decomposition of plant material (primarily *Isolepis digitata*) observed in the impacted location of the Riviersonderend (pers. obs.). However, prolific algal growth in the location at extreme low flows (pers. obs.), could have been a response to increased nutrient availability.

Nitrite concentrations were naturally in a similar overall range for the four sites (0.011-0.239 µmol ℓ^{-1}), but concentrations were variable among months (Figure 10.25). There was no discernible effect of unnaturally low flows on nitrite concentrations (Figure 10.25).

The Elands River exhibited markedly higher (and variable) soluble reactive phosphate (SRP) values than the other three sites, reaching a maximum of 2.365 µmol ℓ^{-1} (Figure 10.26). Examination of phosphate values for the Molenaars and Du Toits sites showed that values were lower for impacted locations than for the controls during the impact phase, and then higher for the impact location of the Du Toits in the post-impact phase (Figure 10.26). The degree to which the recorded changes might be significant requires assessment (Tharme in prep.). There was no obvious change in phosphates at the Riviersonderend site with unnatural flow reduction (Figure 10.26).

Silicon concentrations were similarly low in December and January at all sites, increasing to peak levels in February/March for the Elands site, and April/May for the other sites (Figure 10.27). Concentrations were highly variable (a total range of 0.320-113.879 µmol ℓ^{-1}). Although a dramatic increase in concentration in April, for the impacted location of the Du Toits River, was associated with unnaturally low baseflows (Figure 10.27), similar sharp increases in concentration were evident under natural conditions (Figure 10.27).

10.4 EFFECTS OF NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS ON PHYSICAL BIOTOPE CHARACTER AND DYNAMICS

In order to understand the significance of any changes in BMI assemblages under conditions of both natural and unnatural dry-season baseflows, it is essential to relate the observed distribution patterns and assemblage composition to changes in the physical biotopes inhabited by them. Changes in physical biotopes are potentially manifested in two main ways. Firstly, there can be a change in hydraulic character, notably in terms of depth and velocity. In addition, there are likely to be dynamic changes in the proportions of various biotopes, with a change in discharge. For clarity, in the following sections, 'average velocity' refers to a mean value derived from several velocity figures. Where the terms 'velocity' or 'maximum velocity' are used, they pertain to velocity measured at 0.6 depth from the water surface (also known as average or mean column velocity).

The following section (Section 10.4.1) addresses the effects of natural and unnatural, dry-season baseflows on the hydraulic characteristics of riffle and run transect biotopes (Figure 9.9). It is important to note that these and other biotopes were identified subjectively in the field, on the basis of the characteristics given in Section 9.5.1. Discriminant analysis will be undertaken in Tharme (in prep.) to validate the subjective categorisations of biotopes. In Section 10.4.2, the dynamics of the proportions of the various biotopes observed at each site are described. In both cases, the results are based on an examination of only the cross-

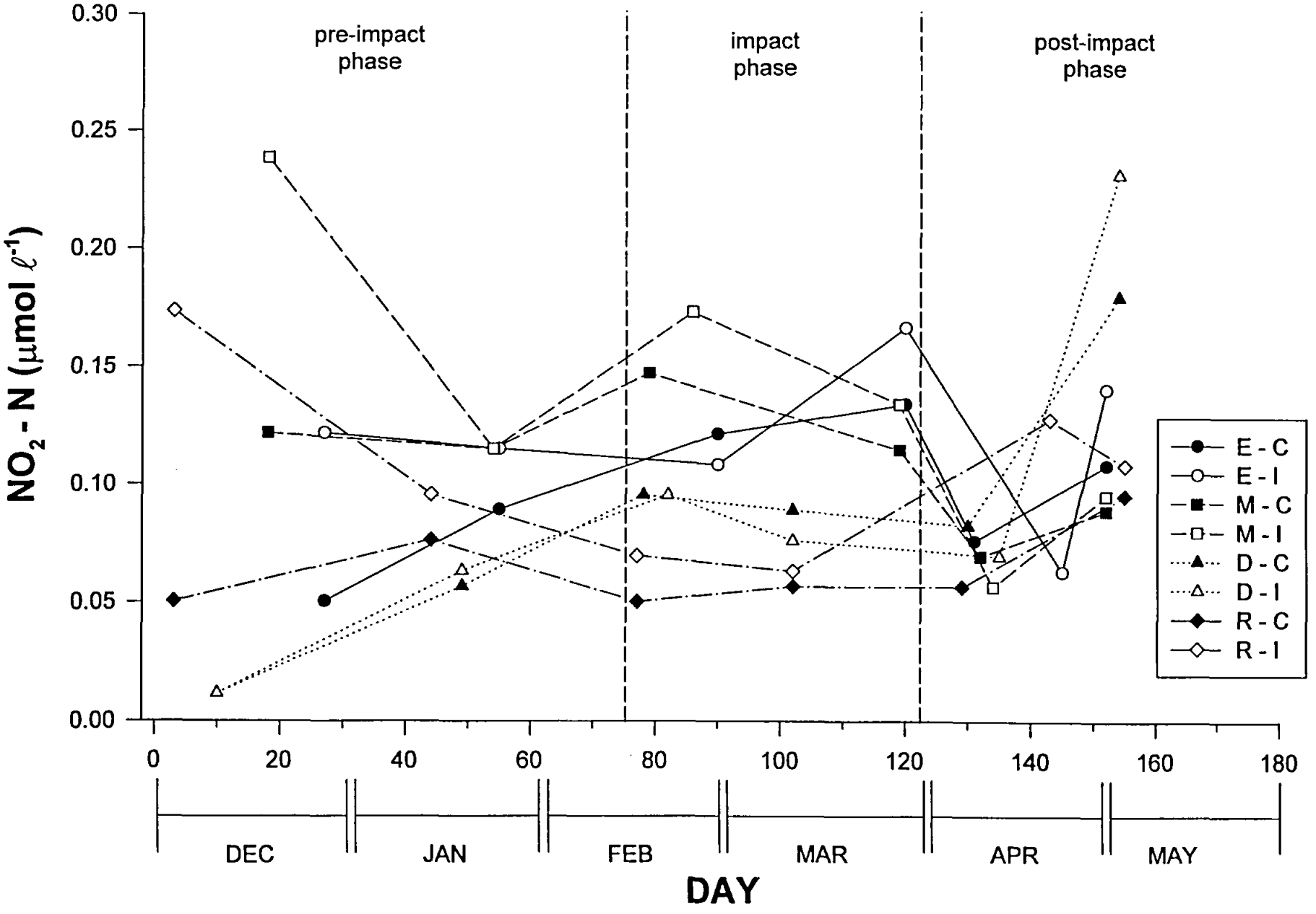


Figure 10.25 Changes in nitrite (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

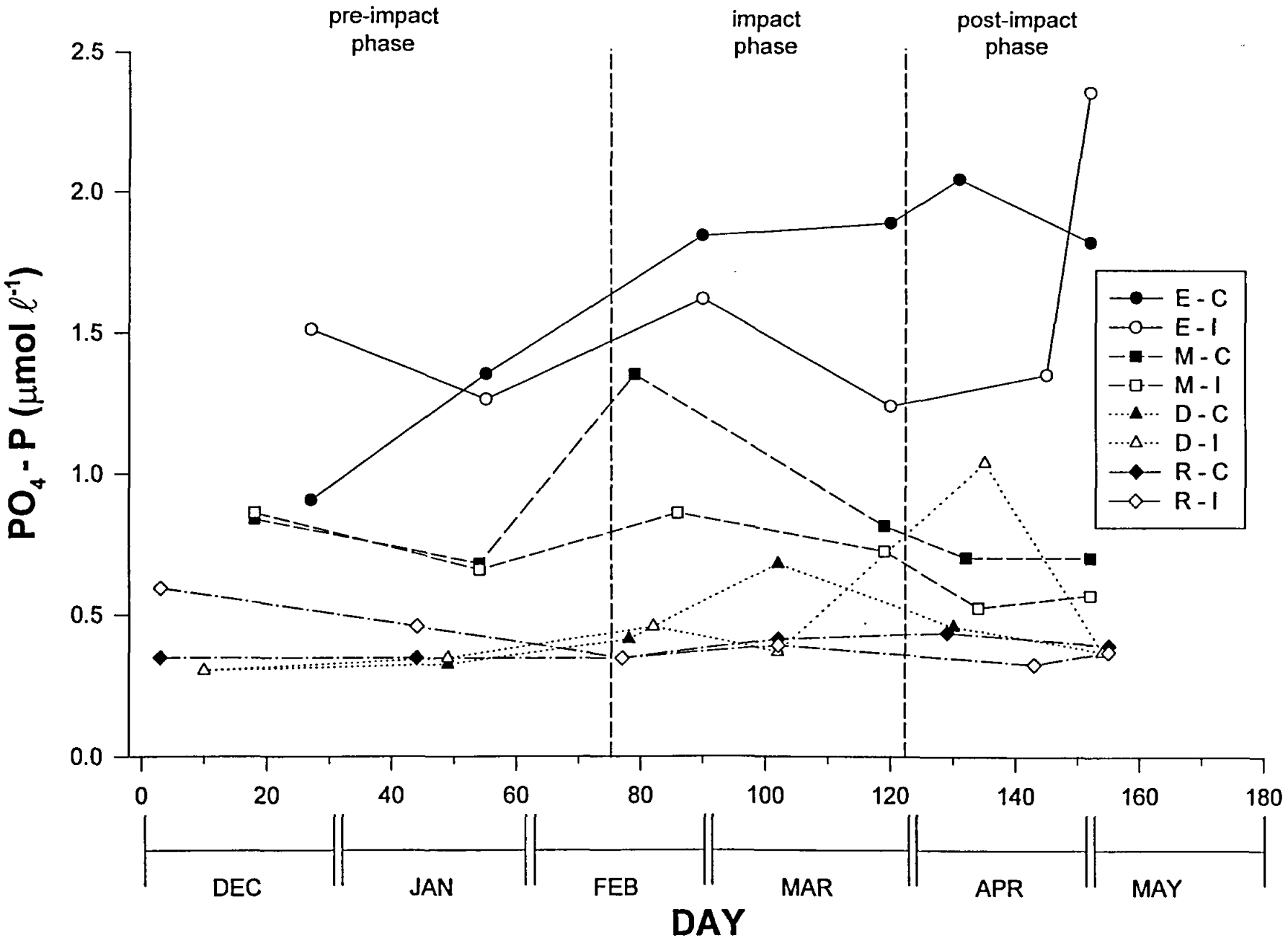


Figure 10.26 Changes in phosphate (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

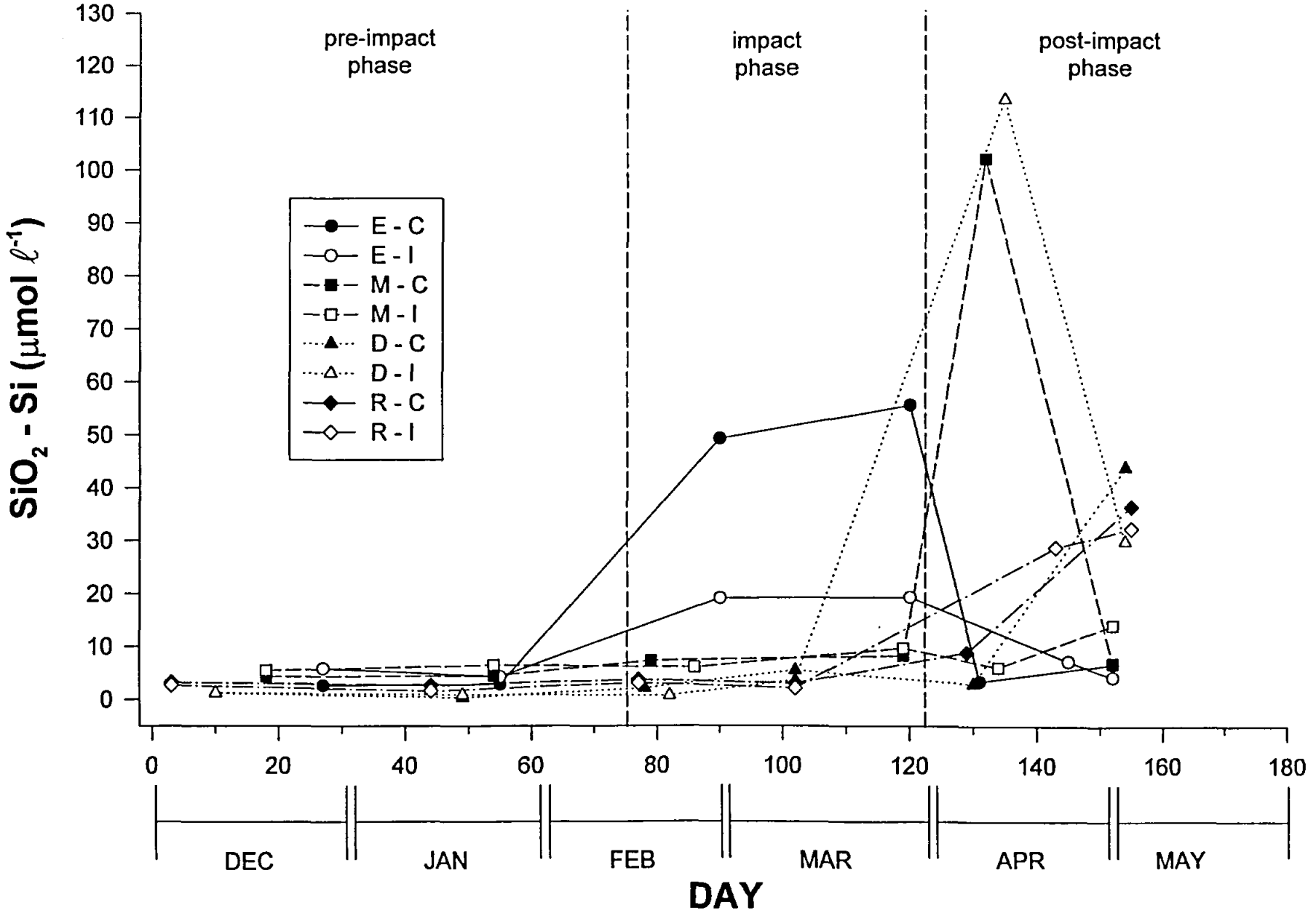


Figure 10.27 Changes in silicon (µmol ℓ^{-1}) at the four sites, during the three phases of the low-flow study. The demarcation of the impact phase is approximate (see Table 9.3 for precise dates).

section data, and do not include assessments of biotope maps. The latter information is addressed in Tharme (in prep.). Further analyses of the hydraulic data presented in the next sections, such as percentage losses in wetted area and connectivity with decreases in baseflow, are also described in Tharme (in prep.).

10.4.1 Effects of natural and unnatural dry-season baseflows on the hydraulic character of physical biotopes, across riffles and runs

In this section, observed changes in depth and average velocity profiles across cross-sections, are discussed first for runs and then for riffles, with fluctuations in both natural and unnatural baseflows. Changes in near-bottom velocities and Froude numbers (an index of turbulence) are not described in the current report, but are in Tharme (in prep.).

The discharges indicated on the various cross-section profiles are those recorded for the specific transect biotope (Figure 9.9), in all instances except the riffle cross-section, in the impact location, Riviersonderend site. In the latter instance, high experimental error in discharge measurement necessitated the use of the discharge recorded for the run. Changes in the type of transect biotope represented by each cross-section are indicated where they occurred.

Effects of natural dry-season baseflows on the hydraulic character of transect runs

Figures 10.28-10.32 illustrate the natural hydraulic character of transect runs at each of the four sites. Figures 10.28 and 10.29 both reflect natural run profiles for the Elands, as the control site.

Comparing all four sites, hydraulically, runs were typically deeper and exhibited a lower range of velocities than riffles within the same reach (see next section for a hydraulic description of riffles). Depths and average velocities naturally decreased with a decrease in baseflow and the converse happened with increased discharge (Figures 10.28-10.32, see below). However, the differences in hydraulics were generally minor due to the stability of baseflow magnitudes throughout the dry season. There was a natural increase in the width of channel exposed at lower flows, although dissection of the wetted profile into additional small channels, or wetted sections, of the channel was fairly low. Even though the degree of dissection increased with naturally lower flows, fairly wide sections of deep water and broad velocity range generally remained intact. Large scale increases in depth and velocity required large increases in baseflow. Such elevated baseflow conditions were only apparent at the beginning of the dry season (December), for the Molenaars (Figure 10.30), and in early autumn (April) for the Riviersonderend site (Figure 10.32). In the case of the Du Toits River (Figure 10.31), the shallow nature of the channel (most of the channel was less than 0.25 m deep) resulted in smaller natural differences in baseflow, between February and December, having a more pronounced influence on hydraulics, notably depths, than similar magnitude fluctuations in discharge at the other sites. Shifts in the location of areas of greatest depth and highest velocity seldom occurred with fluctuations in discharge, especially in the case of depths, but were observed on occasion. For example, there was a shift in velocities across the control run at the Molenaars site (Figure 10.30).

The Molenaars site had the deepest runs, with a maximum depth of 0.58 m in December, at highest baseflow (Figure 10.30). Highest maximum velocity was recorded across the control run, Riviersonderend, at 1.006 m s^{-1}, during the high April baseflow (Figure 10.32). During the dry months of February (/early March), average depths ranged from 0.12 m at the Du Toits site, to 0.24 m at the Elands site (Figures 10.28-10.32). For the same period (impact phase), the range of natural, average velocities was from only

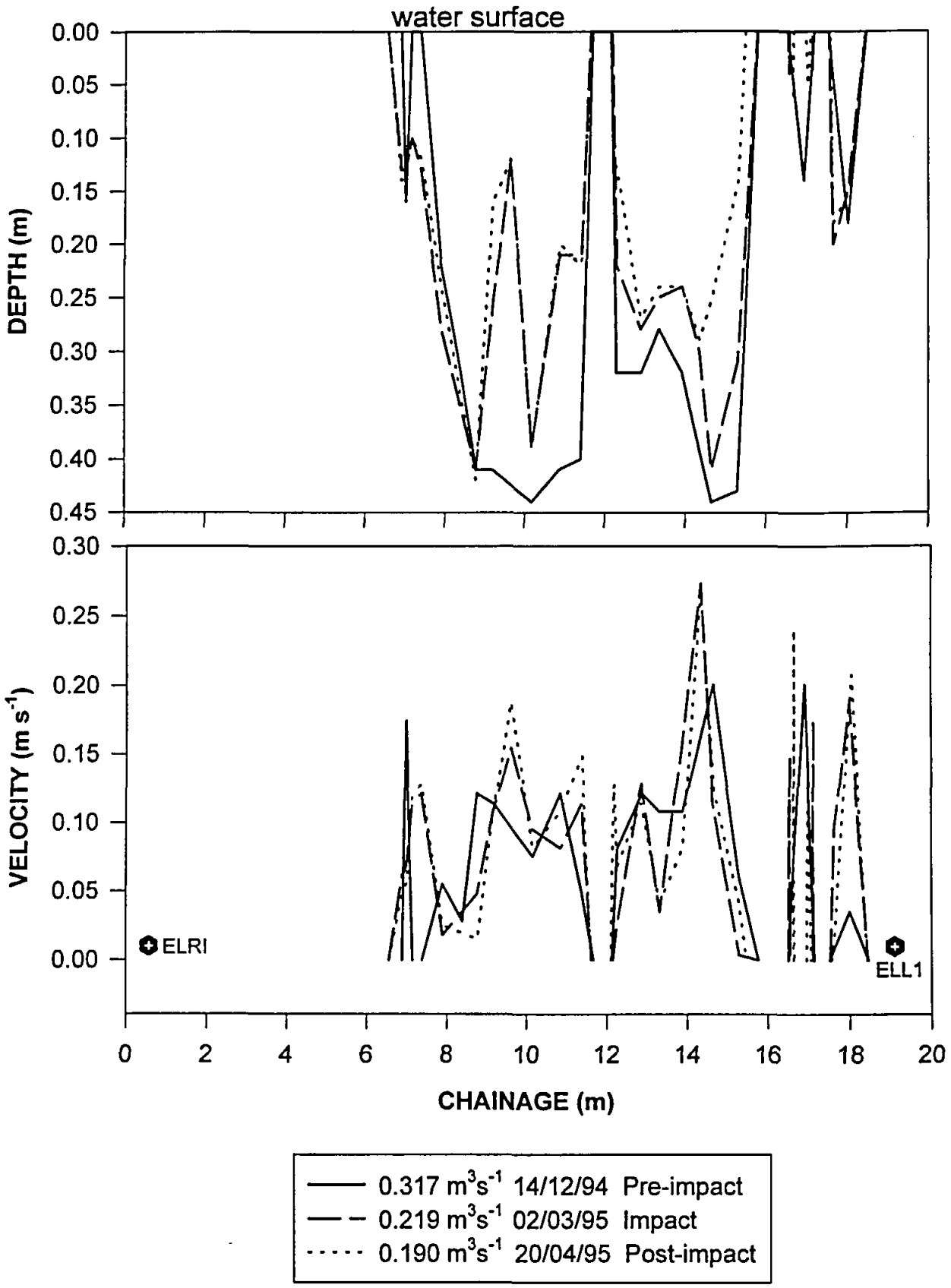


Figure 10.28 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 1) in the control location, Elands River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

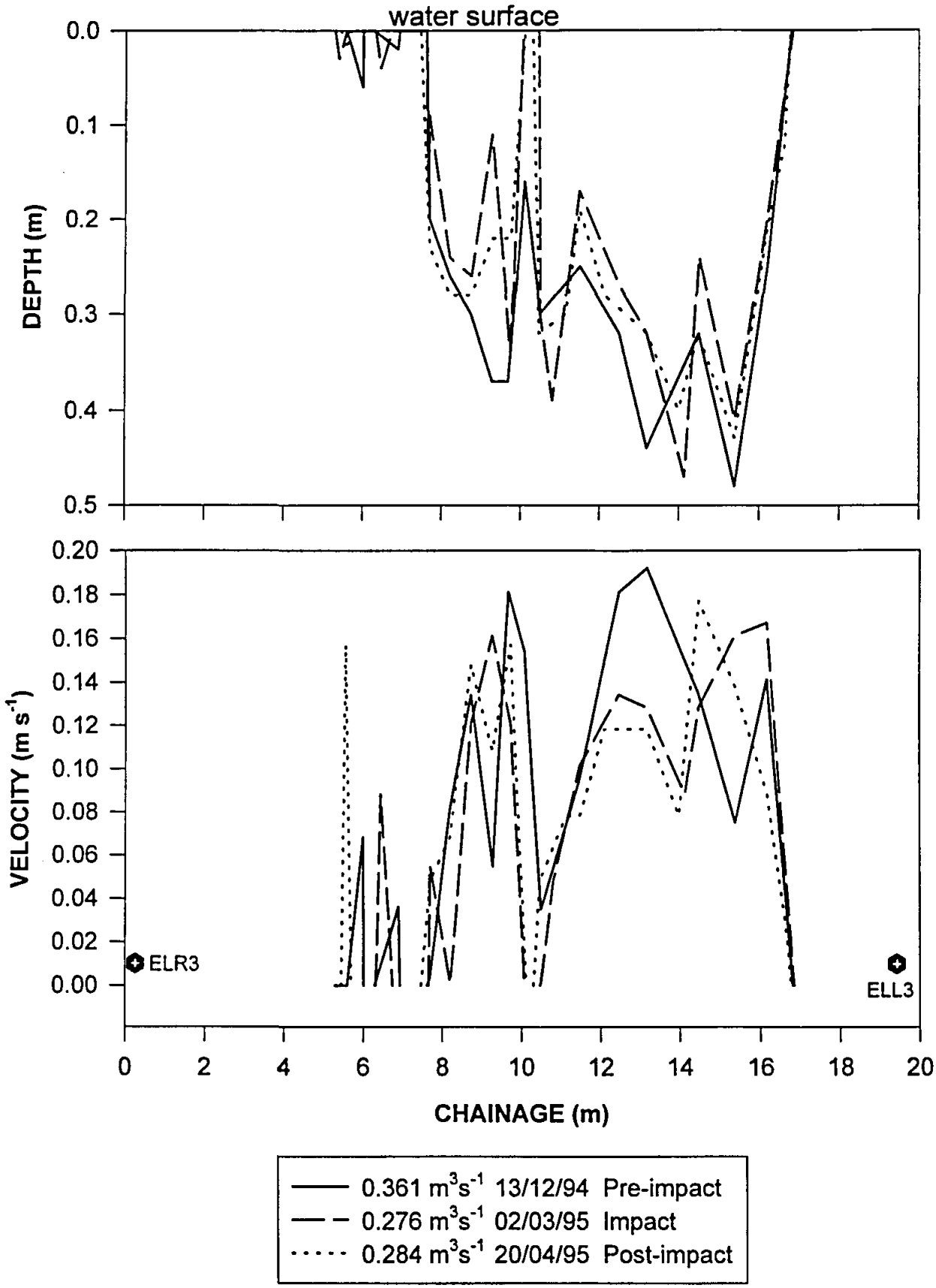


Figure 10.29 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 3) in the impact location, Elands River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

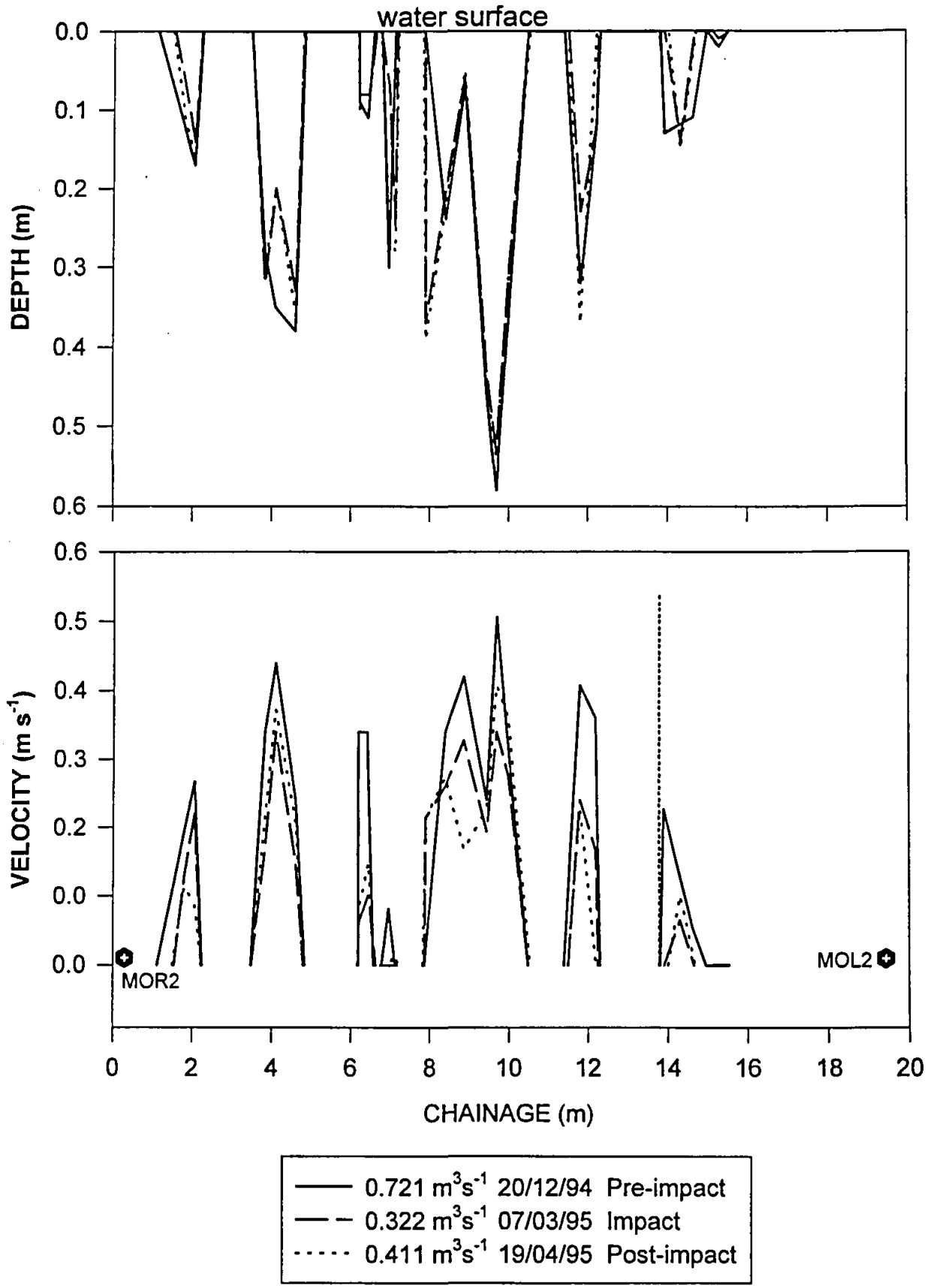


Figure 10.30 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 2) in the control location, Molenaars River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

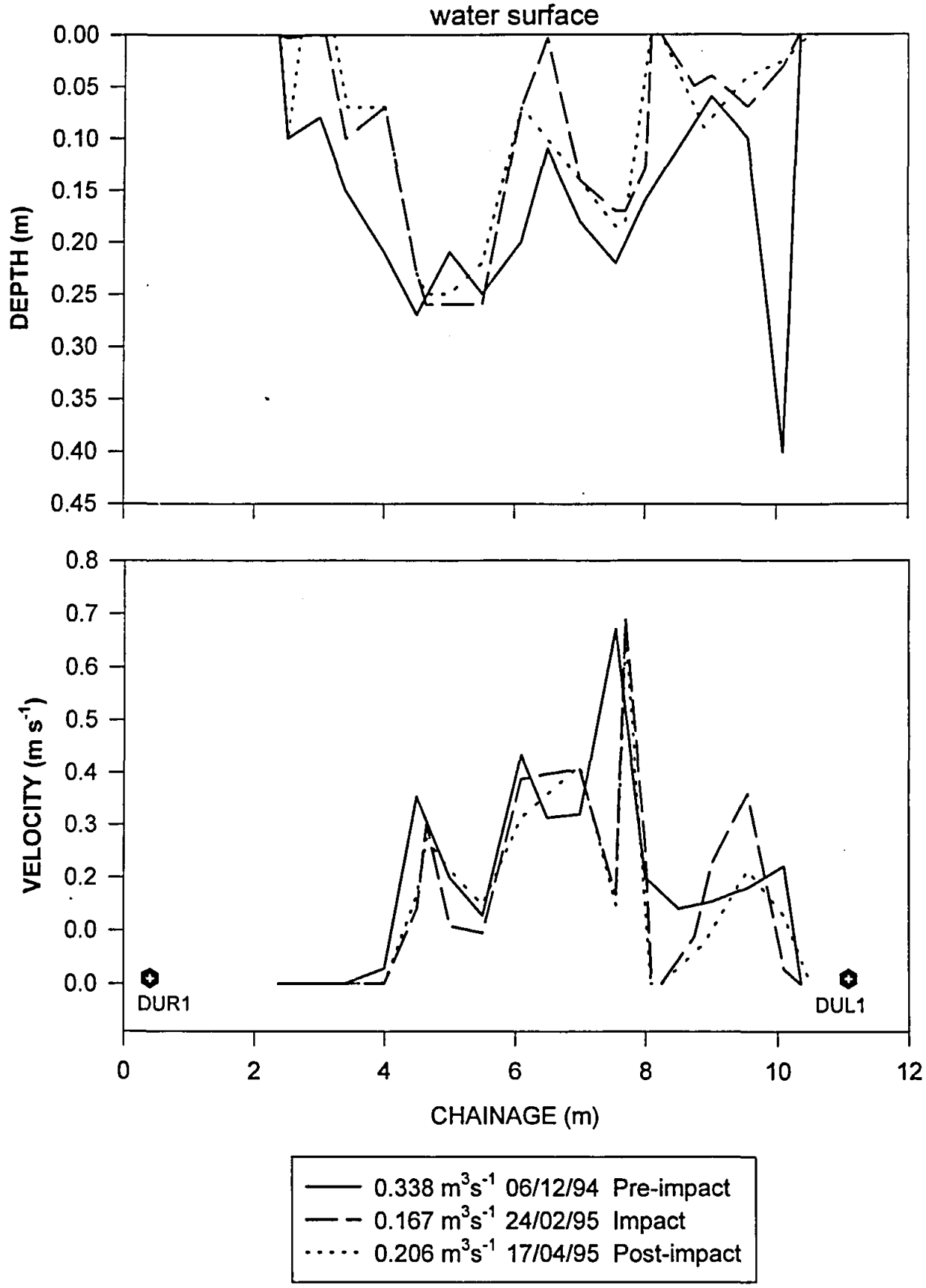


Figure 10.31 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 1) in the control location, Du Toits River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

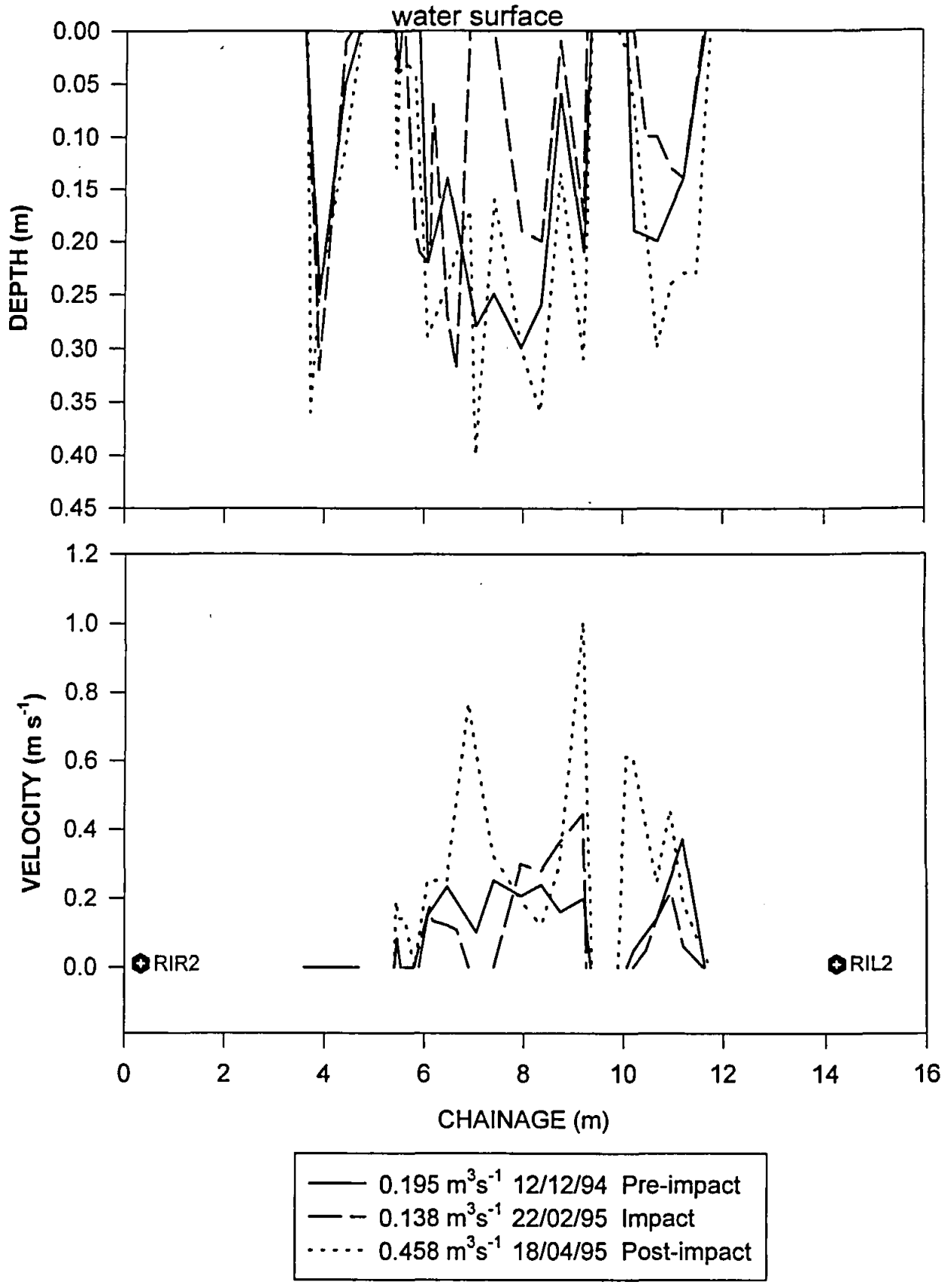


Figure 10.32 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 2) in the control location, Riviersonderend River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

0.088 m s^{-1} for one of the Elands runs, to 0.215 m s^{-1} for the run in the Du Toits control location (Figures 10.28-10.32). Over the duration of the entire low-flow study, average velocities ranged from 0.088 m s^{-1} (Elands, February) to a maximum average velocity of 0.297 m s^{-1}, for the Riviersonderend control run, in April. For the same overall period, average depths ranged from 0.12 m (Du Toits site, February) to 0.33 m (Elands site, December). Minimum depths and velocities were considered arbitrary measures, as depths in the order of only a few centimetres and zero velocities occur under both natural and extreme low flow conditions. Consequently, they are not discussed further here, but are reflected in the figures presented. However, the proportions of zero velocities and of depths below specified minima are potentially significant factors (Tharme in prep.).

Distribution profiles of depths and velocities in runs were naturally different at different locations within the same reach (for example, Figures 10.28 and 10.29) and among reaches (Figures 10.28-10.32), although trends in hydraulic character were sustained irrespective of morphological differences. The Du Toits River exhibited fairly simple, homogenous run profiles, while the Molenaars and Riviersonderend rivers had complex runs. Hydraulic differences between runs and riffles (see next section for a description of riffles) were least pronounced for the mountain-stream reach of the Riviersonderend River (Figure 10.32), where both transect biotopes were similarly highly heterogeneous.

In most instances, under conditions of natural baseflow from December through to May, cross-section runs retained their fundamental biotope type. However, for the Riviersonderend control location, the cross-section representing a run had to be reclassified as transitional to a riffle (riffle/run transition; April) and then as a riffle (May). This reclassification was a result of the natural, large scale increases in discharge, and associated changes in velocity relative to depth, recorded in autumn (Figure 10.1).

Effects of unnaturally low dry-season baseflows on the hydraulic character of transect runs

Molenaars site

Although the proportion of baseflow abstracted upstream of the Molenaars impact location was moderate at 36% (Figure 10.1), it resulted in a pronounced reduction in the proportion of depths greater than 0.20 m for the run cross-section (Figure 10.33). This reduction is marked relative to recorded natural decreases in depth under conditions of naturally low flow for the same biotope (Figure 10.30). During this midsummer impact phase, the average depth decreased by 52% relative to a natural value of 0.21 m (control run), to 0.10 m. An increase in the dissection of the wetted channel was also observed at extreme low flow. Two areas of high velocity were also lost with unnatural flow reduction, in contrast with similar control velocities during summer and in autumn (Figures 10.33 and 10.30). The average velocity of the impact run remained higher than that of the control run, at 0.201 m s^{-1} as compared with 0.171 m s^{-1}. Although there was a decline in average velocity for the impact run, from December to February, of 29%, the percentage decrease for the control run was higher at 40% (Figures 10.30 and 10.33).

Du Toits site

For the impacted run at the Du Toits site, the approximately 86% reduction in baseflow (Figure 10.1) had a pronounced impact on run hydraulics, particularly as the channel exhibits few naturally deep areas (Figure 10.34). The run became extremely shallow throughout the profile, with a recorded average depth of 0.06 m (Figure 10.34), while the average depth in a similar natural run was 0.12 m (Figure 10.31). The change in depth represented a 50% decrease for the impact location. There were also no remaining wide, moderately

deep areas. Both major flowing sections of the channel became less than 0.10 m deep, and the majority of the wetted width was less than 0.05 m deep. There was a concomitant marked loss of connectivity of wetted habitat, with five separate shallow sections interspersed with areas of exposed cobbles/boulders. A loss of high-velocity areas was coupled with a general decline to velocities below 0.200 m s^{-1} (average of 0.123 m s^{-1}, compared to a natural average of 0.215 m s^{-1}; Figure 10.34). These values indicate a 43% decrease in average velocity. The maximum velocity was reduced from a natural peak of 0.690 m s^{-1} (control; Figure 10.31) to 0.313 m s^{-1} (Figure 10.34).

Riviersonderend site

The impacted run at the Riviersonderend site also showed a pronounced reduction in depth with about 86% baseflow reduction, with much of the run shallower than 0.10 m (Figure 10.35). The average depth in February was about 55% lower (0.06 m) than the average depth of 0.12 m in a natural run. The maximum depth, however, remained similar to that of the non-impacted run (Figure 10.32). Although the cross-sectional morphology of the channel was such that a short section of deeper water remained, the velocity of the section was far lower than under natural conditions. Indeed, the average velocity for the run was extremely low, at only 0.029 m s^{-1} (Figure 10.35). In contrast, the average velocity recorded for a naturally flowing run was 0.153 m s^{-1} (Figure 10.32). Hence, the percentage reduction in average velocity at extremely low baseflow was in the order of 81%. Furthermore, an entire flowing secondary channel was lost with extreme flow reduction (Figure 10.35). The loss of all high velocity areas was a major effect of the unnaturally low flow, at this site, especially as even the few remaining deep sections were barely flowing (all velocities were below 0.100 m s^{-1}). The degree of dissection of the run also increased at very low flows, with about six newly exposed areas of the substratum (Figure 10.35).

Comparison of runs subjected to unnaturally low baseflows

The greatest maximum depth at extreme low flows (i.e. during the impact phase) was recorded for the Riviersonderend site at 0.31 m, while the lowest maximum depth was recorded for the Du Toits run, at 0.15 m. Average depths at extreme low flows ranged from 0.06-0.10 m, for the Du Toits and Riviersonderend sites, respectively. The highest and lowest maximum velocities in the same period were 0.505 m s^{-1} and 0.081 m s^{-1}, for the Molenaars and Riviersonderend sites, respectively. Average velocities at extreme low flows ranged from 0.201 m s^{-1} (Molenaars site) to 0.029 m s^{-1} (Riviersonderend site). The percentage decrease in average depth in February, relative to natural conditions at that time, was greatest for the Riviersonderend impact location at 55%. Similarly, the largest percentage decrease in average velocity, of 81%, was recorded for the Riviersonderend site.

A 36% reduction in baseflow, during February and March, did not alter the type of transect biotope of the impacted run at the Molenaars site. Although the impacted run at the Du Toits site was also unchanged in type with 86% baseflow reduction, a similarly high reduction in flow wholly transformed the Riviersonderend run (impact location) to a pool/run transitional biotope, for the entire duration of the impact phase.

Effects of natural dry-season baseflows on the hydraulic character of transect riffles

Changes in the hydraulic character of transect riffles at each of the four sites, as a result of natural fluctuations in discharge over summer and autumn, are illustrated Figures 10.36-10.40. Figures 10.36 and 10.37 reflect two transect riffles under conditions of natural flow, for the Elands as the control site.

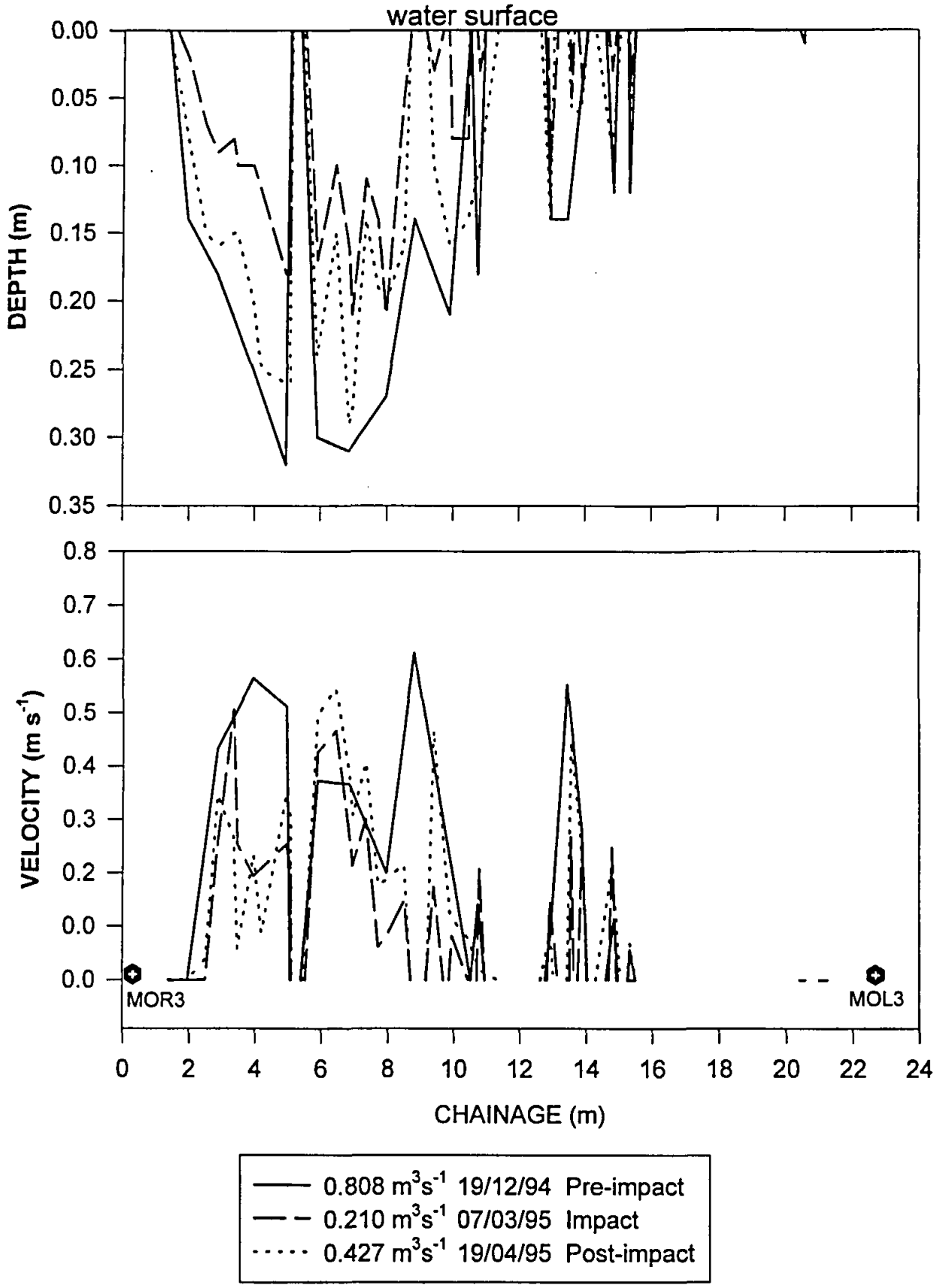


Figure 10.33 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 3) in the impact location, Molenaars River, at three discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study. The discharge recorded during the impact phase constitutes an unnaturally low baseflow (compare with Figure 10.30).

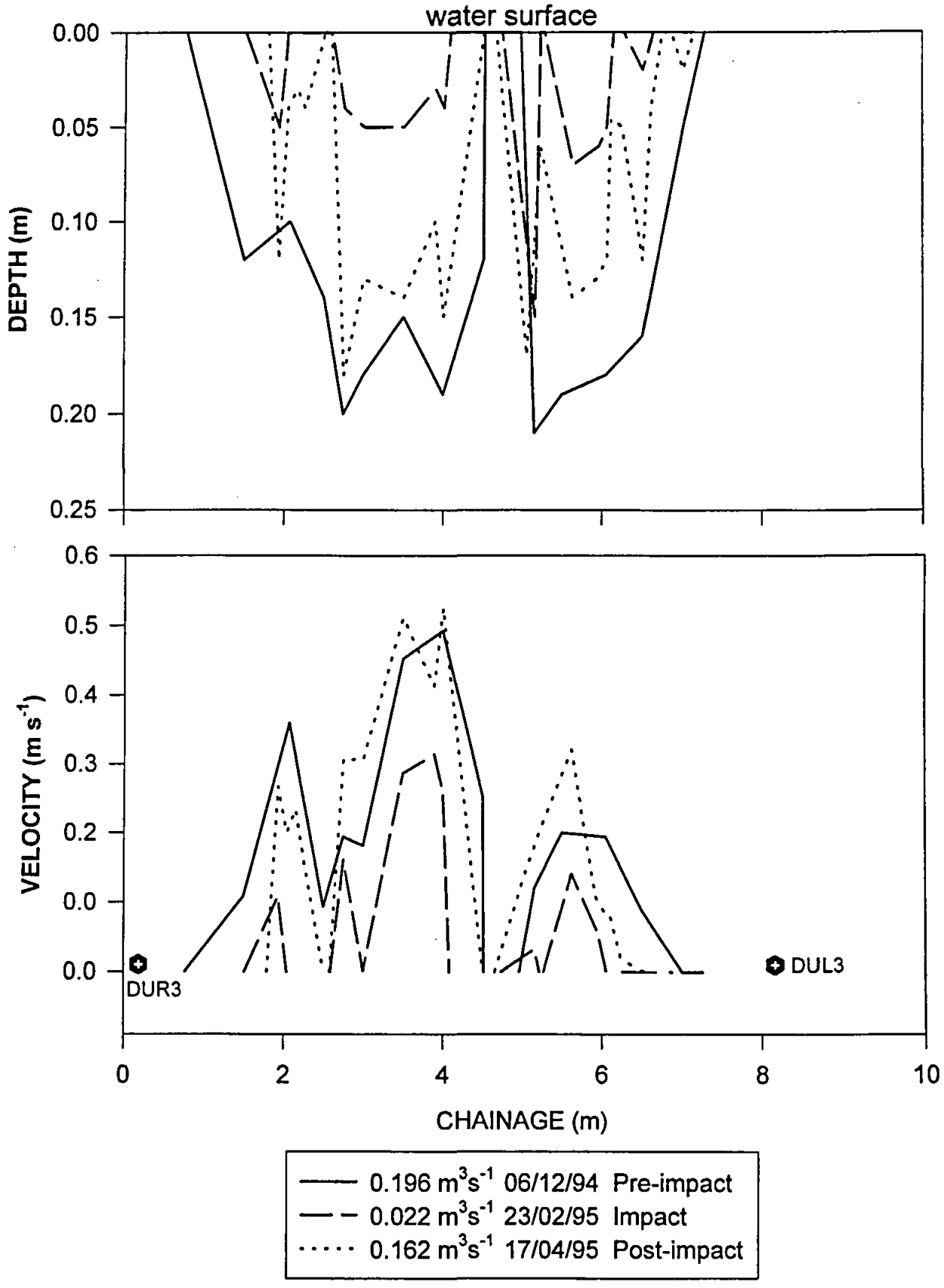


Figure 10.34 **Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 3) in the impact location, Du Toits River, at three discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study. The discharge recorded during the impact phase constitutes an unnaturally low baseflow (compare with Figure 10.31).**

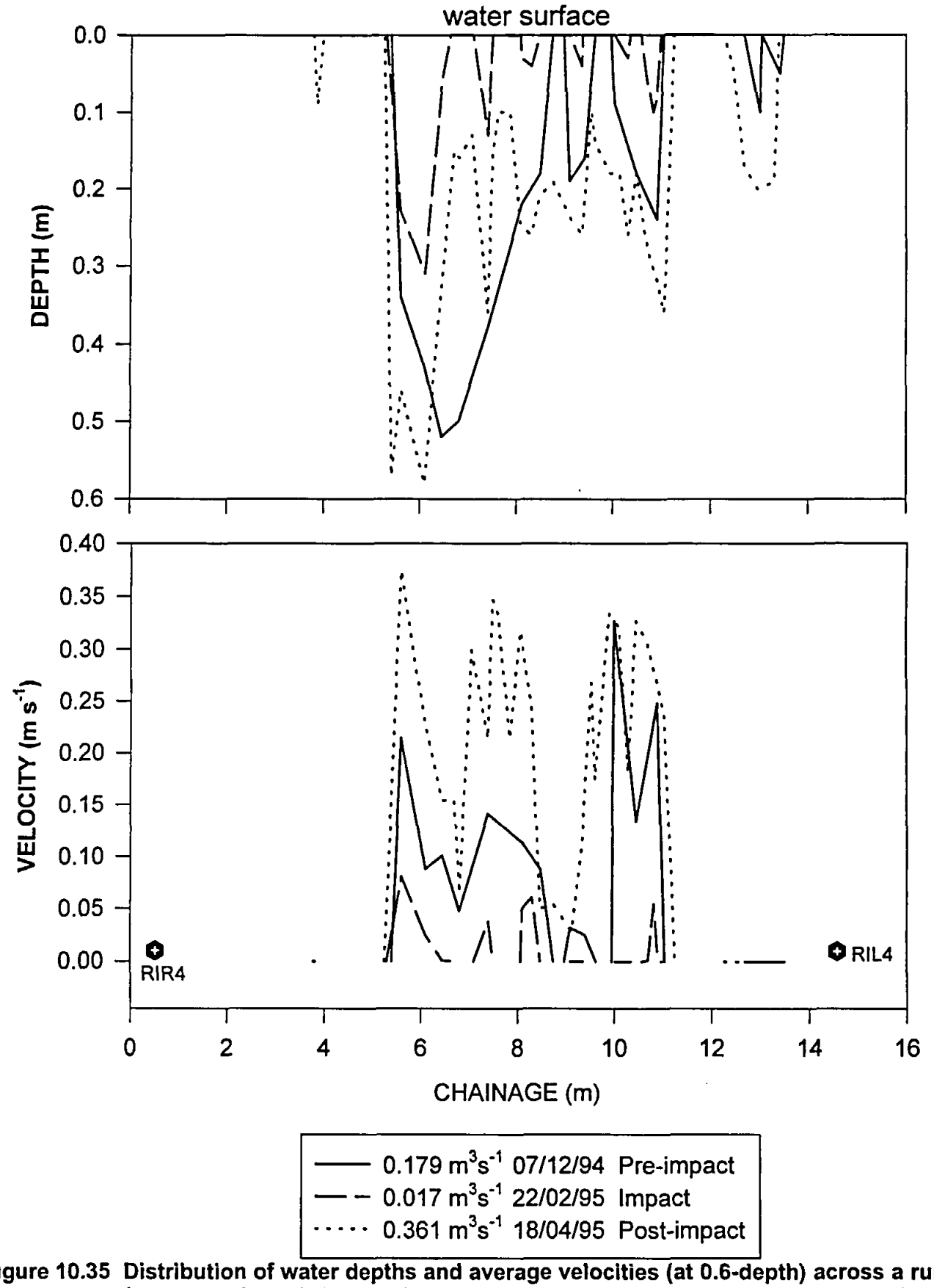


Figure 10.35 Distribution of water depths and average velocities (at 0.6-depth) across a run (cross-section 4) in the impact location, Riviersonderend River, at three discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study. The discharge recorded during the impact phase constitutes an unnaturally low baseflow (compare with Figure 10.32).

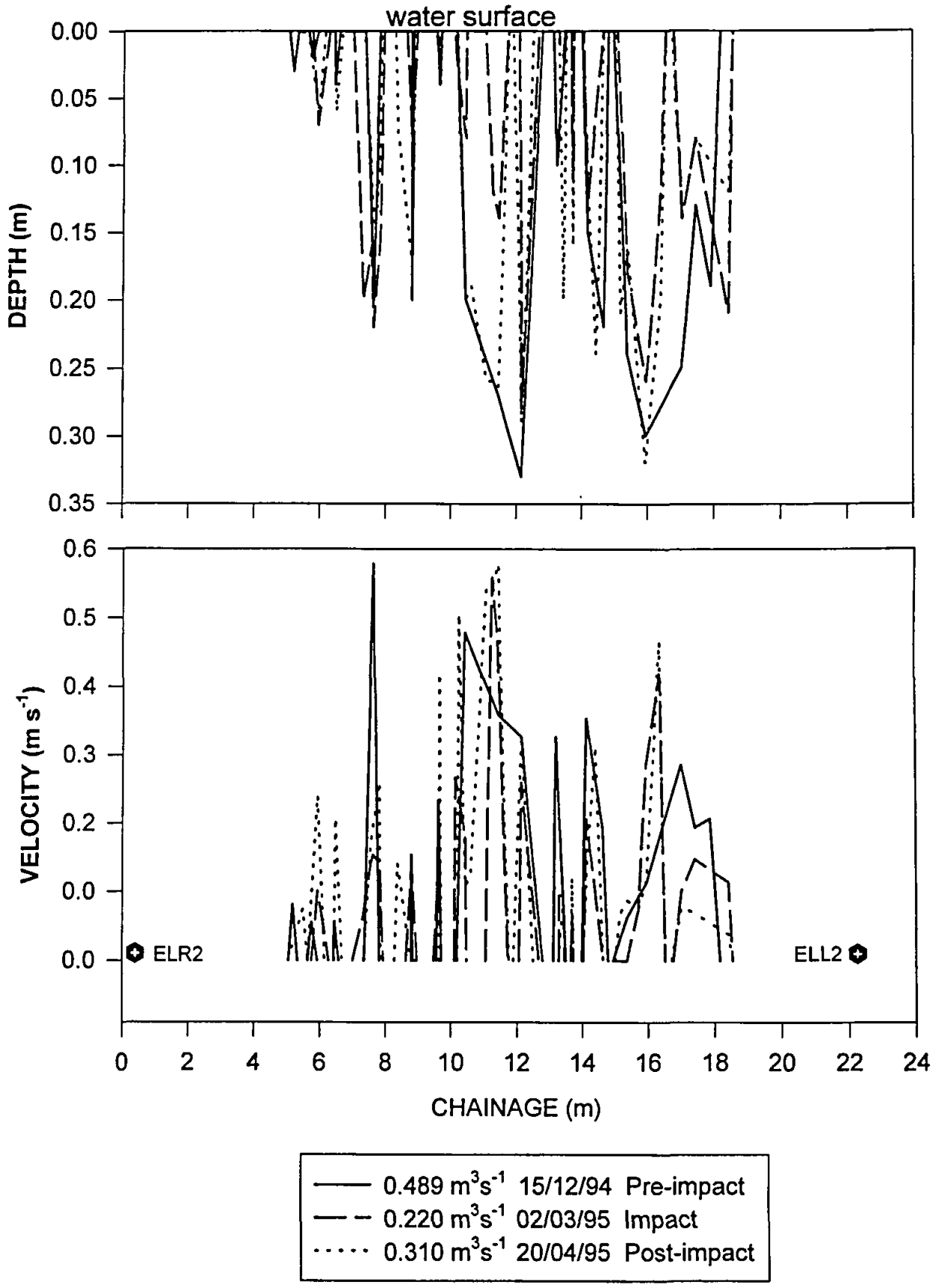


Figure 10.36 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 2) in the control location, Elands River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

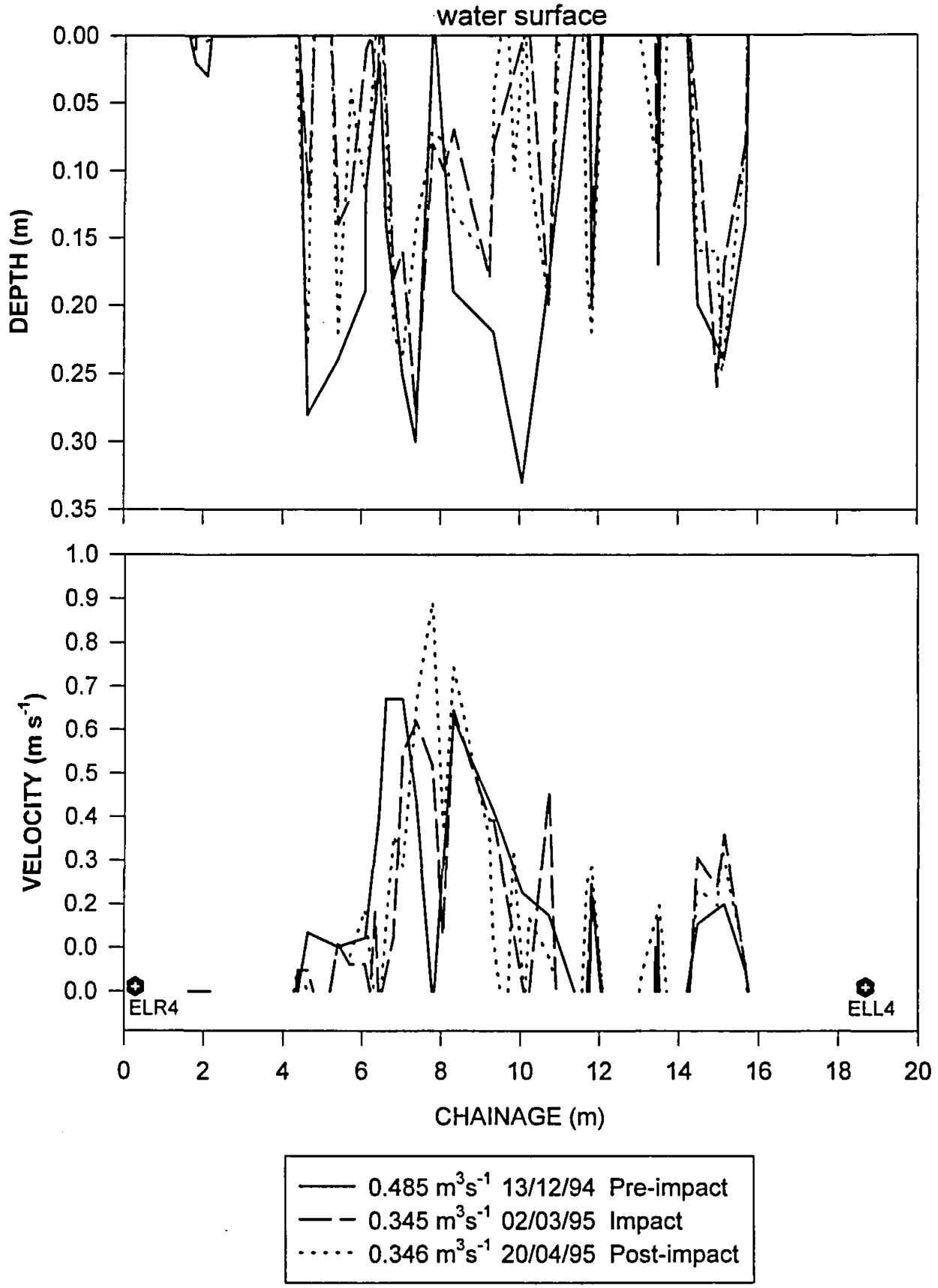


Figure 10.37 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 4) in the impact location, Elands River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

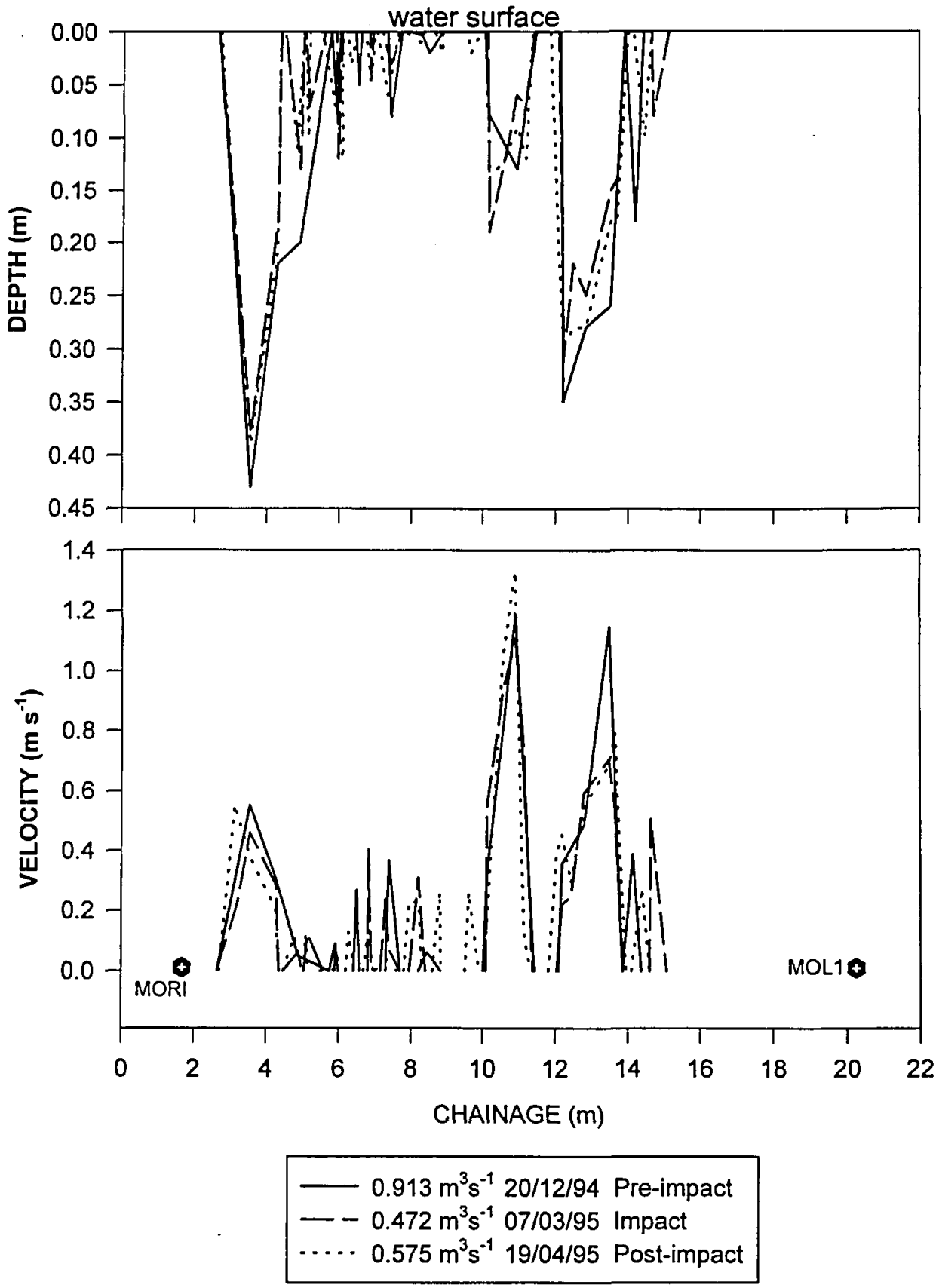


Figure 10.38 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 1) in the control location, Molenaars River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

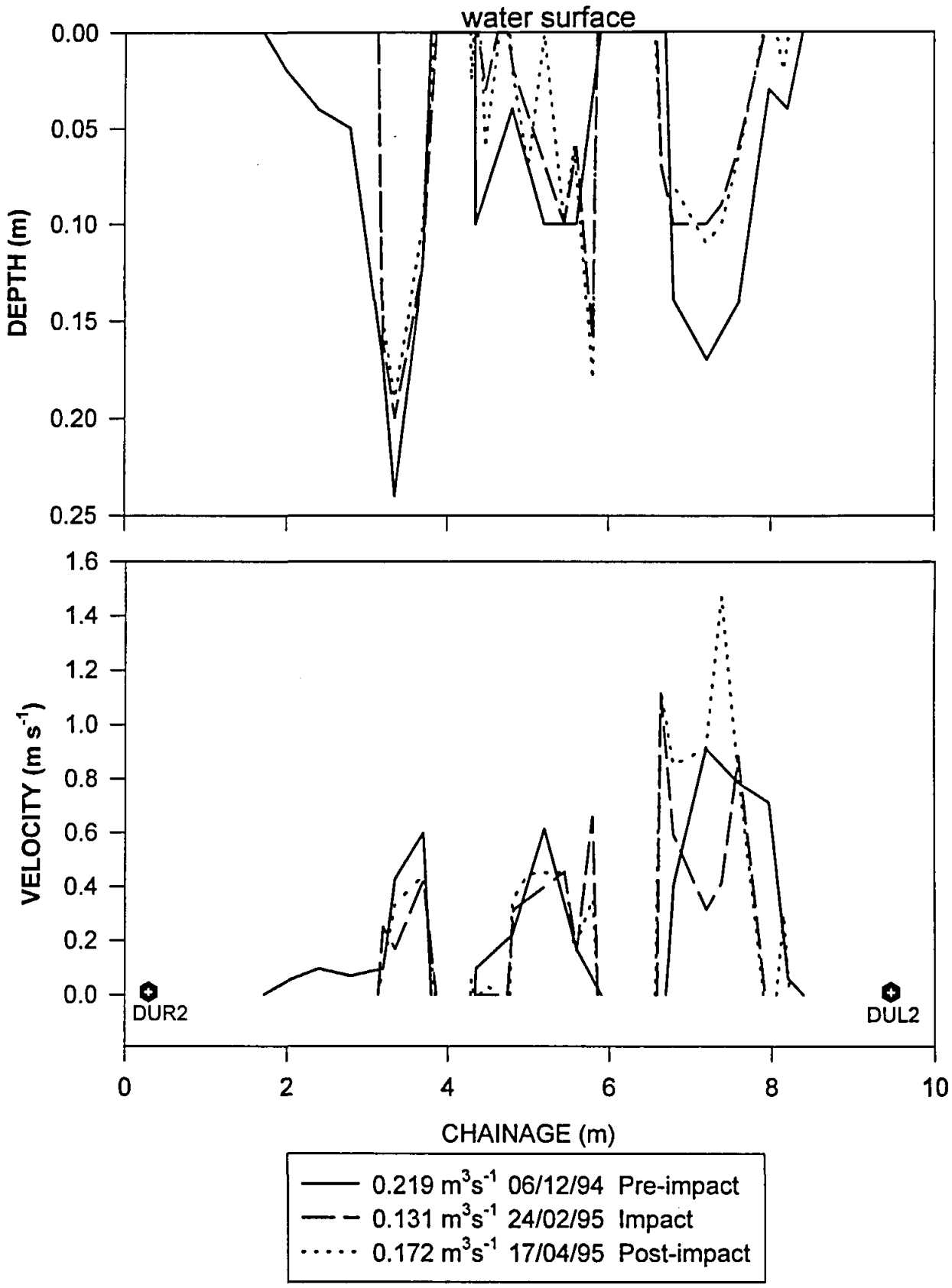


Figure 10.39 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 2) in the control location, Du Toits River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

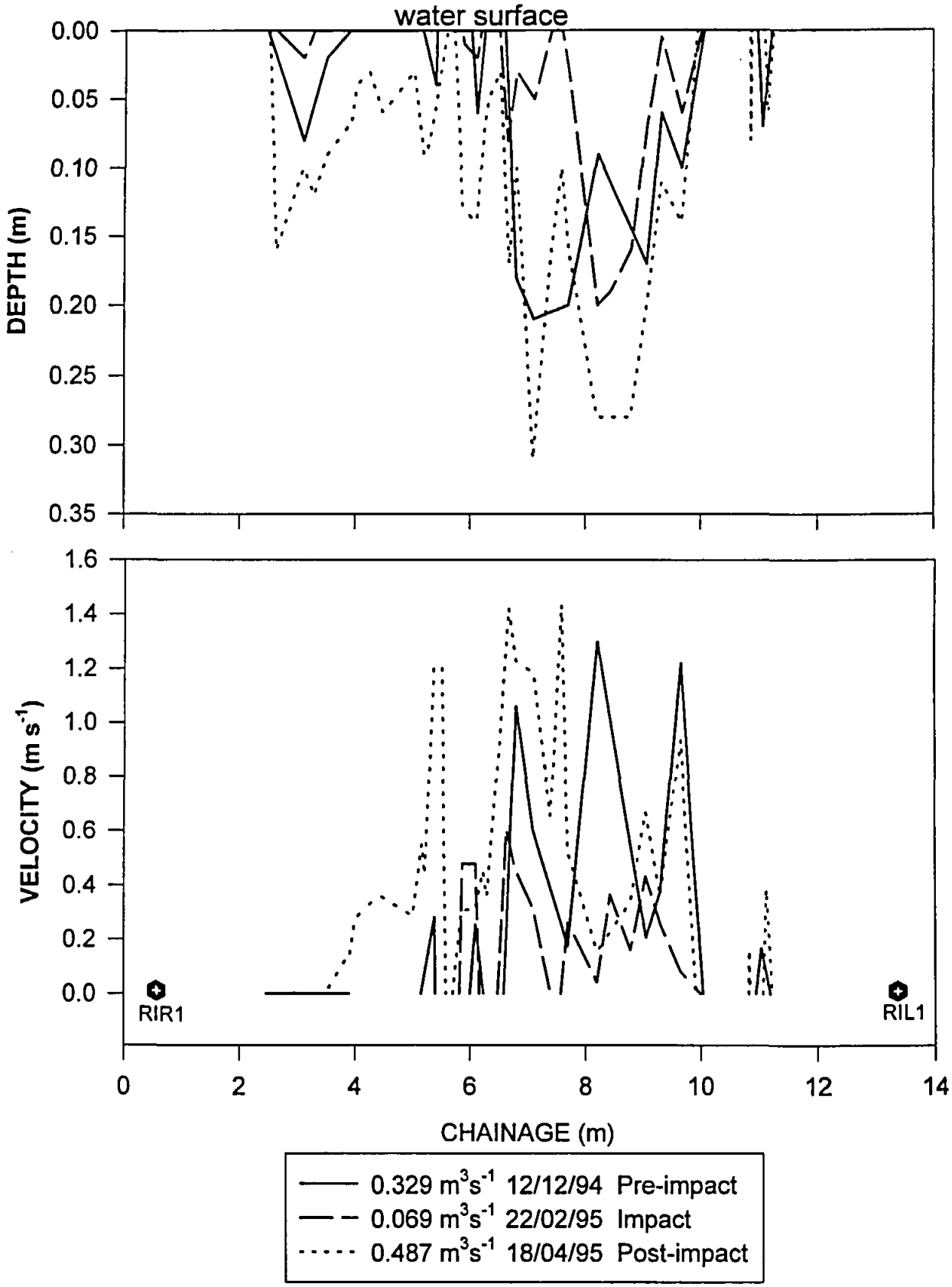


Figure 10.40 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 1) in the control location, Riviersonderend River, at three natural discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study.

Hydraulically, riffles were generally shallower and exhibited far higher maximum velocities than runs in the same reach (see above for a description of runs). The riffles exhibited more variable depth and velocity profiles than runs, due to the naturally higher number of large bed elements relative to depth, compared with runs. There were also fewer deep sections, especially wide ones. Velocities increased or decreased across the channel more sharply than in runs. Depths and velocities naturally decreased with a decrease in baseflow or increased with an increase in discharge (Figures 10.36-10.40). The decrease was typically to a greater extent than observed for runs as similar baseflow magnitudes. At naturally low baseflows, however, there were still narrow fairly deep sections of channel and occasional high-velocity peaks. There appeared to be a less consistent relationship between depth and velocity, than for runs. For example, deep areas in riffles were not slower flowing than shallow ones as often as with runs. The high turbulence and influence of upstream hydraulic refuges characteristic of riffles, are possibly responsible for a less direct relationship between the two hydraulic variables.

There was a natural increase in the width of exposed channel bed at lower flows, and the wetted channel was more dissected with respect to the connectivity of wetted biotope patches (cell biotopes, Figure 9.9) than for runs (Figures 10.28-10.32 and 10.36-10.40). The latter result was primarily due to protruding substratum elements.

As with runs, large increases in depth and velocity required relatively high magnitude baseflows. Such elevated baseflow conditions were only apparent at the beginning of the dry season, for the Molenaars River (Figure 10.38), and in April, for the Riviersonderend site (Figure 10.40). In the latter case, there was a pronounced increased in depth and in the width of the flowing channel.

The Molenaars control riffle (Figure 10.38) had two moderately deep sections, to a depth exceeding 0.35 m, making it morphologically fairly similar to a run in the same location (Figure 10.30). Both run and riffle biotopes as described by the cross-sections, were also quite dissected at this site. This contrasted with the greater differences in hydraulic profile observed between the Elands riffles and runs (Figures 10.28-10.29 and 10.36-10.37). For the Du Toits site, the control riffle (Figure 10.39) exhibited markedly less connectivity than the control run (Figure 10.31), at natural baseflows. Variability in depth and velocity across the cross-section profile was similar for the Riviersonderend run (Figure 10.32) and riffle (Figure 10.40), although the latter biotope was generally shallower and faster flowing.

Shifts in the location of areas of highest velocity, and particularly, greatest depth, occurred infrequently and mostly as a result of a large change in baseflow. For instance, the April increase in discharge in the Du Toits River resulted in a shift in the velocity peak along the profile (Figure 10.39).

The Riviersonderend site exhibited the shallowest riffles, with an average depth under natural low flow conditions of 0.07 m (Figure 10.40; February/March data). The average depth of the Molenaars riffle was greatest, but still shallow relative to runs, at 0.14 m, at the same time of year (Figure 10.38). Natural average velocities in the peak dry season ranged from 0.183 m s^{-1} for one of the Elands riffles to 0.440 m s^{-1} for the Du Toits riffle (Figure 10.39). Overall, for all months of the study, the greatest maximum depth in a riffle, of 0.43 m, was recorded for the Molenaars control location in December. The highest average depth in all months, 0.18 m, was recorded for both the Elands and Molenaars Rivers, in December. The highest maximum velocity recorded in a riffle for the three months assessed, was 1.470 m s^{-1}, for the Du Toits River

(April). The highest average velocity for the same period was 0.521 m s^{-1}, for the Riviersonderend control location, in April.

Riffle depth and velocity profiles were naturally variable both within a reach (Figures 10.36 and 10.37) and among reaches (Figures 10.36-10.40), particularly in relation to the protrusion of instream cobbles/boulders. However, trends in hydraulic character were sustained irrespective of morphological differences.

Natural baseflow fluctuations during the entire period of December to mid-May did not change the type of transect biotope from riffles to any other biotope, at any of the sites.

Effects of unnaturally low dry-season baseflows on the hydraulic character of transect riffles

Molenaars site

An unnatural, roughly 36% baseflow reduction at the Molenaars impact location (Figure 10.1) resulted in a marked decrease in the connectivity of biotope patches along the riffle cross-section. There was a concomitant increased in the width of exposed cobbles/boulders (Figure 10.41). There was a major loss of the widest section of moderately deep riffle, with channel dissection. Although there were still two channels deeper than 0.25 m in early March, they were extremely narrow (Figure 10.41). The average channel depth was 0.09 m (Figure 10.41), compared to 0.14 m under natural flow conditions (Figure 10.38). This represented a percentage reduction in average depth of 36%. The loss of overall wetted channel width, from waters' edge (WE) to WE, at unnaturally low flow, was also strongly apparent. There was a decline in the maximum velocity of the impact riffle, and a decrease in the highest velocities in each of the subsidiary channels (Figure 10.41). Average velocity at natural peak, dry-season baseflows was 0.419 m s^{-1} (Figure 10.38), and 17% lower at 0.347 m s^{-1}, for the impacted riffle (Figure 10.41).

Du Toits site

Similarly, for the riffle in the impact location at the Du Toits site (Figure 10.42), the approximately 86% reduction in baseflow, to only 0.036 m^3 s^{-1} (Figure 10.1), resulted in a major decrease in cross-channel depths, a loss of connectivity, and an increase in the widths of exposed bed. The second most deep channel section was lost, and the remaining moderately deep section narrowed (Figure 10.42). The average depth was almost halved (46%) from the natural depth of 0.10 m (control riffle; Figure 10.39). There was a decrease in maximum velocity to only 0.406 m s^{-1} (Figure 10.42). This was coupled with a dramatic increase in the width of channel sections with zero flow (Figure 10.42). Average velocity in February, at 0.150 m s^{-1}, was 66% lower than the control average in the same month, of 0.440 m s^{-1}.

Riviersonderend site

The riffle within the impact location of the Riviersonderend site was naturally complex, with an isolated secondary channel over bedrock. It was less typical than riffles at the other sites, and was especially more heterogeneous in terms of depth (Figure 10.43). Comparison of average riffle depth (0.088 m), relative to the control value of 0.071 m for the same period, showed that there was actually a slight increase in average depth at extreme low flow. Maximum depth showed the same anomalous trend. In the impact phase, with extreme reduction in baseflow, there was still a wide area of fairly deep water (maximum depth of 0.23 m). However, much of the channel was flowing slowly, with an average velocity of 0.249 m s^{-1} (Figure 10.43). This value represented a decrease of 18% relative to the natural, average velocity in the control riffle of 0.302 m s^{-1}. Although maximum velocity was still high, at 1.001 m s^{-1}, and exceeded the natural maximum

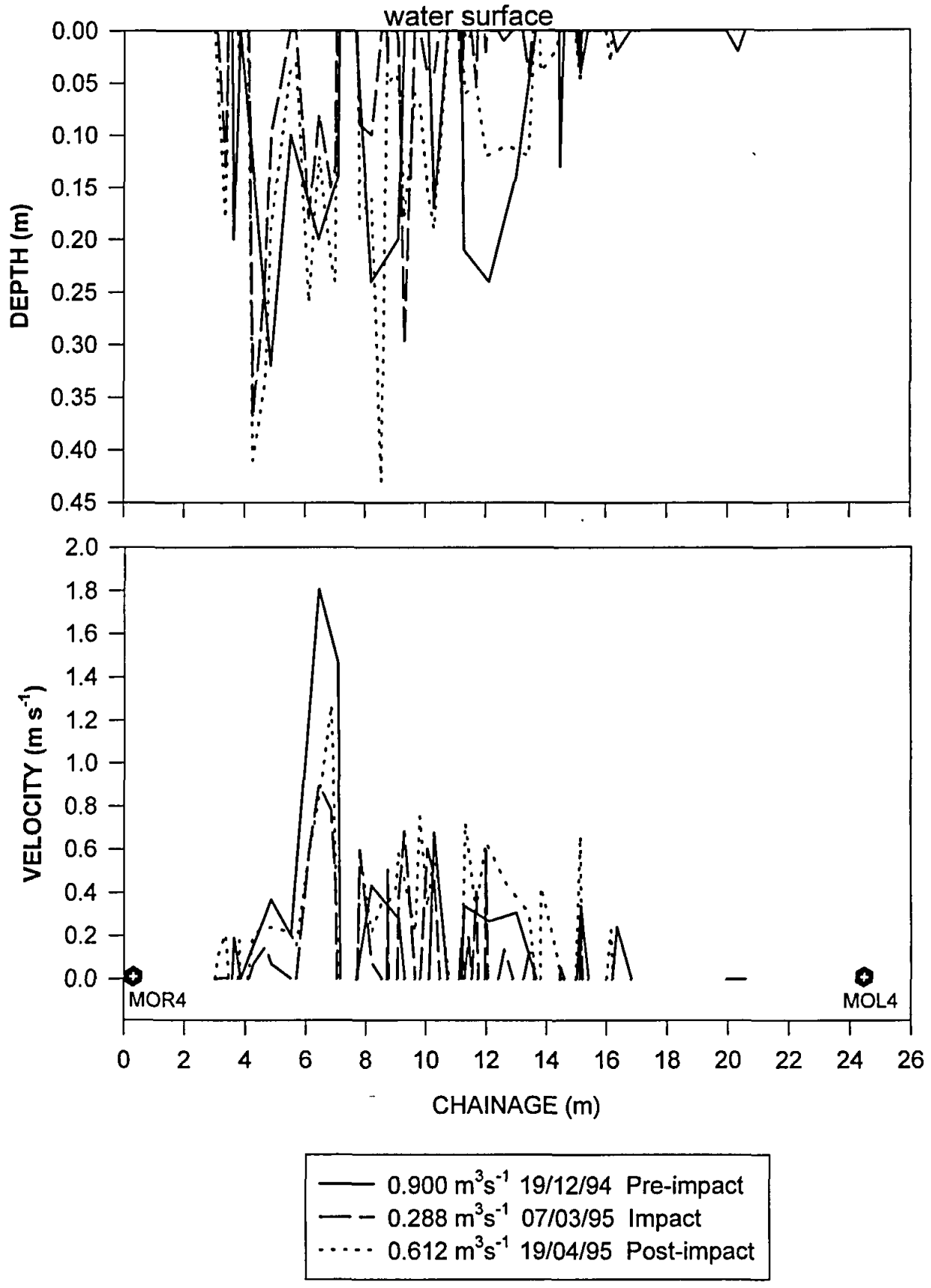


Figure 10.41 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 4) in the impact location, Molenaars River, at three discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study. The discharge recorded during the impact phase constitutes an unnaturally low baseflow (compare with Figure 10.38).

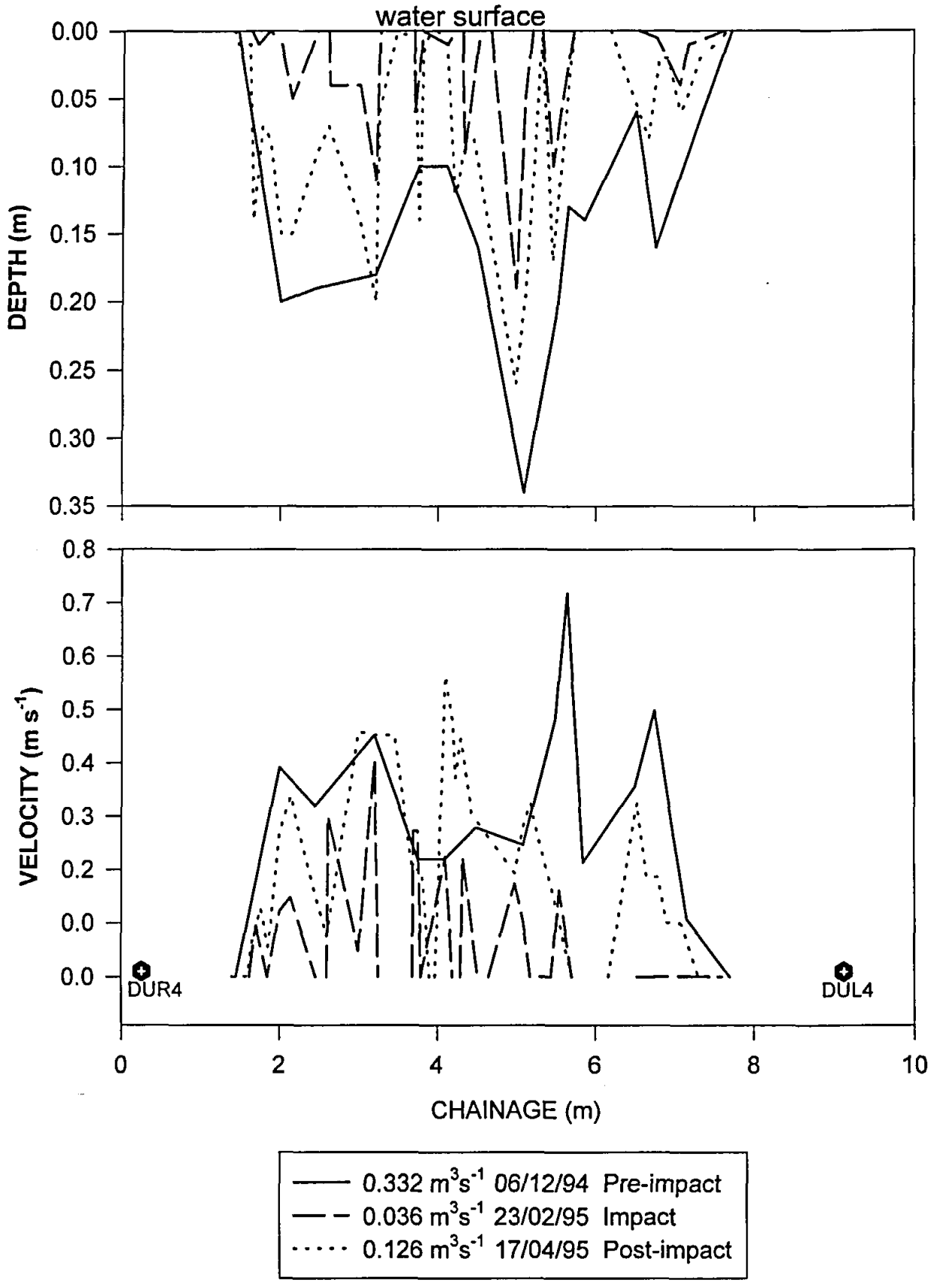


Figure 10.42 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 4) in the impact location, Du Toits River, at three discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study. The discharge recorded during the impact phase constitutes an unnaturally low baseflow (compare with Figure 10.39).

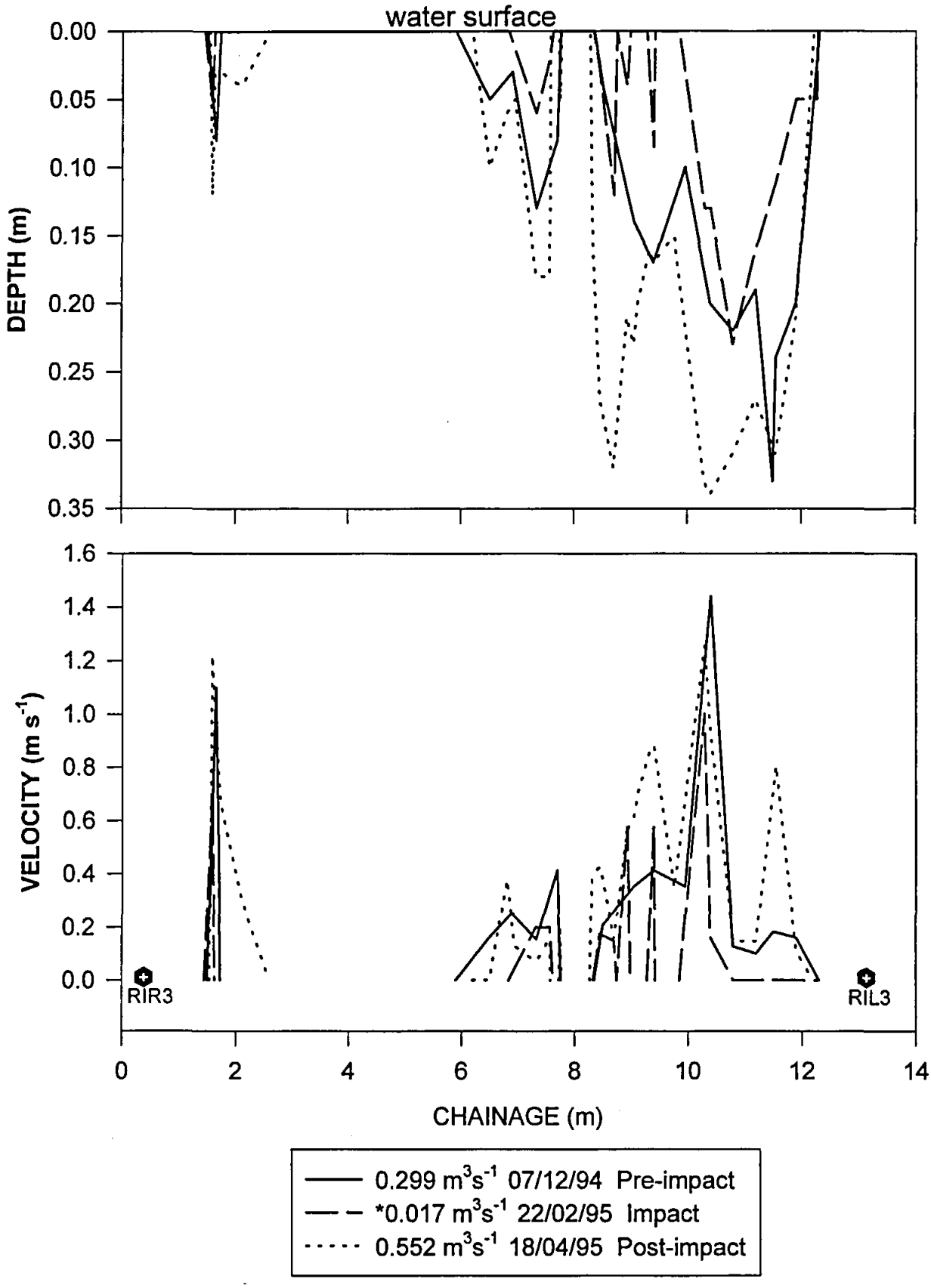


Figure 10.43 Distribution of water depths and average velocities (at 0.6-depth) across a riffle (cross-section 3) in the impact location, Riviersonderend River, at three discharges. The data were recorded during the pre-impact, impact and post-impact phases of the low-flow study. The discharge recorded during the impact phase constitutes an unnaturally low baseflow (compare with Figure 10.40). *The run discharge is indicated, as a high experimental error was associated with the measured riffle discharge.

of 0.604 m s^{-1} (Figure 10.40), there was a marked increase in zero velocity cells and a loss of wetted width, compared with antecedent conditions (Figure 10.43). An increase in the dissection of wetted biotope width was also evident (Figure 10.43).

Comparison of riffles subjected to unnaturally low baseflows

For riffles at extremely low baseflows, in the impact phase, the maximum depth recorded was 0.37 m (Molenaars site), while both the Molenaars and Riviersonderend impact locations had similarly high average depths of 0.09 m. The Du Toits impact riffle had the lowest maximum and average depths, at 0.19 m and 0.05 m, respectively. For the three impacted riffles, the highest maximum and average velocities observed were 1.001 m s^{-1} and 0.347 m s^{-1}, respectively, for the Riviersonderend and Molenaars impact locations. The lowest maximum velocity, of 0.406 m s^{-1} and lowest average velocity of 0.150 m s^{-1}, were obtained for the Du Toits riffle. The percentage decrease in average riffle depth in February, relative to natural conditions at that time, was greatest for the Du Toits impact location at 46%, while an anomalous pattern was observed for the Riviersonderend (see above). The largest percentage decrease in average velocity, of 66%, was recorded for the same site.

A 36% reduction in baseflow at the Molenaars site, during the impact phase, did not result in a change in the transect riffle to any other biotope type. However, extreme reductions in baseflow at the Du Toits and Riviersonderend sites, resulted in changes from transect riffle to trickle run and riffle/run transitional biotope, respectively (Section 9.5.1). With a trickle run, flow trickled among the stones but the water was not sufficiently deep to cover them. The flow was also of low velocity, without the turbulent, broken water surface characteristic of riffles. In the riffle/run transitional biotope, although riffle areas remained, there were also slow-flowing sections where water trickled among stones. These changes in biotope type persisted throughout the impact phase, but recovery to pre-impact riffles immediately followed the reinstatement of natural baseflows.

Typically, the natural physical character of the impacted riffles rendered them more susceptible to large scale losses in connectivity of wetted habitat than runs, and to increases in the proportion of wetted width subjected to flow cessation.

10.4.2 Effects of natural and unnatural dry-season baseflows on the dynamics of physical biotopes

The effects of fluctuations in dry-season baseflows on the dynamics of physical biotopes were examined, both in relation to changes in the total amount of wetted habitat, and for available wetted habitat, to changes in the proportions of the various cell biotopes it represented.

Changes in wetted habitat represented by wetted widths of transect biotopes

Changes in available wetted habitat represented by the total channel widths (i.e. WE to WE) across riffles, runs and for the data from both transect biotopes combined (average wetted widths), were examined for control and impact locations, for all four sites. The data excluded all dry patches along each cross-section, such as exposed boulders. They were used to provide an indication of any large scale changes in the amount of habitat available for the BMI fauna, with fluctuations in discharge (Figures 10.44-10.46). The corresponding discharge data are presented in Figure 10.1.

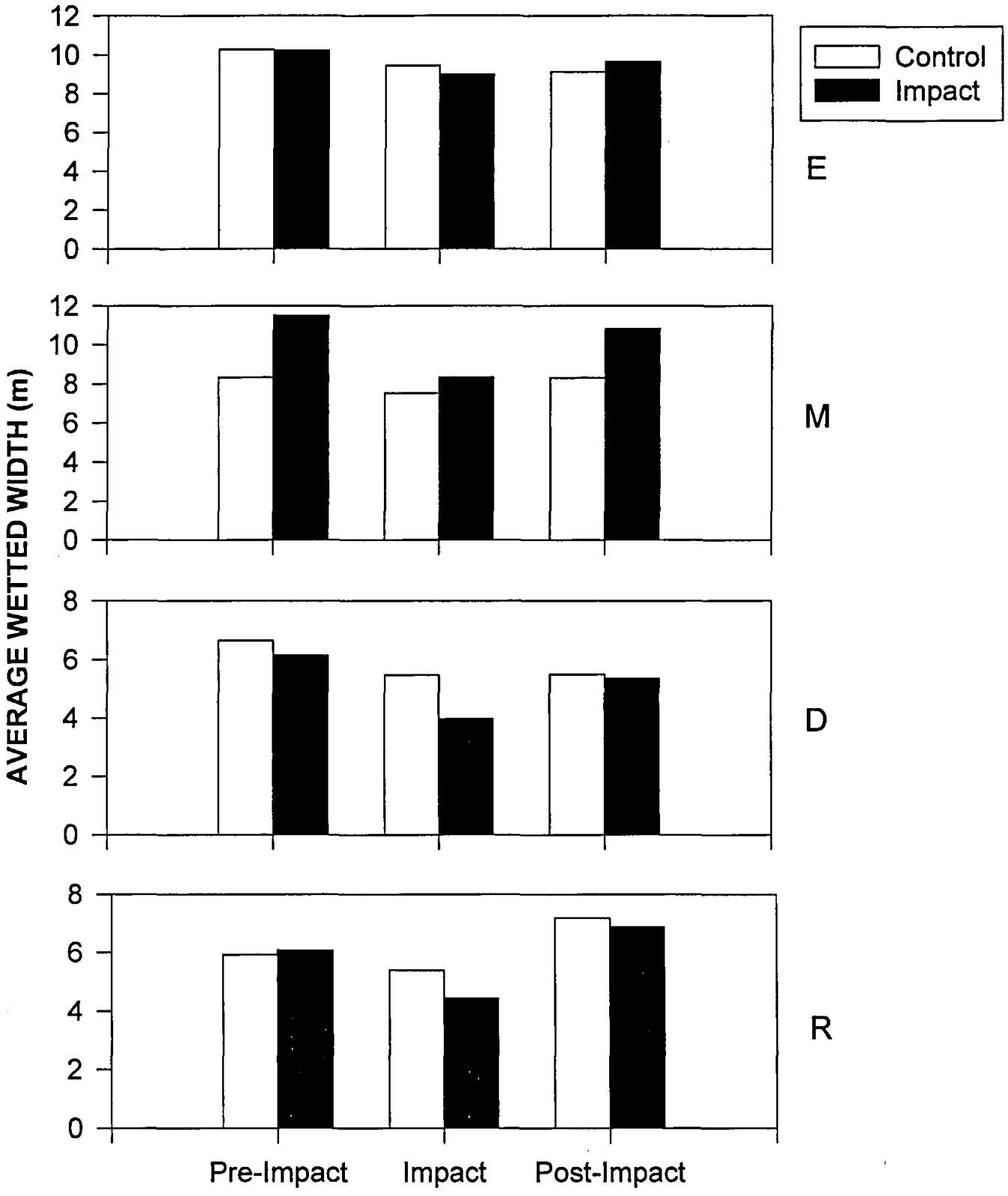


Figure 10.44 Average wetted widths calculated from two cross-sections in each control and impact location, for the four sites. Pre-impact – December 1994; impact – February/early March 1995; post-impact – April 1995. E – Elands; M – Molenaars; D – Du Toits; R – Riviersonderend.

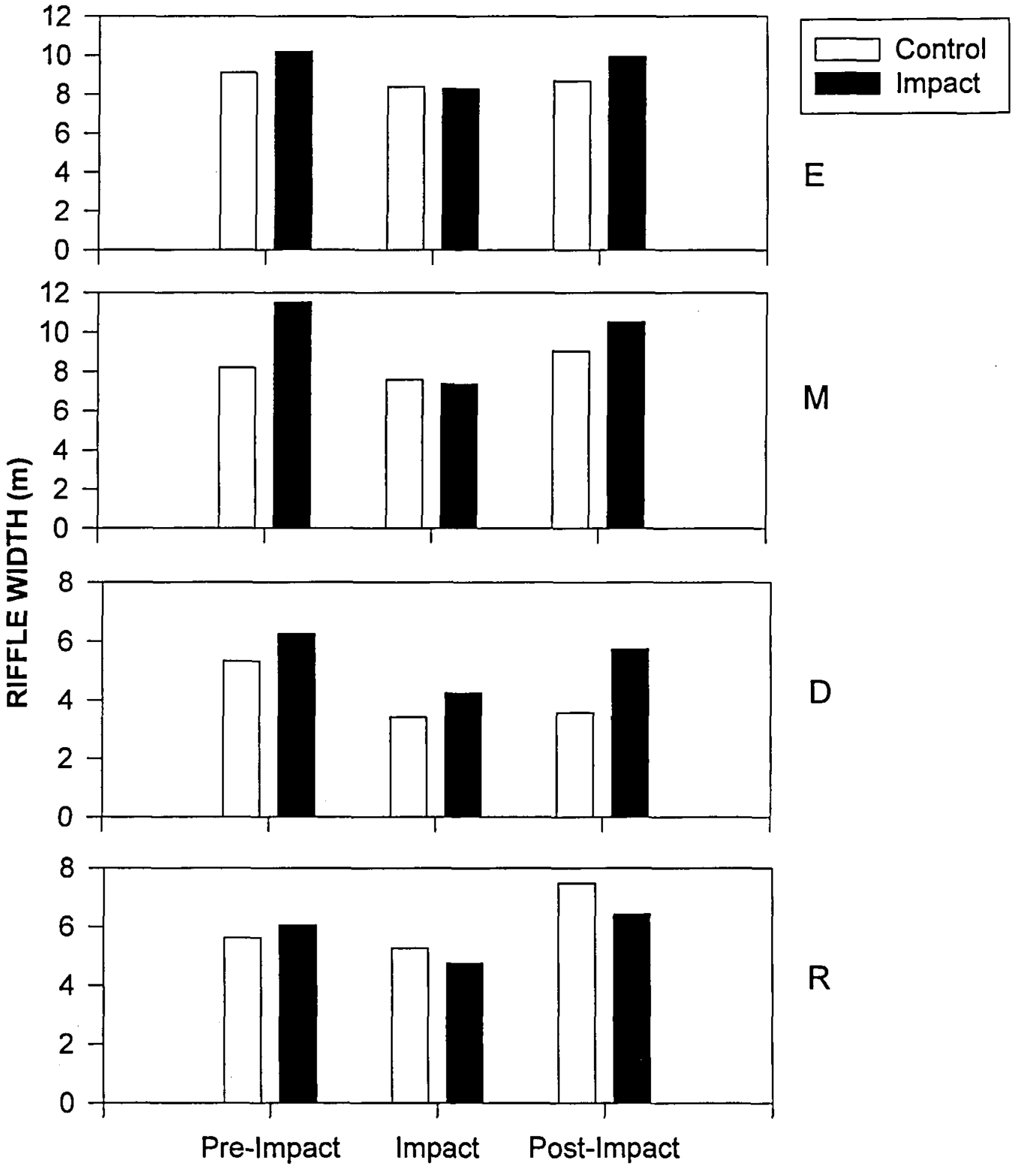


Figure 10.45 Riffle widths calculated from a single cross-section in each control and impact location, for the four sites. Pre-impact – December 1994; impact – February/early March 1995; post-impact – April 1995. E – Elands; M – Molenaars; D – Du Toits; R – Riviersonderend.

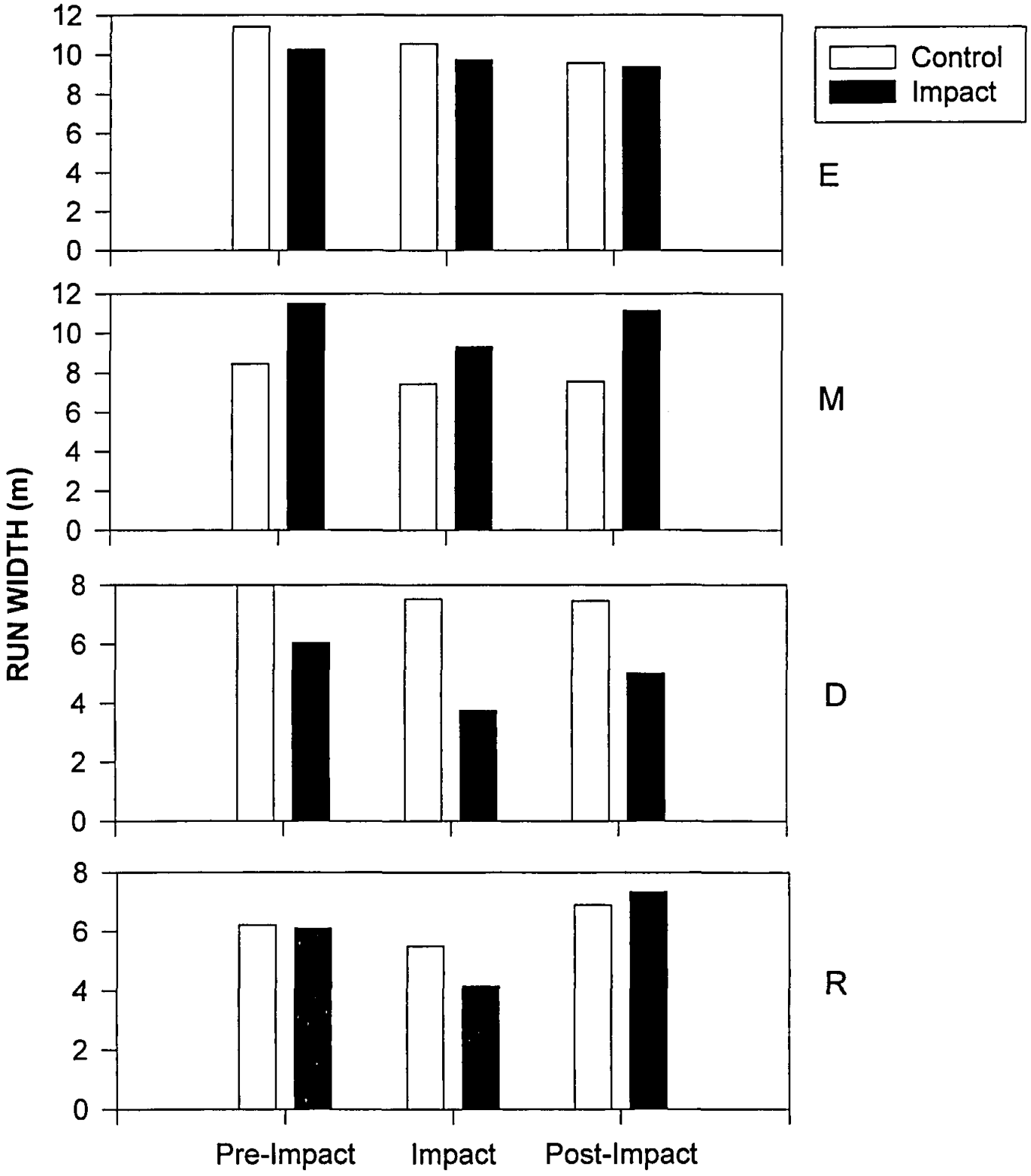


Figure 10.46 Run widths calculated from a single cross-section in each control and impact location, for the four sites. Pre-impact – December 1994; impact – February/early March 1995; post-impact – April 1995. E – Elands; M – Molenaars; D – Du Toits; R – Riviersonderend.

As envisaged, in all instances, changes in average wetted habitat (Figure 10.44), riffle widths (Figure 10.45) and run widths (Figure 10.46), clearly tracked natural increases or decreases in discharge at all sites (Figure 10.1). However, it was apparent that relatively large changes in discharge were dampened in the degree to which they were reflected in terms of changes in available habitat. This effect was probably as a result of cross-sectional shape (see Section 10.4.1). Moreover, the Elands data indicated that there are clear natural differences in the morphology (wetted widths) of both different riffles and runs within the same reach, as did the pre-impact data for the three impacted sites.

Unnatural reductions in discharge resulted in reductions in wetted width that were roughly proportional to the quantities of flow abstracted, but were also dampened relative to the absolute quantity of flow abstracted. Although the relative change in width with unnaturally low flows was small for the Molenaars (Figure 10.44), it corresponded with the lesser amount of flow abstracted compared with the other two experimental sites. It was not small, however, considering the large size of the river. Reductions in average wetted width were similar for the Du Toits and Riviersonderend sites, in accordance with the similar percentages of flow reduction (Figure 10.44). This was despite the differences in channel morphology between the two sites (Section 9.2). Decreases in wetted riffle width with flow reduction (Figure 10.45) were less marked than decreases in run widths (Figure 10.46). Hence, most of the difference between average wetted widths for impacted versus non impacted locations was a function of changes in run dimensions. This was an unexpected result, as it was envisaged that riffle areas would shrink the most at extremely low flows. During the post-impact phase, habitat widths for the impact locations were similar, relative to natural widths, to the widths calculated prior to unnatural flow reduction, showing the expected, immediate reinstatement of natural biotope dynamics.

The dynamics of cell biotopes with natural and unnatural changes in dry-season baseflow

The data presented below represent changes in the combined proportions of cell biotopes (see definition in Section 9.5.1, Figure 9.9) across the riffle and run wetted cross-sections (or transect biotopes; see definition in Section 9.5.1, Figure 9.9). These data were used as an index of biotope dynamics (Figures 10.47-10.50). More comprehensive information on biotope dynamics, calculated from biotope maps, is presented elsewhere (Tharme in prep.). The results reflect the percentages of various biotopes in relation to the total amount of wetted habitat provided at the two cross-sections, in each location. Particularly in the case of the locations subjected to unnatural reductions in flow, in addition to any observed changes in biotope proportions, the total amount of available habitat has decreased (as noted above).

Elands site

Under conditions of natural baseflow, as represented by the Elands site (Figure 10.47), cross-sectional biotopes varied highly in their proportions. The proportions were affected by the physical morphology of the cross-sections, which differed for different riffles and runs within the site (Section 10.4.1). The number of biotope types was high at 6-7, and remained stable irrespective of natural reduction in discharge during the dry season. Runs were the dominant biotope in all three months, representing an average of 57-71% of total wetted habitat, followed consistently by riffles at an average of 11-40%. With a natural decrease in baseflow in the peak summer months, there was a marked decrease in the quantity of riffle habitat. However, changes in run proportions varied according to the nature of the cross-sections. Transitional habitats between riffles and runs increased, as did shallow areas of water trickling between cobbles. Biotopes that were present in naturally small proportions, such as pools and isolated standing waters varied

in terms of their presence or absence in any month or part of the reach represented by the site. These factors made such biotopes poor indicators of changes in habitat with changes in discharge. There was a negligible increase in baseflow from March to April (March average discharge of 0.231 $m^3 s^{-1}$, and April average discharge of 0.237 $m^3 s^{-1}$; Figure 10.1), yet some redistribution of biotopes was recorded. Moreover, the natural baseflow of the Elands was only marginally lower in the months of March (impact location) and April (control location) than in the other study months. Much of the change in biotope proportions in different months was, therefore, quite possibly a result of subjective, misclassification of riffle/run transitional areas as riffles or runs.

Within the reach represented by the Elands site, there was a fairly high variability in the proportions of cell biotopes at the two locations, throughout the six-month experimental period. Differences between the two locations were encountered for both minor and dominant biotopes (Figure 10.47). For instance, runs represented 42% and 90% of total biotope availability, at the control and impact locations respectively, in December. For smaller biotopes, pools represented 6% of the total in the control location in December, while there were no pool patches across the cross-sections in the impact location at that time (Figure 10.47).

Molenaars site

For the Molenaars River (Figure 10.48), under conditions of natural baseflow, cross-sectional biotope proportions varied highly, as a result of the naturally varied morphologies of the transect biotopes (Section 10.4.1). Runs were the dominant biotope in all three months, representing 60-66% (February and December, respectively) of total wetted habitat, followed consistently by riffles at 16-31% (April and December, respectively). With a natural decrease in baseflow in the peak summer months, there was a decrease in riffle proportions that extended into early autumn. A slight decrease in runs was paralleled by an increase in pool/run transitional areas. Pools and standing waters were present in small proportions, and varied in terms of their presence or absence in any month or part of the reach represented by the site. Hence, they were poor indicators of changes in habitat with changes in discharge. A small increase in runs and higher-velocity areas tracked the slight increase in baseflow from March to April (Figure 10.1).

With a moderate, unnatural reduction in discharge (36%) below natural summer magnitudes, there was a decrease in the proportion of riffles, as well as a concomitant increase in runs and areas transitional between the two biotopes (Figure 10.48). Changes in biotope proportions accentuated the dominance by runs. Although riffles decreased in availability, they remained the second most dominant biotope. Although there was an increase in the percentage of pool habitat, it was such that natural variability could not be discounted as the reason for the increase. The number of biotope types was fairly variable both under natural and unnatural baseflow conditions and bore no consistent relationship with increases or decreases in baseflow. Recovery of biotope proportions to natural levels occurred in April, with increases in both riffle and run percentages reflecting the slight natural increase in baseflow.

Du Toits site

For the Du Toits site (Figure 10.49), under conditions of natural baseflow, runs were typically in slightly higher proportions (32-47%; February and April, respectively) than riffles (22-38% of total wetted habitat; February and December). This trend was maintained throughout the study period, except in December. With a natural decrease in baseflow in the peak summer months, there was a decrease in riffle proportions and a slight decrease in habitat provided by runs. These changes were paralleled by an increase in run/riffle

transitional areas. Pools and standing waters were present in naturally small proportions, and varied in terms of their presence or absence in any month or part of the reach represented by the site. An increase in runs and riffles, and a concomitant decrease in transitional combinations of these two biotopes, occurred with a natural increase in baseflow in early autumn (Figures 10.49 and 10.1).

The most pronounced effect of the large, unnatural reduction in discharge (86%) in February, was the complete loss of all cell riffles. Indeed, within the entire impact location, only a few very small patches of riffle remained (pers. obs.). This marked decline in high velocity habitat was accompanied by a pronounced shift in biotope proportions towards low velocity habitats, with a particularly large increase in the availability of runs (to 79% of total wetted habitat). It should also be borne in mind that there was a pronounced overall loss in wetted habitat (Figure 10.43). Although runs remained the dominant biotope, the second most abundant habitats were pool/run transitional areas (Figure 10.49). That is, there was a shift in dominance at the level of the subdominant biotope. Hence, although an exaggeration of natural trends occurred with extreme reduction in dry-season baseflows, there was also an important shift in biotope dominance. The number of biotope types was fairly variable under conditions of both natural and unnatural baseflow, and bore no consistent relationship with increases of decreases in baseflow. Recovery of biotope proportions to natural levels occurred in April, with the natural increase in baseflow (Figure 10.1). This primarily took the form of reinstatement of riffles as the second most dominant biotope, associated with a simultaneous decrease in run proportions to more natural levels.

Riviersonderend site

The Riviersonderend site (Figure 10.50) did not exhibit a clear dominance between riffle and runs within the reach, at natural baseflows. Natural variability, between months, in percentages of biotopes was also high. With a natural decrease in baseflow in the peak summer months, there was a decrease in both riffle and run proportions, paralleled by an increase in pools (Figure 10.50). The naturally higher baseflows in April than in the preceding months (Figure 10.1), and higher amounts of total wetted habitat, resulted in different biotope dynamics in the two locations. However, there was a mutual shift towards higher-velocity biotopes.

The large, unnatural reduction in discharge (86%) in the impact location, in February, resulted in the loss of almost all riffle biotopes. Only 6% of total wetted habitat remained as riffles in contrast to 32% for the control cross-sections (Figure 10.50). A total of only two, small patches of riffle remained during the impact phase (pers. obs.). Moreover, the decline in high-velocity habitat was compounded by the additional loss of runs, and a greater shift to areas transitional to pools. As a result, biotope dominance was entirely altered, with pool/run transitional areas dominant at 29%, in contrast to natural dominance in this location by riffles during the pre-impact and post-impact periods (Figure 10.50). Pools and standing waters also increased markedly (Figure 10.50). This was in contrast to the response at the Du Toits site, where the channel morphology differed, with few naturally deep pools. The number of biotopes, however, bore no consistent relationship with increases or decreases in baseflow magnitude.

The changes in biotope proportions observed at the Riviersonderend site were particularly dramatic because, as was the case at the Du Toits site, they occurred within a lower amount of remaining wetted width (Figure 10.44).

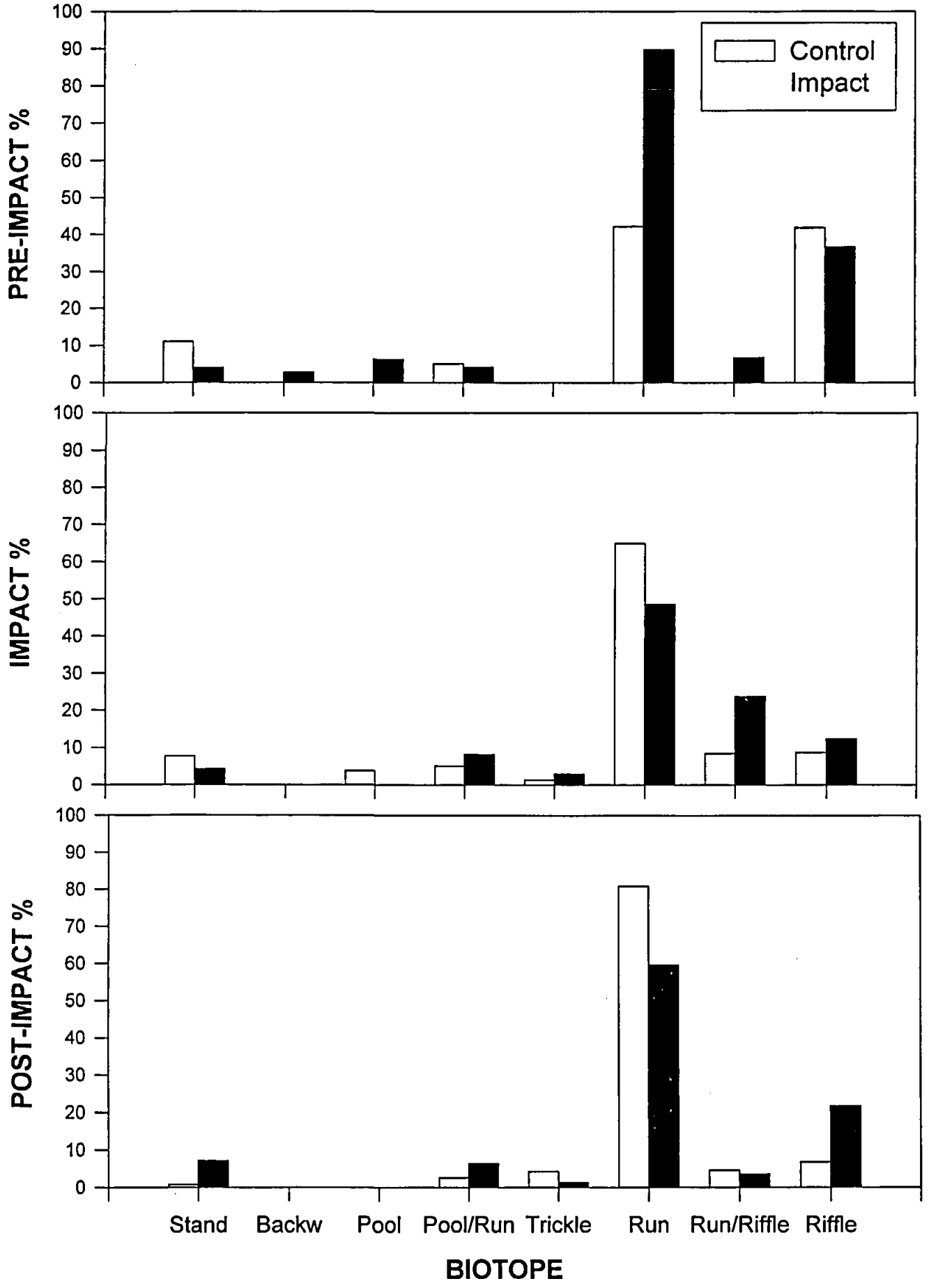


Figure 10.47 Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Elands *River. Pre-impact – December 1994; impact – February/early March 1995;* post-impact – April 1995. Stand – standing water; backw - backwater; pool/run – pool/run transition; trickle – trickle run; run/riffle – run/riffle transition (See Section 9.5.1).

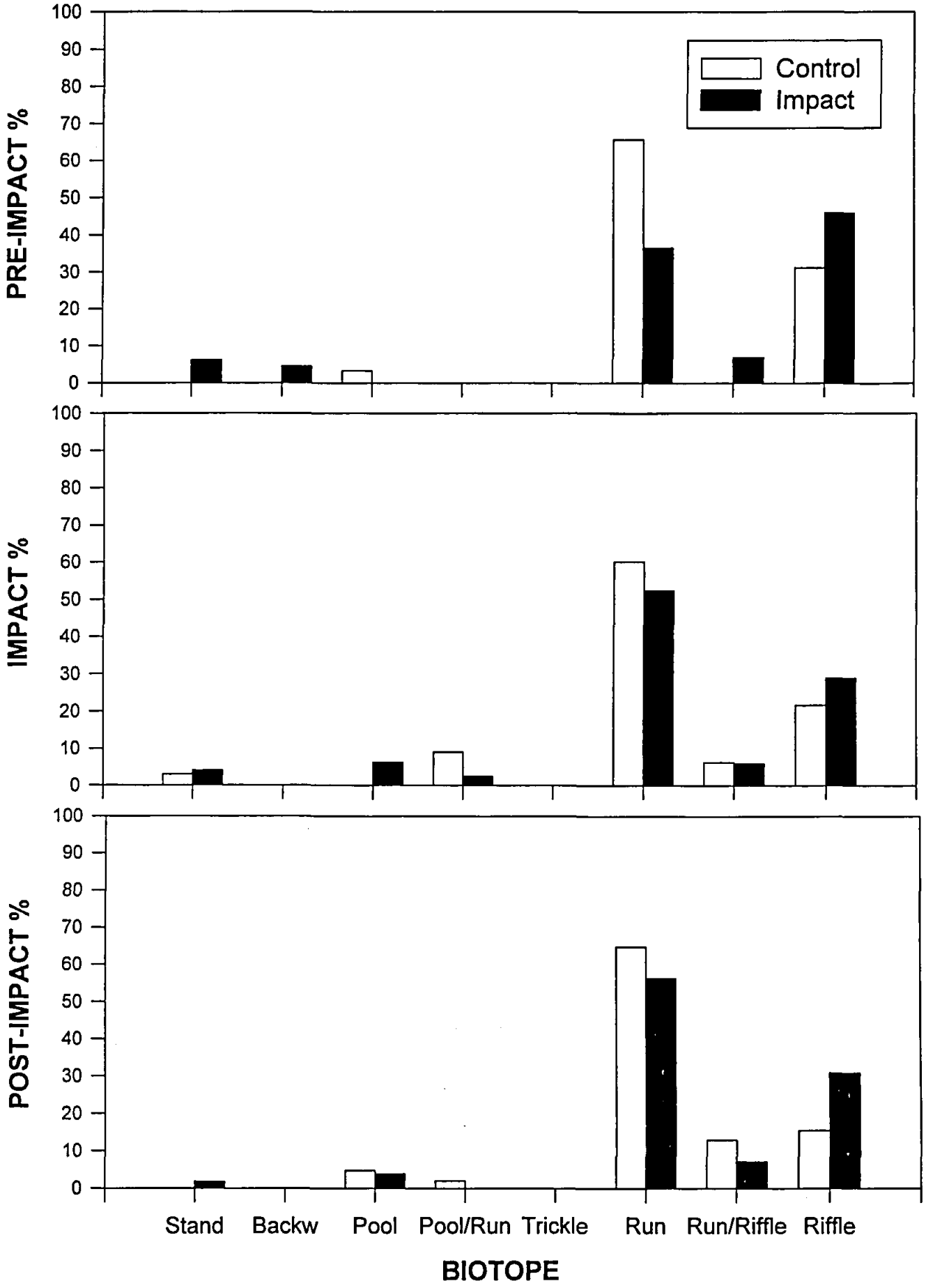


Figure 10.48 **Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Molenaars River. Pre-impact – December 1994; impact – February/early March 1995; post-impact – April 1995. Stand – standing water; backw - backwater; pool/run – pool/run transition; trickle – trickle run; run/riffle – run/riffle transition (See Section 9.5.1).**

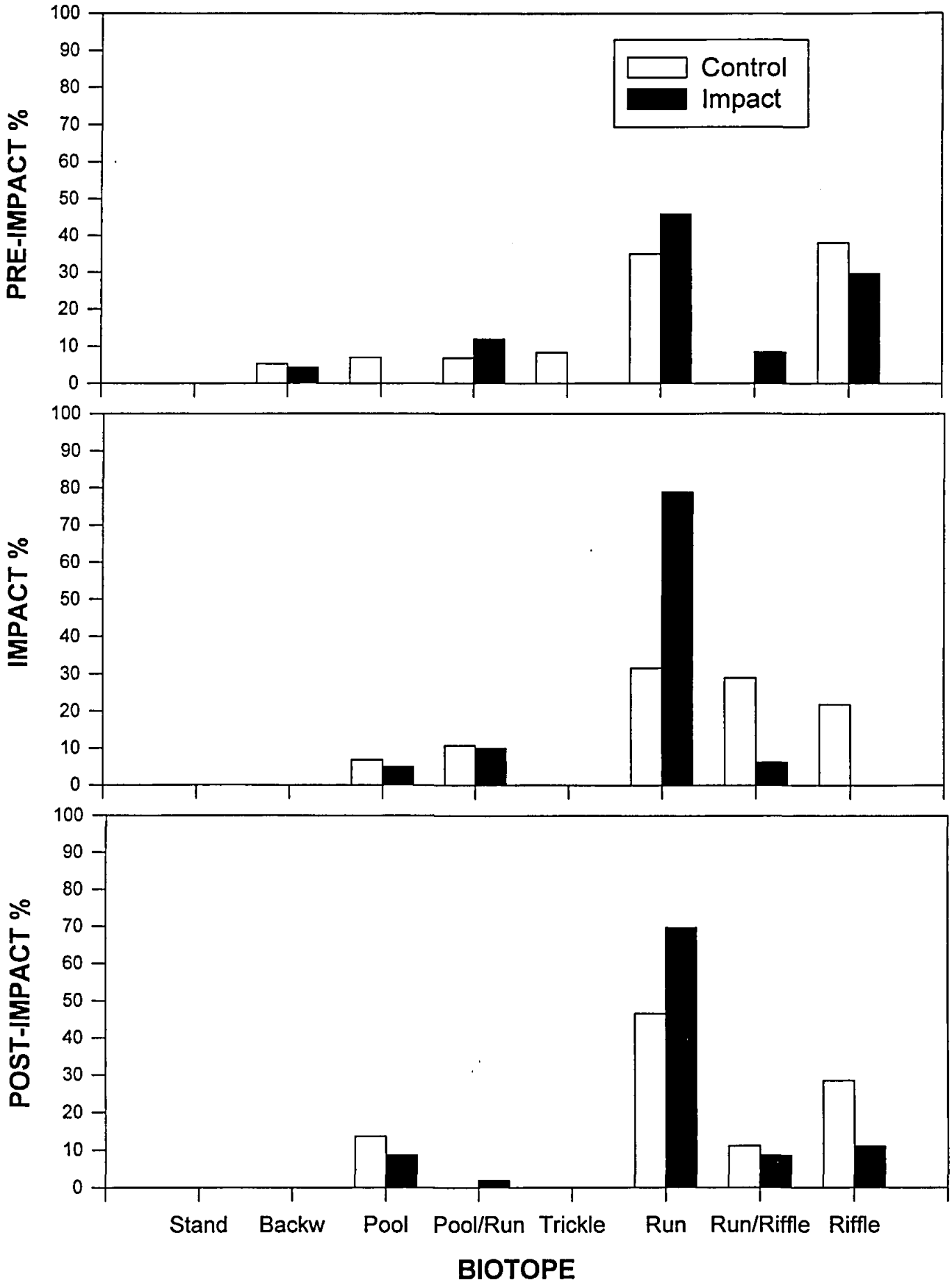


Figure 10.49 Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Du Toits River. Pre-impact – December 1994; impact – February/early March 1995; post-impact – April 1995. Stand – standing water; backw - backwater; pool/run – pool/run transition; trickle – trickle run; run/riffle – run/riffle transition (See Section 9.5.1).

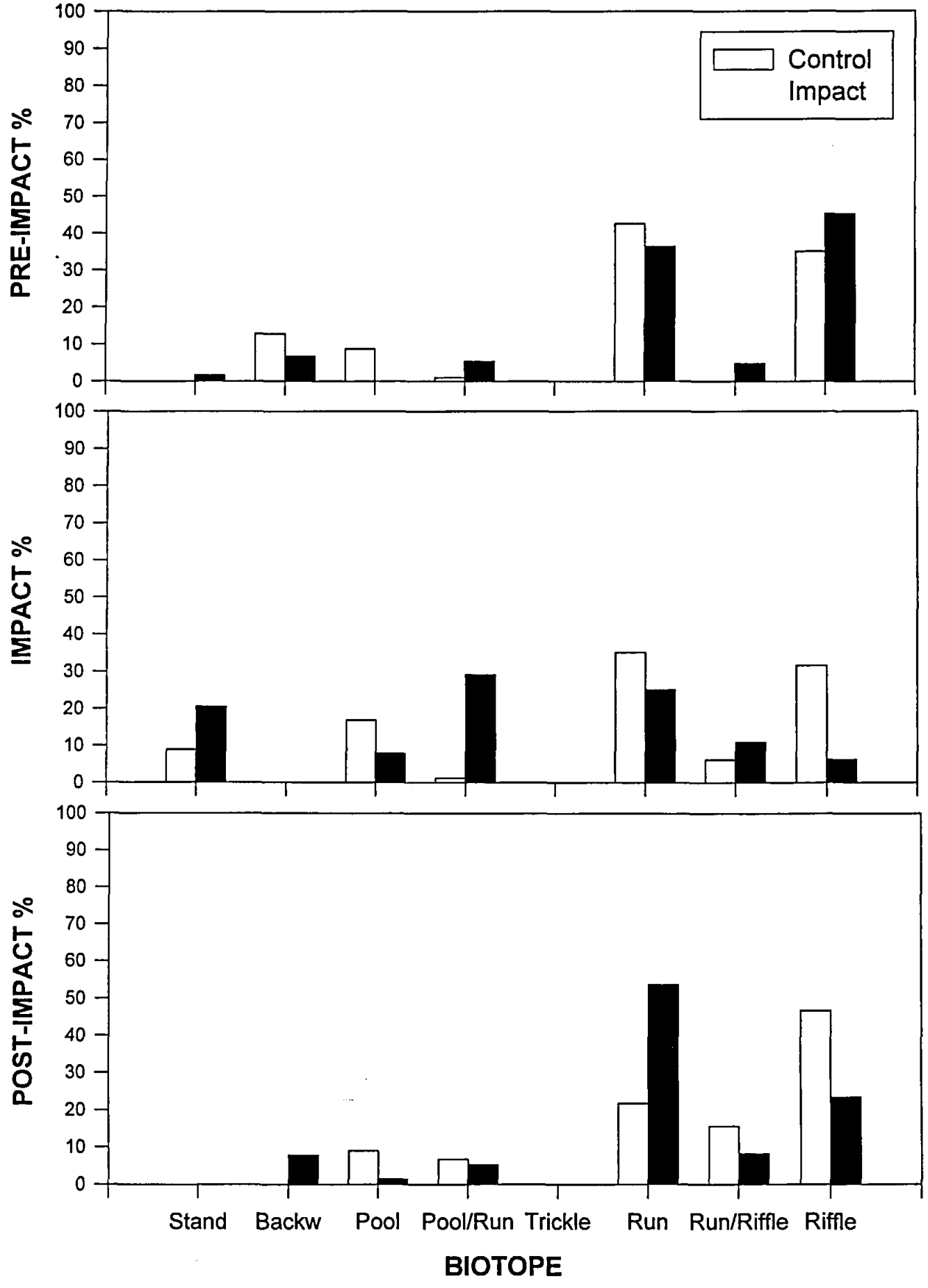


Figure 10.50 Cell biotope proportions as percentages of the total wetted cross-section distance (calculated from two cross-section per location), for the Riviersonderend River. Pre-impact – December 1994; impact – February/early March 1995; post-impact – April 1995. Stand – standing water; backw - backwater; pool/run – pool/run transition; trickle – trickle run; run/riffle – run/riffle transition (See Section 9.5.1).

10.5 RESPONSES OF BENTHIC MACROINVERTEBRATE ASSEMBLAGES TO NATURAL AND UNNATURAL DRY-SEASON BASEFLOWS

The results of a preliminary assessment of changes in the composition of the BMI assemblages of riffle and run biotopes, in relation to both natural and unnatural, dry-season baseflow regimes, are presented in this section. The relevant corresponding discharge data are given in Figure 10.1, while information on methods of BMI data collection and analysis are described in Section 9.5.3. The data presented below represent averages of three BMI samples, per 0.1 m^2 unit area. Averages of samples and individual samples did, however, typically exhibit the same trends. Assessments based on individual samples are presented in Tharme (in prep.). Responses of pool invertebrate assemblages to dry-season baseflow regimes are presented elsewhere, as are species-level data (Tharme in prep.). Direct relationships between BMIs assemblages and the kinds of changes in physical habitat and water chemistry documented in Sections 10.3 and 10.4, are also described in Tharme (in prep.).

In the following sections, the responses of BMI assemblages to baseflow regime were assessed using the following variables: the number of families or higher taxa (hereafter simply referred to as taxa); abundances, as total numbers of individuals; the percentage contribution of various taxa to the overall assemblage, where an arbitrary threshold of 5% or greater was used to identify dominant taxa; diversity, using the Shannon-Weiner diversity index; and Bray-Curtis percentage similarities, for entire assemblages. Benthic macroinvertebrate response was assessed using the data at several levels of resolution. Overall changes in the BMI assemblages inhabiting each location were firstly addressed, for the four sites. Responses were then documented on a biotope-specific basis, and in relation to the various components of the substratum, as illustrated in Figure 9.10.

10.5.1 Changes in the number of benthic macroinvertebrate taxa in response to natural and unnatural dry-season baseflows

Effects of natural and unnatural baseflows on the overall number of taxa at the four sites

Changes in the number of taxa comprising the BMI assemblages of each location, in relation to both natural and unnatural baseflow conditions, before, during and after flow manipulation, are illustrated in Figure 10.51.

Generally, at natural baseflows, similar numbers of taxa (20-37 0.1 m^{-2}) were collected from each site and location, in all three sampling months (Figure 10.51). Numbers were fairly stable over the range of natural baseflows, and there was no consistent pattern among rivers of an overall increase or decrease in taxa with a decrease in baseflow magnitude (Figure 10.51). For the Elands and Riviersonderend sites, there was a slight decrease in the number of taxa during the months of naturally lowest flow, while the converse was true of the other two sites. Numbers of taxa were typically lowest at the end of summer/early autumn (Figure 10.51). Across all four sites, the lowest number of taxa (20 0.1 m^{-2}) was recorded for the Riviersonderend site in April, when baseflows increased with the shift to autumn instream conditions (Figure 10.1). The highest number of taxa, 37 0.1 m^{-2}, was recorded from the Elands, in December (Figure 10.1). Some natural variability in numbers of taxa within the same reach was clearly shown by the Elands data, where the impact

location had consistently higher numbers of taxa than the control location for all three sampling trips (Figure 10.51).

For the three experimental sites, the difference between data for the two locations in any month, during the pre- and post-impact phases, was similar or not much greater than the difference between natural and impacted locations. This made it difficult to establish whether or not the differences in the number of taxa between control and impact locations for these sites were significant, without further analyses. During the impact phase, the number of taxa increased for all the locations at which extreme flow reduction took place (Figure 10.51). However, there did not appear to be a direct relationship between the proportion of flow reduction and the increase in number of taxa. After the reinstatement of natural baseflows, in April, the Molenaars showed complete recovery in terms of the number of taxa, while numbers were still marginally elevated, relative to natural trends, at the Du Toits and Riviersonderend impact locations (Figure 10.51). However, the differences could well fall within the realm of the natural variability evident from the sites. It should be borne in mind that observed changes in riffle and run biotopes might have counterbalanced each other, obscuring any overall trends. Also, there is likely to have been a variable overlap in non biotope-specific taxa between the two biotopes, which would influence the observed overall patterns.

Effects of natural baseflows on the number of taxa in riffle and run biotopes

The numbers of taxa recorded independently for riffles and runs, including the numbers of taxa inhabiting the tops and bottoms of stones, and the underlying substratum, are given in Figures 10.52-10.55, for the four sites.

Elands site

Under conditions of natural, dry-season baseflow, variability in the numbers of taxa inhabiting both biotopes, and various parts of the substratum, was shown to be high, especially on the basis of the Elands data (Figure 10.52). At this site, overall, numbers of taxa between locations and for different months were most divergent for runs. Tops of riffle and run stones clearly provided microhabitat for fewer taxa than stone undersides or the substratum underneath the stones (Figure 10.52), with 6-17 taxa 0.1 m^{-2} and 2-13 taxa 0.1 m^{-2}, respectively. This contrasted with similar, higher numbers for the other two microhabitats, with 14-27 riffle taxa 0.1 m^{-2} and 6-17 run taxa 0.1 m^{-2} for stone bottoms, and 19-26 riffle taxa 0.1 m^{-2} and 11-27 run taxa 0.1 m^{-2} for underlying cobble (Figure 10.52).

No consistent relationship between the number of taxa and baseflow magnitude could be detected at the Elands site over the experimental period, for the entire assemblages from either biotope (Figure 10.52). For both riffles and runs, the number of taxa on stone tops was lowest in February, and lower at the end of the dry season than in December, indicating a tendency for a decrease in numbers of taxa with natural decreases in baseflow. However, no clear or consistent relationships between baseflows and numbers of taxa were evident for riffle or run stone bottoms. Similarly, for the underlying substratum, although there tended to be a trend towards a reduction in numbers with a decrease in baseflow, it was not consistent.

Molenaars site

At the Molenaars site, there was a slight increase in the number of taxa in both riffles and runs at naturally low baseflows (Figure 10.53). Numbers remained elevated for runs during April, but decreased with increasing baseflow for riffles. The overall pattern for riffles was reflected by the tops and particularly the

bottoms of riffle stones while, in contrast, the number of taxa in the underlying substratum was lowest at the peak of the dry season. Under natural conditions, the maximum number of taxa inhabiting riffles (28 0.1 m^{-2}) was recorded for stone bottoms in February, while only 9 taxa 0.1 m^{-2} occurred on stone tops in both December and April. The same general trends in the distribution of riffle taxa were observed in runs. However, the numbers of taxa on the tops of stones in runs remained elevated above December values late in the dry season. The range of numbers of taxa recorded for runs was from 6 0.1 m^{-2} in December (tops of stones) to 23 0.1 m^{-2} in February and December, for stone bottoms and the underlying substratum, respectively.

Du Toits site

The Du Toits control data showed a slight increase and decrease in the number of taxa in riffles and runs, respectively, at naturally lowest, dry-season baseflows (Figure 10.54). These marginal trends were reversed for the two biotopes with the slight increase in baseflow in April (Figure 10.1). However, overall, numbers of taxa were stable over the whole summer in both instances. The overall pattern of response of riffle taxa was most strongly reflected by the data from stone tops, and to a lesser degree the underlying substratum. The converse was recorded for the bottoms of the riffle stones. For runs, there was a natural decrease in the number of taxa inhabiting all parts of the substratum in the peak dry season, followed by a partial increase towards December values, in April. The maximum number of taxa recorded from riffles in at the Du Toits control location was 27 0.1 m^{-2}, for the underlying substratum in February. A minimum of 7 taxa 0.1 m^{-2} occurred on the tops of riffle stones, in December. The same general trend was observed in the distribution of run taxa, where the most taxa were recorded from the underlying substratum in December (25 0.1 m^{-2}), and the minimum of 6 0.1 m^{-2} occurred on stone tops in February.

Riviersonderend site

For the Riviersonderend control location, there was a decrease in the number of taxa, for both riffles and runs, from December to naturally lowest flows in February (Figure 10.55). The decrease was observed for all components of the substratum. Numbers decreased further with the autumn increase in baseflow above December magnitudes. A maximum of 26 riffle taxa 0.1 m^{-2} was recorded for both stone bottoms and underlying stones, in December. A minimum of only 6 taxa 0.1 m^{-2} occurred on riffle stone tops, in February. The range of numbers of taxa recorded for runs was from 5 0.1 m^{-2} in February (stone tops) to a relatively low maximum of 16 0.1 m^{-2}, for the underlying substratum, in December.

Effects of unnatural baseflows on the number of taxa in riffle and run biotopes

Molenaars site

With a moderate (36%) unnatural reduction in baseflow in the Molenaars impact location, there was an increase in the overall number of riffle taxa above control numbers during the impact phase (34 taxa 0.1 m^{-2} as compared with 31 0.1 m^{-2} for the control location). The increase was sustained into the recovery phase. In contrast, numbers of run taxa dropped markedly (18 0.1 m^{-2}) relative to the natural situation (26 0.1 m^{-2}), but full recovery was apparent after the reinstatement of natural flows in early April (Figure 10.53). Examination of the distribution of riffle taxa revealed that the majority of the observed change could be attributed to an increase in taxa in the underlying substratum to 28 0.1 m^{-2}, compared with 18 0.1 m^{-2} under natural conditions. There were no obvious changes in numbers of taxa, relative to the control riffles, for the stone tops or bottoms. For impacted runs, the majority of change in terms of a reduction in the number of taxa, occurred on stone bottoms, with a very low average of 8 taxa 0.1 m^{-2}, as opposed to 23 0.1 m^{-2} in

control samples. There was no concomitant increase in taxa for stone tops or underlying stones, suggesting that the movement of taxa was out of run biotopes. Increases in taxon numbers in the impact location took place for all three components of runs, during the recovery phase of the study. Such trends were not apparent for riffles.

Du Toits site

With about 86% discharge reduction in February, at the Du Toits impact location, there was a decrease in the overall number of riffle taxa from 28 to 23 taxa 0.1 m^{-2}. The number of taxa increased by two (to 30 0.1 m^{-2}) over the same period under natural conditions (Figure 10.54). The opposite response was observed for runs, where the numbers of run taxa increased by 12 0.1 m^{-2} (to 30 taxa 0.1 m^{-2}) relative to pre-impact conditions, while the natural change was a loss of two taxa, to only 23 taxa 0.1 m^{-2}. For both biotopes, recovery towards natural levels appeared partial. From the recorded distribution of the numbers of riffle taxa, it was apparent that the observed decrease in taxa for riffles was mostly a result of a marked decline in numbers on the tops of stones to only 4 taxa 0.1 m^{-2} (control tops had 15 taxa 0.1 m^{-2}), with a small decrease for stone bottoms. This contrasted with the natural increase in numbers of taxa in the same month. Although there was an increase in taxa inhabiting the underlying stones, it was slight, providing little evidence for a redistribution of taxa within the same biotope. However, for runs, the number of taxa increased above control figures for all substratum parts. For instance, the number of taxa in the underlying impacted stones rose from 14 0.1 m^{-2} in December to 25 0.1 m^{-2}, while the natural trend was a decrease from 25 to 19 taxa 0.1 m^{-2}. The response of run BMIs to unnaturally low flows was particularly marked, as the number of run taxa in the impact location was naturally consistently lower than the number from control runs, prior to flow manipulation. The tops of impacted run stones still had relatively high numbers of taxa during the experimental recovery phase, while numbers for the other stone parts were the same for the control and impact locations.

Riviersonderend site

A marginally higher proportion flow reduction in the impact location, Riviersonderend site, in February appeared to result only in a slight increase in riffle taxa and decrease in run taxa (Figure 10.55). Moreover, the differences were within the natural variability shown for the control site (see above). For both biotopes, post-impact numbers were highly similar for the control and impact locations. For riffles, examination of the distribution of the numbers of taxa among the various parts of the substratum showed that although there was no obvious change in the number of taxa on the bottoms of riffle stones from December to February, the tops and underlying substratum showed increases in numbers. The latter changes were in contrast to the decline in numbers recorded under conditions of natural baseflow (see above). For instance, the natural change for stone tops was from 17-6 taxa 0.1 m^{-2} (December to February), while the numbers for impacted tops increased from 7-9 0.1 m^{-2} for the same period. Changes in numbers of taxa inhabiting runs were even less evident than those observed for riffles, under unnaturally low flows. Although a slight increase and decrease in numbers was recorded for tops and bottoms of run stones, respectively, the changes could be attributed to natural variability. No obvious response to unnatural baseflows was apparent for the underlying substratum, relative to natural trends.

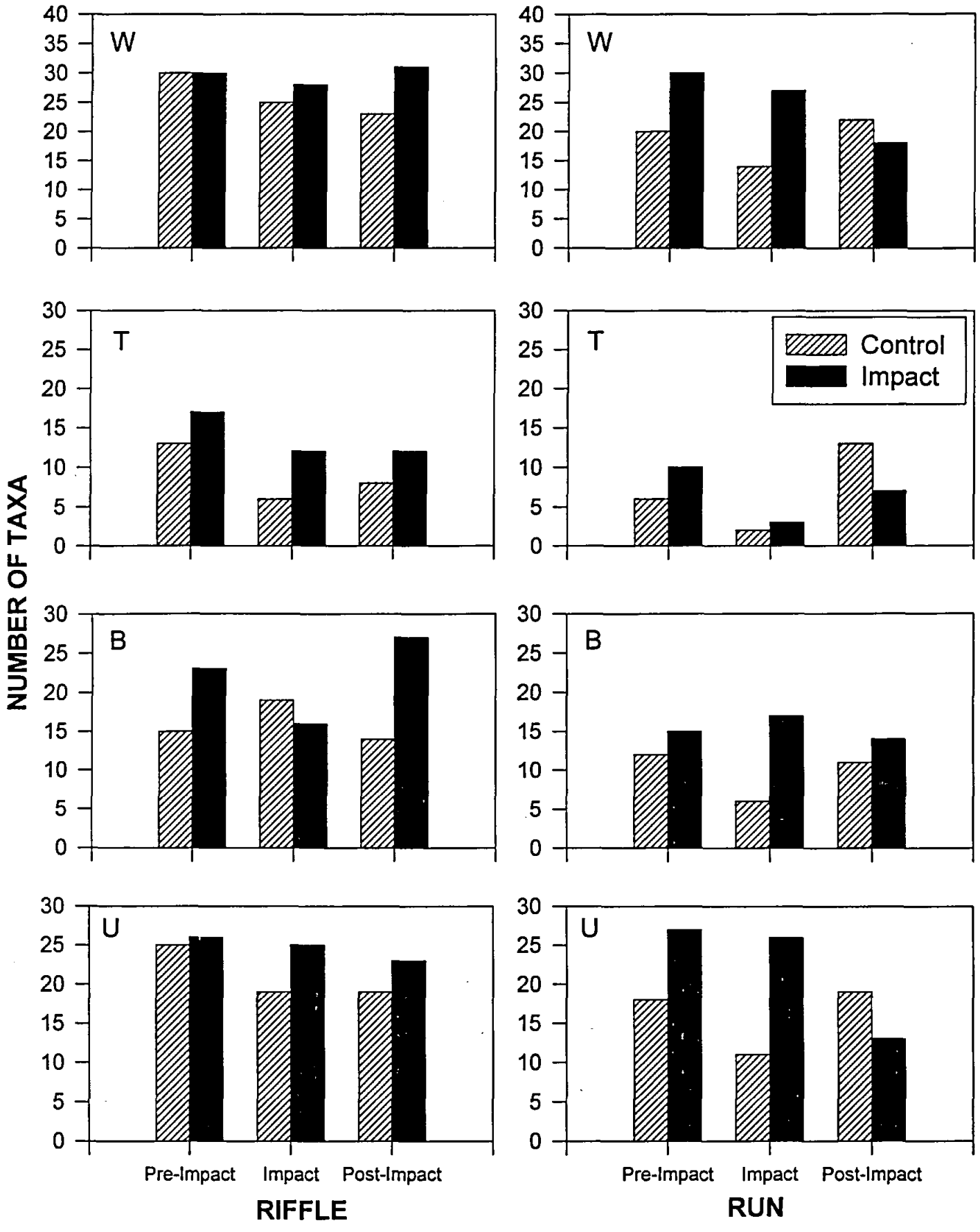


Figure 10.52 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Elands site, at natural baseflows. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

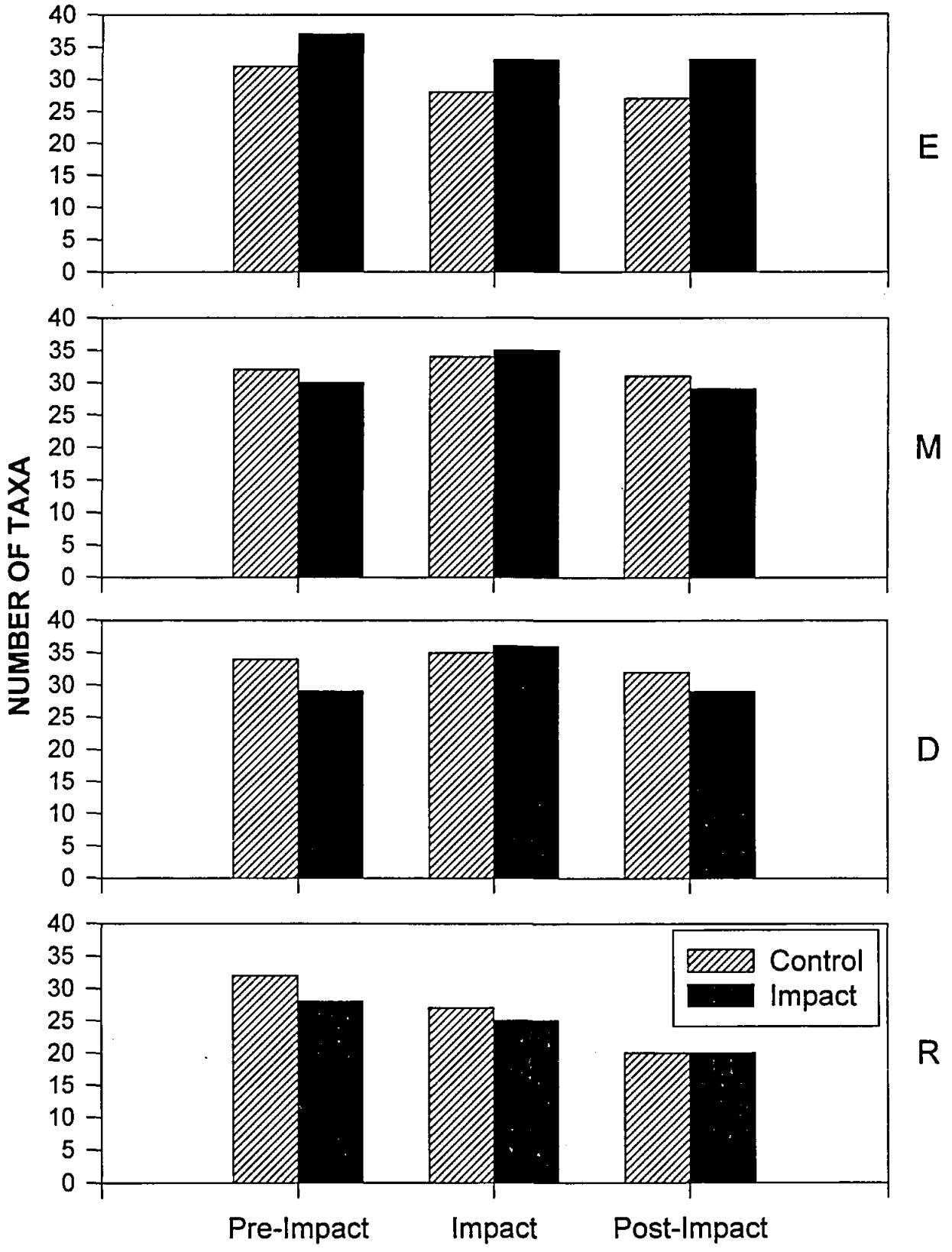


Figure 10.51 Numbers of families and higher taxa recorded in the control and impact locations at the four sites, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. E – Elands; M – Molenaars; D – Du Toits; R – Riviersonderend.

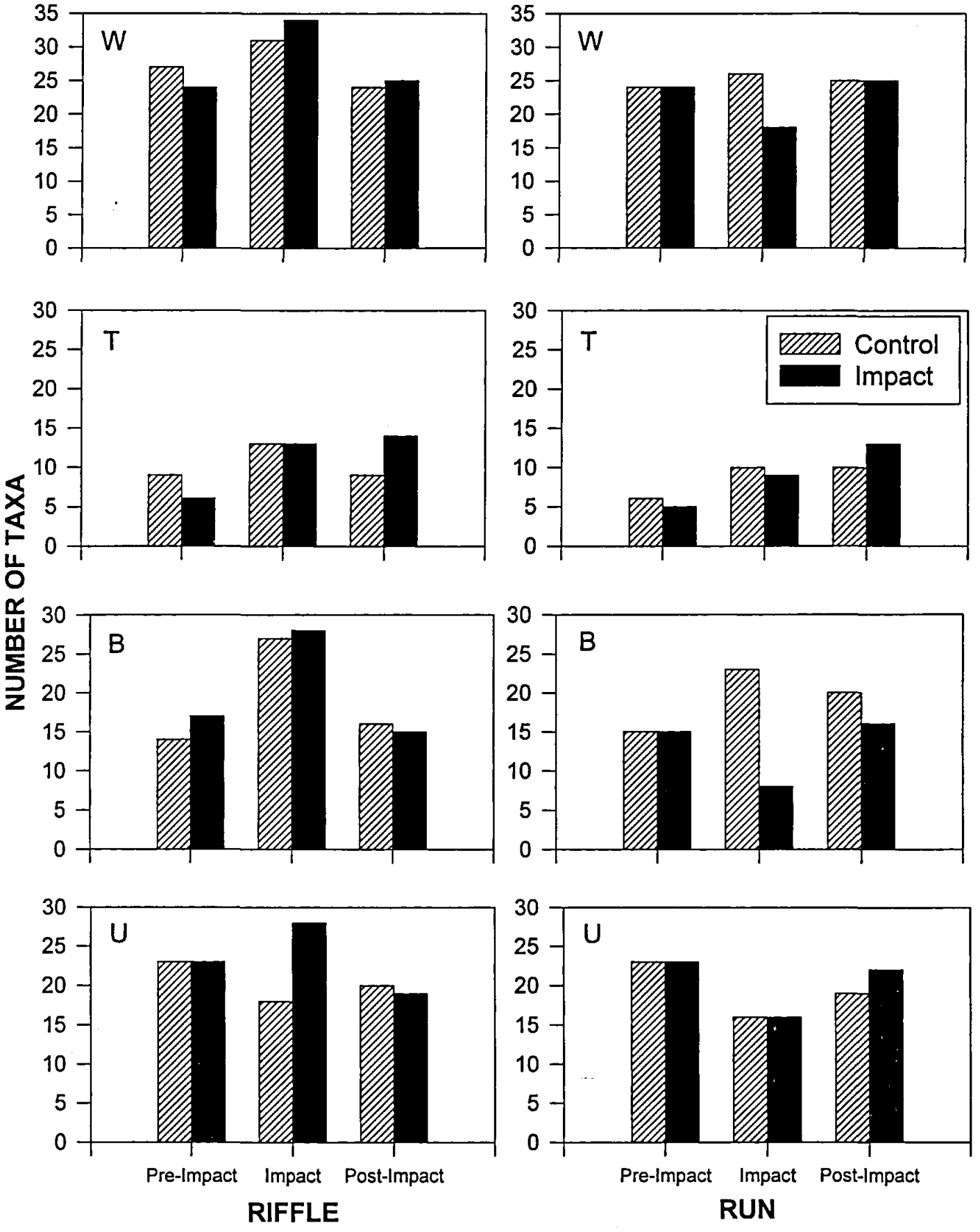


Figure 10.53 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

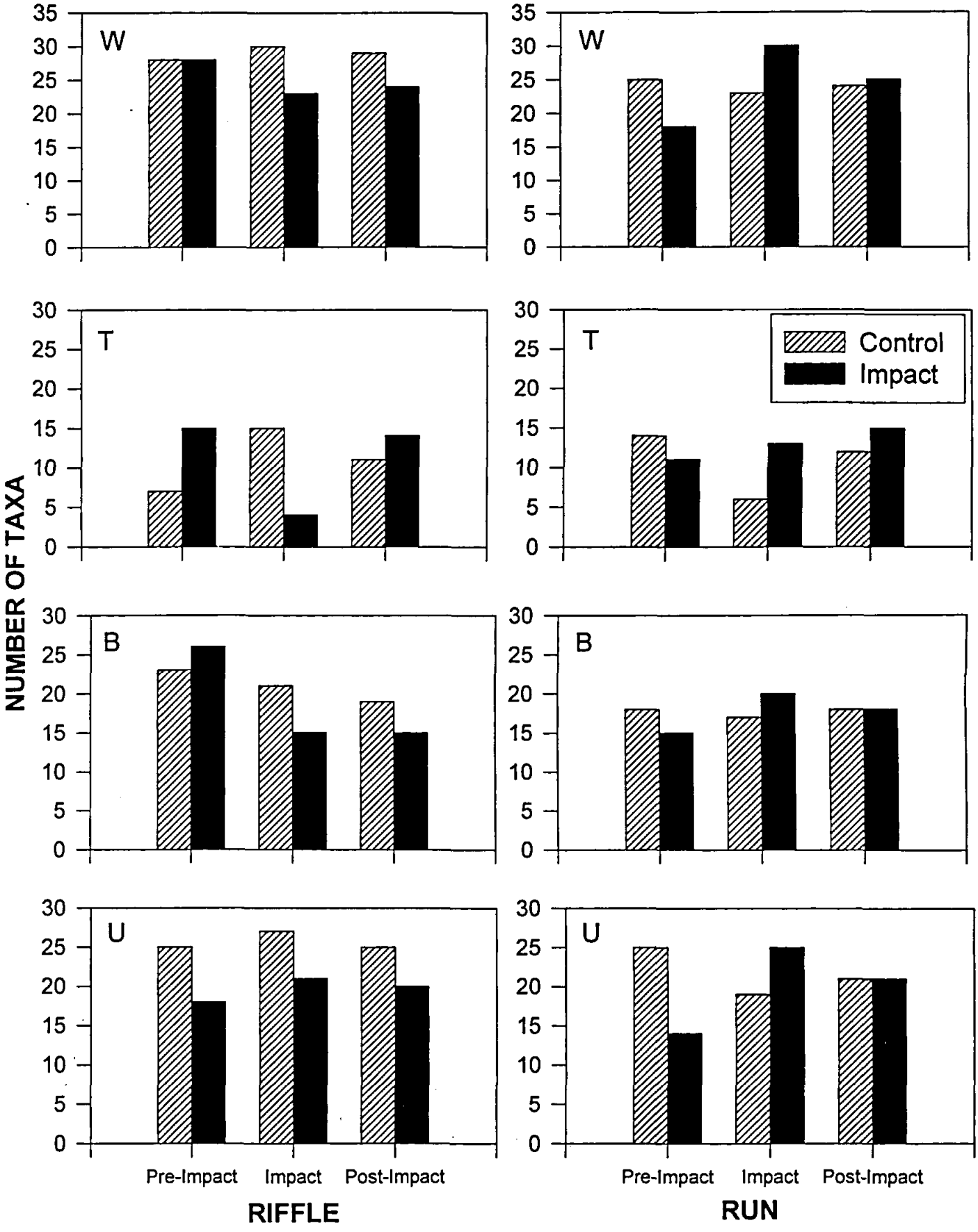


Figure 10.54 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

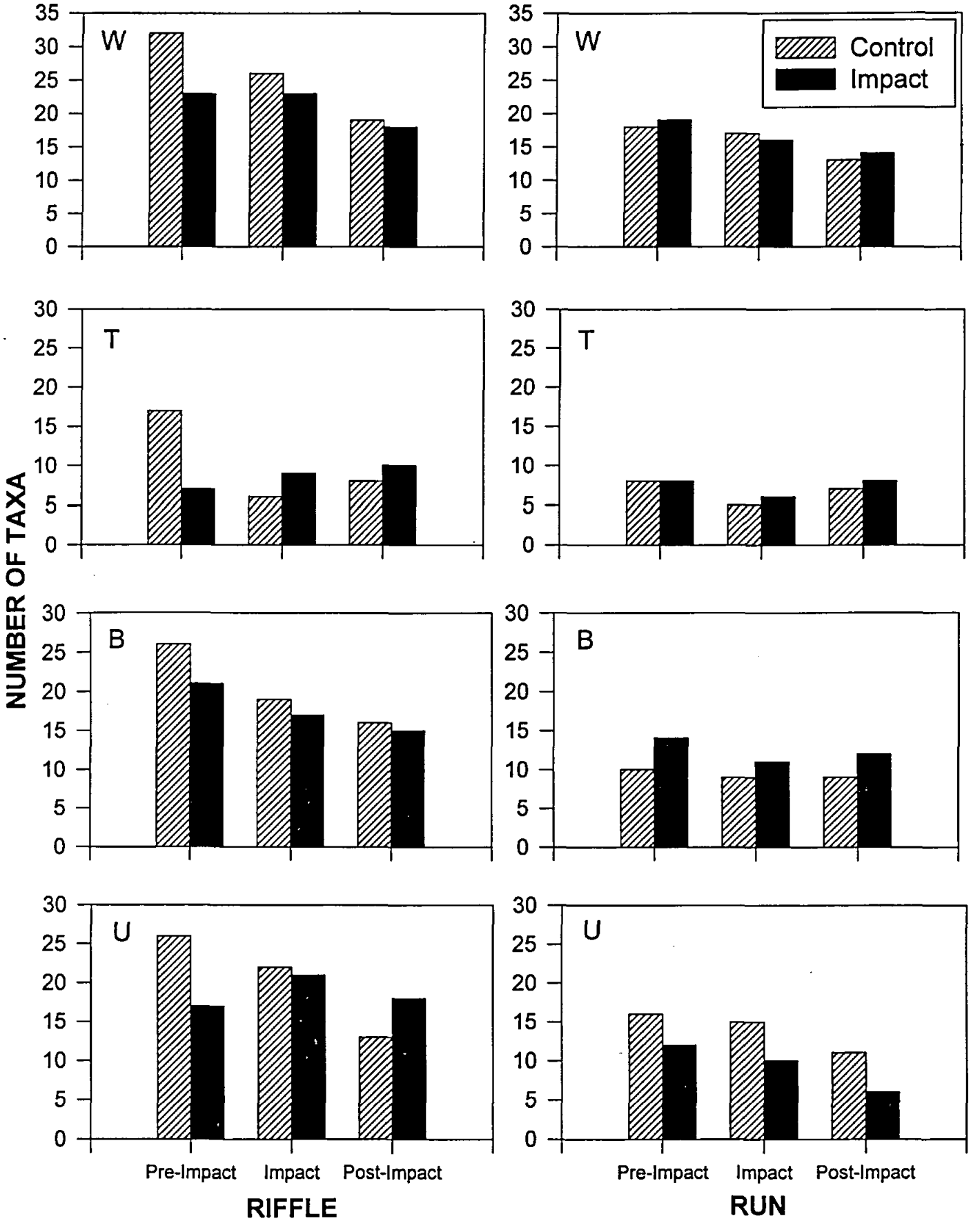


Figure 10.55 Numbers of families and higher taxa recorded from riffles and runs in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

Summary of the effects of natural and unnatural baseflow regimes on numbers of taxa

At the four sites, overall numbers of taxa were fairly stable over the range of natural baseflows, although some natural variability was apparent. Moreover, there was no consistent pattern among rivers of an overall increase or decrease in taxa with a decrease in baseflow magnitude. Under natural baseflow regimes, numbers of taxa were typically lowest at the end of summer or in early autumn (April). During February, with reductions in baseflow to levels far below natural magnitudes, the numbers of taxa increased in all three impacted rivers. However, there did not appear to be a direct relationship between the proportion of flow abstracted and the increase in number of taxa. After the reinstatement of natural baseflows, in April, the site at which the least baseflow had been abstracted, the Molenaars, showed complete recovery in numbers of taxa. However, for both the Du Toits and Riviersonderend impact locations, where flow reduction exceeded 86% of the natural magnitudes, numbers were still marginally elevated in April.

Among sites, under natural baseflow regimes, numbers of taxa were typically higher in riffles than in runs. The difference was sometimes slight, however, as in the case of the Molenaars River. Lowest numbers of taxa were recorded from the tops of riffle and run stones, while stone bottoms and the underlying stones, in both biotopes, exhibited similar, higher numbers of taxa.

With natural decreases in baseflow from December to February, there were no clear overall patterns in the number of taxa inhabiting riffles or runs. For the Elands site, there was no trend for either biotope, while increases in the numbers of taxa in riffles and runs were recorded for the Molenaars site. For the Du Toits site, numbers increased in riffles and decreased in runs, while numbers in both biotopes decreased in the Riviersonderend River. Changes in numbers of taxa among the various substratum components of each biotope with natural decreases in discharge, were highly varied for both riffle and run assemblages. However, for the underlying substratum in runs, there was a consistent decline in numbers with a decrease in flow. Responses in numbers of invertebrate taxa, associated with the shift to autumn instream conditions, varied among both sites and biotope components.

For the Molenaars and Riviersonderend sites, in February, with moderate and extreme baseflow reduction below natural levels, respectively, numbers of taxa increased in riffles but decreased in runs. The opposite response was observed in riffles and runs at the Du Toits site where the river was subjected to extreme low flows. Consistent changes in numbers of taxa were not evident for most substratum components. However, numbers of taxa in the underlying substratum of riffles increased in all impact locations. In addition, the tops of stones in runs showed increases in numbers of taxa though only at extreme low flows (*c.* 86% reduction of natural baseflow).

10.5.2 Changes in total benthic macroinvertebrate abundance in response to natural and unnatural dry-season baseflows

Effects of natural and unnatural baseflows on abundances at the four sites

The results of an assessment of the effects of a natural and an unnatural, dry-season baseflow regime on the total numbers of individuals of BMIs, standardised per unit area of 0.1 m^2, at each site and location, are given in Figure 10.56.

Effects of natural baseflows on abundances

The total number of individuals varied markedly among sites and with baseflow regime, under natural conditions (Figure 10.56). The Molenaars River had the highest number of individuals (3183-1019 0.1 m^{-2}), followed by the Elands and Du Toits, in order of decreasing abundances. The Riviersonderend River had considerably lower numbers than the other three sites, throughout the study period, ranging from 272-876 0.1 m^{-2}.

Numbers of individuals varied over the range of natural baseflows, at all four sites. For all sites except the Riviersonderend, there was a decrease in abundance from December to the lowest values in February, corresponding with low baseflow magnitudes (Figures 10.56 and 10.1). The relative decrease in numbers was lowest for the Du Toits site. The number of individuals increased at these three sites in April, with the various increases in baseflow from February magnitudes. The Du Toits control data for April showed a higher number of individuals in April than December, despite both months experiencing similar instantaneous discharges. This pattern differed from that for the Elands and Molenaars, although it might simply be a function of natural variability. For the Riviersonderend site, there was a natural small increase in abundance in February, with similar peak dry-season numbers to those of the Du Toits site. The lowest numbers of individuals at the Riviersonderend site was associated with the large increase in baseflow with the shift to autumn flow conditions in April, which was early relative to large scale baseflow increases at the other sites (Figure 10.1). Some natural variability in the numbers of individuals within each reach was evident for all four sites, on the basis of pre-impact data and the results for the control site.

Effects of unnaturally low baseflows on abundances

There was no obvious effect of unnaturally low flows in February at the Molenaars impact location, with a similar response in terms of abundance for both locations. However, after flow diversion ceased, April numbers of individuals were greater for the impact location. Statistical analysis is required to ascertain whether or not the response can be attributed to natural intra-reach variability (Tharme in prep.). This is necessary as full recovery to natural abundances was recorded for the other two impacted sites. Of the two impact locations at which the proportion of flow reduction was high, only the Du Toits location showed a clear response, with a major increase in the total number of individuals above natural levels. The number of individuals increased by 291%, from 835 0.1 m^{-2} in December to 3263 0.1 m^{-2} in February. In contrast, the natural invertebrate response was a decline in abundance (21%), from a December figure of 1148 0.1 m^{-2} to a February abundance of 907 0.1 m^{-2}. Numbers of individuals at the Riviersonderend location increased only marginally above natural levels during the impact phase, so there was no marked response to unnaturally low baseflows (Figure 10.56).

Effects of natural baseflows on abundances in riffle and run biotopes

The numbers of individuals (per 0.1 m^2) recorded for riffles and runs, including abundances on the tops and bottoms of stones, and on the surface of the underlying substratum, are given in Figures 10.57-10.60, for the four sites.

Elands site

For the Elands (control) site, from December to April, although there was an extremely high degree of variability and a lack of consistency in patterns between the two locations for both riffles and runs, there was a consistent decrease in the total number of individuals at naturally lowest baseflows in February

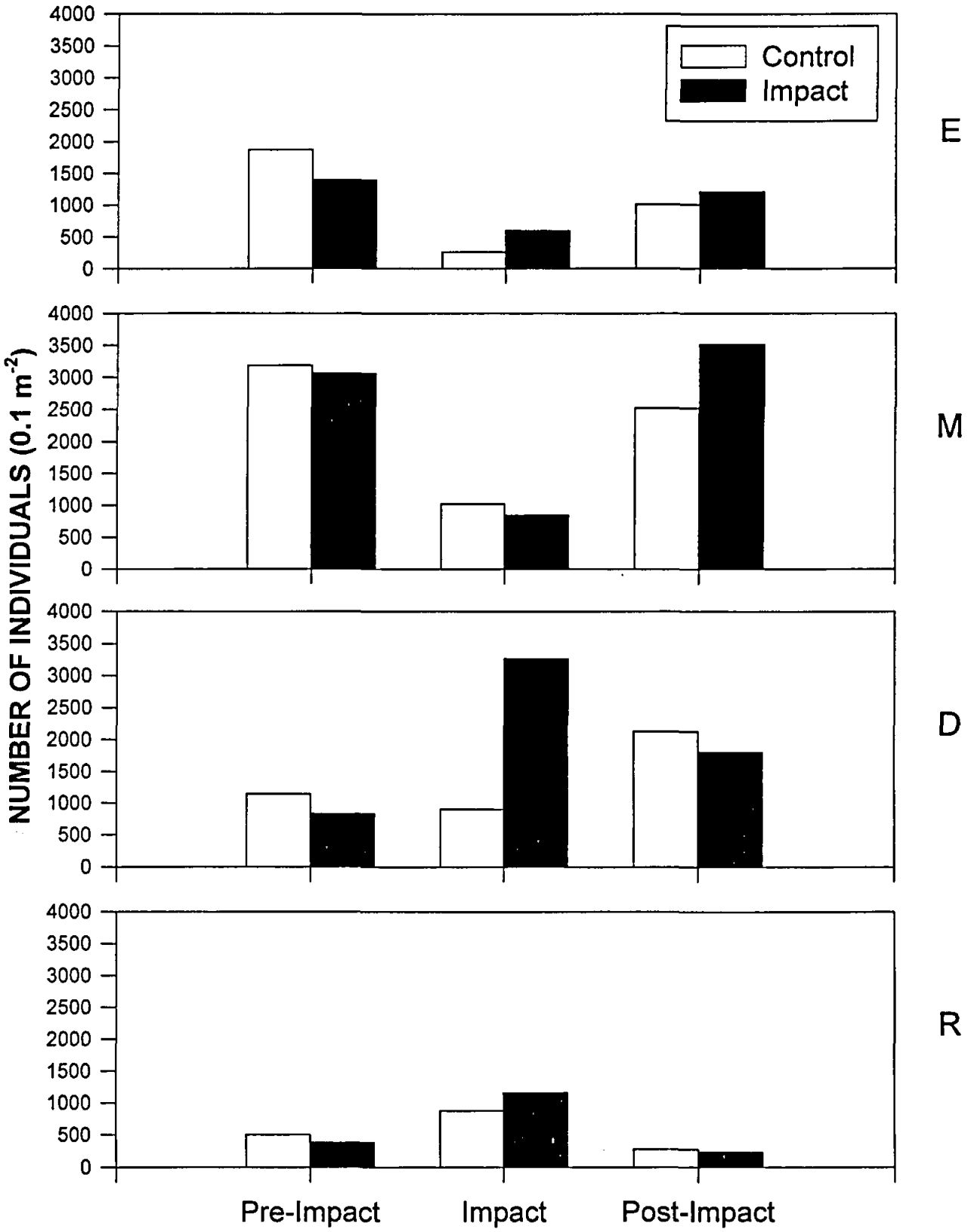


Figure 10.56 Total numbers of individuals (0.1 m^{-2}) recorded in the control and impact locations at the four sites, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. E – Elands; M – Molenaars; D – Du Toits; R – Riviersonderend.

(Figure 10.57). Moreover, numbers of individuals in each biotope were similarly low for all parts of the substratum in the peak of the dry season. Typically, for both biotopes, numbers increased again in April, though either to a greater or lower extent than December values. However, in the impact location, for the underlying substratum of runs, there was a further decrease in numbers in April. This decrease was sufficiently large that it affected the overall trend in abundance for runs (Figure 10.57 W). From the data on the distribution of individuals among the various parts of the substratum, it was clear that the tops of both riffle and run stones had the lowest abundances, while the numbers of individuals were similarly higher for stone bottoms and underlying substratum. Runs had consistently lower numbers of individuals than riffles. From combined data from both locations, the range in abundance for riffles was from 332 (stone tops, February)-3870 individuals 0.1 m^{-2} (underneath stones, December). In contrast, runs showed a very low minimum of 36 individuals 0.1 m^{-2} (stone tops, February) and a maximum of 2315 individuals 0.1 m^{-2} (underneath stones, December).

Molenaars site

At the Molenaars site, there was also considerable natural variability within the reach, as shown by the relative differences between the numbers of individuals in the two locations before the flow impact (Figure 10.58). This variability was most pronounced for samples taken from the substratum underneath stones (Section 9.5.3). Typically, for both biotopes, numbers naturally decreased from December to midsummer minima. The decrease appeared greater for runs than riffles. Numbers then increased again in April, generally to lower values than for December. However, for the tops of riffle and run stones, numbers were greater in April than December. The only recorded increase in the numbers of individuals from December values was for the tops of riffle stones in February. The same trend was not evident for runs. Examination of the total numbers of individuals from the different parts of the substratum showed that the tops of both riffle and run stones had the lowest abundances, while the numbers of individuals were similarly higher for stone bottoms and underlying substratum. Riffle abundances ranged from a minimum of 544 0.1 m^{-2} (stone tops, December) to a maximum of 5751 0.1 m^{-2} (stone bottoms, December) individuals. Runs had consistently lower numbers of individuals than riffles, exhibiting a range of total number of individuals of 137 0.1 m^{-2} (stone tops, February)-5749 0.1 m^{-2} (underneath stones, December).

Du Toits site

Under conditions of natural baseflow at the Du Toits control location, there were somewhat different responses in terms of numbers of individuals during the study period, for riffle and run biotopes (Figure 10.59). Overall, there was a small decrease in abundances from December to February, with a subsequent marked increase to peak numbers in April. This pattern was reflected for data from the bottoms of the stones, and to a lesser degree for the underlying substratum, which was more stable in numbers. Riffle assemblages followed the overall pattern, although there was a slight increase in numbers on stone tops in February. For runs, abundances increased slightly from December to the peak of the dry season, with another slight increase from February to April. This pattern was evident for both the tops and bottoms of stones, while, for the underlying substratum, the numbers of individuals declined slightly in February compared to December figures (Figure 10.59).

Riviersonderend site

Under conditions of natural baseflow, there was an increase in numbers of individuals from December to February, for both riffles and runs at the Riviersonderend site (Figure 10.60). This was followed by a

consistent natural decrease to lower abundances in April than December for riffles, with the autumn increase in baseflow (Figure 10.1). For runs, the pattern differed in that April abundances were marginally higher than those recorded for December.

Effects of unnatural baseflows on abundances in riffle and run biotopes

Molenaars site

The data for the Molenaars impact location revealed an overall decrease in the total number of individuals with unnaturally low flows, for riffles (Figure 10.58). Recovery to near-natural numbers occurred post impact. The decrease occurred entirely in the underlying substratum, with a change from 5577 individuals 0.1 m^{-2} in December to only 1572 individuals 0.1 m^{-2} during the impact phase, while the corresponding natural change was a decline from 2360-1260 individuals 0.1 m^{-2}. The tops of impacted riffle stones had similarly low numbers of individuals both before and during unnatural flow reduction, in contrast to the natural pattern (see above). There was no obvious difference in trends in numbers for bottoms of riffle stones under conditions of natural and unnatural baseflow, throughout the study period. For runs, with an unnatural reduction in flow, there was a decrease in overall numbers from December figures, as under conditions of naturally low baseflow, but February numbers were greater than control figures. As for riffles, the observed change in runs took place in the substratum underneath the sample stones, with a decrease from 3691 individuals 0.1 m^{-2} (pre-impact) to 785 individuals 0.1 m^{-2} (February), compared with 5749-176 individuals 0.1 m^{-2} for the control location. That is, the trend was the same as the natural one, but to a lesser degree. Numbers were still elevated above control values after the impact was removed.

Du Toits site

With *c.* 86% reduction in flow at the Du Toits site, there was a clear response by the BMIs of riffle biotopes, and a lesser response for runs (Figure 10.59). While numbers naturally decreased from December to February under natural conditions in riffles, a major increase in abundance was recorded at the impact location. The increase occurred for all parts of the substratum, but was particularly marked for stone tops with a 664% difference between control and impact abundances. The natural number of individuals was 1070 0.1 m^{-2}, while 8176 individuals 0.1 m^{-2} were collected from impacted stones. Recovery data conflicted with these results, with lower numbers than natural for stone tops and still elevated numbers for the underlying substratum. The small increase in numbers of individuals (301-1330 individuals 0.1 m^{-2}) recorded for impacted runs, compared with a natural decrease from 879-610 individuals 0.1 m^{-2}, could be attributed entirely to a response by BMIs inhabiting the underlying substratum. For this sample component, numbers were still elevated after natural baseflows were reinstated.

Riviersonderend site

With a greater than 86% reduction in discharge in the Riviersonderend impact location, there was a clear response by the assemblages of riffle biotopes, and a lesser response for runs (Figure 10.60). There was a pronounced increase in the number of individuals in all substratum components of riffles, from December to February. The greatest increase was observed in the underlying substratum, from 526-3301 individuals 0.1 m^{-2}, in contrast with a smaller increase from 1334-1569 individuals 0.1 m^{-2} for the control location. Recovery appeared complete in April. Impacted runs showed the same trends as control runs for all parts of the substratum, but the increases were relatively low. Overall, recovery appeared to occur, although numbers were still low at the impact location when compared with the pre-impact ratio for the two locations.

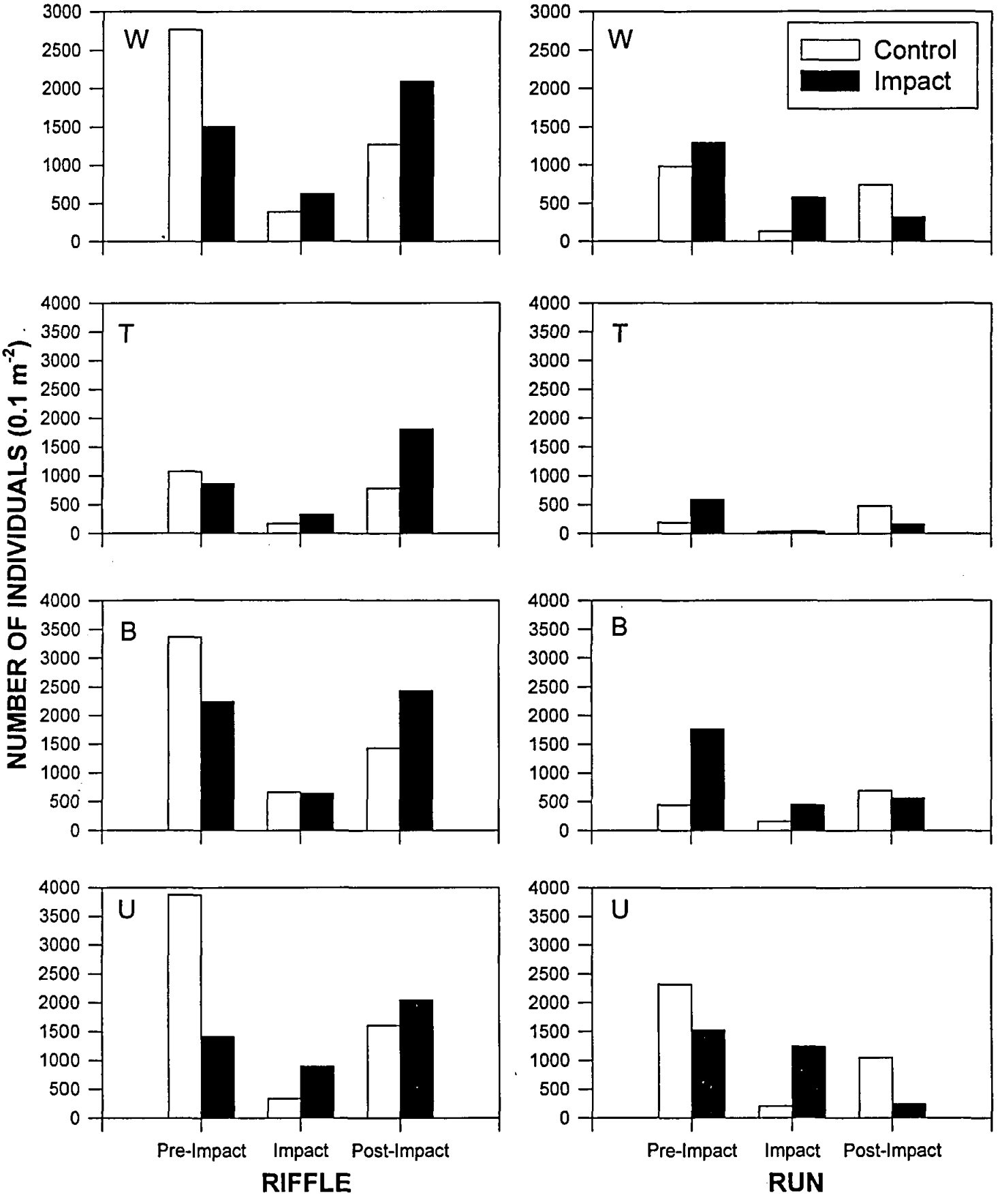


Figure 10.57 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Elands site, at natural baseflows. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

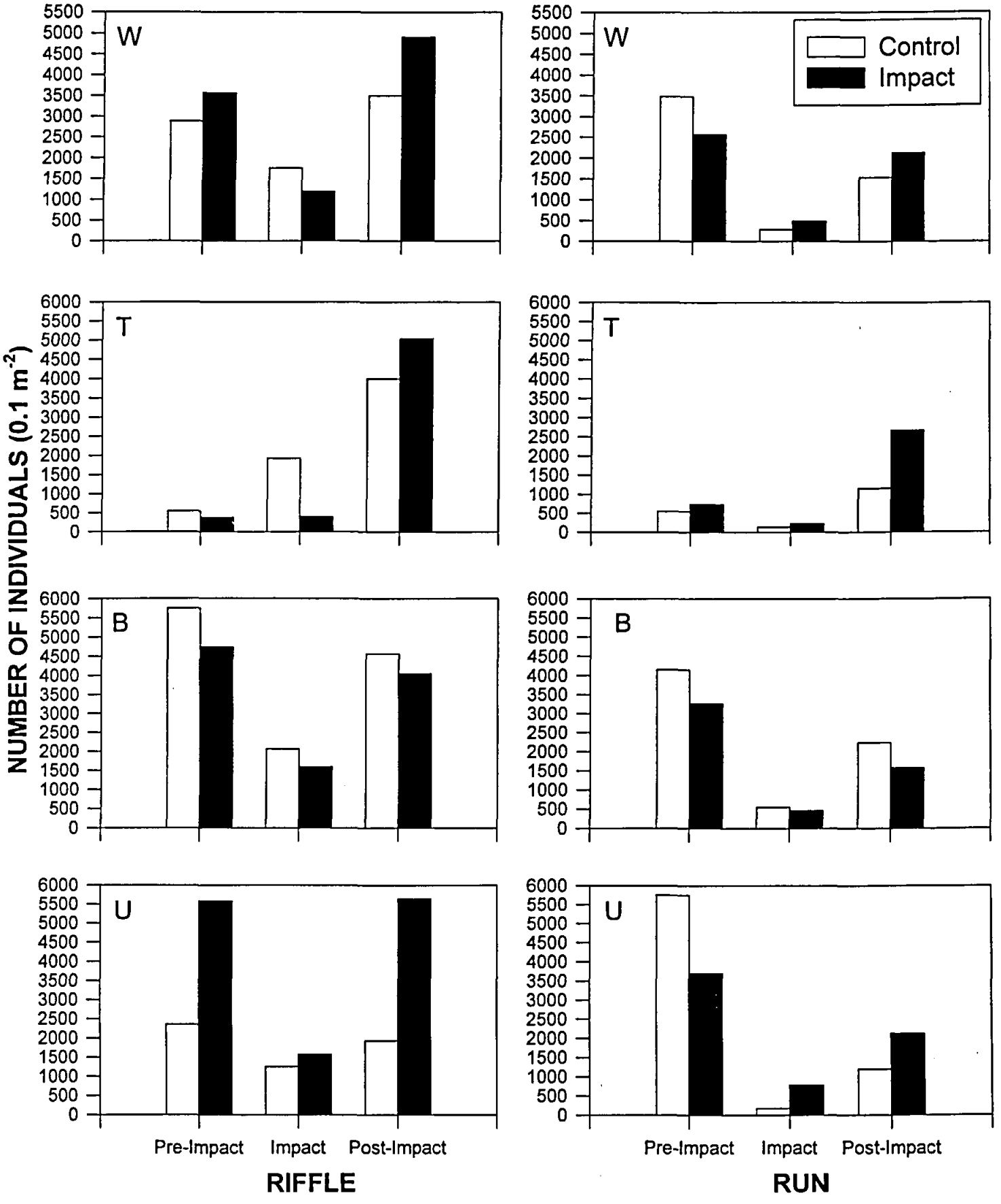


Figure 10.58 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

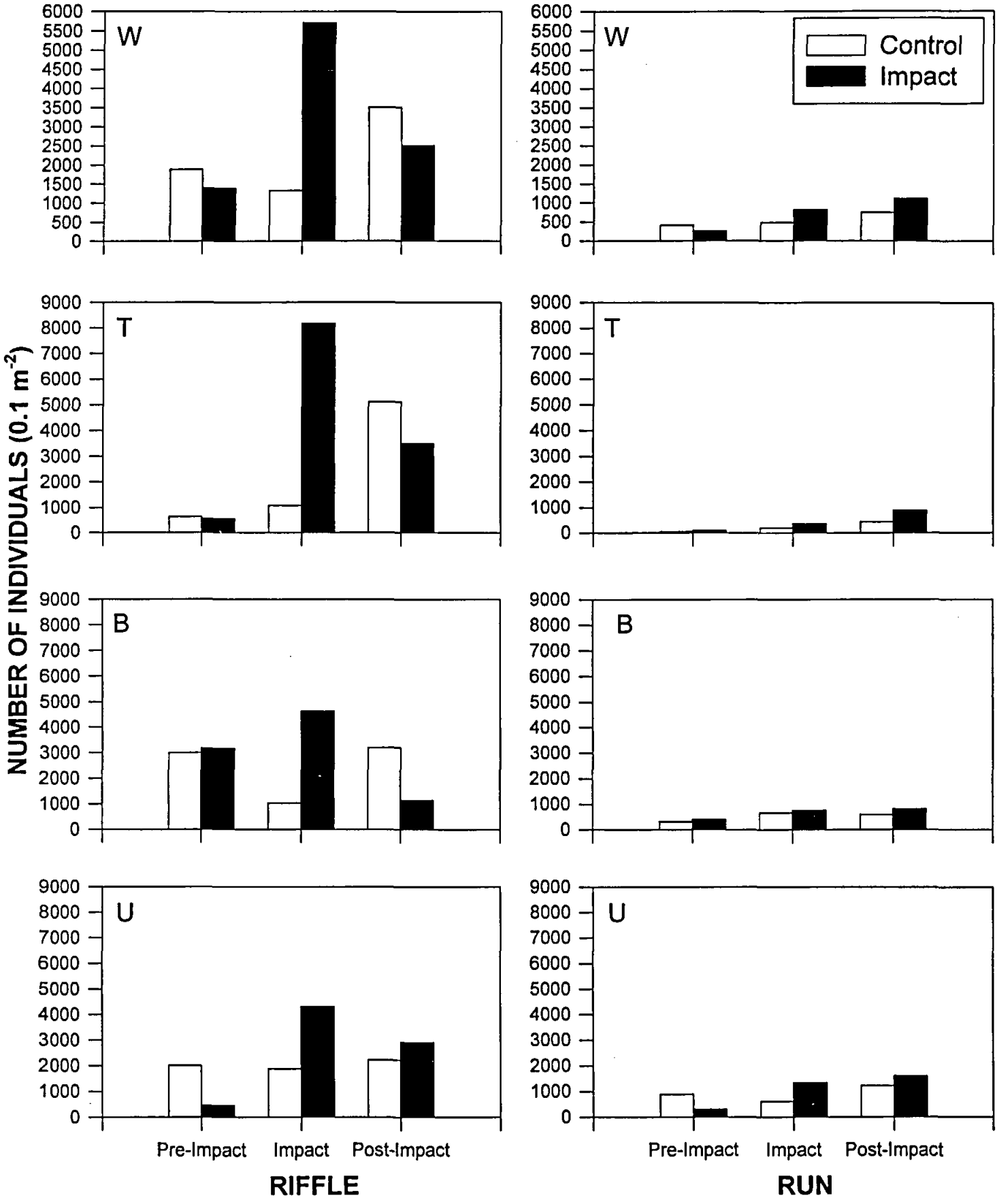


Figure 10.59 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

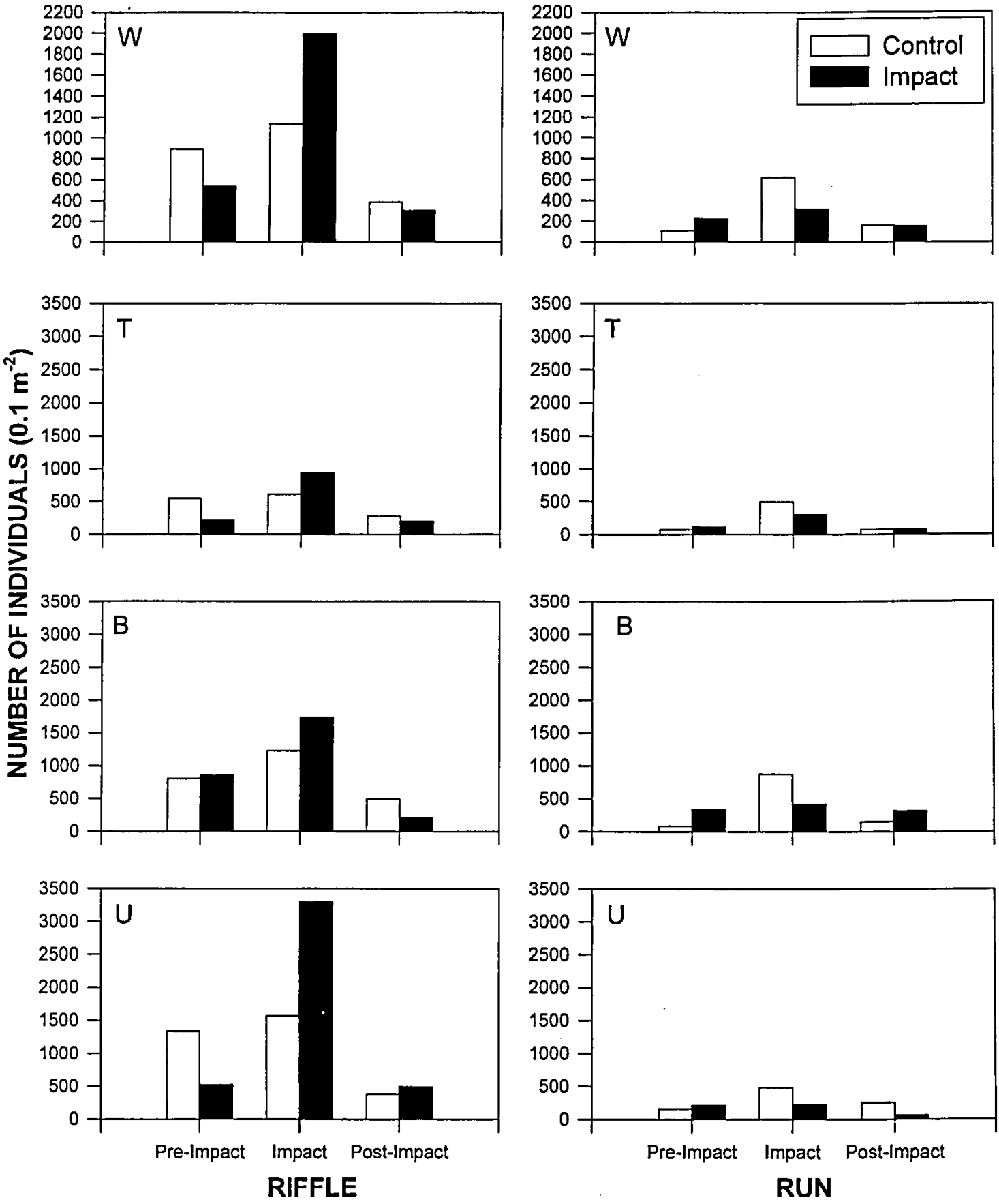


Figure 10.60 Total numbers of individuals (0.1 m^{-2}) recorded from riffles and runs in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

Summary of the effects of natural and unnatural baseflow regimes on numbers of individuals

Under natural conditions, the total number of individuals varied markedly among sites, within each reach and with baseflow regime. For all sites except the Riviersonderend, there was a decrease in abundance from December to lowest numbers of individuals overall, in February. This decrease corresponded with the naturally low baseflow magnitudes in this month. The Riviersonderend site had considerably lower numbers of individuals than the other three sites throughout the low-flow study, and showed a natural, small increase in abundance in February. The lowest numbers of individuals at this site were associated with elevated baseflows in autumn.

There was no obvious effect of a 36% reduction in baseflow, in February, on the overall numbers of individuals in the Molenaars impact location. However, after flow diversion ceased, April numbers of individuals were greater for the impact than the control location. Of the two impact locations at which the proportion of flow reduction was considerably higher, at about 86%, only the Du Toits location showed a marked response. At this site, there was an increase in the total number of individuals by 291% above natural levels, in contrast with the natural response of a decline of 21% in abundance. In the Riviersonderend impact location, numbers increased only marginally above natural levels during the impact phase. For both sites, full recovery to natural abundances was recorded in April.

For all sites, there was a high degree of variability in abundances in riffle and run biotopes under natural baseflow regimes. The tops of both riffle and run stones characteristically had the lowest numbers of individuals, while the numbers of individuals were similarly higher for stone bottoms and the underlying substratum. Runs had consistently fewer individuals than riffles.

Under natural baseflow regimes, all sites except the Riviersonderend showed a decrease in the numbers of individuals in the BMI assemblages of riffles and runs, from December to midsummer minima in February. Typically, again for all sites except the Riviersonderend, numbers increased again in April. Within the various substratum components of the two biotopes, there were some divergences from this characteristic pattern. Firstly, at the Molenaars site, a natural increase in the numbers of individuals from December to February was recorded for the tops of riffle stones. The same trend, though slight, occurred for the overall assemblage inhabiting the tops of stones in the Du Toits River. In addition, at the Du Toits site, in runs, overall abundances increased slightly from December to the peak of the dry season, except in the underlying substratum. At the Riviersonderend site, BMI assemblages of riffles and runs exhibited opposite trends to those at the other three sites, under conditions of natural baseflow. Specifically, there were increases in numbers of individuals from December to February, overall and for riffles and runs. Further, there was a consistent natural decrease to lower abundances in April.

Assessment of the effects of unnatural baseflows on abundances in riffle and run biotopes showed different patterns according to the percentage flow reduction below natural magnitudes. In the Molenaars impact location, for riffles and runs, there was an overall decrease in the total number of individuals with 36% reduction in baseflow in February, where the decrease occurred primarily in the underlying substratum. Hence, the general direction of change in abundance was the same as the natural one. Recovery to near-natural numbers occurred post impact. With *c.* 86% reduction in flow, there were distinct responses by the BMI assemblages of riffle biotopes, and lesser responses for runs. For both the Du Toits and Riviersonderend sites, from December to February, a major increase in numbers of individuals was recorded

in riffles for all parts of the substratum. In the Du Toits River, the increase was particularly marked for stone tops with a 664% difference between control and impact abundances. The greatest increase in the Riviersonderend impact location was observed in the underlying substratum of impacted riffles. For both sites, there were only small increases in numbers of individuals recorded for impacted runs. In the Du Toits River, the increase could be attributed entirely to a response by BMIs inhabiting the underlying substratum, while all substratum components showed increases in abundance in the Riviersonderend impact location. For both riffles and runs, recovery appeared to occur by April for the Riviersonderend, but not entirely for the Du Toits impact location.

10.5.3 Changes in taxon dominance in benthic macroinvertebrate assemblages in response to natural and unnatural dry-season baseflows

Taxonomic dominance was assessed by depicting the BMI families or higher taxa that contributed 5% or greater to the total assemblage, on the basis of abundances per standard unit area of 0.1 m^2. The threshold of ≥ 5% was selected on the basis that the majority of taxa each represented less than 5% of the assemblage, while only a small number of taxa were responsible for the greatest number of individuals. Dominant taxa are represented graphically, using taxon-specific patterns and shadings. The remaining taxa (i.e. those taxa < 5% of the overall assemblage) are grouped together as a single percentage, and are referred to as "other taxa". Larvae and adults are specified when a particular taxon can include members of both life history stages. In the other instances, the predominant aquatic life stage pertains. For example, Leptophlebiidae represents only nymphs. Only the major trends are noted below, and the extent to which such trends represent significant changes in assemblage dominance will be explored by statistical means, once further BMI data have been included in the analysis (Tharme in prep.).

Effects of natural and unnatural baseflows on overall benthic macroinvertebrate taxonomic dominance for the four sites

The dominant taxa for each location, for the four sites, are illustrated in Figure 10.61, for the three phases of the low-flow study. Each site is dealt with independently below, due to the complexity of the responses in dominance that were observed with differences in baseflow regime.

Elands site

Data on dominant taxa at the Elands, control, site (Figure 10.61E) showed clearly that there is a high natural spatial and temporal variability in assemblage composition in a single reach, over a dry season and into early autumn. Such variability under conditions of natural baseflow renders the assessment of responses to unnaturally low baseflows more difficult (see below). Over the duration of the low-flow study, only Baetidae and Chironomidae were always present, in both locations, as dominant taxa, and in varying proportions. Furthermore, in December and February, there were additional dominant taxa in the assemblage in each location. However, the same three taxa, namely Baetidae, Ephemerellidae and Chironomidae dominated the assemblages at both locations in April. At naturally low baseflows in the Elands, in peak summer, there was a marked shift in dominance, as Ephemerellidae increased to dominate the overall invertebrate assemblage. In contrast, Baetidae decreased in prevalence, relative to both December and April figures. Chironomidae attained greatest dominance in early autumn.

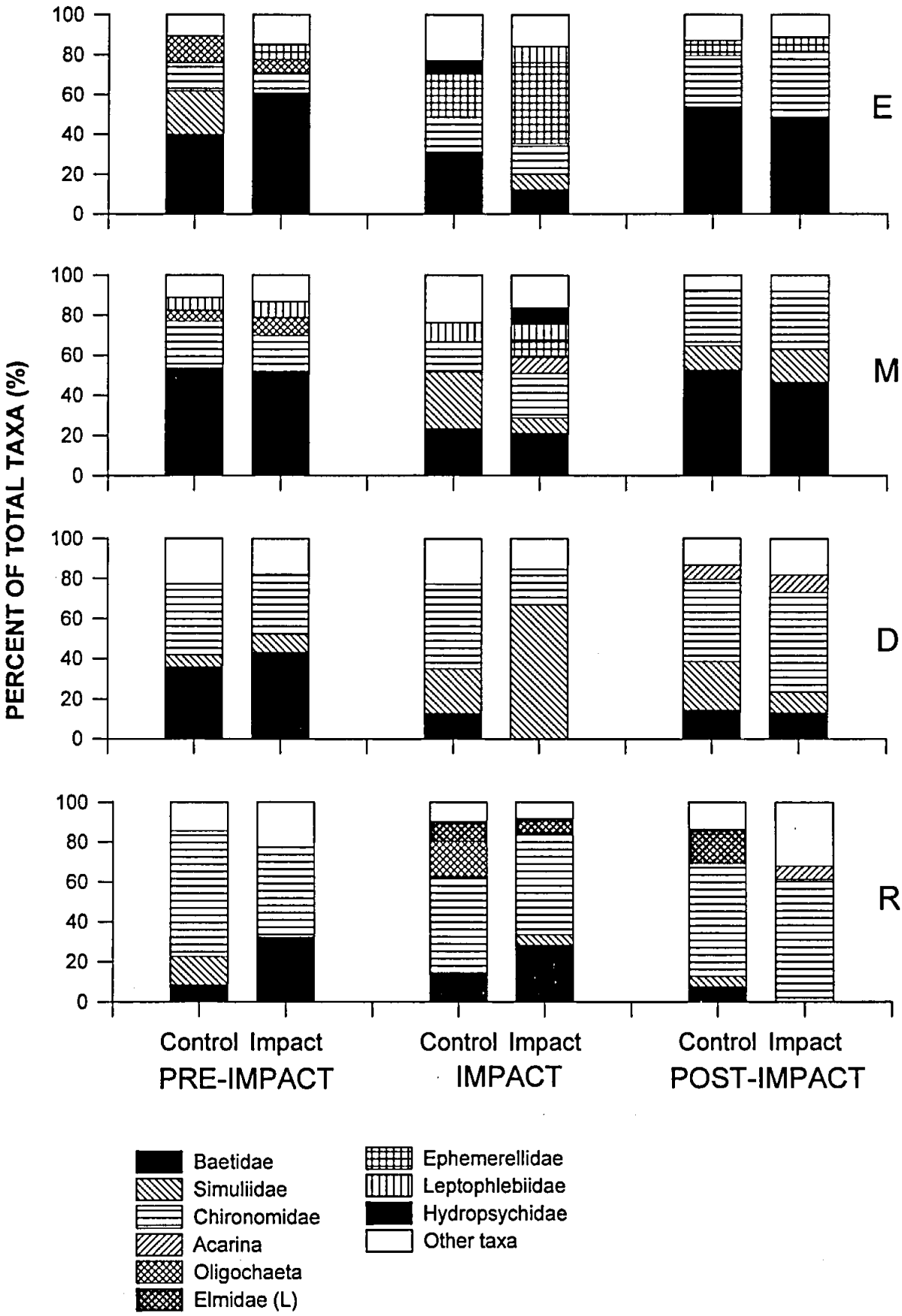


Figure 10.61 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) in the control and impact locations at the four sites, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. E – Elands; M – Molenaars; D – Du Toits; R – Riviersonderend. L – larvae.

Molenaars site

At the Molenaars, the dominant taxa consistently present under conditions of natural dry-season baseflow (control location) were baetids and chironomids (Figure 10.61M). During both the pre-impact and post-impact phases of the study, there were no major differences in the dominant taxa between the control and impact locations, with an increase in chironomids in both instances. The latter result indicates that any impacts of unnaturally low flow on overall assemblage dominance did not continue after natural baseflows were reinstated. In early summer (December), at natural flows, Leptophlebiidae and oligochaetes were abundant. In contrast, at peak low flows, simuliids became one of the dominant taxa; baetids decreased in abundance and oligochaetes were no longer prominent.

With 36% flow reduction at the impact location, more taxa were present at abundances exceeding 5% than at the control location, namely Hydropsychidae, Ephemerellidae and acarinids. In addition, simuliids were present at a lower percentage of the total, at 8% 0.1 m^{-2}, compared with 28% 0.1 m^{-2} under natural conditions.

Du Toits site

At the Du Toits site, under natural, summer flow conditions, baetids, simuliids and chironomids were consistently present in relatively high proportions (Figure 10.61D). Both locations showed similar taxon proportions during the pre-impact phase, and basically similar percentage contributions by taxa after the impact was removed. In the latter instance, the order Acarina gained prominence at both locations. Under naturally low flows in February, the same taxa dominated the assemblage as in December although percentages altered, with a decrease in baetids and a concomitant increase in percentage contributions by simuliids, chironomids and other taxa (as a general group).

With a high percentage flow reduction in the impact location, in February, there was a pronounced increase in dominance by Simuliidae, which contributed 67% 0.1 m^{-2} to the overall assemblage as compared with 22% 0.1 m^{-2} under control conditions. Chironomidae decreased in abundance, at 17% 0.1 m^{-2}, while this taxon represented 42% 0.1 m^{-2} of the BMI assemblage at natural low flows. Baetidae, representing 13% 0.1 m^{-2} of the assemblage under natural conditions, were absent as a dominant taxon in the impact location, at only 2.5% 0.1 m^{-2}.

Riviersonderend site

For the Riviersonderend, only chironomids and baetids were naturally dominant throughout the low-flow study (Figure 10.61R). There was some natural variability in assemblage composition in December, with Simuliidae present as a dominant taxon in only the control location, as well as differences in the relative percentage contributions of taxa in each location. Under conditions of naturally low baseflows (February) simuliids were present in relatively low abundances, while Elmidae larvae and oligochaetes attained dominance.

There were no major shifts in dominance with flow reduction to unnaturally low magnitudes, but Simuliidae gained prominence. In the recovery phase, there were several differences between the previously impacted and control locations. Of the four taxa dominant at the control location, only Chironomidae were also present, as a similarly high percentage, in the impact location. Baetidae, a dominant family under natural conditions, represented a small percentage of the impact assemblage, at 4.3% 0.1 m^{-2}. Acarina were present

only in the impact location, and a high proportion of the BMI assemblage (32% 0.1 m^{-2}) comprised a variety of taxa present at percentages lower than 5% 0.1 m^{-2}.

Changes in the taxonomic dominance of riffle assemblages in response to natural and unnatural dry-season baseflows

Examination of the data on percentage contributions of various taxa to the BMI assemblages of riffles and their substratum components (tops, bottoms and underlying substratum elements), in relation to natural and unnatural baseflows, showed the following main trends (Figures 10.62-10.65). Each site is discussed independently, due to the site-specific nature of many of the observed changes in assemblage dominance.

Elands site

Riffle data for the Elands site, showed a high degree of intra-reach variability for all substratum components in both dominant taxa and their relative percentages (Figure 10.62). The only families dominant for all riffle components were Baetidae and Chironomidae. Fewer taxa, namely baetids, chironomids and ephemerellid mayflies, dominated the riffle assemblage in early autumn (April) than in summer. Moreover, the assemblage was most consistent in terms of the most common taxa in this month. It was apparent from comparison of the data from the two locations throughout the study period that several taxa, notably Simuliidae, had a patchy distribution, as they were often present in large numbers in one location and in low proportions in the other. For the overall riffle assemblage, in early summer (December), the most common taxa for both locations were similar, but simuliids and Ephemerellidae were only prominent in one location. With a natural decrease in baseflow in February, Baetidae decreased in their percentage contribution to the riffle assemblage, while Chironomidae and particularly Ephemerellidae increased in dominance. Water mites, Hydropsychidae, Leptophlebiidae and simuliids were also dominant taxa at this time of year, though not in both locations. There was no strongly consistent pattern in dominant taxa in relation to changes in natural baseflow, or between the two locations.

The tops of riffle stones were inhabited by the broadest number of common taxa in December, with a particularly high number of simuliids in the control location (Figure 10.62T). In February and April, Baetidae was the most abundant family on stone tops, followed by chironomids and then simuliids (February). Throughout the study period, there was only a small proportion of the assemblage that was represented by taxa at percentages lower than 5% 0.1 m^{-2}. The bottoms of stones in riffles exhibited the greatest number of dominant taxa in the peak dry season, when baetids declined and ephemerellids increased in abundance (Figure 10.62B). Acarinids and Hydropsychidae were also abundant, though not consistently in both locations. The underlying riffle substratum varied greatly in the composition of most common taxa, in summer, and Elmidae (larvae) and juvenile mayflies were only common in this substratum component, in February and December, respectively (Figure 10.62U). Ephemerellids and chironomids comprised the greatest proportion of this BMI assemblage at the time of lowest baseflow, while baetids also decreased in this substratum component.

Molenaars site

Riffle assemblages at the Molenaars site were naturally fairly variable with regards dominant taxa, mostly during the peak dry season and to a lesser extent in early summer (Figure 10.63). In contrast, in April, the assemblage had the same dominant families for all riffle components, namely Baetidae, Simuliidae and Chironomidae. Furthermore, both the control and impact locations had remarkably similar proportions of

these families in this month, indicating that the impacted assemblage had completely recovered, specifically in terms of taxon dominance, after the reinstatement of natural baseflows. For the entire riffle assemblage, in December, the most common families for both locations were Baetidae and Chironomidae, respectively (Figure 10.63W). At the impact location, at this time, oligochaetes and Leptophlebiidae were also common. Both locations shared the following common taxa in February, despite unnatural low baseflows in the impact location: Baetidae, Simuliidae, Chironomidae, Acarina and Letophlebiidae. Baetidae had decreased in their percentage contribution to the assemblage in both instances, compared with December values. The main changes in dominance apparent at the impact location were a relatively lower percentage contribution to the assemblage by simuliids, 11% 0.1 m^{-2} compared with 33% 0.1 m^{-2} under natural flows, and the presence of Hydropsychidae (12% 0.1 m^{-2}). The latter taxon was dominant on the stone bottoms and particularly in the underlying substratum of riffles (see below).

For the tops of Molenaars riffle stones, Baetidae were naturally highly dominant in the pre-impact phase (77% 0.1 m^{-2} and 85% 0.1 m^{-2} in the control and impact locations, respectively), followed by chironomids (Figure 10.63T). Simuliids were only present at one location in early summer. The same three families remained dominant throughout the rest of summer and early autumn. During February, with a reduction in baseflow by 36%, baetids were more abundant in the impact location (45% 0.1 m^{-2}) than under natural baseflows (14%), while the converse trend was apparent for simuliids (34% 0.1 m^{-2} versus 65% 0.1 m^{-2} in the control location). Stone bottoms from both locations showed a similar taxon dominance before and after flow reduction (Figure 10.63B). The only clear difference in taxon dominance between the control and impact locations during the impact phase was the presence of Leptophlebiidae in the former location. Differences in the percentage contributions of taxa common to both locations, for instance Hydropsychidae and Acarina, were small. Before experimental flow reduction, taxon dominance in the underlying substratum was similar for both locations, with the sole difference being the presence of Leptophlebiidae in the impact one (Figure 10.63U). For the two locations, the greatest variability in common taxa and their relative contributions to the assemblage in the underlying cobbles/gravel was observed during the impact phase. Limnichidae were only dominant in this component of riffles. Six taxa were common under conditions of natural flow, with simuliids and ephemerellids only present at this location. With unnaturally low baseflows, a large contribution to the assemblage of 18% was made by hydropsychid trichopterans, simuliids were not dominant, and there were fewer Leptophlebiidae than were naturally present. Additionally, the contribution of taxa at percentages below 5% 0.1 m^{-2} was greatest for this location in this month.

Du Toits site

Under natural conditions, riffles assemblages at the Du Toits site were dominated by Baetidae, Chironomidae and, for some substratum components, Simuliidae (Figure 10.64). Natural variability in taxon dominance was moderately low at this site, for both the pre- and post-impact experimental phases. For the overall riffle assemblage, baetids, simuliids and chironomids predominated in early summer, while water mites were also present in high abundances in early autumn in all riffle components (Figure 10.64W). Under conditions of natural baseflow in February, oligochaetes contributed to the dominant taxa, largely as a result of contributions from stone bottoms and the underlying substratum (Figure 10.64). The other three common taxa were chironomids, simuliids and baetids. Taxon dominance with unnaturally low flows was confined to chironomids and simuliids, with the latter contributing a considerable 77% 0.1 m^{-2} to the assemblage (compared with a natural figure of 26% 0.1 m^{-2}). Chironomids represented a smaller percentage

of the impacted assemblage, at 16%, than at the control location (38% 0.1 m^{-2}). In addition, baetids were absent as a dominant taxon in the impact location, while representing 16% 0.1 m^{-2} of the natural riffle assemblage (Figure 10.64W).

Under conditions of natural baseflow, stone tops comprised predominantly Baetidae in December (>76% 0.1 m^{-2} of the assemblage). This taxon was followed by Chironomidae and simuliids, though only in one location (Figure 10.64T). At naturally low flows, there was a decrease in baetid numbers, a small increase in chironomids and an overall dominance by Simuliidae. At the impact location, with unnaturally low baseflows, baetids decreased dramatically in proportion to only 1.1% 0.1 m^{-2}. In contrast, there was a major increase in simuliids, to 92% 0.1 m^{-2}, compared with 51% 0.1 m^{-2} for the control location. Chironomids were the second most common family. This pattern of dominance in the impacted assemblage persisted in all riffle components (Figure 10.64T, B and U). At both locations, for riffle stones, there was little contribution by other taxa to the composition of the BMI assemblage. For stone bottoms, at natural baseflows, there was a decrease in baetid numbers from December to February, and an increase to >5% 0.1 m^{-2} by both Acarina and oligochaetes (Figure 10.64B). In April, at natural baseflows, Hydroptilidae were common, along with the consistently present baetids, chironomids and simuliids and acarinids. The underlying substratum was naturally dominated by baetids, chironomids and acarinids in December. In February, simuliids and oligochaetes gained prominence (Figure 10.64U). With extreme flow reduction, for this component of riffles, leptocerids were an additional dominant family.

Riviersonderend site

At the Riviersonderend site, natural variability was high and obscured any major patterns. Two trends were evident. Elmid larvae were particularly abundant in all parts of riffles at natural baseflows, as were oligochaetes (Figure 10.65). Chironomids were the only family consistently present as a dominant taxon under both natural and unnatural baseflow regimes. Overall, the early summer riffle assemblage was dominated by chironomids, and baetids and simuliids were in fewer numbers (Figure 10.65W). During the impact phase, at natural baseflows, elmids and particularly oligochaetes (27% 0.1 m^{-2}) increased to become dominant taxa, in addition to baetids and chironomids. In early autumn, simuliids were once again common, oligochaetes declined in numbers and elmids increased in abundance. With extreme low flows, in February, chironomids were still the major BMI component, as at natural baseflows. The assemblage differed from the natural one in that oligochaetes were not dominant (only 0.8% 0.1 m^{-2}) and simuliids were fairly abundant (at 6% 0.1 m^{-2}). In early autumn, with a baseflow exceeding the December magnitude (Figure 10.1), elmids increased in abundance and simuliids occurred in greater numbers than midsummer. The assemblage composition of the impact location had changed relative to February, but still differed from the control one. The main difference was the presence of water mites (9% 0.1 m^{-2}). Simuliids and baetid mayflies comprised only marginally lower than 5% 0.1 m^{-2} of the total assemblage, so their absence relative to the dominant control taxa was not remarkable. Moreover, chironomids remained the major taxon in both locations.

Examination of the relative contributions by riffle taxa to stone tops and bottoms, and the substratum underneath, did not reveal any pronounced differences between the two locations in response to natural versus unnatural baseflow regime (Figure 10.65). During the pre-impact phase, riffle components were dominated by baetids, chironomids and simuliids, although there was some natural variability in both the dominant taxa and their relative proportions (Figure 10.65T, B and U). The same riffle taxa were common

on stone tops under conditions of natural and extremely low baseflows, although fewer elmids and more baetids were present in the impact location. In April, the assemblages at the two locations closely resembled the February ones, but simuliids were no longer dominant in the impact location (Figure 10.65T). Stone bottoms in natural and impacted riffles differed in February in that oligochaetes were common in the former case, while Acarina were more abundant in the impact location. From February to April, there was a clear increase in elmid numbers on stone bottoms, and water mites became one of the dominant taxa in the control location. In contrast, Acarina declined in the impact location, and simuliids and oligochaetes gained prominence (Figure 10.65B). Chironomids dominated the BMI assemblage of the underlying substratum in December and again in April. During February, the dominant taxa in the control and impact locations were quite different, except for chironomid larvae (Figure 10.65U). Elmids and large numbers of oligochaetes were present under natural conditions, while baetids were not a dominant group. In contrast, simuliids and baetids were common at extreme low flows. After natural flows were reinstated, the two locations still differed markedly, with only chironomids in common. From February to April, the control assemblage gained simuliids as a dominant taxon, while oligochaetes decreased in abundance. In contrast, numbers of baetids and simuliids decreased in the impact location, while Leptophlebiidae and Acarina increased in their percentage contribution to the assemblage.

Changes in the taxonomic dominance of run assemblages in response to natural and unnatural dry-season baseflows

Changes in the contributions of various taxa to the composition of BMI assemblages inhabiting runs appeared less consistent than those of riffles (see above). Although run and riffle assemblages were similar in having baetids and chironomids as the most consistently dominant families, simuliids were not a dominant group in runs (Figures 10.66-10.69). A number of other taxa were present in high abundances in runs at the sites, typically on a site-specific basis (see below). Heptageniids and Leptophlebiidae appeared to be interchangeably dominant in locations within the same reach, indicating possibly patchy distributions.

Elands site

Under entirely natural, dry-season baseflows, at the Elands site (Figure 10.1), taxon dominance was highly variable in run biotopes at an intra-reach scale. Moreover, few consistent trends could be detected for each substratum component (Figure 10.66). Overall, however, runs were consistently dominated by baetids, chironomids, oligochaetes and ephemerellids in early summer (Figure 10.66W). With lower baseflows midsummer, there was a shift in overall dominance from baetids to Ephemerellidae, with the latter increasing to represent numerically more than half the assemblage (51% and 54% in the two locations). Leptophlebiidae and Heptageniidae also increased in abundance, though not in both locations (Figure 10.66W). A similar assemblage inhabited runs in April, possibly reflecting the highly similar baseflows in these two months (Figure 10.1). However, ephemerellids had declined, relative to baetids and chironomids, at this time.

The tops of stones located in runs were dominated by baetids in both December and February (> 83% 0.1 m^{-2}), while chironomids increased in abundance in April (Figure 10.66T). Ephemerellids increased in abundance in February and April, but were not consistently dominant in the control and impact locations. Stone bottoms were similarly dominated by baetids in December, with the other dominant taxa being Heptageniidae and chironomids (Figure 10.66B). With a gradual, natural decrease in baseflow from December to February (Figure 10.1), there was a pronounced increase in the numbers of ephemerellids, and

Acarina and Heptageniidae reached proportions exceeding 5% 0.1 m^{-2} of the total assemblage (Figure 10.66B). A similar assemblage composition was recorded in April, although acarinids were no longer a dominant group. The underlying substratum of runs clearly differed in assemblage composition from that of the overlying stones, though mostly in early summer (Figure 10.66U). Baetids represented a lower percentage of the overall assemblage, and ephemerellids and chironomids were also in lesser proportions than oligochaetes. At naturally low baseflows (February), Ephemerellids represented the most abundant family of the assemblage, at > 62% 0.1 m^{-2}. Other dominant taxa were chironomids, elmid larvae, heptageniid mayflies and Leptophlebiidae, although these taxa were not all common to both locations. With a change to early autumn conditions, chironomids and baetids increased in number, while ephemerellids declined in their percentage contribution to the assemblage.

Molenaars site

Benthic macroinvertebrate assemblages of runs at the Molenaars site showed a high degree of intra-reach variability, both in dominant taxa and their relative contributions to the composition of the assemblage (Figure 10.67). Overall, Baetidae and Chironomidae were the only two taxa consistently present in all months, under both natural and unnaturally low baseflows (Figure 10.67W). In December, oligochaetes, acarinids, ephemerellids and leptophlebiids were also important faunal components. From December to February, at natural baseflows, there was an increase in the percentage contribution by Ephemerellidae and Leptophlebiidae, and Heptageniidae were also dominant. The latter taxon was not dominant (3.8% 0.1 m^{-2}) at extreme low flows during the impact phase, and ephemerellids were in slightly higher numbers than at the control location. However, overall, there was no marked change in taxon dominance associated with extremely low baseflows. Furthermore, run assemblages in the two locations were highly similar in taxon dominance during the post-impact phase.

The tops of run stones in both locations at the Molenaars site were highly dominated by baetids, with percentages of 90% 0.1 m^{-2} and 93% 0.1 m^{-2}, in the control and impact locations respectively, in December (Figure 10.67T). Chironomids were also fairly abundant, but only dominant in the control location. Dominance by baetids continued into February, while chironomids increased in abundance. Heptageniidae were also found on stone tops at natural, low flows, but were not as abundant in impacted runs. During the recovery phase, both locations were highly similar in the percentage contributions by the two main taxa, baetids and chironomids. On stone bottoms, baetids were present in highest numbers in December and, like chironomids, were consistently present in both locations irrespective of baseflow magnitude (Figure 10.67B). Ephemerellidae increased markedly with flow reduction, and was the most abundant family at extremely low baseflows. Leptoceridae were only dominant in impacted runs, and the converse trend was apparent for leptophlebiids. Heptageniidae were a dominant family in both locations. In April, percentage contributions by the dominant taxa, baetids and chironomids, were highly similar for both locations for stone bottoms and the underlying substratum. In both substratum components, ephemerellids persisted as a dominant group only in the impact location (Figure 10.67B and U). In early summer, the underlying substratum of runs in both locations was naturally dominated by baetids, chironomids, oligochaetes, ephemerellids and leptophlebiids, while acarinids were only present in one location (Figure 10.67U). With a natural decrease in dry-season baseflows, in February, the composition of the assemblage was markedly altered only by the addition of Limnichidae as a dominant family. The composition of the impacted substratum was highly similar to that of the control location, with the exception that limnichids were less

common (1% 0.1 m^{-2}, compared with a natural 6% 0.1 m^{-2}) than elmid larvae (7% 0.1 m^{-2}, compared with a control figure of 2% 0.1 m^{-2}).

Du Toits site

At the Du Toits site, it was difficult to detect any clear trends especially resulting from the 86% baseflow reduction midsummer, due to the inherent variability of run assemblages. Run assemblages were consistently dominated by chironomids under the natural, dry-season baseflow regime (Figure 10.68). The underlying substratum additionally had Leptoceridae and notonemourids as consistently dominant taxa. An assessment of the entire run assemblage in early summer showed an overall dominance by Chironomidae, with other shared dominant taxa being baetids, Leptophlebiidae and Leptoceridae. Several other taxa were also abundant, though not common to both locations. These taxa were oligochaetes, juvenile ephemeropterans, Acarina and Notonemouridae (Figure 10.68W). The natural decline in baseflow in February resulted in a change in the overall run assemblage. Although chironomids remained the greatest faunal component, simuliids and ephemerellids became more abundant, and baetids decreased in numbers. Taxon dominance was shared among a greater number of taxa under conditions of extremely low baseflow, although chironomid larvae were still the most common element. These taxa were baetids, Acarina, ephemerellids, leptocerids and notonemourids. Control and impact locations were most similar in terms of dominant taxa during the post-impact phase. The main difference observed was the presence of ephemerellids and leptophlebiids in the control runs, in contrast to oligochaetes in the impact location.

Examination of the various components of runs, showed a marked variety of a maximum of nine dominant taxa (control location) inhabiting the tops of run stones, under natural, early summer conditions (Figure 10.68T). The greatest percentage contribution to the assemblage was by chironomid larvae in both locations. The impact location was naturally less variable in taxon dominance, but included simuliids. With naturally low baseflows during the impact phase, only two taxa dominated the tops of run stones, namely simuliids and chironomids. Moreover, the percentage contribution by other taxa was low. At extremely low baseflows, the two taxa common in the control location were present in lower numbers, while baetids and Acarina were also dominant. The greatest degree of similarity between locations was found in April, where chironomids again dominated overall, while marginally more simuliids were recorded from the control location. In early summer, the bottoms of stones in runs were predominantly inhabited by chironomids, baetids and Leptoceridae, while leptophlebiids, simuliids and Acarina were present in only one of the two locations (Figure 10.68B). Comparison of the bottom assemblages at natural and extremely low baseflows indicated that the following dominant taxa were shared, baetids, chironomids and ephemerellids. The latter family was in greater abundances in the impact location. Simuliids were present as a dominant family only under conditions of natural baseflow. Similarly, leptocerids, Acarina and Caenidae were only abundant at very low flows. The presence of caenids contributing 7% 0.1 m^{-2} to the overall assemblage at extremely low baseflows, compared with a lesser 1% 0.1 m^{-2} at naturally lowest baseflows, was noteworthy. The family is recognised as preferring slow-flowing conditions and pool biotopes. During the post-impact phase, the two locations were still fairly divergent in terms of dominant taxa on the bottoms of run stones, having only chironomids and leptocerids in common. Hydroptilidae were also dominant in the control location, while baetids, Acarina and ephemerellids were dominant in the previously impacted location. For the underlying substratum, Leptoceridae, chironomids and notonemourids were naturally common throughout the study period, while baetids were in relatively low numbers (Figure 10.68U). Variability in percentage composition of the assemblage was high in December and the number of dominant taxa

remained elevated into February. At naturally low baseflows three taxa, Athericidae, ephemerellids and simuliids increased in numbers to >5% 0.1 m^{-2}. Comparison of the run assemblage at extremely low flows with control data revealed several differences in taxon dominance. In the former case, baetids and Acarina were abundant, while Athericidae represented a marginally lower percentage of the assemblage than under natural flows, at 2.3% 0.1 m^{-2}.

Riviersonderend site

Natural variability in taxon dominance of runs at the Riviersonderend site appeared lowest at the peak of the dry season (Figure 10.69). Baetidae and Chironomidae were naturally common throughout the study period, with the latter family in highest abundances in most instances. In early summer, Acarina, leptophlebiids, Leptoceridae and Athericidae were also dominant taxa. The overall assemblage changed from early to midsummer, with a natural decline in baseflow magnitude, to one dominated by only three main families, namely elmids (larvae), baetids and mostly chironomids (Figure 10.69W). There was no apparent difference between the control and impact locations, suggesting that the overall assemblages were not altered in dominance by extreme flow reduction. During the post-impact phase, both the natural and impact assemblages gained ephemerellids as an abundant family. However, the impact location showed a reduction in the numbers of baetids and elmids relative to the control location, as well as a gain in the proportion of Hydroptilidae.

Throughout the various components of runs in the Riviersonderend, there was less of a difference between the control and impact locations during the flow reduction phase of the study, than during the recovery phase (see below). The assemblage composition of the tops of stones in runs was fairly similar within the same reach of the Riviersonderend. Leptocerids, Acarina and elmids were common, in addition to baetids and chironomids (Figure 10.69T). The two locations were highly similar in taxon dominance with only the two consistently dominant taxa present, irrespective of the different baseflow magnitudes. The greatest differences in stone tops between locations were evident during the recovery phase. In the impact location, baetids were less and chironomids more abundant than in the control location. The only other common family was Elmidae. Ephemerellids occurred in high numbers only in the control location. In contrast, hydroptilids, oligochaetes and water mites were dominant taxa in previously impacted runs. Stone bottoms had similar dominant taxa to stone tops, in early summer, with an additional increase in Athericidae (Figure 10.69B). As for stone tops, there was no evidence of a difference in taxon dominance with naturally and unnaturally low, dry-season baseflows, with a marked increase in elmid numbers in both instances. Greater divergence in taxon dominance was observed after natural flows were reinstated in the impact location, with fewer dominant taxa in this location and an overall dominance by chironomid larvae. Under conditions of natural baseflow, Leptophlebiidae and Athericidae were most abundant underneath run stones, in early summer (Figure 10.69U). At peak low flows (February), three taxa were most common naturally, namely chironomids, followed by baetids and elmids. At extremely low flows, the former two taxa were still abundant, while Ephemerellidae replaced elmids as the third most common BMI family. Although the dominant elements of the control assemblage persisted through to April, despite naturally elevated baseflows (Figure 10.1), there was a shift in dominance in the impact location. In the latter case, oligochaetes, elmids and hydroptilids increased in abundance, chironomids remained dominant, and only a small percentage of the assemblage comprised other taxa.

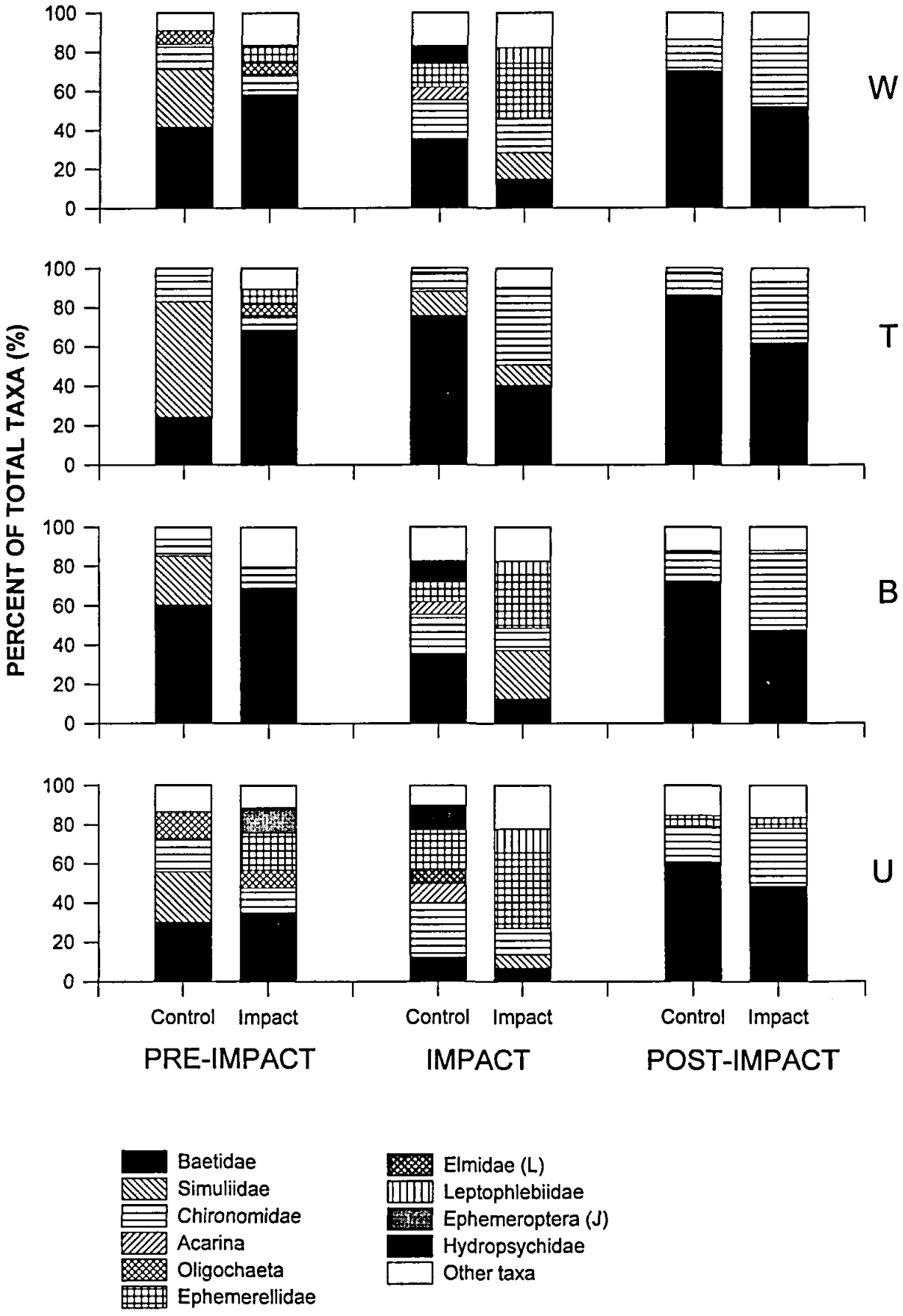


Figure 10.62 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Elands site, under a natural baseflow regime. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum. L – larvae; J – juveniles.

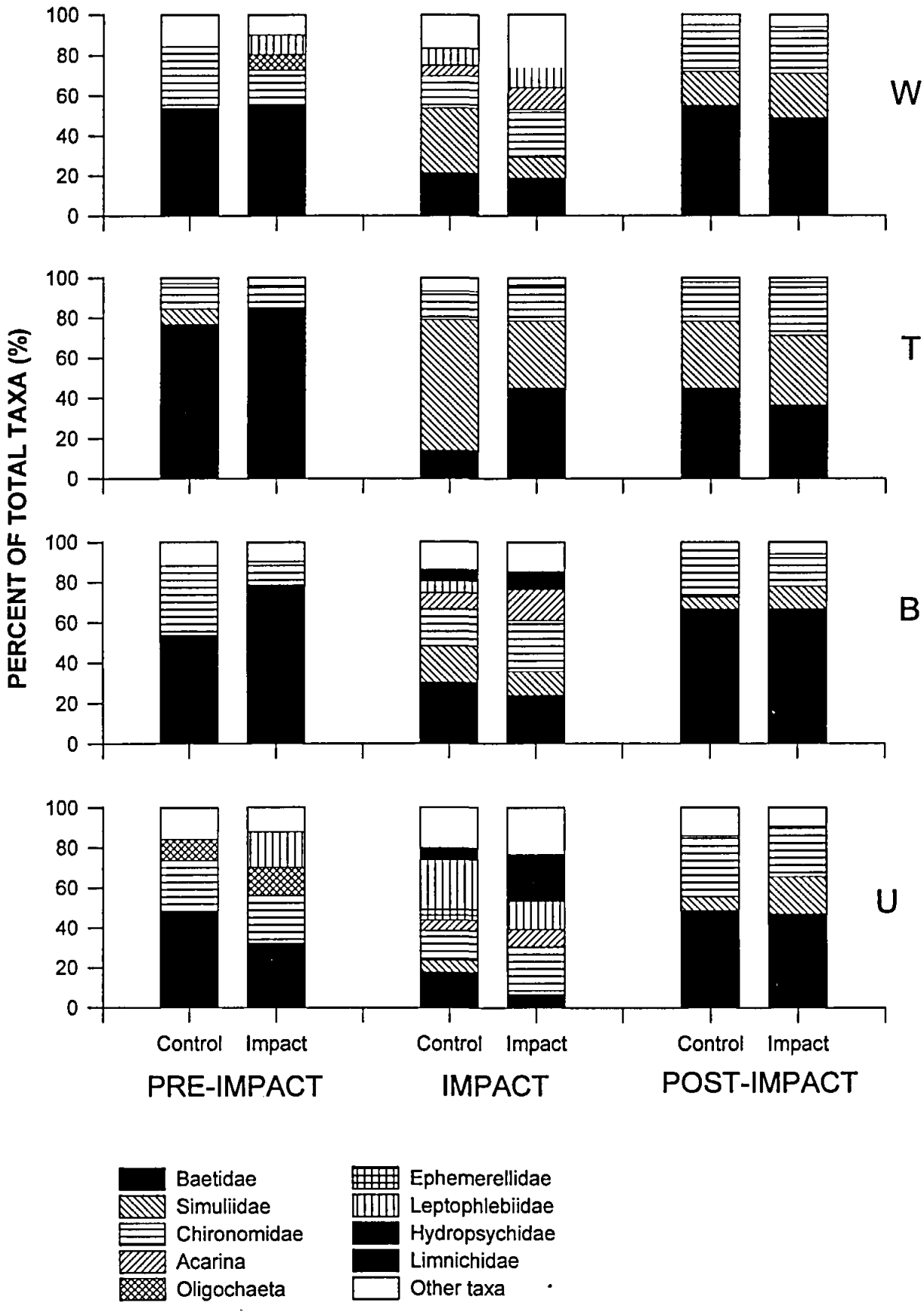


Figure 10.63 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

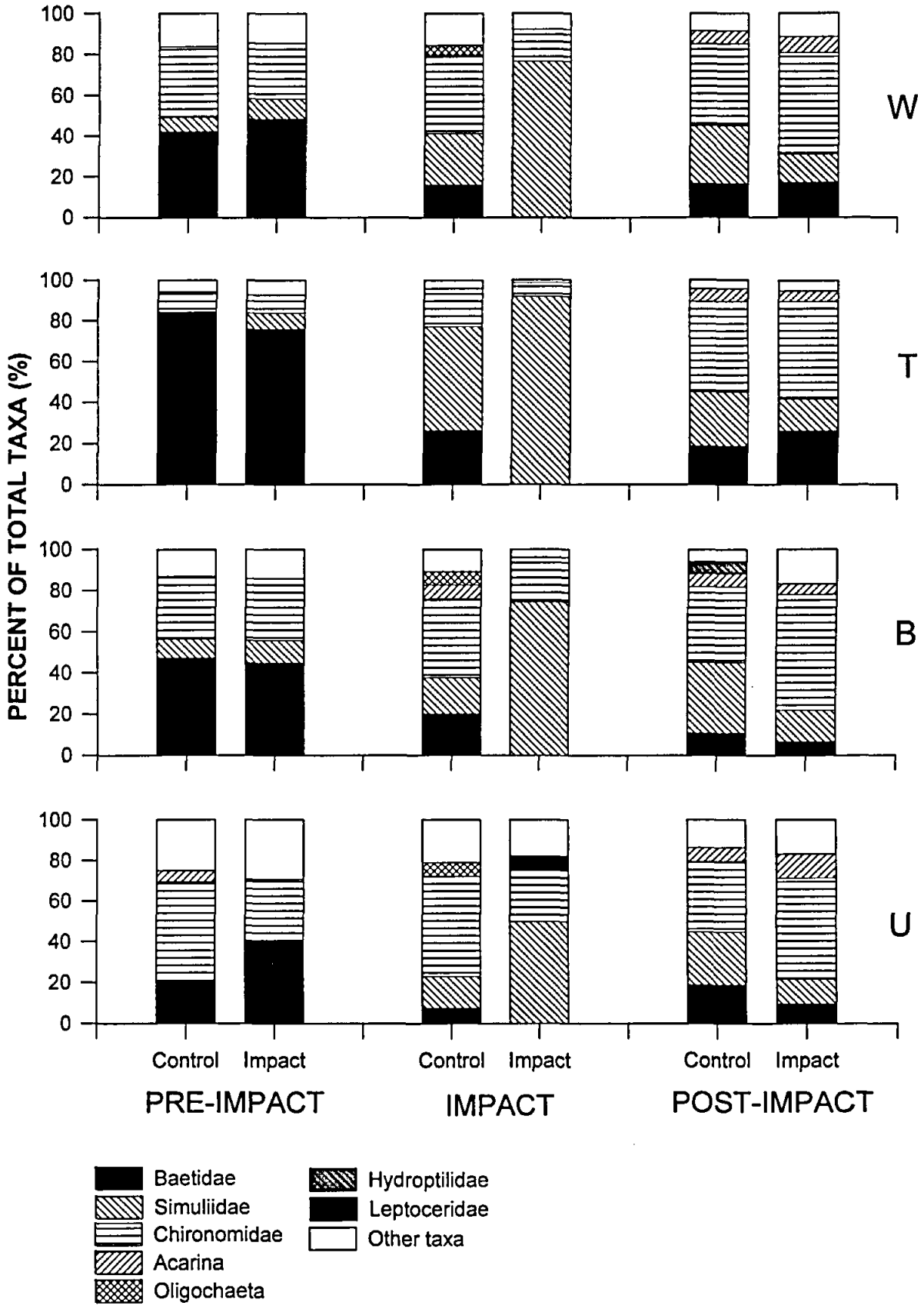


Figure 10.64 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

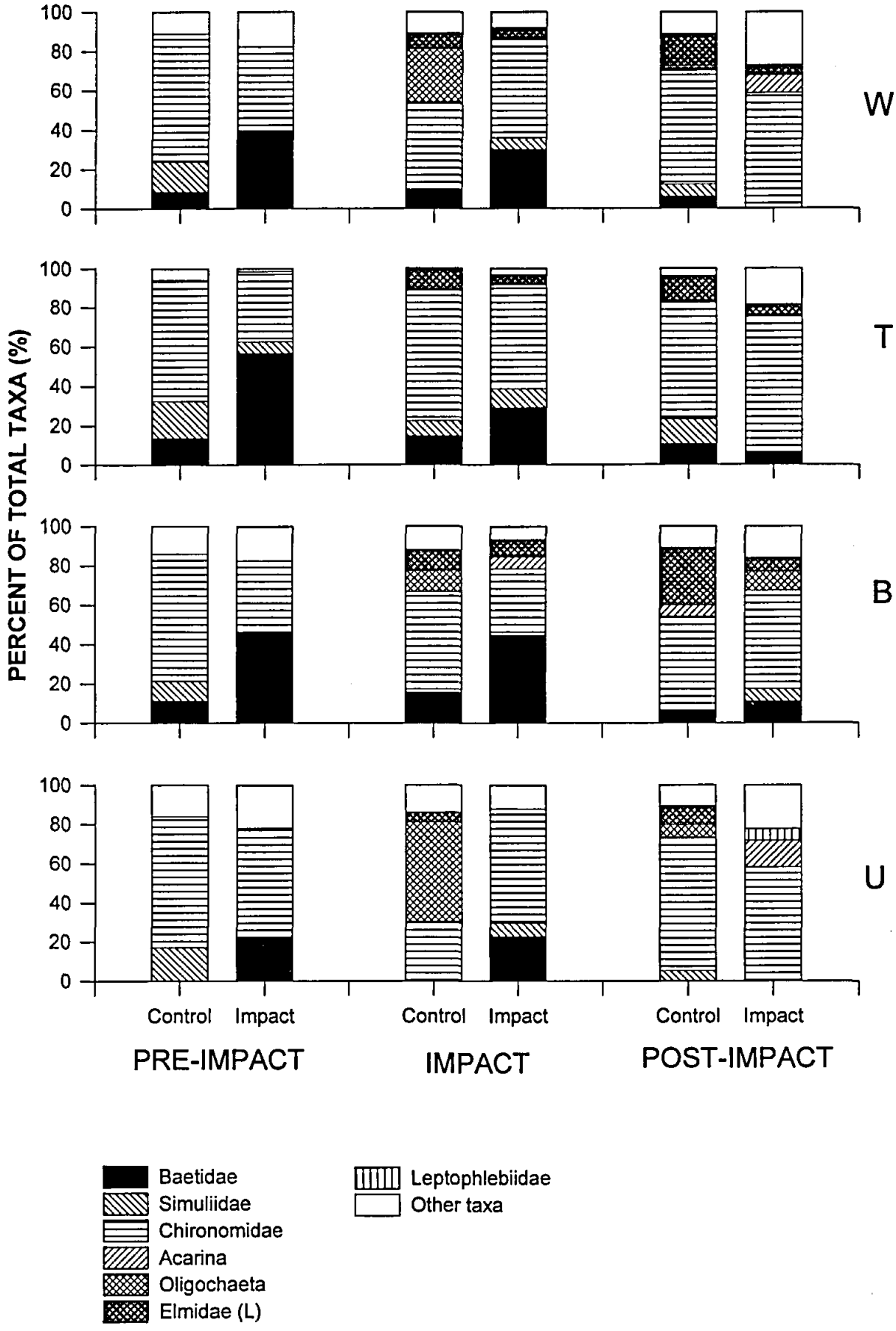


Figure 10.65 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from riffles in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum. L – larvae.

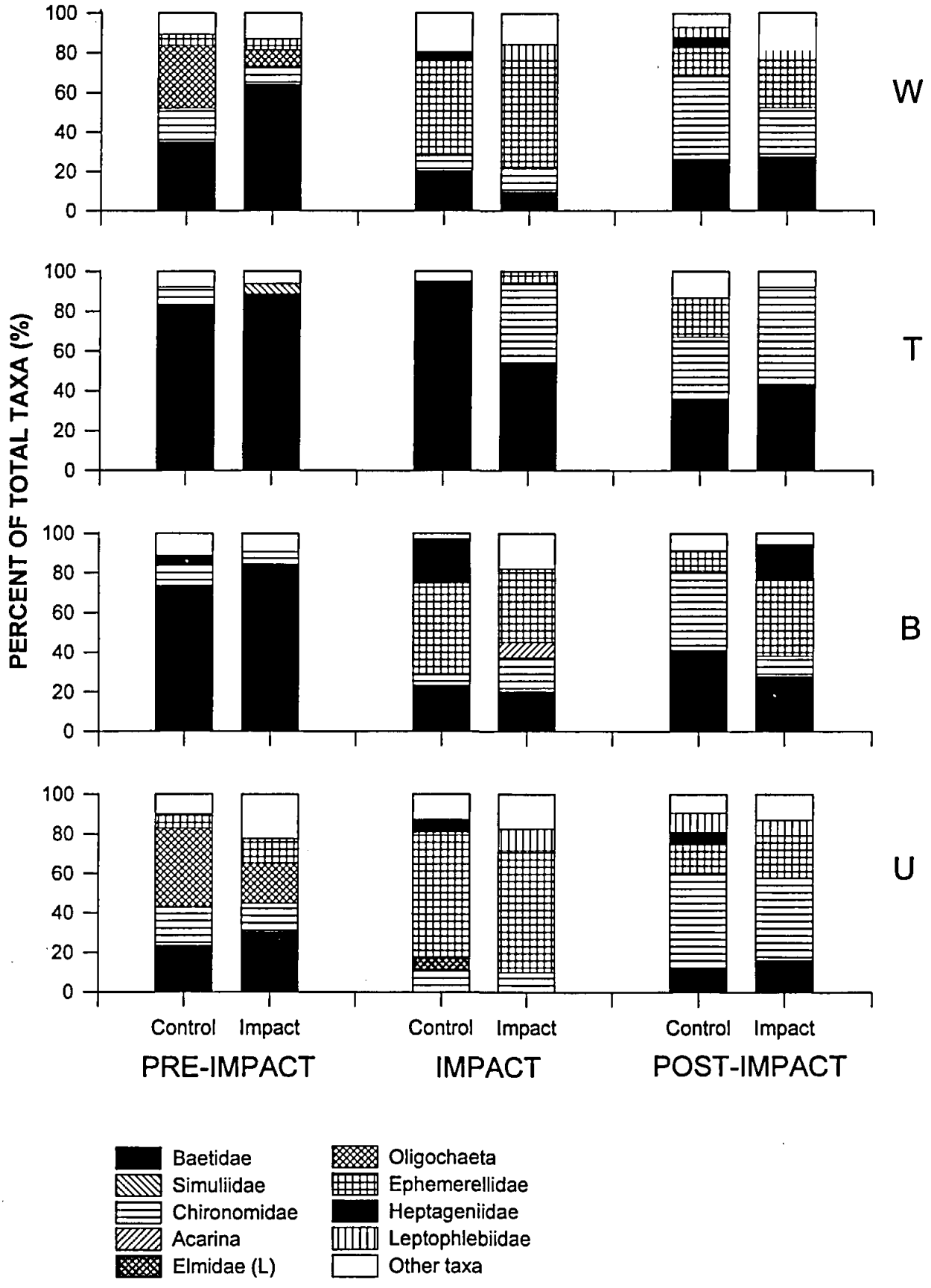


Figure 10.66 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Elands site, under a natural baseflow regime. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum.

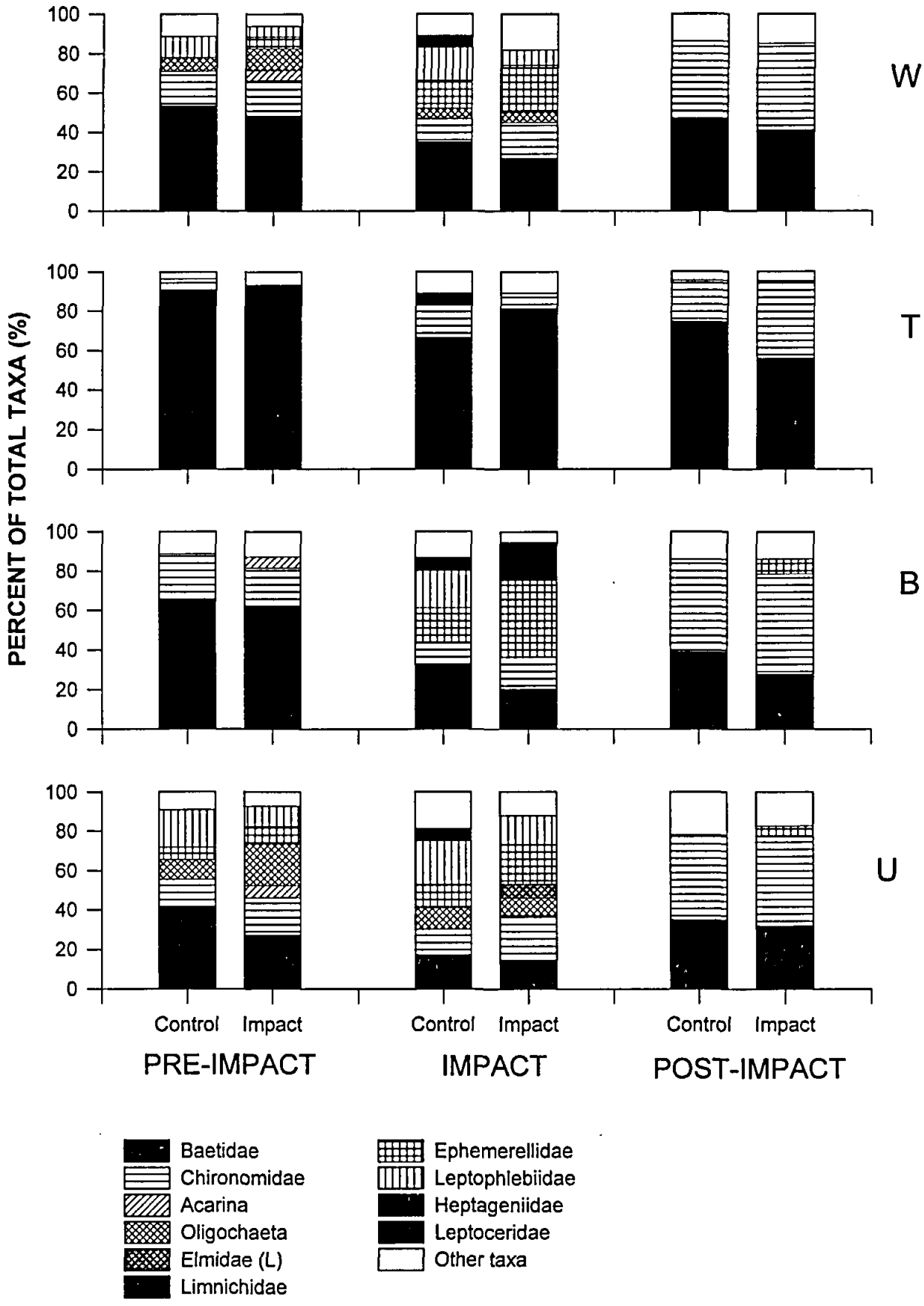


Figure 10.67 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Molenaars site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum. L – larvae.

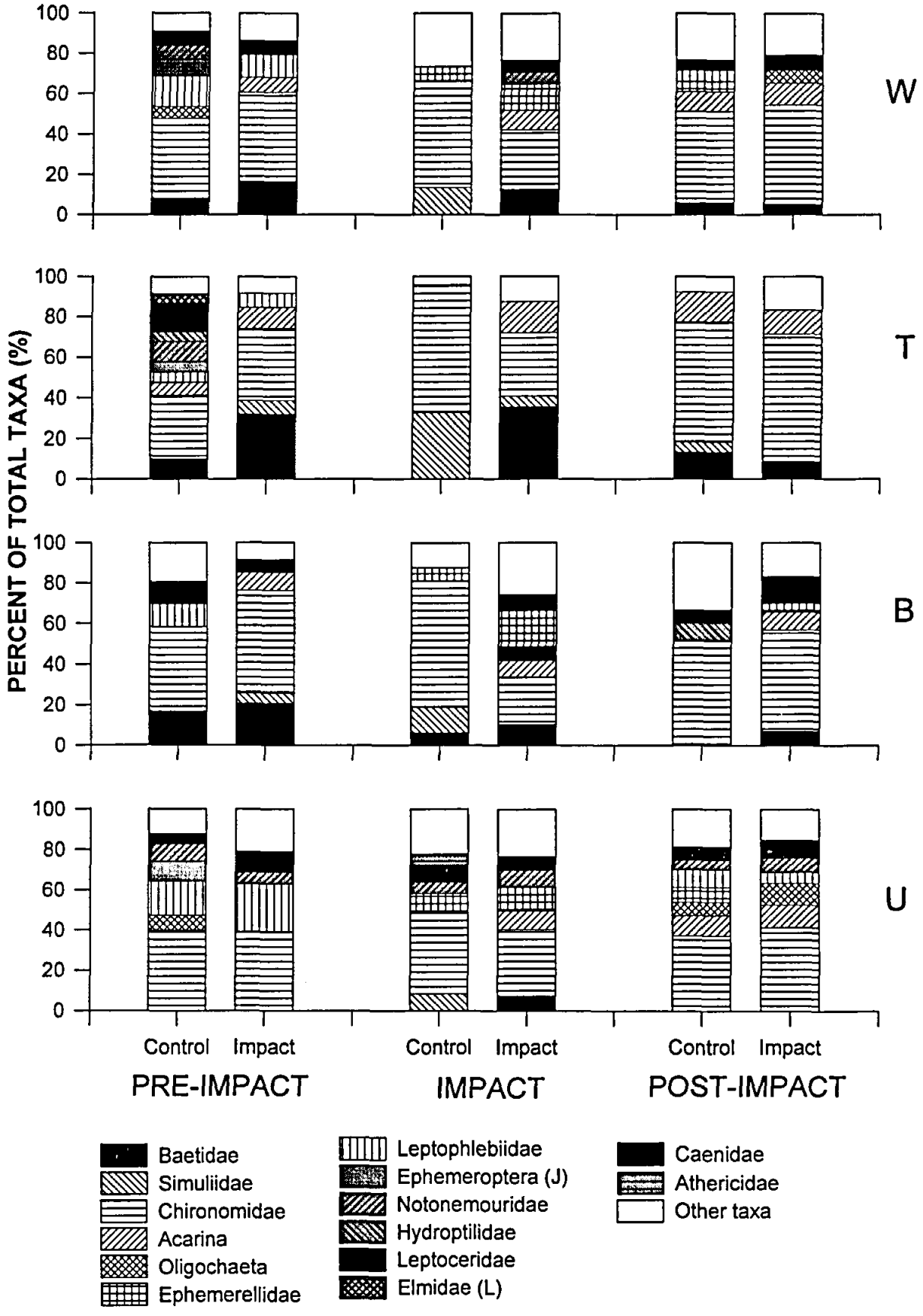


Figure 10.68 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Du Toits site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum. L – larvae; J – juveniles.

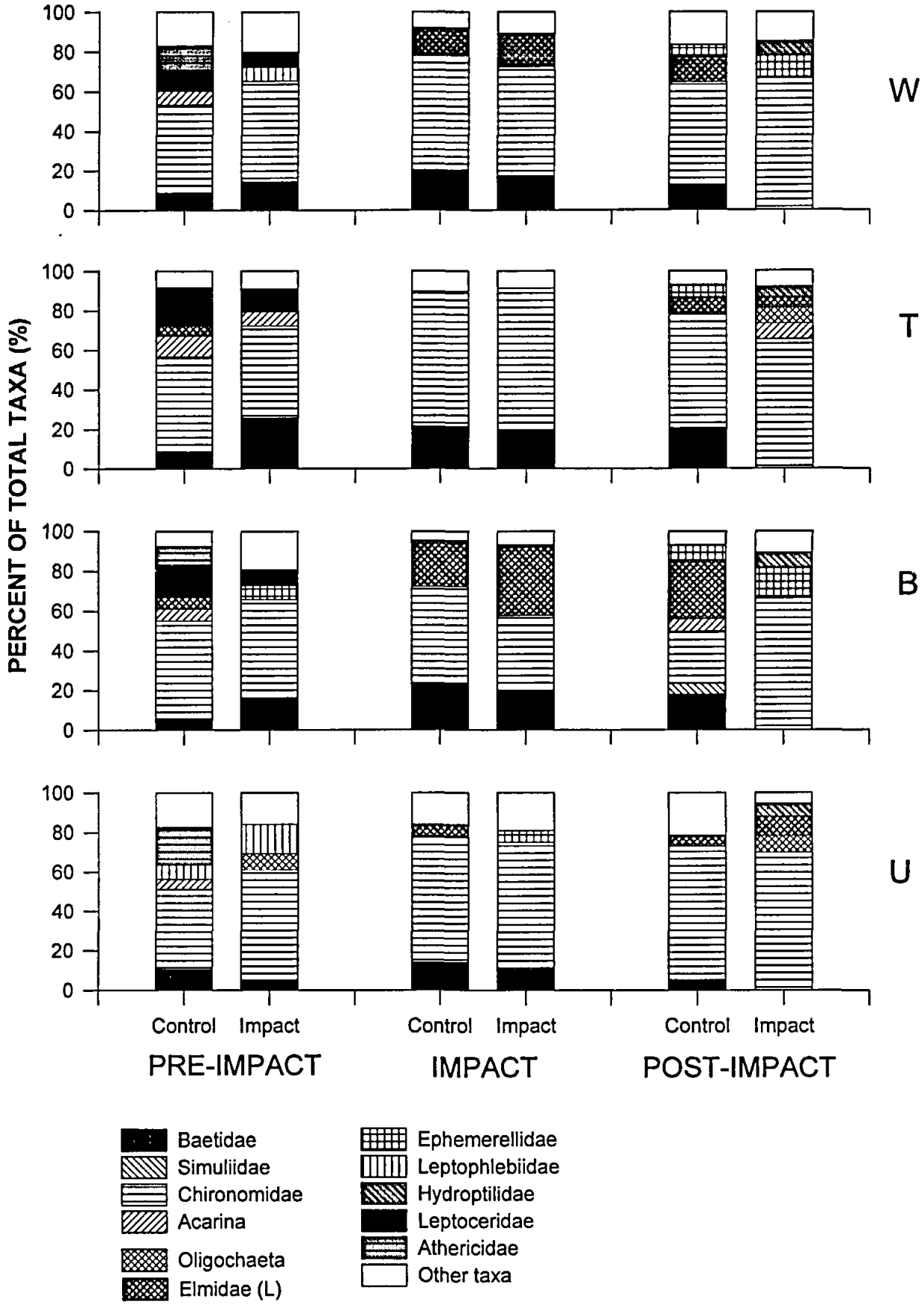


Figure 10.69 Percentage contributions by dominant benthic macroinvertebrate taxa (numerically ≥ 5% of total assemblage) from runs in the control and impact locations at the Riviersonderend site, under conditions of natural and unnatural baseflow. Pre-impact – December 1994; impact – February 1995; post-impact – April 1995. W – whole biotope; T – tops of stones; B – bottoms of stones; U – underlying substratum. L – larvae.

Summary of the effects of natural and unnatural baseflow regimes on taxon dominance

Comparison of all four sites, under natural baseflows, revealed that, excluding the group comprising "other taxa", typically 2-7 taxa 0.1 m^{-2} comprised the dominant ones in numbers. On the basis of an analysis of taxonomic dominance of the overall benthic macroinvertebrate assemblage in each location, the most common families throughout summer in all four rivers were Baetidae, Chironomidae, and Simuliidae. However, there were considerable differences among the rivers in the other taxa recorded at abundances ≥ 5% of the total assemblage. Natural variability in assemblage composition was also high within the same reach and between months.

There were no entirely consistent trends among the four sites, either with the natural decrease to low flows in February, or the subsequent transition to autumn. However, numerous small and larger-scale shifts in dominance were observed. For instance, with the natural decreases in baseflow magnitude from December to February, Baetidae typically decreased in abundance while Simuliidae increased, although the opposite two trends were evident at the Riviersonderend site. A number of other taxa showed distinct changes in abundance with a natural decrease in flow, though the responses were most often site-specific. For instance, the following taxa increased in numbers at the sites specified in parentheses: Ephemerellidae (Elands), Chironomidae (Du Toits), "other taxa" (Du Toits), Elmidae and oligochaetes (Riviersonderend). The numbers of individuals of some other taxa decreased noticeably, such as oligochaetes (Molenaars control location). There were fewer clear trends in taxon dominance with the natural change from February to April flow conditions. However, increases in taxa such as Acarina, Baetidae and Chironomidae, among others, were often observed.

Also at the scale of the entire location, with unnatural flow reduction in February, there were few major shifts in taxon dominance that could be identified as markedly different from natural ones. A decrease in the number of simuliids, and increases in numbers of Ephemerellidae and Hydropsychidae were recorded with a 36% reduction in baseflow. With an 86% flow reduction, numbers of simuliids increased in the Du Toits and to a lesser extent the Riviersonderend impact locations. At the Du Toits impact location, chironomids and baetids declined in numbers, while few changes in dominance were evident at the Riviersonderend site. Recovery appeared to have taken place in April in the case of the Molenaars and Du Toits rivers. However, the status of the Riviersonderend impact assemblage was difficult to assess, due to several differences in the most common taxa in the control and impact assemblages.

Trends in taxon dominance in riffles and runs, under natural and unnatural baseflow regimes, were also without consistent trends, and were most frequently site-specific. Hence, changes in dominance were summarised on a river-by-river basis. Patterns evident for riffle and run assemblages in their entirety sometimes differed from those observed for individual components of the substratum in each biotope. These differences were particularly due to the strong differences in assemblage composition recorded between the tops of stones and the often highly similar sets of taxa from stone bottoms and the underlying substratum. It was not possible to draw generalised, summary trends from the data for various substratum components. The distribution of taxa among these components merits further attention, however, especially due to the differences in the direction of change observed among components. For instance, an overall increase in the dominance of a particular taxon might be a function of a vast increase in its numbers on the

bottoms of riffle stones and the underlying substratum, while a smaller decrease in its dominance was recorded for stone tops.

Under conditions of natural baseflow in riffles at the four sites, the assemblages were most consistent in terms of common taxa in April, with generally fewer dominant taxa in this month than in midsummer. As the focus of the work was on responses to low flows in the peak of the dry season, in February, only these trends are briefly summarised below for the individual rivers. Responses in riffle assemblages to changes in the baseflow regime from summer to autumn are detailed in previous sections.

At the Elands site, with a natural decrease in baseflow in February, Baetidae decreased in their percentage contribution to riffle assemblages, while Chironomidae and particularly Ephemerellidae increased in dominance. Despite unnaturally low baseflows in the impact location of the Molenaars site in February, both impact and control riffles shared several common taxa, namely Baetidae, Simuliidae, Chironomidae, Acarina and Leptophlebiidae. Baetidae had decreased in their percentage contribution to the assemblage in both instances, compared with December values. The main changes in dominance possibly associated with a 36% reduction in baseflow magnitude, relative to the natural assemblage, were a decline in numbers of simuliids and the presence of Hydropsychidae as a common family. At the Du Toits site, under conditions of natural baseflow in February, the most common taxa were chironomids, simuliids, oligochaetes and baetids, with a decrease in the latter family relative to December. In stark contrast, taxon dominance at unnaturally low flows was confined to chironomids and simuliids. The latter taxon contributed a considerable 77% 0.1 m^{-2} to the assemblage, compared with a natural figure of 26% 0.1 m^{-2}, while chironomids represented a smaller percentage of the impacted assemblage than the control one. In addition, baetids were absent as a dominant taxon in the impact location, while representing 16% 0.1 m^{-2} of the natural riffle assemblage. At the Riviersonderend site, at natural baseflows during the impact phase, elmids and particularly oligochaetes increased from December numbers to become dominant taxa, in addition to baetids and chironomids. The natural decrease in Baetidae from December to February, observed at the other three sites, was not apparent at the Riviersonderend site. With unnaturally low February baseflows in the impact location, chironomids remained the main component of the assemblage, as at natural baseflows. The assemblage differed from the natural one in that oligochaetes were not dominant and simuliids were fairly abundant.

Changes in the contributions of various taxa to the composition of BMI assemblages inhabiting runs appeared less consistent than changes in riffles. Although run assemblages were similar to those of riffles in having baetids and chironomids as the most common families, simuliids were never a dominant group in runs. As the focus of the work was on responses to low flows in the peak of the dry season, in February, only these trends are briefly summarised below for the individual rivers. Changes in the run assemblages from summer to autumn are described in previous sections.

In the Elands River, with lower natural baseflows midsummer, the main response in run assemblages was a shift in dominance from baetids to Ephemerellidae, with the latter increasing to represent numerically more than half the assemblage. Leptophlebiidae and Heptageniidae also increased in abundance, though not in both locations. Similarly, in the Molenaars River, there were natural increases in the percentage contributions by these three taxa, while Baetidae, chironomids and oligochaetes were also common. In the Molenaars impact location during February, with a 36% reduction in dry-season baseflow, heptageniids

were no longer a dominant group and ephemerellids were in slightly higher numbers than at the control location. However, overall, there was no marked change in taxon dominance in runs. With a natural decrease in discharge magnitude from December to February, at the Du Toits site, although chironomids remained the greatest faunal component of runs, simuliids and ephemerellids became more abundant, and baetids decreased in numbers. Taxon dominance was shared among a greater number of taxa under conditions of extremely low baseflow (86% reduction), although chironomid larvae were still the most common element. The other taxa were baetids, Acarina, ephemerellids, leptocerids and notonemourids. At the Riviersonderend site, run assemblages changed from early to midsummer, with a natural decline in baseflow magnitude, to one dominated by only three main families, elmids, baetids and mostly chironomids. There were no apparent differences between the most abundant taxa in control and impact assemblages, suggesting that there were no major alterations in dominance as a result of an extreme flow reduction of 86%.

10.5.4 Changes in the diversity of benthic macroinvertebrate assemblages in response to natural and unnatural dry-season baseflows

Within the scope of the current report, changes in the diversity of BMI assemblages in relation to natural and unnatural, dry-season baseflow regimes were only addressed at the scale of whole locations. Further trends in Shannon-Weiner diversity and related indices are examined in Tharme (in prep.), as are the merits of the various indices. It should be noted that the indices calculated below pertain to BMI families and higher taxa, and that different values would be expected for a species level analysis (Tharme in prep.). In addition, the data presented here represent a combination of BMI data from riffle and run biotopes.

Shannon-Weiner diversities (H') for the four sites and locations, during the low-flow study are given in Figure 10.70.

Elands site

From the Elands data, it is evident that the natural variability in BMI assemblages was reflected by small differences in diversity between locations, particularly in early summer (December) (Figure 10.70). Diversity reached a maximum (2.04) at naturally lowest baseflow in February, and subsequently decreased to an overall minimum (1.43) in April (autumn).

Molenaars site

Of the four sites, diversity was highest at the Molenaars River (Figure 10.70). Under conditions of natural baseflow, it showed the same trend as the Elands, with a midsummer peak in diversity of 2.15, compared with an April minimum of only 1.32. Despite a roughly 36% reduction in baseflow in February, the same trend was observed for the impact location. Although diversity of the assemblage inhabiting the impact location was naturally higher than that in the control location (December values), the increase to 2.38 was marginally higher relative to the pre-impact ratio. However, statistical analysis would be required to ascertain whether or not the difference is significant (Tharme in prep.).

Du Toits site

For the Du Toits site, although the natural trend observed above was maintained, the difference in diversity between months was low (Figure 10.70). At extremely low baseflows in February (86% reduction in

discharge magnitude), diversity was the lowest recorded overall, at only 1.25. Although the impact location exhibited a naturally lower diversity than the control location (December), the decline in diversity recorded during the impact phase was marked. Moreover, during the recovery phase, in April, diversity was higher in the impacted location. The degree to which the observed changes are significant requires examination (Tharme in prep.).

Riviersonderend site

At the Riviersonderend site, a natural peak in diversity in February (1.60) as observed for the other sites occurred, but April diversity exceeded that in December (Figure 10.70). At a high reduction in baseflow below natural levels (*c.* 86%), there was a decrease in diversity (1.44), this despite the relatively higher natural diversity at the impact location than the control one (as shown in December). Also, the diversity recorded for April was marginally lower than that of December, unlike the established natural pattern.

Summary of the effects of natural and unnatural baseflow regimes on family-level diversity

Typically, during a natural dry season, diversity was greatest at lowest baseflow, in February. It was generally lowest in early autumn (April). However, the latter trend did not hold for the Riviersonderend site, a mountain-stream reach, where diversity was lowest at the onset of summer in December.

With a moderate, unnatural reduction in dry-season baseflow in February, diversity increased marginally from natural levels, following the natural trend. However, with extreme reductions in baseflow below natural magnitudes in midsummer, diversity decreased markedly.

10.5.5 Effects of natural and unnatural dry-season baseflows on the composition of benthic macroinvertebrate assemblages

Exploratory classification and ordination analyses (Section 9.5.3) of overall patterns in the composition of the BMI assemblages, at the four sites, was undertaken at several levels of resolution. These ranged from the overall presence or absence of taxa in the locations at all four sites, through to the distribution of taxa, on the basis of abundances, among the various substratum components of riffles and runs during only the impact phase of the study.

The results were generally inconclusive, and stress values for the ordinations reflected a high degree of variability among sample groups. In most instances, marked changes in the composition of the BMI assemblages, in relation to both natural and unnatural, dry-season baseflows were not apparent. Moreover, analyses based on only the presence or absence of taxa were typically less useful in showing trends than those based on their abundances. Therefore, in this section, results are only presented for two analyses, using abundance data. Firstly, the results of the broadest level of assessment, of the changes in BMI composition for each location in its entirety, for all sites combined, are presented. On the basis of these results, the results of an analysis of the BMI data for riffle and run biotopes of the Du Toits River, following extreme flow reduction in February, are discussed.

Trends for the other sites and at other times of the study, in various components of riffle and run biotopes, will be described elsewhere, after the incorporation of additional BMI data (Tharme in prep.). For example,

classification of presence/absence BMI data for riffle and run biotopes in all locations showed no clear trends, but the addition of further data might increase data resolution.

Effects of natural and unnatural baseflows on the overall composition of benthic macroinvertebrate assemblages

Classification of the BMI abundance data for all site locations, for all three phases of the low-flow study, showed that between-river differences in assemblage composition were more pronounced than differences related to unnatural baseflow regimes (Figure 10.71). The major division into two groups in the dendrogram (Figure 10.71A), at a Bray-Curtis percentage similarity of 65%, separated the Riviersonderend site from the other three sites. The second major division in the data resulted in all the Du Toits samples clustering separately from a combination of Molenaars and Elands samples, at a Bray-Curtis similarity of 67%. This division partly reflects the fact that the Elands and Molenaars sites are part of the same system, although they are within different zones. Internal groupings of samples for each site were all greater than 70% similarity. For all four sites, a seasonal pattern was evident, with summer and autumn grouping separately. In April, the control and impact locations grouped together at fairly high levels of similarity, ranging from 75-89%, for the Riviersonderend and Molenaars sites, respectively. Within-dry season groupings of the two locations, for December and February, were less consistent, with month-by-month groupings only occurring for the Molenaars and Riviersonderend sites. This indicated low differentiation among samples throughout the dry season proper.

Although three locations were subjected to unnaturally low baseflows in February, those at the Molenaars and Riviersonderend sites nevertheless grouped with the corresponding control location, at high percentage similarities of 84% and 81%, respectively (Figure 10.71A). In contrast, the impacted location for the Du Toits site separated from all other samples for the river, including data for the February control location, at 74% similarity. However, the relative differences in Bray-Curtis similarity among samples were small, as was also found for the trends discussed above. Moreover, for the Elands site, the percentage similarity computed between the control and impact locations for the impact phase was only 69%. The absence of clear changes in overall composition of the BMI assemblages, in response to extremely low baseflows, extended into the post-impact phase. The classification results suggest a stronger seasonal trend (see above) than any pattern associated with recovery from flow reduction.

Ordination of the same data set supported the above relationships among locations (Figure 10.71B), although the relatively high degree of stress (0.14) indicates that the relationship is not well represented in only two dimensions (Section 9.5.3). Analysis of the computed stress values showed that a good ordination of the data with no real prospect of a misleading interpretation would be achieved only in 3-d. The only obvious difference between the classification and ordination results is that the February and April assemblages for the Du Toits impact location appeared more similar in the ordination than on the basis of the dendrogram. This could, in part, be attributed to the fact that the dendrogram groups are not "fixed" spatially one to the other, as depicted.

Effects of natural and unnatural peak dry-season baseflows on the composition of benthic macroinvertebrate assemblages in riffles and runs at the Du Toits site

Examination of trends in both presence/absence and abundance data for the Du Toits site, for all phases of the low-flow study, did not show marked differentiation of impact and control samples. Moreover,

classification and ordination of data from only the impact phase, on the basis of the presence and absence of taxa showed no major trends. Analysis of abundance data for the peak dry season, February, however, showed some initial patterns in the groups of samples representing assemblages from various substratum parts of riffles and runs (Figure 10.72).

The dendrogram indicated that the main division in the data, at a low Bray-Curtis similarity of 40%, separated all BMI assemblages recorded from impacted riffle tops and, mostly, a number of assemblages from the tops of non-impacted run stones, from the rest of the data (Figure 10.72A). Although the highest similarity was recorded for two of the impacted riffle top samples (92%), the third sample of this kind grouped more closely with assemblages from the tops of stones from control runs, in what constituted the second major division of the data (at 48%). Hence, biotope type and substratum component appeared to have at least as much of an influence on the composition of the BMI assemblage as baseflow magnitude. At further groupings of the data, trends between impacted and non-impacted assemblages were weak. For instance, although impacted assemblages from the tops of some runs grouped with those from tops of non-impacted riffle stones, it was at a low similarity of 58%. Although two groups of impacted run samples, from all three substratum components, were formed that were fairly similar in composition at 75% and 76% similarities, they grouped at a lower similarity with an array of non-impacted run and riffle assemblages.

The ordination primarily showed the same trends as the dendrogram, but the stress value for representation in 2-d was fairly high (Figure 10.72B). It was evident from the distances among samples, that impacted and control riffle assemblages grouped closer than assemblages from impacted and control runs. However, some separation of impacted and natural assemblages on the basis of Bray-Curtis similarity was evident for each biotope. There were no distinct groupings of various substratum components, under conditions of either natural or unnatural baseflow.

Summary of the effects of natural and unnatural baseflow regimes on the overall composition of benthic macroinvertebrate assemblages

In most instances, there were no distinct changes in the composition of BMI assemblages, in relation to either natural or unnatural, dry-season baseflows. Classification and ordination of BMI abundance data for all site locations, for all three phases of the low-flow study, showed that between-river differences in assemblage composition were more pronounced than differences related to unnatural baseflow regimes. The trend was evident from the peak of the dry season in February, into autumn, with a stronger seasonal trend than any pattern associated with recovery from flow reduction.

Analysis of BMI abundance data for February, from riffles and runs in the Du Toits River, showed few possible trends on the basis of Bray-Curtis percentage similarities. The 86% reduction in baseflow below natural magnitudes showed some grouping of assemblages from the tops of impacted riffle stones with control assemblages from the tops of run stones, as well as the converse pattern. Although these results suggest a loss of taxa from riffles and concomitant gains in taxa in runs, at extreme low flows, further analyses are required to substantiate the trend. Other similarities between impacted and non-impacted assemblages were weak, and biotope type and substratum component appeared to have at least as much of an influence on the composition of the BMI assemblages as baseflow magnitude.

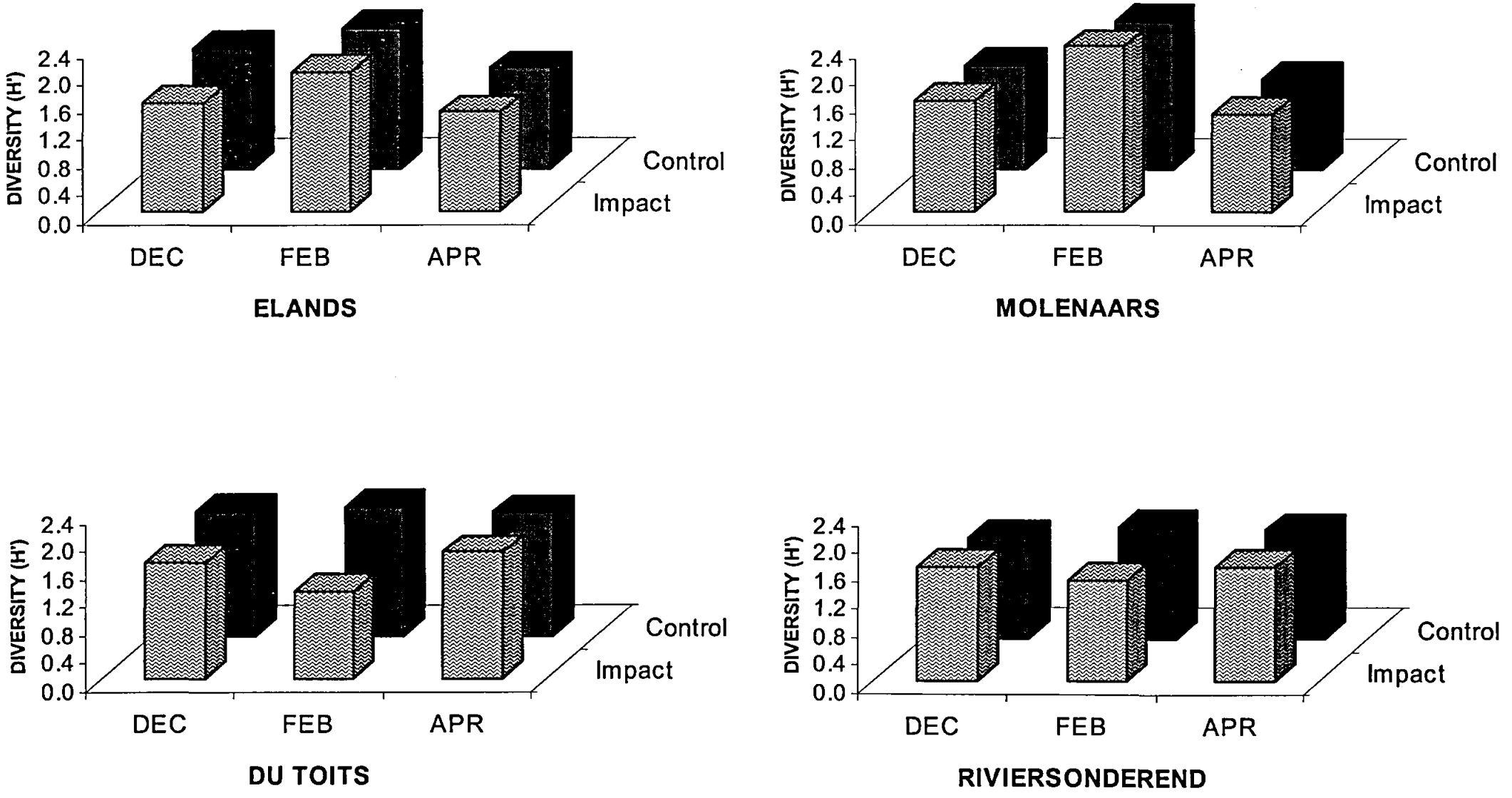


Figure 10.70 Shannon-Weiner diversities (H') for the control and impact locations at all four sites, for the pre-impact (December), impact (February) and post-impact (April) phases of the low-flow study.

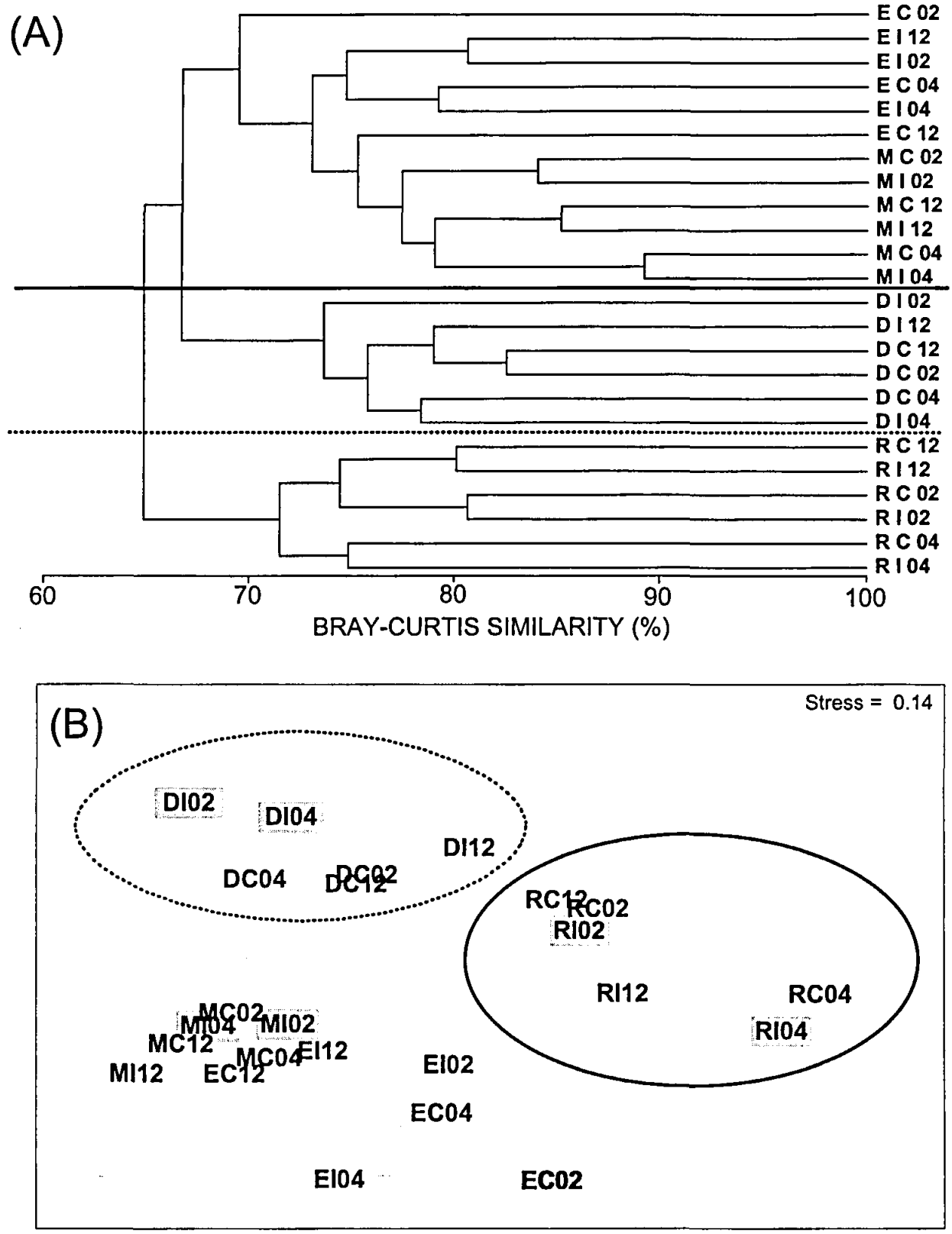
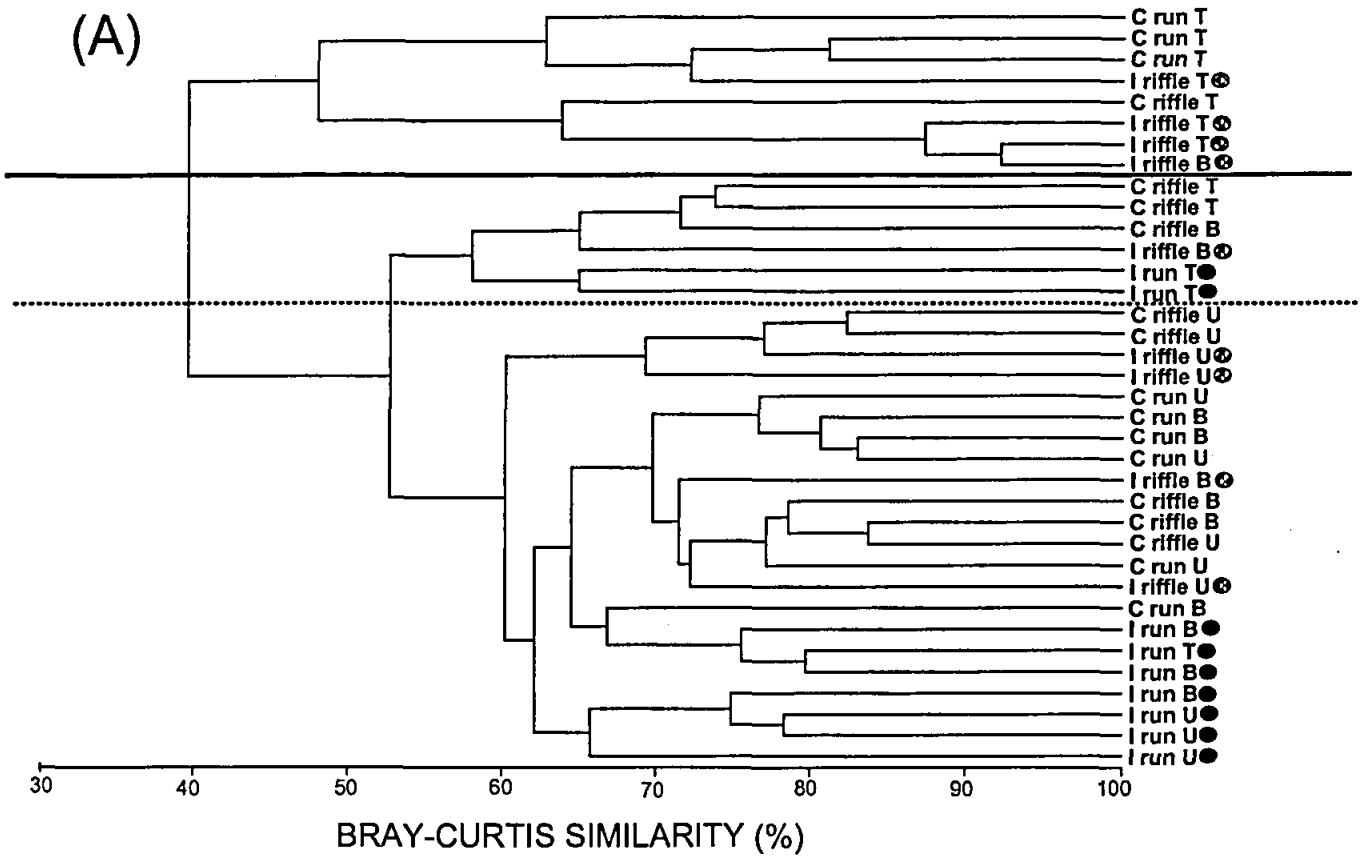


Figure 10.71 (A) Classification of benthic macroinvertebrate assemblages at all four sites, during the pre-impact (December, 12), impact (February, 02) and post-impact (April, 04) phases of the low-flow study. Average taxon abundances 0.1 m^{-2} for control (C) and impact (I) locations are represented. The solid line delimits the two main groups, and the dashed line defines the second set of groups. E - Elands; M - Molenaars; D - Du Toits; R - Riviersonderend. (B) Ordination of the same data set, by non-metric multidimensional scaling. The entire circle represents the first division shown in the dendrogram, and the dashed circle represents the second one. Assemblages that were subjected to unnaturally low baseflows in February, or were collected from the impact location after natural baseflows were reinstated in April, are shaded.



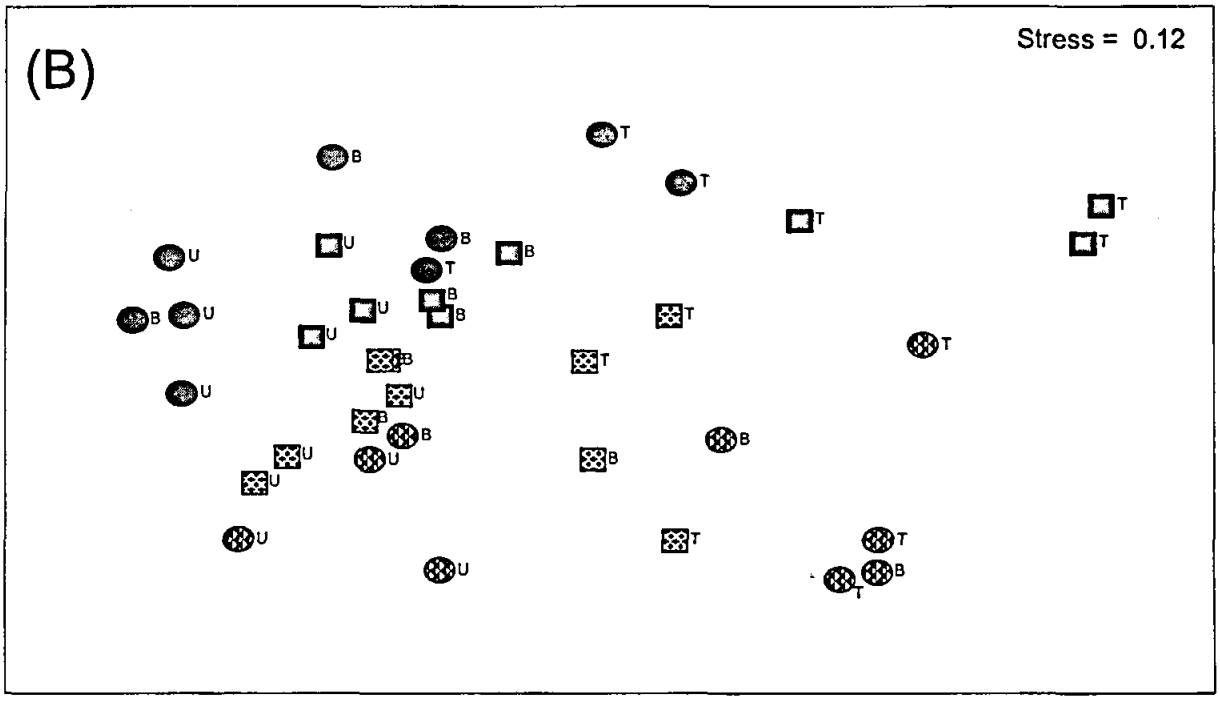


Figure 10.72 (A) Classification of benthic macroinvertebrate assemblages of riffles and runs in the control and impact locations, at the Du Toits site during the impact phase (February). Individual sample taxon abundances 0.1 m^{-2} are represented. Biotope assemblages were differentiated as follows: control location riffles (▩) and runs (□); impact location riffles (⊗) and runs (●); stone tops (T); bottoms (B); and underlying substratum (U). The two main groups of samples are indicated by the solid line, and the second division of groups is shown by the dashed line. (B) Ordination of the same data set, by non-metric multidimensional scaling.

11. SYNOPSIS AND DISCUSSION OF THE PRELIMINARY RESULTS OF THE LOW-FLOW STUDY

Written by R. Tharme

11.1 INTRODUCTION

11.2 CHANGES IN DRY-SEASON BASEFLOW REGIME WITH SPECIFIC REFERENCE TO HYDROLOGICAL DISTURBANCE HISTORY

11.3 ECOLOGICAL RESPONSES OF UPPER RIVERS TO NATURAL DRY-SEASON BASEFLOW REGIMES AND REDUCTIONS IN BASEFLOW MAGNITUDE BELOW NATURAL LEVELS

11.3.1 Changes in river physical and chemical variables in relation to dry-season baseflows

11.3.2 Effects of dry-season baseflow regimes on physical biotopes

11.3.3 Effects of dry-season baseflow regimes on benthic macroinvertebrate assemblages

11.1 INTRODUCTION

The following sections provide an overview of the salient findings of the low-flow study to date. Section 11.2 summarises the observed natural baseflow regimes at the four study sites during the dry season, and the hydrological implications of the series of experimental reductions in baseflow. The next section provides a synopsis of the most apparent effects of unnaturally low flows on water chemistry, the character of physical biotopes and the composition of BMI assemblages. The observed responses are compared with the recorded natural trends in these three components of the riverine ecosystem. Direct comparisons with the scientific literature are not made here, for two reasons. Firstly, the majority of available literature does not pertain explicitly to the effects of low dry-season baseflows on perennial rivers (Section 7.1.4). In addition, a comprehensive review of the various effects of low flows on rivers is underway (Tharme in prep.). A brief discussion of the ways in which the kinds of results obtained in the low-flow study might be of use in the development of ecologically-relevant low flow indices, particularly within instream flow methodologies, is given in Chapter 16, as are recommendations for future work.

11.2 CHANGES IN DRY-SEASON BASEFLOW REGIME WITH SPECIFIC REFERENCE TO HYDROLOGICAL DISTURBANCE HISTORY

The following basic trends were observed in the natural baseflow regime during the low-flow study, from December to May (see also Section 10.2.1):

- On the basis of the field measurements of instantaneous discharge, the summer month of lowest baseflow differed for the four sites. For the Elands and Molenaars rivers, discharge measurements for the third field trip took place in early March, but it is likely that the flows were representative of the latter part of February. For the Elands site, baseflow was lowest in March/April, and the Molenaars site

experienced its lowest discharge in early March. February was the month of naturally lowest flow for both the Du Toits and Riviersonderend sites.

- On the same basis, December was the month of highest recorded baseflow for all sites, except the Riviersonderend. For the latter site, the highest magnitude discharge was recorded in May.
- On the basis of average daily discharges from the near-natural historical flow record, the months of highest and lowest baseflow differed markedly, in many instances, from the months of highest and lowest flow identified from instantaneous measurements. The relevant months also differed according to whether or not the average or median daily discharge, from the hydrological record, was used.
  - On the basis of average daily discharge, the pattern was consistent for all four sites. The months of lowest and highest baseflow were February and May, respectively.
  - On the basis of median daily discharge, the trend in month of lowest baseflow was the same as for average daily discharge at only the Elands River. In the case of the Molenaars River, February and March had similarly low discharges. At the Du Toits site, there was a shift to March as the month of lowest flow. In contrast, at the Riviersonderend site, January was identified as the month of lowest median discharge. The month of highest discharge remained May, for all four sites.
- Fluctuations in observed discharge over the six-month period were greatest for the mountain-stream reach of the Riviersonderend River.
- The six-month experimental period probably encompassed a naturally dry summer.

In order to identify links between the long-term hydrological history of each of the study sites and observed flow data from the present study, the discharge magnitudes represented by various hydrological indices derived from the near-natural, hydrological record first needed to be assessed. Flow percentiles were derived from 1-day FDCs for the months concerned. These indices showed the following main trends in terms of baseflow regime for each of the sites during the six months over which the low-flow study was undertaken (see also Section 10.2.2):

- For each site, (mean) mean, (median) median, median 7-day low and (minimum) minimum indices were fairly similar to one another, in terms of the corresponding discharges, for the months of December to March. However, these indices were increasingly divergent in April and May. In these latter months, higher flow events occur more frequently, with the transition to a winter rainfall climate.
- For all sites, average daily discharge was the hydrological index most influenced by the occurrence of fresh (higher flow) events, which tended to occur towards the very beginning and end of the dry season.
- For each site, Q_{95} values were highly similar in all months, except December and sometimes May, when they were higher.
- For each site, Q_{95} values were not markedly lower than Q_{75} figures, except in December and to a greater extent in May.

For the four study sites, natural instantaneous discharges measured in each month at the control locations corresponded with a range of the above hydrological indices, depending on both the month and the site. The following trends were observed (see also Section 10.2.2):

- For all sites and months, natural instantaneous discharges were within an envelope of discharges that were lower than the corresponding historical average daily discharge and higher than the absolute minima on record.
- At the Elands site, recorded instantaneous baseflows approximated the long-term, median 7-day low flows, but were slightly higher than discharges corresponding with them in January and early March. Generally, the measured discharges were in the order of or slightly higher than the 90-95th percentile range of magnitudes.
- At the Molenaars site, measured baseflows were typically at lower than or similar magnitudes to the long-term median 7-day low flows, except in December when they were higher. For February, the instantaneous discharges fell between the median 7-day low flow and the absolute minimum. They approximated the Q_{75} for December to January, decreased to about the Q_{95} in February, and increased again to between the Q_{75} and Q_{95} values from March onwards.
- At the Du Toits site, instantaneous discharges were similar to long-term median flows and median 7-day low flows in December. Observed baseflows from January to April approximated the discharges corresponding with the median index, but were closer to the median 7-day low flow index in May. None of the discharges corresponded with Q_{95} values.
- At the Riviersonderend site, observed baseflows were similar to median 7-day low flows in December and February, greater than the long-term median discharge in January and similar to the median discharges from March to May. None of the instantaneous discharges corresponded with Q_{95} values.

These results indicate that the rivers each had a different hydrological character in the dry-season.

The experimental, unnatural reductions in baseflow during February and March (the impact phase of the low-flow study) represented the following hydrological impacts on the three experimental sites:

- A decrease in the magnitude of baseflow during two of the peak months of the dry season, February and March. The magnitude of the decrease in discharge from natural baseflow was approximate for the three experimental sites, as follows: 36% at the Molenaars site; 86% at the Du Toits site; and 86% at the Riviersonderend site. Although the latter two sites had roughly the same proportion of flow abstracted, evaluation of all observed discharge data indicated that a marginally higher proportion of flow was diverted at the Riviersonderend site.
- A duration of the period of flow reduction of roughly two months.

These unnaturally low baseflows corresponded with various calculated long-term hydrological indices of natural flow, in the period February to March, as follows:

- The 36% flow reduction in the Molenaars impact location resulted in baseflows that were of lower magnitude than the long-term, natural Q_{95} values for the river in February and March, and similar in magnitude to the absolute minima on record for these months.

- An 86% reduction in discharge in the Du Toits impact location resulted in baseflows that were significantly lower in magnitude than both the Q_{95} values and absolute minima on record for the river in these two months.
- An 86% reduction in discharge in the Riviersonderend impact location resulted in discharges that were far lower than natural Q_{95} values and just below the absolute minima on record for the river in these two months.

Hence, all three experimental sites were subjected to unnaturally low flows in February and March that represented marked physical disturbances to the riverine ecosystem. In all three instances, the observed discharges were well beyond those that the riverine ecosystem had been subjected to historically over a known period of almost 30 years.

The results summarised above had the following preliminary, broad implications for the study and for research on dry-season low flows in general:

- The use of a single hydrological index in instream flow assessments, for example the Q_{90} index, across all months of the dry season is probably inappropriate. Natural, instantaneous baseflows did not bear a consistent relationship with any single flow index over the December to May period for all four sites, or even within the recognised low-flow months of January to March. Each month exhibited an individual hydrological character, even within a season that is well defined on a hydrological basis. Although baseflows are relatively constant throughout the dry season, it is really only the peak dry month, typically February, that is highly stable. This month has the lowest range of discharges, indicating that it is least influenced by fresh events.
- In addressing low flow recommendations, it is essential to use low flow indices that are stable indicators of dry season flow conditions, in that they do not incorporate the effects of fresh events to a high degree. Indices like the average and possibly even the median daily discharge appear unsuitable in that they generally do not represent the dry-season baseflow *per se*.
- Clearly, from the above results, the observed natural range of present-day instantaneous discharges, measured at each site, is sufficiently variable that several commonly computed hydrological indices have potential merit as standard low flow indices, on a month-by-month basis. These indices are the median 7-day low flow, and 90th and 95th FDC percentiles from 1-day FDCs for each month. In the selection of appropriate low flow indices representing suitable dry-season flows for particular months, cognisance would need to be taken of which values of flow percentiles or other types of indices represented flows that a particular river usually experienced.
- The results highlight the importance of assessing additional hydrological factors such as the coefficient of variation (CV) and other measures of inherent, natural flow variability, predictability and constancy. These hydrological measures appear to be important not only in temporary systems, but possibly also in perennial rivers with seasonal flow patterns, such as the rivers used in this study. Certainly, hydrological variability or the rate of change in discharge magnitude might be at least as important as magnitude itself, in directing ecosystem response. The degree to which such factors are or are not of relevance has yet to be assessed.
- The results of this study highlighted the critical importance of antecedent events, especially hydrological events such as freshes, in directing the extent and direction of ecosystem response. For instance, the duration and intensity characteristics of individual, storm-generated rainfall events had a

marked effect on stream flow response. Such events occurred within the catchments of the individual study sites during the experimental period, often at different times and even in the characteristic low-flow months.

- It was clearly essential that the natural baseflow regime and specific low flow impacts were placed in the context of the natural, long-term hydrological disturbance history of each river. The actual magnitudes of reduction in flow were not meaningful indices, in themselves, of the degree of hydrological disturbance they represented for the river ecosystem. Similar amounts of flow (86%) were diverted from the Du Toits and Riviersonderend impact locations, although marginally more flow was abstracted in the latter location. Yet, a comparison of the remaining discharge magnitudes with flows from the available record hydrological record revealed that the Du Toits River had never experienced such baseflows historically, while similar albeit slightly higher discharges had been recorded at the Riviersonderend site. Moreover, although a markedly smaller quantity of flow was diverted in the Molenaars impact location, the resultant baseflows in February and March were also similar in magnitude to the absolute minima on record for these months.
- The unnatural flows were extreme events, not only in magnitude but also in duration. They continued over a period of roughly two months, which appears a far greater duration than natural drought events apparent from calculated flow indices.
- The study summer appeared to be a dry one relative to the long-term hydrological record. This factor might have important implications with respect to the responses of the riverine ecosystem, especially in conjunction with the character of antecedent hydrological events, both intra- and inter-annually. For instance, the biota might already be experiencing stress associated with lower than usual summer flows, or alternatively, they might already be partly prepared to cope with unnaturally low baseflows. This highlights the importance of identifying dry, wet and moderate years of record and relating observed responses to longer-term climatic trends.

11.3 ECOLOGICAL RESPONSES OF UPPER RIVERS TO NATURAL DRY-SEASON BASEFLOW REGIMES AND REDUCTIONS IN BASEFLOW MAGNITUDE BELOW NATURAL LEVELS

11.3.1 Changes in river physical and chemical variables in relation to dry-season baseflows

There were several inherent limitations and issues in the sampling of water chemistry that had implications for the reliability of the results obtained (Section 10.3), as follows:

- Single, instantaneous measurements of physical and chemical variables were used to describe the monthly status of this component of the ecosystem.
- Repeated sampling at short time intervals, during the flow reduction phase and immediately after the temporary weirs were removed, when poor quality water was flushed downstream, was absent. Hence, there may have been short-lived changes in water quality with flow reduction that were not recorded. Continuous recording would have also enabled the identification of natural background ranges in variables on a daily and monthly basis. Hence, it would have been possible to develop clearer relationships between baseflow magnitude and natural fluctuations in water chemistry.

- The focus of sampling was on still-flowing sections of river, during the impact phase of the experiment, in order to characterise the general condition of the river at natural and unnatural baseflows. Hence, there were likely areas of altered water quality within the main channel that were not sampled. Records of certain variables, like conductivity, in non-flowing, isolated pools showed some changes in chemistry in such areas.
- It was evident that cognisance needed to be taken of the biotope within which water chemistry data were recorded. Data were collected from run biotopes in all cases, particularly as this biotope was typically dominant.
- The time of day was also an important consideration, in that single measurements were made of variables that are known to fluctuate on a diel basis.

The main preliminary findings of the assessment of changes in water chemistry under natural and unnatural baseflow regimes were as follows:

- Natural variability in chemistry variables was high, rendering it difficult to ascertain clear trends in most variables in relation to flow regime. Work is underway in Tharme (in prep.) to identify any statistically significant trends from the observed data.
- Although each of the four rivers exhibited a unique chemical character, the differences among sites were generally small and broad similarities could be shown. However, there was a tendency for the two more anthropogenically disturbed sites, namely the Elands and Molenaars rivers, to pair due to elevated concentrations of variables, notably nutrients, relative to the Du Toits and Riviersonderend sites.
- Under natural summer baseflow regimes, only a few of the recorded physical and chemical variables, such as temperature, TA and conductivity, among others, showed characteristic increases to peak levels in midsummer (February), and subsequent decreases in concentration with the onset of higher flows in April.
- At natural discharges, differences in values of many chemical variables between summer months were often fairly low, indicating high overall stability during a dry season with no clear relationships with baseflow magnitude. This was evident for variables such as sodium, magnesium and DO. In contrast, a few variables fluctuated highly between months, such as calcium, nitrites and silicon.
- Several variables showed considerable natural inter-reach variability at some sites, for instance, minimum temperature (Riviersonderend), phosphate (Elands), TA (Elands) and potassium (Elands). Dissolved oxygen showed within-reach differences in range, but specifically at the level of different biotopes. Other variables, like conductivity, showed little fluctuation within a reach.
- The main, natural changes in chemistry appeared to occur at the very end of the dry season, in late March, with the initiation of a natural shift to autumn conditions. Such changes were not necessarily associated with large-scale increases in baseflow in April, however, and are likely to be a function of broader climatic conditions. However, the relationships between individual variables and discharge require further analysis (Tharme in prep.).
- It was evident from the chemistry results that a better understanding of natural variability is required before statements can be made on responses in different variables to summer baseflow regimes. Examination of long-term trends in water chemistry, specifically system variables, in relation to baseflow regimes should provide useful information in this regard (Tharme in prep.).

- Unnaturally low baseflows in February and March had the following implications for the riverine ecosystem in terms of water chemistry (see also Sections 10.3.2-10.3.7):
  - An unnatural 36% reduction in peak dry-season baseflow did not appear to markedly alter any of the measured water chemistry variables.
  - Although an extremely high reduction in baseflow magnitude of 86% resulted in possible changes in a few variables, the changes were generally small relative to the observed natural ranges. Moreover, the changes were not consistently evident at both sites at which similarly high magnitudes of flow were diverted. However, a clear change in water chemistry was observed for the Riviersonderend impact location, specifically:
    - An increase in conductivity was recorded in isolated pools that were previously linked within the main channel. Moreover, flushing of the main channel with the reinstatement of natural baseflows, in April, resulted in pulses of highly elevated levels of the correlated variables conductivity and TDS, relative to natural concentrations. These two variables did not show pronounced increases in the still-flowing section of the main channel, during the flow reduction phase of the experiment.
  - The impacts of unnaturally low discharges on the above variables were short-lived, with a recovery to natural values soon after the reinstatement of natural flows in April.
  - The direction of change in conductivity at unnaturally low flows was the same as under the natural baseflow regime at this site, but was exaggerated.
- There were some physical and chemical variables, such as temperature, TA and pH, *inter alia,* for which lesser differences were apparent between natural values and those recorded in the impact locations at unnaturally low baseflows. However, the resultant trends were not marked and would require statistical validation of their significance (Tharme in prep.).
- Several indirect effects of unnaturally low flows on water chemistry were observed, though not quantified, including:
  - Flushing out of high levels of humic acids associated with the observed desiccation and subsequent decomposition of plant matter, especially *Isolepis digitata*, was noted in the Riviersonderend impact location at unnaturally low flows. The colour of the water changed from pale gold to dark orange-brown in isolated pools in February to March, and during April flushing of the leachates.
  - A proliferation of green filamentous algae was recorded at extreme low flows, in the slow- and non-flowing areas of the Riviersonderend impact location.
  - Widespread total desiccation of non-filamentous algae on stone surfaces in the Du Toits impact location.
  - Several of these effects of flow reduction might have altered water quality in the affected reaches. For instance, an increase in humic acids might have been responsible, in part, for an observed decrease in pH. The increase in Chlorophyta might have altered nutrient levels and the concentration of DO. Desiccation of natural algal communities might have reduced or altered food availability for instream biota.
- The observed changes in chemistry during the impact and post-impact phases, in locations impacted by unnatural flow reduction, illustrated the important action of elevated baseflows or fresh events in flushing accumulated water of poor quality from the reach. This action was a vital reset mechanism for the river.

11.3.2 Effects of dry-season baseflow regimes on physical biotopes

Natural and unnatural baseflow regimes were shown to have several effects on the hydraulic character of cross-section biotopes and on biotope dynamics, as briefly summarised below (see also Section 10.4). In the majority of cases, extreme low flows resulted in an exaggeration of the main trends recorded in midsummer with natural decreases in baseflow magnitude.

The following main trends were noted under natural dry-season baseflow regimes:

- Relationships between habitat and discharge were made complex by the natural variability in both riffle and run morphology observed within the same reach of each river and between the different rivers. The geomorphological differences between mountain stream and foothill zones contributed particularly to the latter variability. However, major trends in hydraulic character with flow were usually sustained, irrespective of morphological differences within each cross-section biotope.
- Riffles differed hydraulically from runs at natural baseflows, as has been shown in several other studies. Riffles were typically shallower on average than runs, with a wider range of velocities, higher maximum velocities and greater turbulence. They also exhibited more variable depth and velocity profiles, largely as a result of an increased number of exposed bed elements. Runs typically showed more consistent relationships between depth and velocity than riffles.
- Differences in the relative proportions of biotopes within a reach at a single discharge, were functions of variations in natural channel morphology.
- At natural baseflows, runs and riffles were usually the dominant and subdominant biotopes of the four upper rivers, respectively. The mountain stream reach of the Riviersonderend, and to a lesser extent the mountain-foothill transitional Elands site, showed more biotope heterogeneity than the other two sites, without a clear dominance by runs.
- At natural baseflows, most runs did not alter entirely in type, even in the higher flow months of December and May. Instead, in the specified months, the hydraulic variables changed in magnitude, with runs becoming deeper and/or faster flowing. The only observed exception was a heterogeneous run, with a high proportion of cell riffles, in the Riviersonderend mountain-stream site. With natural elevations in baseflow in autumn, the run became transitional to a riffle and transformed entirely at high baseflow.
- At natural baseflows, none of the riffles changed in biotope type at any time during the low-flow study. However, the hydraulic variables changed in magnitude according to whether discharges increased or decreased.

A reduction of 36% of natural dry-season baseflow exerted a major impact on the available quantity and hydraulic suitability of biotopes, while discharge reductions in the order of 86% had even greater effects. The following general effects on biotopes were observed, with an increasing degree of change occurring with decreasing baseflow magnitude, as indicated. In a few instances similar changes occurred, though to a far lesser degree, under conditions of naturally low flow during February and March:

- A loss of wetted width, and hence of wetted perimeter, occurred. Changes in average wetted habitat, riffle widths and run widths clearly tracked natural and unnatural changes in discharge at all sites, although the effects of large changes in magnitude were dampened as a result of cross-section

shape. At unnaturally low flows, runs appeared to shrink in from the channel edges more than did riffles.

- A large-scale change in biotope type sometimes took place at the scale of transect (cross-section) biotopes. At low to moderate reductions in baseflow below natural magnitudes, there was no whole-scale transformation in biotope type for riffles or runs. However, at reductions in discharge of about 86% of natural baseflow, runs showed a tendency to change to pool/run transitional biotopes, primarily where depth was high. At similarly high reductions in flow, riffles either transformed partly to become riffle/run transitional biotopes, where depths were high, or transformed entirely to trickle 'runs' in shallow reaches. The latter biotope was characterised by an entire loss of turbulence, so that water trickled shallowly between cobbles, many of which were partially or wholly exposed.
- Under conditions of natural flow, even during the dry season, riffles and runs typically remained the two dominant biotopes, although the relative proportions of these and other biotopes were noticeably altered. Midsummer, unnatural reductions in baseflow resulted in various changes in the natural patterns of dominance by biotopes. With a 36% reduction in baseflow, there was no major shift from one of natural dominance by runs, and although riffles decreased in availability they remained the second most dominant biotope. At the Du Toits impact location, runs remained dominant with extreme reduction in flow, but there was a change in the subdominant biotope from riffles to pool/run transitional biotopes. In the case of the Riviersonderend impact location, the shift in dominance was marked at all levels, including the most common biotope. The shift was from naturally dominant riffles to a dominance by pool/run transitional biotopes, while pools and standing waters also increased markedly.
- Under conditions of unnaturally low baseflow, transitional biotopes assumed a more important role in the provision of physical habitat.
- Typically, there was a shift towards slow-flowing biotopes with flow reduction. Unnatural baseflow reductions of $\geq$ 86% resulted in an exaggeration of this observed natural shift, with an extreme reduction in high velocity habitats.
- At manipulated, extreme low flows there was a severe to total loss of riffles.
- A reduction in average and maximum depth was recorded at unnaturally low flows. With a 36% reduction in baseflow (Molenaars site), decreases in average depth of about 52% and 36% were evident in runs and riffles, respectively. With extreme reductions in baseflow magnitude, run depths decreased by 50% (Du Toits) and 55% (Riviersonderend) relative to natural depths. Riffle depths decreased by 46% in the Du Toits impact location relative to natural conditions, but those at the Riviersonderend site did not markedly differ from natural depths.
- An increase in the proportion of extremely shallow areas (< 0.05 m) occurred with a decrease in discharge magnitude, especially at extremely low flows.
- A reduction in average (0.6-depth velocities) and maximum velocities was sometimes observed at low flows. With an unnatural baseflow reduction of 36%, average velocities in control and impacted runs remained similar. However, average velocities in riffles were 17% lower than natural. With extreme reductions in baseflow magnitude of 86%, the average velocity in runs decreased by 43% in the Du Toits River and drastically by 81% for the Riviersonderend impact

location, relative to natural magnitudes. At extremely low baseflows, the average velocity in riffles was 66% lower in the Du Toits River, but only 18% reduced in the Riviersonderend impact location.

- At extreme low flows particularly, there were marked increases in the cross-channel proportions of non-flowing sections.
- A complete loss or pronounced narrowing of the deepest sections of the channel was recorded at unnaturally low flows.
- For runs, where the natural channel morphology included a deep main channel, as for the mountain-stream reach (Riviersonderend), effects of decreased velocities at extreme low flows were more pronounced than the impact on depth. That is, although there were remaining deeper areas, they had been transformed entirely into pool habitats.
- With flow reduction below natural February/March magnitudes, increased exposure of the substratum on the channel bottom was evident, both with shrinkage of the wetted channel margins and instream with increased channel dissection. Hence, not only edge habitat was lost through drying, but perhaps more importantly also instream habitat. This resulted in a marked loss of connectivity of wetted biotopes of various kinds at extreme low flows, as well as total isolation of some marginal and instream pools.
- At low flows, a number of alternative types of biotope were generated for potential utilisation by the biota. These included entirely isolated standing pools, trickle runs, and partially and fully exposed pockets of substratum. Such biotopes may be only partially suitable as habitat, if at all.
- Typically, the natural physical character of impacted riffles rendered them more susceptible to large-scale losses in connectivity of wetted habitat than runs, and to increases in the proportion of wetted width subjected to flow cessation.
- After the reinstatement of natural baseflows, recovery to biotope conditions similar to those in the control locations was immediate.

Other general points emerging from the assessment of changes in biotopes with increases or decreases in discharge were as follows:

- Although direct comparisons between biotopes of the same type between control and impact locations provided a useful measure of the differences associated with unnatural flow reduction, inter-reach differences in morphology within one biotope type made such comparisons less accurate.
- The wetted width of cross-section biotopes was not a very useful surrogate of changes in biotope proportions within a reach, particularly as single cross-sections undoubtedly provided a simplistic picture of changes in the availability of biotopes with flow.
- Riffles did not appear to show necessarily greater losses in the quantity of wetted habitat with flow reduction than runs. Moreover, they sometimes developed low velocity areas or retained fairly deep sections, depending on their natural morphology.
- Biotopes that were present in naturally small proportions under natural dry-season baseflow regimes, such as pools and isolated standing waters varied in terms of their presence or absence in any month or

part of the reach represented by the site. These factors made such biotopes poor indicators of changes in habitat with extreme increases or decreases in discharge.

- Biotopes recovered in physical character immediately from the impacts of flow reduction once natural flows were reinstated. Although such recovery of hydraulic conditions is likely to have accelerated recovery of the instream BMI assemblages, other biotic interactions associated with extreme low flows were likely at play (Section 11.3.3). Hence, it was found to be essential to relate changes in biotope availability and suitability to biotic change. However, biotic response also needed to be addressed independently of biotope response, especially in terms of recovery from flow reduction as a disturbance.

The main changes observed in various biotope-related factors documented above, both at naturally and unnaturally low flows, are likely to be extremely important from the perspective of the instream biota.

11.3.3 Effects of dry-season baseflow regimes on benthic macroinvertebrate assemblages

From the preliminary types of assessment of the effects of natural and unnatural baseflow magnitudes on upper rivers (Section 10.5), it is clear that the relationships between discharge and biotic response are highly complex. In most instances, the types of analyses conducted did not reveal distinct or consistent changes in the composition of BMI assemblages, in relation to either natural or unnatural, dry-season baseflows.

Possibly, more sensitive kinds of analyses are required to elucidate such patterns, including species-level analyses and assessments based on individual samples rather than on averages of sample groups. However, for the purposes of this study, the emphasis was on the identification of any broad-scale responses of invertebrates to flow reduction, using relatively simple biotic measures of response. The strengths and weaknesses of such an approach were clear from the results obtained. Although univariate indices, such as abundance and diversity, appeared to have potential in establishing trends, preliminary multivariate analyses were less successful. Classification and ordination analyses of patterns in the composition of the BMI assemblages, at the four sites were generally inconclusive at several levels of resolution (Section 10.5.5). Further analyses approaching the data in different ways, including canonical correspondence analysis and assessment of the presence and absence of rare or flow-sensitive taxa (Tharme in prep.), may enable the identification of more meaningful trends in biotic response.

In this study, attention was focused on changes in the BMI assemblages in relation to natural peak low flows and unnaturally low flows, in February. Brief discussions of various characteristics of the assemblages in early summer (December) and early autumn (April) are presented in Section 10.5. From the results of the comparative assessment of BMI responses to dry-season baseflows in the four rivers, it could be inferred that there were several possibly fundamental trends that had implications for the observed responses detailed further below:

- Between-river differences in BMI assemblage composition appeared to be more pronounced than differences related specifically to natural or unnatural baseflow regimes.
- Biotope type and substratum component appeared to have at least as much of an influence on the composition of the assemblages as baseflow magnitude.
- Patterns evident for riffle and run assemblages in their entirety sometimes differed from those observed for individual components of the substratum in each biotope. Furthermore, it was not

possible to draw generalised, summary trends from the data for various substratum components within each biotope.

The following main responses of BMI assemblages to natural dry-season baseflow regimes were recorded:

- Overall at the four sites, numbers of families and higher taxa were fairly stable over the range of baseflows, although some natural variability was apparent. There was a greater degree of variability in abundances.
- There was no consistent pattern among rivers of an overall increase or decrease in the numbers of taxa with a natural decrease in baseflow magnitude. However, numbers of taxa were typically lowest at the end of summer or in early autumn (March/April).
- The total number of individuals varied markedly among sites, within each reach and with baseflow regime.
- Numbers of taxa and individuals were typically higher in riffles than in runs.
- Lowest numbers of taxa and individuals were recorded from the tops of riffle and run stones, while stone bottoms and the underlying stones, in both biotopes, exhibited similar, higher abundances and numbers of taxa.
- With natural decreases in baseflow from December to February, there were no clear overall patterns in the numbers of taxa inhabiting riffles or runs. Moreover, changes in numbers among the various substratum components of each biotope with natural decreases in discharge were highly varied.
- Responses in numbers of invertebrate taxa, associated with the shift to autumn instream conditions, varied among both sites and biotope components.
- For all sites except the Riviersonderend, there was a decrease in the numbers of individuals in riffles and runs from December to lowest numbers of individuals overall, in February. This was followed by an increase in numbers again in April. However, within the various substratum components of the two biotopes, there were some divergences from this characteristic pattern.
- At the Riviersonderend site, the numbers of individuals in riffles and runs exhibited opposite trends to those at the other three sites, under conditions of natural baseflow. Specifically, there were increases in numbers of individuals from December to February, overall and for riffles and runs. Further, there was a consistent natural decrease to lower abundances in April.
- Comparison of all four sites, under natural baseflows, revealed that, excluding the group comprising "other taxa", typically 2-7 taxa 0.1 m^{-2} comprised the numerically dominant ones.
- The most common families throughout summer in all four rivers were Baetidae, Chironomidae, and Simuliidae. However, there were considerable differences among the rivers in the other taxa recorded at abundances $\geq$ 5% of the total assemblage. Natural variability in assemblage composition was also high within the same reach and between months.
- There were no entirely consistent trends in taxon dominance among the four sites, either with the natural decrease to low flows in February, or the subsequent transition to autumn. However, numerous small and larger-scale shifts in dominance were observed.

- A number of common taxa showed distinct changes in abundance with a natural decrease in flow, though the responses were most often site-specific. For instance, *inter alia*, Baetidae typically decreased in abundance while the numbers of Ephemerellidae increased.
- There were also few clear trends in taxon dominance with the natural change from February to April flow conditions. However, increases in taxa such as Acarina, Baetidae and Chironomidae, among others, were often observed.
- Trends in taxon dominance in riffles and runs, under natural baseflow regimes, were without consistent trends and were most frequently site-specific.
- In runs, natural changes in the contributions of various taxa to the composition of BMI assemblages were less consistent than changes in those contributions in riffles. Although both biotopes had baetids and chironomids as the most common families, simuliids were never a dominant group in runs.
- Typically, during a natural dry season, Shannon-Weiner diversity was greatest at lowest baseflow, in February, and lowest in April. However, the latter trend did not hold for the Riviersonderend mountain-stream site, where diversity was lowest at the onset of summer in December.

The following main effects of unnaturally low baseflows in February on BMI assemblages were observed:

- The numbers of families and higher taxa increased in all three impacted rivers. However, there did not appear to be a direct relationship between the proportion of flow abstracted and the observed increases.
- After the reinstatement of natural baseflows, in April, the site at which the least baseflow had been abstracted, the Molenaars, showed complete recovery in numbers of taxa. However, for both the Du Toits and Riviersonderend impact locations, where flow reduction exceeded 86% of the natural magnitudes, numbers of taxa were still marginally elevated in April.
- For the Molenaars and Riviersonderend sites, in February, with moderate and extreme baseflow reduction below natural levels, respectively, numbers of taxa increased in riffles but decreased in runs. The opposite response was observed in riffles and runs with extreme reduction in flows at the Du Toits site. Also, consistent changes in numbers of taxa were not evident for most substratum components.
- There was no obvious effect of a 36% reduction in baseflow, in February, on the overall numbers of individuals. In contrast, with an 86% reduction in discharge, there was a pronounced increase in the total number of individuals per 0.1 m^2 by 291% above natural levels, in comparison with the natural response of a decline of 21% in abundance. However, this trend was only apparent at the Du Toits site. In the Riviersonderend impact location, numbers increased only marginally above natural levels during the impact phase.
- The effects of unnatural baseflows on abundances in riffle and run biotopes showed different patterns according to the percentage flow reduction below natural magnitudes. For riffles and runs, there was an overall decrease in the total number of individuals with 36% reduction in baseflow in February (Molenaars), where the decrease occurred primarily in the underlying substratum. Hence, the general direction of change in abundance was the same as the natural one. With *c.* 86% reduction in flow, there were distinct increases in numbers of individuals in riffles, especially on the tops of stones. However, in impacted runs, there were only small increases in the numbers of individuals recorded.

- Full recovery to natural numbers of individuals was recorded at all three impacted locations, with the reinstatement of natural baseflows in April, with the possible exception of the Du Toits impact location.
- At the scale of the entire location, with unnatural flow reduction in February, there were few major shifts in taxon dominance that could be identified as markedly different from natural ones.
- A decrease in the number of simuliids, and increases in numbers of Ephemerellidae and Hydropsychidae were recorded with a 36% reduction in baseflow. With an 86% flow reduction, numbers of simuliids increased in the Du Toits and to a lesser extent the Riviersonderend impact locations. At the Du Toits impact location, chironomids and baetids declined in numbers, while few other changes in dominance were evident at the Riviersonderend site.
- Recovery in terms of dominant taxa appeared to have taken place in April in the case of the Molenaars and Du Toits rivers, but the status of the Riviersonderend impact assemblage was difficult to assess, due to several differences in the most common taxa in the control and impact assemblages.
- Trends in taxon dominance in riffles and runs, under unnatural baseflow regimes, were without consistent trends and were most frequently site-specific.
- In riffles, the main changes in dominance possibly associated with a 36% reduction in baseflow magnitude, relative to the natural assemblage, were a decline in numbers of simuliids and the presence of Hydropsychidae as a common family. At the Du Toits site, with an 86% reduction in baseflow, taxon dominance was confined to chironomids and simuliids, with the latter taxon contributing 77% of the assemblage. At the Riviersonderend impact location, chironomids remained the main component of the assemblage, as at natural baseflows, but oligochaetes were not dominant and simuliids were fairly abundant.
- In runs, with a 36% reduction in dry-season baseflow (Molenaars impact location), there was no marked change in taxon dominance. At the Du Toits site, taxon dominance was shared among a greater number of taxa under conditions of extremely low baseflow (86% reduction), than at natural baseflows. At the Riviersonderend site, there were no major alterations in taxon dominance as a result of an extreme flow reduction of 86%.
- With a moderate (36%), unnatural reduction in dry-season baseflow in February, Shannon-Weiner diversity increased marginally from natural levels, following the natural trend. However, with extreme reductions in baseflow below natural magnitudes midsummer, diversity decreased markedly.
- Results of classification and ordination of BMI abundance data for the impact phase (February), from riffles and runs at the Du Toits site, showed few trends on the basis of Bray-Curtis percentage similarities. However, there appeared to be a loss of taxa from riffles and concomitant gains in taxa in runs, at extreme low flows. Further analyses would be required to substantiate this result.

The following more general issues emerged from an assessment of the results of dry-season baseflow regimes on the BMI fauna:

- Apparent trends in variables such as numbers of taxa or abundances may be a result of factors other than or in addition to changes in the magnitude of baseflow. Moreover, such other factors might not be flow-related. As yet, the observed biotic responses to low flows are not well understood.

- It is important to consider that additional data from the low-flow study, particularly from the second month of impact, have not yet been analysed. It is possible that these data might show more pronounced or different types of response, with extension of the duration of the low flow impact into a second month. In particular, the intensity of biotic interactions would be expected to potentially increase during such a protracted period of unnaturally low flow, even though similar proportions of different biotopes would still be available.
- Changes at the species-level within the various families might be important, especially in the dominant families, Baetidae, Chironomidae and Simuliidae, that typically comprise the greatest proportion of the overall BMI assemblage in any month.
- Examination of the loss of families and/or species with specific hydraulic requirements might allow the identification of sensitive or biotope-specific taxa, in comparison with more generalist groups tolerant of widely fluctuating dry-season baseflows (Tharme in prep.).
- A more detailed examination of the responses of the BMI assemblages to the first major increase in discharge magnitude with reinstatement of natural baseflows might provide information on the paths by which recovery was achieved at each of the experimental sites (Tharme in prep.)
- Observed changes in the assemblage composition of riffles and runs, for instance an increase in the number of riffle taxa and concomitant decrease in numbers of run taxa at unnaturally low baseflows, are likely to be at least partially a function of changes in the hydraulic character of these biotopes. At low flows, many riffle patches exhibited ranges in hydraulics transitional or similar to those of runs. Certainly, several of the BMI samples collected from riffles were potentially representative of riffle/run transitional biotopes. This factor highlights the need to ascertain whether or not transitional biotopes have their own suite of BMIs or merely provide more marginal benthic habitat.
- Changes in the distribution of taxa between biotopes with flow reduction appear to be occurring, with shifts in assemblages in response to changes in hydraulic characteristics of individual biotopes and shifts in the dominance of the major biotopes. For example, with a 36% reduction in baseflow (Molenaars data), taxa appeared to have moved from runs into riffles, possibly because the hydraulics of riffles more closely matched those of runs at low flows. Moreover, fewer riffle areas and more runs were available as habitat. With a far higher percentage reduction in flow (Du Toits data) the converse response was, however, observed. This was possibly a result of a major loss in available riffle areas and a vast increase in the proportion of runs.
- It is unclear as to why a far lesser response to extreme low flows was evident at the Riviersonderend than at the Du Toits site. A few possible factors proposed here might be partly responsible for this trend. All of these factors would require considerable further work in order to assess their significance:
    - Examination of the long-term historical record indicates that this site has been subjected to a more variable hydrological regime in summer than the other three sites, with very low natural flows. Hence, a reduction in baseflow of 86% might have less dramatic effects in this river than at the Du Toits site.
    - As the Riviersonderend is the only site located entirely within a mountain-stream zone, it is more heterogeneous in biotope composition, with less differentiation between smaller-scale biotopes than the other three sites.

  - Large instream and marginal pools occur at this site, and therefore possibly assume a greater importance as refugia than at the other sites.
  - Possibly, biotic response occurred more at the level of species tolerant of low-velocity conditions at this site. Such responses would not necessarily be detected at the family level.

- The responses of the assemblages to extreme flow reduction might have been extremely rapid, in which case, durations of a few days before the first sampling event after flow reduction would have been too long a time to wait before sampling. In contrast, the responses may have been indirectly associated with changes in the composition of the assemblage, at a functional rather than structural level. For example, the large-scale loss of rare (low abundance) or long-lived taxa that have important roles as predators could gradually have altered the assemblage composition beyond the time frame of the study. There is little existing information on the temporal scales of invertebrate response to low flow impacts in rivers to assist in this regard.

- Sampling issues such as missing taxa due to over-sampling, under-sampling in relation to natural variability, sampling methods, subjective identification of sampling areas and the availability of suitable sampling points at low flows might have contributed to the difficulties in ascertaining clear trends in BMI assemblage composition with changes in baseflow.

- Changes in BMI composition with changes in flow were probably more difficult to detect due to various poorly understood compounding factors. These would include factors such as variable access to the hyporheos and associated differences in substratum layering in each of the four rivers, facultative utilisation of biotopes other than riffles and runs, drift, life cycle generation times and emergence patterns.

- Factors such as the resistance and resilience of the BMI assemblages in relation to flow-related physical disturbance are likely to have played a role, in the extent to which responses to low flows were observed. Unnaturally low flows might also result in longer-term stresses on the biota that might not be detectable using the types of data collected in this study.

- Although comparison of observed flow data with the long-term record suggests that the BMIs are unlikely to be adapted to cope with the extreme low flow conditions they were subjected to during the low-flow experiment, there is little evidence at this stage of major impacts on the assemblages. However, analysis of the observed biotic responses in conjunction with the pronounced changes in the quantity and quality of available habitat, is likely to show marked alterations to assemblage composition at a broader reach scale.

In conclusion, although various changes were observed in all ecosystem components at low flows, much as expected, they were not very obvious. Further investigation underway in the more detailed study (Tharme in prep.) might show clearer trends, and provide reasons for the kinds of responses observed.

12. A SEARCH FOR THE LINK BETWEEN WETTED PERIMETER AND EASILY COMPUTED HYDROLOGICAL INDICES OF BASEFLOW CHARACTERISTICS

Written by J. Hollands (then at the University of Stellenbosch)

12.1 INTRODUCTION
12.2 OBJECTIVES
12.3 DESCRIPTION OF THE DATA BASE
12.4 METHODS
12.4.1 Fieldwork
12.4.2 Statistical analysis
12.5 RESULTS
12.5.1 Mean annual runoff
12.5.2 Percentile of the flow duration curve
12.5.3 1:2 year maximum average d5aily flow
12.6 DISCUSSION

This Chapter summarises a one-year Masters degree project done by Mr Jacques Hollands. The work was funded by this project and registered at the Department of Civil Engineering, University of Stellenbosch. The project thesis is entitled ***Investigation towards the linking of ecologically meaningful low flows and hydrological indices for south-western Cape rivers*** *(Hollands 1996). Project supervisors were Prof. A. Görgens of the University of Stellenbosch, and one of the authors of this report (JMK). The objective of the research was to establish, in several rivers, the discharge at which the loss of wetted perimeter accelerates as the wetted channel shrinks in from the banks, and to link this to easily computed hydrological indices.*

12.1 INTRODUCTION

When compiling a modified flow regime for management of a river subjected to a water resource development, there may be a need to reduce the baseflow of the river from its pristine condition, so that some of the water can be allocated to offstream users. Reducing baseflows without causing undue damage to the river ecosystem is a management challenge, for which there are as yet no objective guidelines in South Africa. When this management decision is needed for a stream which has not been monitored in terms of biota, channel morphology or flow patterns (as often happens in planning studies), the problem threatens to become intractable.

The study described here is a tentative step towards finding a guideline that has its roots in the abiotic realm, by focusing on the wetted perimeter-discharge characteristics of streams. Essentially, as the discharge of a river decreases, there is a discharge at which the loss of wetted perimeter accelerates as the wetted channel

shrinks in from the banks (King and Tharme 1994). The project was designed to identify this discharge in several western Cape headwater rivers and to attempt to link it to one or more easily computed hydrological indices. If such a link could be made, it could act as a guideline in future situations where the water requirements for river maintenance were being considered but fine-resolution (daily) hydrological data were unavailable. Hydrological indices of a common nature are often available on a regional basis or can be generated from regional relationships or via catchment models. It was hoped that the general availability of such indices would provide a key to making reasonable first-approximation inferences about critically low discharges at unmonitored sites.

12.2 OBJECTIVES

At the start of this study it was assumed that accelerated loss of wetted bed through lateral shrinkage would occur at a critically low discharge which would detrimentally affect the riverine ecosystem (Gordon *et al.* 1992). The focus of the study was the hydrological-hydraulic part of this assumption, while the biotic part of this assumption is reported on in Chapters 7-11. The objectives reported on here were:

- for several anthropogenically undisturbed rivers in the south-western Cape, to seek an inflection point at the lower end of the wetted perimeter-discharge curve, which would reflect the low discharge at which the shrinking of the wetted channel accelerated;
- to search for a relationship between this discharge and easily quantified hydrological indices.

For each river, the discharge indicated by the inflection point was termed the *Wetted Bed Flow (WBF)*. This WBF became the focus of the research. In the context of this chapter, "flow" is equivalent to "discharge".

12.3 DESCRIPTION OF THE DATA BASE

The hydrological records held by DWAF were searched for long-term daily flow records of relatively undisturbed streams in the south-western Cape. These would be needed to link the WBF with hydrological indices. Following the selection of 40 historical hydrological record sets identified in the report by King and Tharme (1994) as statistically homogeneous, field visits were conducted by the author of this chapter and one of the authors of the report (RET) to further assess anthropogenic disturbance, state of the gauging weirs and potential of the river channels for surveying purposes. Finally, eleven rivers were chosen for the research (Table 12.1). These rivers are scattered over a wide area of the south-western Cape. Some of their major features are summarised in Table 12.1, as are the initial survey dates.

The following data for each river were assembled:

- long-term daily flow records at DWAF flow gauges, acquired from the Computing Centre for Water Research (CCWR), Pietermaritzburg;
- cross-sectional data in the vicinity of each DWAF flow gauge, obtained through field surveys;
- discharge data obtained through direct stream measurements with an AA Price current meter.

Table 12.1 Characteristics of the study rivers.

	Langrivier	Zachariashoek	Bakkerskloof	Kasteelskloof	Du Toits	Riviersonderend	Molenaars	Hex	Little Berg	Elands	Banhoek
Weir no.	G2H007	G1H014	G1H018	G1H016	H6H007	H6H008	H1H018	H2H006	G1H008	H1H017	G1H019
First survey date	24/04/94	11/04/94	13/04/94	07/04/94	04/04/94	03/05/94	02/03/94	06/04/94	18/04/94	28/02/94	05/05/94
Latitude	33°59'13"	33°49'39"	33°49'21"	33°49'19"	33°56'19"	33°03'44"	33°43'24"	33°34'39"	33°18'41"	33°44'00"	33°54'44"
Longitude	18°58'13"	19° 02'08"	19°02'50"	19°03'35"	19°10'17"	19°04'23"	19°10'13"	19°30'12"	19°94'31"	19°04'31"	18°56'36"
MAR ($m^3 \times 10^6$)	3.719	1.173	2.570	2.173	32.571	59.635	132.00	104.200	72.640	89.000	18.801
Catchment area (km^2)	2.5	2.8	3.4	3.3	46.0	38.0	113.0	703.0	395.0	61.0	25.0
Length of flow record (y)	32	29	29	29	29	29	24	13	31	24	28
Vegetation	Indigenous riparian fynbos	Indigenous riparian fynbos	Indigenous riparian fynbos	Indigenous vegetation	Indigenous riparian fynbos	Indigenous riparian fynbos	Indigenous riparian fynbos	Alien riparian vegetation	Alien riparian vegetation	Indigenous riparian fynbos	Alien riparian vegetation
Type of stream	Near pristine mountain stream	Lower mountain stream	Mountain stream	Mountain stream	Upper foothill zone	Upper foothill zone	Foothill zone	Foothill zone	Lower foothill zone	Lower mountain stream	Foothill zone

12.4 METHODS

There were two main activities: field work and statistical analysis.

12.4.1 Fieldwork

Four cross-sections representing two biotopes, two riffles and two runs, were surveyed in at each river. On the first visit the channel morphology was surveyed in fine detail at each cross-section. This was later repeated for the smaller and some of the larger rivers, so that any changes that might have occurred to channel shape due to the winter floods could be considered.

On the first visit, and on at least two other visits, the water surface elevation was surveyed in several times at each cross-section, and the discharge measured. Discharge values obtained from DWAF's hydrological records of the gauging station were also used to check the measured discharges.

With this information, the hydraulic roughness variations of each cross-section were calibrated. A backwater analysis was done for each river, in order to calculate the wetted perimeter for a range of discharges, and wetted perimeter-discharge curves were then generated. It should be noted that for the lower end of the range of discharges, hydraulic roughness values required for the backwater model calibration were considerably higher than roughness values required for the higher discharges. This phenomenon has also been reported by Dr George Heritage (ÇWE, University of the Witwatersrand pers. comm.) for other hydraulic modelling for ecological purposes.

The WBF was, by definition, related to a low discharge and was identified as the point on these curves where, mathematically, the rate of shrinkage of wetted perimeter with decreasing discharge accelerated sharply (Figure 12.1). For some of the rivers, the nature of particular cross-sections was such that they yielded wetted perimeter-discharge curves that were relatively flat. Identification of the WBF for such cross-sections was considered too speculative to be acceptable. In all cases, however, the final choice of the WBF for any one river was based on at least three cross-sections.

12.4.2 Statistical analysis

The hydrological data acquired from CCWR were subjected to a range of statistical analyses in order to determine a selection of indices commonly used by hydrologists, geomorphologists and ecologists. Indices calculated ranged from intuitive ones such as MAR, to more unusual ones such as the 7-day cumulative low flow.

Bearing in mind that the hydrological descriptors most easily obtained by hydrological modellers are the MAR and flow duration curves, the WBF was first linked to these coarser measures. The data were then subjected to monthly and seasonal analyses which included, for each month or season, the mode, the median, and the median value of the 7-day consecutive lowest flow spell.

The data were transformed in order to eliminate scale effects and to compare different indices with one another. As an example, the data manipulations related to MAR are explained. Firstly, for each river, the MAR was expressed as an instantaneous discharge in $m^3 s^{-1}$. Then the WBF was expressed as a proportion

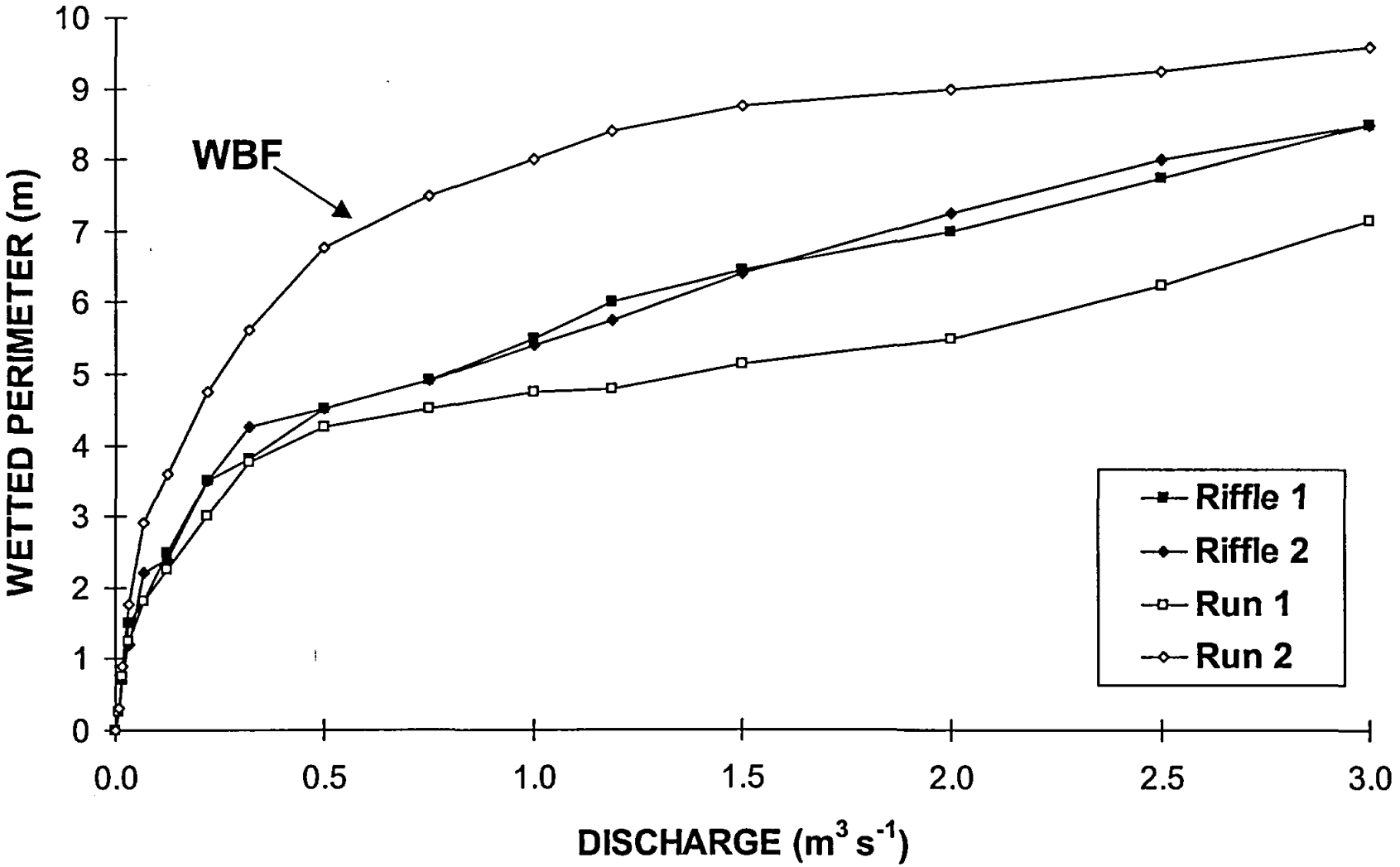


Figure 12.1 ***Wetted perimeter-discharge curves for four cross-sections of the Du Toits River, showing the inflection point or Wetted Bed Flow (WBF).***

of the converted MAR. After doing this for all the rivers, the average proportion of the MAR was calculated. The ratio of WBF to this average was then determined for each river. On the y-axis (Figure 12.2), 1.00 will represent the average of the eleven rivers while a narrow range of deviations of the individual values from 1.00 will indicate a relatively consistent index (and therefore a useful index), in WBF terms. For re-scaling, the absolute average of each index is also provided on each graph.

12.5 RESULTS

The most promising indices in terms of consistency across the full sample of rivers were the MAR, the percentile of the Flow Duration Curve represented by the WBF, and the 1:2 Year Maximum Average Daily Flow (MADF) (Table 12.2). Each of these indices is discussed briefly. Certain promising values shown graphically in Figure 12.2.

12.5.1 Mean annual runoff

The distribution of the WBF: MAR values lies between 0.70 and 1.40 (Figure 12.2). This is quite a large range, but the outliers (the Hex and Little Berg Rivers) are known to have abstractions for irrigation upstream of the gauging stations, despite their having passed the statistical screening reported in King and Tharme (1994). For these rivers, then, the observed MAR at the gauging weir will be smaller than the natural MAR, resulting in higher values for the WBF ratio than would be the case if the rivers were undisturbed. When these two rivers were omitted from the analysis, the mean WBF across the range changed, and a usefully tight envelope of values ranging from 0.78 (29% of converted MAR) to 1.13 (42% of converted MAR) was found.

12.5.2 Percentile of the daily flow duration curve

For each river, the flow duration curve was calculated from all the average daily discharges in the year. The WBF was then read off as a percentile from that curve, and the percentiles averaged as for the MAR. As with the MAR, a promising range of values for the WBF ratio was found, ranging from 0.75 to 1.18 (Figure 12.2). Again the Little Berg and Hex were outliers, surmised to be due to the upstream irrigation impacts. When these two rivers were omitted, the remaining values ranged more tightly between 0.86 (40th percentile) and 1.12 (53th percentile).

12.5.3 1:2 year maximum average daily flow

As the higher flows, including floods, play a large role in dictating channel shape, it was thought that the WBF might be linked to one or more of the high-flow statistics. From the flow record, the maximum average daily flow for each year was taken and ranked to determine the 1:2 year recurrence interval event. These were then averaged as for the MAR.

The envelope of values in WBF ratio ranged from 0.63 to 1.18, with Bakkerskloof River as an outlier (Figure 12.2). This river has a bedrock bed, which will dictate channel shape, whereas the other rivers are cobble-bed rivers, the shapes of which can be moulded by flows. Thus, in a bedrock river, the relationship between channel shape and discharge could be expected to be less obvious than in a sand-bed or cobble-bed river. With the Bakkerskloof River excluded, the remaining values for the WBF ratio range more tightly

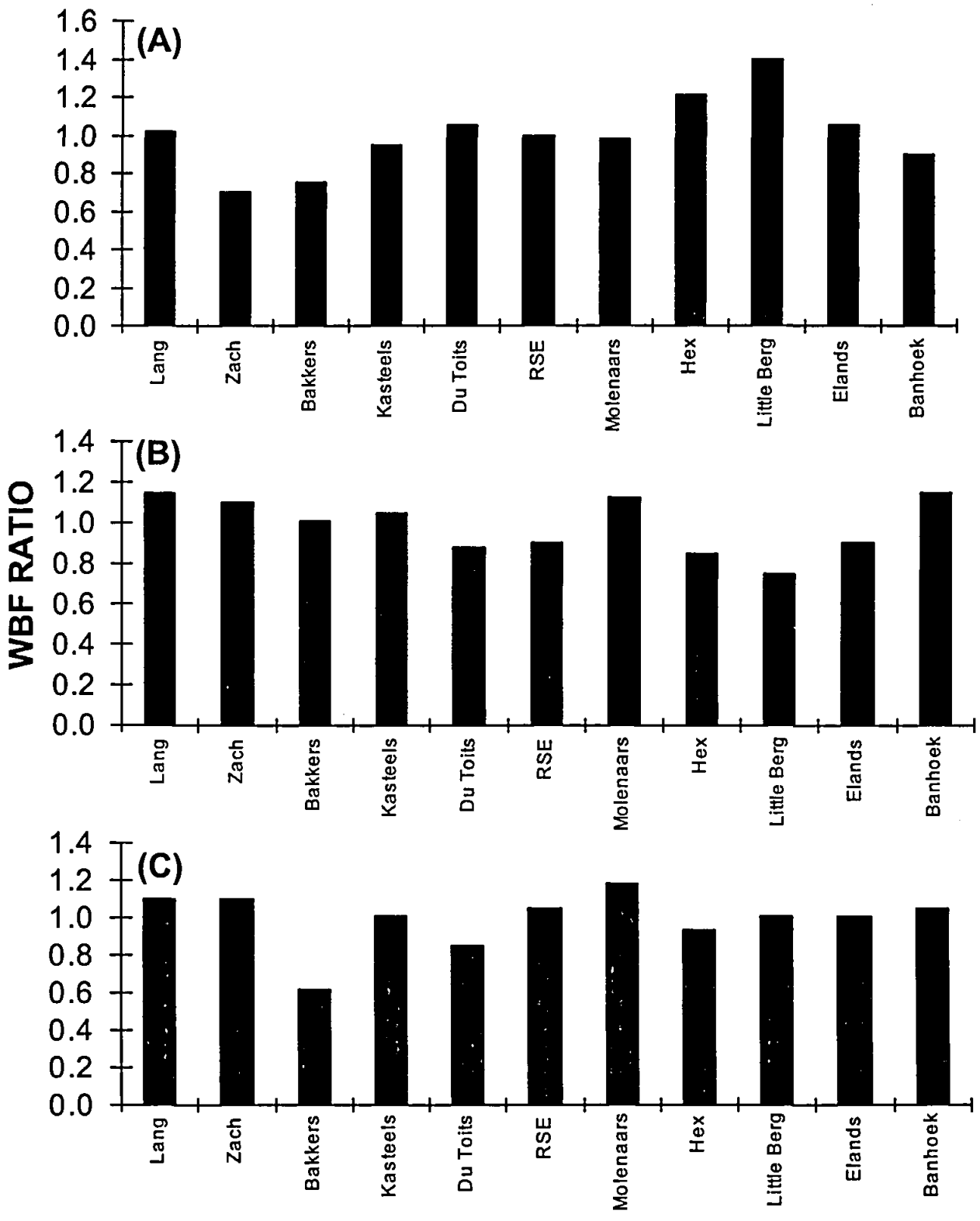


Figure 12.2 Wetted Bed Flow (WBF) as a ratio of: (A) mean annual runoff (MAR); average = 37.2%; (B) percentile of flow duration curve; average = 46.9%; (C) 1:2 year maximum average daily flow (MADF); average = 2.5%. See Table 12.1 for the full names of rivers. RSE – Riviersonderend.

Table 12.2 Hydrological indices calculated for the eleven study rivers.

			Langrivier	Zachariashoek	Bakkerskloof	Kasteelskloof	Du Toits	Riviersonderend	Molenaars	Hex	Little Berg	Elands	Banhoek
Weir no.			G2H007	G1H014	G1H018	G1H016	H6H007	H6H008	H1H018	H2H006	G1H008	H1H017	G1H019
WBF	Value ($m^3 s^{-1}$)		0.045	0.010	0.020	0.030	0.400	0.700	1.500	1.500	1.200	1.100	0.200
	mm/d		1.555	0.309	0.508	0.785	0.751	1.592	1.147	0.184	0.262	1.700	0.691
1:2 Yr	Value ($m^3 s^{-1}$)		1.624	0.358	1.260	1.160	17.888	26.300	49.200	62.500	45.700	41.800	7.390
MADF	%		2.8	2.8	1.6	2.6	2.2	27.0	3.0	2.4	2.6	2.6	2.7
1:5 Yr	Value ($m^3 s^{-1}$)		2.102	0.879	1.940	2.190	24.966	28.800	57.300	101.000	69.200	60.900	11.200
MADF	%		2.1	1.1	1	1.4	1.6	2.4	2.6	1.5	1.7	1.8	1.8
1:10 Yr	Value ($m^3 s^{-1}$)		2.895	1.260	2.400	2.380	25.835	29.400	59.100	107.000	113.000	74.300	18.801
MADF	%		1.6	0.8	0.8	1.3	1.5	2.4	2.5	1.4	1.1	1.5	1.1
MAR	Value ($m^3 x 10^6$)		3.719	1.173	2.257	2.713	32.571	59.635	132.000	104.200	72.640	89.000	18.801
	%		38.2	26.9	27.9	34.9	38.7	37.0	35.8	45.4	52.1	39	33.5
Median	Annual	Value	0.055	0.011	0.016	0.032	0.430	0.550	1.590	1.040	0.550	0.880	0.270
		%	82	91	125	94	93	127	94	144	218	125	74
MSR	Value (m^3)		129749	21486	28082	43222	1039420	1899905	3594636	2095515	1029996	2343156	450338
	%		90	121	185	180	100	95	108	186	302	122	115
MMR	Value m^3	Jan	99301	18406	18662	33116	904900	1179612	2499248	2318293	519297	1473674	286488
		Feb	75700	13383	8650	21264	552337	857959	1815835	1611032	300948	888738	151426
		Mar	94632	15040	16698	31835	778604	1697202	3034052	1658059	663296	2058904	264158
		Apr	190855	18838	34573	54447	1223208	3432044	5144759	2174869	1418872	4548222	886090
		May	348395	61588	196694	190586	2912386	7589921	16444489	10735494	5673247	11570523	1889477
		Jun	627137	192598	444646	442241	4352998	9690883	23004263	14156450	11347876	20528950	2893603
		Jul	643386	278128	539168	482853	5072785	10181702	24274151	22089283	15100323	15460389	3930168
		Aug	620044	259407	493543	427584	7343478	10025132	20925855	20885679	16997394	13578196	3207629
		Sep	406939	159294	288678	285052	3877834	6442250	15277436	17276975	9770747	8876445	2434915
		Oct	294659	86280	140090	140662	2636910	4305249	8643768	6502879	5856989	4632532	1743547
		Nov	173640	38155	54790	68500	1493770	2322970	4706820	2375819	2354771	2713219	643512
		Dec	144366	25093	35117	50168	1283702	1909443	4367099	2435020	922794	2376181	470352
	%	Jan	121	146	287	243	118	159	161	173	619	218	1122
		Feb	144	181	559	341	175	197	200	225	965	327	1917
		Mar	127	178	321	252	138	110	132	242	485	156	1217
		Apr	61	138	150	143	85	53	76	179	219	68	351
		May	35	43	27	42	37	25	24	37	57	28	170
		Jun	19	13	12	18	24	19	17	27	27	15	107
		Jul	19	10	10	17	21	18	17	18	21	21	82
		Aug	19	10	11	19	15	19	19	19	19	24	100
		Sep	29	16	18	27	27	28	25	23	32	35	128
		Oct	41	31	38	57	41	44	46	62	55	69	184
		Nov	67	68	95	114	69	78	83	164	132	115	483
		Dec	83	107	153	160	83	98	92	165	348	135	683

Table 12.2 Continued.

			Langrivier	Zachariashoek	Bakkerskloof	Kasteelskloof	Du Toits	Riviersonderend	Molenaars	Hex	Little Berg	Elands	Banhoek
Weir no.			G2H007	G1H014	G1H018	G1H016	H6H007	H6H008	H1H018	H2H006	G1H008	H1H017	G1H019
WBF			0.045	0.010	0.020	0.030	0.400	0.700	1.500	1.500	1.200	1.100	0.200
FDC (annual)		%	54	52	48	50	42	43	53	40	35	44	55
Flow Duration Curve (monthly)	%	Nov	70	60	25	30	30	25	35	13	13	14	42
		Dec	28	20	13	13	15	15	20	8	6	12	15
		Jan	12	10	8	8	8	8	12	2	4	8	5
		Feb	8	7	1	0	3	8	8	3	0	1	3
		Mar	10	5	5	8	10	12	10	5	2	6	6
		Apr	23	15	9	21	22	35	36	15	9	25	27
MODE	Value	Nov	0.079	0.010	0.029	0.048	0.591	0.774	1.074	0.620	2.285	2.279	0.102
		Dec	0.024	0.008	0.018	0.035	0.337	0.435	0.754	1.071	0.472	0.870	0.121
		Jan	0.026	0.006	0.007	0.012	0.087	0.576	0.843	1.733	0.173	0.710	0.130
		Feb	0.026	0.005	0.002	0.010	0.125	0.341	0.495	0.486	0.040	0.272	0.092
		Mar	0.023	0.006	0.010	0.021	0.358	0.500	1.103	0.549	0.193	0.550	0.119
		Apr	0.151	0.006	0.025	0.028	0.228	0.578	1.415	1.336	0.489	0.932	0.265
	%	Nov	57	100	69	63	68	90	140	242	53	48	196
		Dec	188	125	111	86	119	155	199	140	254	126	165
		Jan	173	167	286	250	460	122	178	87	694	155	154
		Feb	173	200	1000	300	320	205	303	309	3000	404	217
		Mar	196	167	200	143	112	140	136	273	622	200	168
		Apr	30	167	80	107	175	121	106	112	245	118	75
MEDIAN	Value	Nov	0.052	0.012	0.013	0.025	0.258	0.389	1.240	0.739	0.316	0.663	0.178
		Dec	0.039	0.007	0.006	0.016	0.199	0.252	0.912	0.877	0.141	6.000	0.068
		Jan	0.029	0.006	0.003	0.012	0.146	0.172	0.646	0.841	0.070	0.335	0.048
		Feb	0.023	0.005	0.002	0.010	0.133	0.174	0.551	0.554	0.092	0.280	0.032
		Mar	0.021	0.005	0.002	0.009	0.130	0.195	0.538	0.345	0.111	0.291	0.029
		Apr	0.025	0.005	0.003	0.013	0.141	0.419	0.812	0.467	0.214	0.463	0.082
	%	Nov	87	83	154	120	140	180	121	203	380	166	112
		Dec	115	143	333	186	201	278	164	171	851	18	227
		Jan	155	167	667	250	274	407	232	178	1714	328	417
		Feb	196	200	1000	300	301	402	272	271	1304	393	625
		Mar	214	200	1000	333	306	359	279	435	1081	378	690
		Apr	180	200	657	231	284	167	185	321	561	238	244
7 Day Low Flow Analysis (median)	Value	Nov	0.043	0.009	0.009	0.019	0.233	0.255	1.020	0.638	0.191	0.511	0.126
		Dec	0.033	0.006	0.005	0.013	0.163	0.175	0.725	0.568	0.099	0.396	0.052
		Jan	0.025	0.005	0.002	0.011	0.129	0.135	0.560	0.641	0.034	0.288	0.028
		Feb	0.021	0.005	0.002	0.009	0.126	0.132	0.474	0.394	0.051	0.242	0.018
		Mar	0.018	0.004	0.002	0.008	0.120	0.146	0.491	0.300	0.077	0.254	0.018
		Apr	0.018	0.005	0.002	0.010	0.121	0.175	0.550	0.357	0.129	0.272	0.030
	%	Nov	105	111	222	156	172	275	147	235	628	215	159
		Dec	136	167	400	231	245	400	207	264	1212	278	385
		Jan	180	200	1000	273	310	519	268	234	3529	382	714
		Feb	214	200	1000	333	317	530	316	381	23523	455	1111
		Mar	250	250	1000	375	333	479	305	500	1558	433	1111
		Apr	250	200	1000	300	331	400	273	420	930	404	667

between 0.83 (2% of 1:2 year average daily flood peak) and 1.14 (2.9% of 1:2 year average daily flood peak).

12.6 DISCUSSION

The relative consistency of the three sets of WBF ratios over the range of rivers holds much promise that the WBF in south-western Cape rivers can be calculated from some coarser hydrological statistic. However, it should be noted that the WBF approximates the median flow of each river (on average, the 47th percentile of the flow duration curve). Another interesting factor is that the average WBF expressed as a factor of the Mean Dry Seasonal Runoff lies in the order of 120%. These two findings indicate that, on average, for half the year the flow is lower than the WBF.

It had been expected that the WBF would be a very low flow, which had tentatively been named the critical minimum. The critical minimum flow had been seen as a value below which, for ecological purposes, discharges should never drop. It had been expected that some higher flow would be more acceptable as the low flow necessary for ecosystem maintenance, with the WBF or critical minimum being the level that flow could drop to in drought years only.

This research has shown that not to be so. A parallel study in the Freshwater Research Unit (Chapters 7 to 11) in which ecologically relevant low flows from a biological perspective are sought, might now be expected to show that flows that are **lower** than the WBF, and **not higher,** will be acceptable for ecosystem maintenance. This is because the rivers reported on here are relatively undisturbed, and must experience flows lower than the WBF for about half the year, probably during the dry season. They must therefore experience wetted-bed shrinkage in the normal course of events and biota must be adapted for such seemingly adverse conditions. Nevertheless, the WBF is a useful first point when considering ecologically-acceptable low flows, as it provides an order-of-magnitude indicator of a low flow that is higher than that where serious impacts become obvious.

A consideration for immediate further investigation is that a purely abiotic view of the flow coverage of the river bed might still yield an ecologically useful critical low flow index. This might be determined by the inter-connectivity of the lowest portions of the wetted bed at flow stages well below the WBF identified here, rather than the extent of bed coverage *per se*.

Until the results of such parallel studies are available, it has been concluded that the definition of a flow lower than the WBF for ecosystem maintenance should **not be based solely on wetted perimeter, but primarily on biological considerations.**

13. SPAWNING BEHAVIOUR AND EARLY DEVELOPMENT OF THE CLANWILLIAM YELLOWFISH (*BARBUS CAPENSIS*; CYPRINIDAE), LINKED TO EXPERIMENTAL DAM RELEASES IN THE OLIFANTS RIVER, SOUTH AFRICA

Written by J. Cambray (Albany Museum), J. King and C. Bruwer (DWAF)

13.1 INTRODUCTION

13.2 STUDY SITE

13.3 THE STUDY SPECIES

13.4 METHODS
13.4.1 Water release strategy and associated hydraulic measurements
13.4.2 Water chemistry
13.4.3 Biological data collection
13.4.4 Age determination of *Barbus capensis* embryos and larvae
13.4.5 Definitions of terms used

13.5 RESULTS
13.5.1 Local hydraulics
13.5.2 Water quality
13.5.3 General overview of biological results
13.5.4 Fish movements observed (December 1994)
13.5.5 Spawning activity (December 1994)
13.5.6 Egg development (1993 and 1994)
13.5.7 Larval and juvenile habitat (1993 and 1994)
13.5.8 Larval development (1993 and 1994)
13.5.9 Summary of developmental stages linked to dam releases

13.6 DISCUSSION
13.6.1 Spawning season of the genus *Barbus*
13.6.2 Spawning and reproductive style of *Barbus capensis*
13.6.3 Linking dam releases with spawning of *Barbus capensis*
13.6.4 River regulation and spawning in other southern African river systems
13.6.5 Management of flows in the Olifants River for conservation purposes
13.6.6 Management of spawn beds at Clanwilliam
13.6.7 Future work

13.7 CONCLUSIONS

This chapter describes a short collaborative research project designed to investigate if a serial-spawning fish species could be induced to spawn by freshes released from an upstream dam. A revised version of this chapter is published by the same authors under the same title in Regul. Rivers: Res. Mgmt. 13: 579-602 (1997).

13.1 INTRODUCTION

This chapter deals with initial research on the importance of one of the flow components addressed in the BBM, the freshes. The objective of the research was to investigate the importance of freshes to the breeding behaviour of the Clanwilliam yellowfish *Barbus capensis* Smith 1841. This was done by analysing past records of releases from the Clanwilliam Dam, devising and releasing experimental freshes, and recording the results in terms of water quality, fish behaviour, and egg production and development.

13.2 STUDY SITE

The Olifants River in the Western Cape of South Africa (Figure 13.1) was chosen for the study because it has two large, instream dams and has the highest conservation priority of any river in the region, ranking in the top few nationally. It rises in a high-rainfall (1400 mm y^{-1}) mountain range, and its mainstem is a powerful, perennial river 260 km long (King and Tharme 1994). Most of its catchment of 46 000 km^2 (Morant 1984) lies in a semi-arid area, however, and overall it thus contributes only 2% of South Africa's MAR (King *et al.* 1979). The river flows through a winter rainfall region, with about 85% of MAP occurring between the months of May to September (King and Tharme 1994). Before development of the catchment and construction of the two dams, showers in spring and early summer (October-December) brought small pulses of flow, or freshes, downstream, with these gradually decreasing in number and size to be followed by about four dry months of baseflow with relatively little fluctuation.

The two instream impoundments are in the middle reaches of the river (Morant 1984). Bulshoek Weir, 23 km downstream of the town of Clanwilliam, was built in 1919 and for ecological purposes effectively acts as an instream dam wall. With a storage capacity of 7.5 x 10^6 m^3, its reservoir feeds water to the lower Olifants River irrigation scheme through an irrigation channel. The main flows in the river below the weir are winter spillage, controlled releases for irrigation and summer return flows from agricultural land. Upstream, at Clanwilliam, Clanwilliam Dam wall was constructed in 1932, raised in 1966 and the reservoir has a storage capacity of 127 x 10^6 m^3 (Pitman *et al.* 1981). Water is stored there for downstream release on demand to Bulshoek Weir. The two dams have effectively fragmented the river's fish populations along the mainstem into three semi-isolated communities. Fish have been seen massing below their walls (Harrison 1939; Scott 1982), but cannot move further upstream as fish ladders have not been incorporated into the structures. Downstream movement may still occur through fish being swept over the dam walls or through the release structures.

The investigations reported on here resulted from observations made during a fish survey of the Olifants system between 20-31 January 1992 for a different purpose (King and Tharme 1994), when a few eggs and free embryos of *B. capensis* were found 1.5 km downstream of Clanwilliam Dam by one of us (JAC). The site is 125 km from the source of the river, at a latitude of 32° 10' 26"E and longitude of 18° 52' 16"S. It may be unique in terms of fish spawning, because it is the only known area of cobble bed in the stretch of river between the two dams. A road bridge arching over it allows excellent viewing of fish behaviour on the cobble bed.

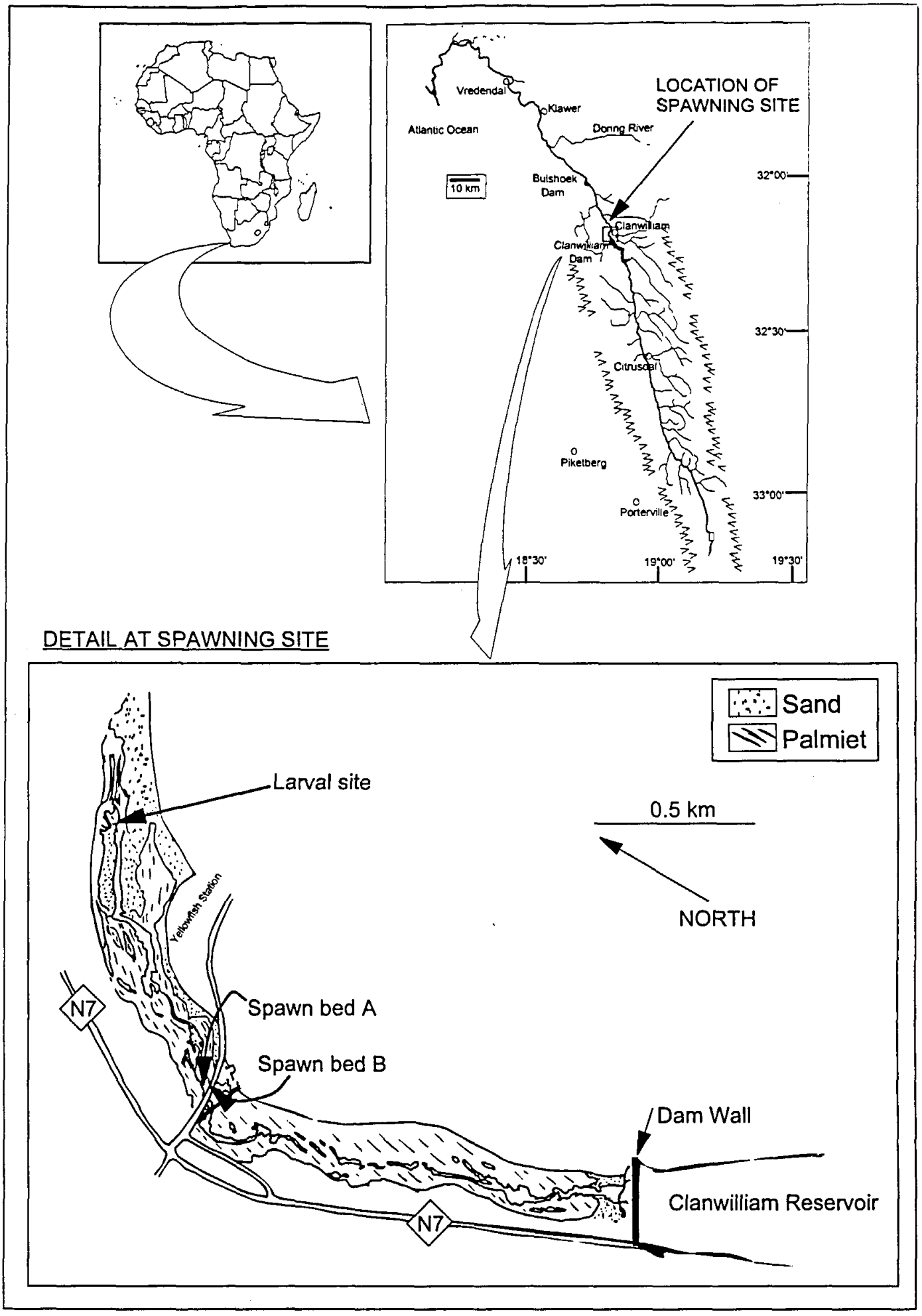


Figure 13.1 **Olifants River system (western Cape) showing the *Barbus capensis* spawn beds A and B, the road bridge and the larval fish site below the Clanwilliam Dam wall.**

The river at this spawning site is about 30 m wide, with multiple channels separated by dense belts of palmiet, *Prionium serratum* (Juncaceae). The eggs and free embryos were found in a shallow, minor channel characterised by a lack of instream or overhead vegetal cover and a substratum of 50% small boulder, 30% small cobble and 20% large gravel (Figure 13.1: spawn bed B). They were lodged under a large boulder between the cobbles and gravel, in a current speed of 0.2 m s^{-1} and water depth of 0.22 m. Upstream towards the dam wall, the anastomosing channels are more pool-like with finer substrata, and downstream the river becomes a wide, shallow, single channel with a sandy bed and little vegetation. In this general area, Dr A. Bok (Eastern Cape Nature Conservation (ECNC) pers. comm.) also located *B. capensis* eggs on 19 December 1990. They were in crevices or under rocks and away from the main current; in a water depth of 0.3-0.5 m, estimated current speed of 0.5 m s^{-1} and midday water temperature of 20 °C. Bok suggested, from his knowledge of captive propagation of this species, that a water temperature of at least 20-22 °C is required for *B. capensis* to spawn; summer temperatures near the dam are known to be lower than those further downstream (King and Tharme 1994).

Measurements taken at that site at intervals throughout 1992 (King and Tharme 1994) revealed that the water was of high quality, clear and with low conductivity (8.5-32.4 mS m^{-1}). Water temperatures in 1992 were somewhat lower or higher than those in neighbouring upstream and downstream reaches, depending on the time of the year, due to hypolimnetic dam releases. Farming practices in surrounding areas, mostly related to orchards and vineyards and including extensive water abstraction, must have an impact on the site, but an unknown number of mature *B. capensis* still inhabit the reach between the two dams (Mr D.Impson, CNC pers. comm.).

At a second site about 2 km downstream from the spawning site (Figure 13.1), larval *B. capensis* were present in 1992. Flow at the larval site was placid, with extensive shallow edge areas over sand. It appears likely that, in common with some other *Barbus* species, *B. capensis* larvae disperse from the spawning site in the late free-embryonic stage, through a swim-up period when they are swept downstream. One of the authors (JAC) felt, judging by the developmental stages present at the spawning site and the larval fish habitat, that at least three different spawns had occurred during December/January 1992.

13.3 THE STUDY SPECIES

The river's eight endemic fish species (Table 13.1) are listed in the country's Red Data Book of Fish (Skelton 1987), and several are close to extinction. Principle threats are habitat destruction, fragmentation of the river continuum by dams, massive abstraction of water, which is changing the middle reaches of the river from perennial to seasonal, and predation by the alien smallmouth bass, *Micropterus dolomieu* (Skelton 1987; King and Tharme 1994).

B. capensis is the largest endemic fish species in the system; the following is a summary of recorded information on its spawning behaviour and early life stages. A game fish which can reach lengths of 99 cm, the species is vulnerable because it takes five to seven years to mature (Scott 1982). It was once plentiful but is now declining in numbers due to the introduced predators, and the impacts of the dam and associated anthropogenic factors (Gaigher 1973; Gaigher *et al.* 1980; Scott 1982). *B. capensis* is listed as rare by Skelton (1987), and very little detail is known of its natural spawning habitats or the triggers for spawning. Before the

two dams were built it used to undertake mass upstream migrations (Harrison 1950; Jubb 1974) between September and December. The fish congregated below natural rapids and then ran upstream "with great agility during the months of September and October" (Harrison 1950). Hey (1947) noted "In spring the yellowfish ascend the rivers in large numbers in search of suitable spawning beds, (which) would appear to be clean gravel beds in the shallow portion of the river. The fish mass in countless numbers on the spawning beds and the eggs and milt appear to be shed indiscriminately. After spawning, the adults drift back to the deeper waters and the eggs are left to their own devices". Bok (ECNC pers. comm.) observed an accumulation of large adults at the spawning site downstream of Clanwilliam Dam in January 1990.

Table 13.1 Fish species occurring in the Olifants River system, their distributions and Red Data Book status as of 1987. Endemic species are confined to the Olifants River system, indigenous species are native to southern Africa, and alien species are introduced. Information from Skelton (1987) and de Moor and Bruton (1988).

SPECIES	DISTRIBUTION	RED DATA BOOK
FAMILY Bagridae		
Austroglanis barnardi	endemic	endangered
Austroglanis gilli	endemic	rare
FAMILY Cyprinidae		
Barbus capensis	endemic	rare
Barbus serra	endemic	vulnerable
Barbus calidus	endemic	rare
Barbus erubescens	endemic	vulnerable
Barbus anoplus	indigenous	
Pseudobarbus phlegethon	endemic	endangered
Labeo seeberi	endemic	rare
FAMILY Galaxiidae		
Galaxias zebratus	indigenous	
FAMILY Anabantidae		
Sandelia capensis	translocated indigenous	
FAMILY Centrarchidae		
Lepomis macrochirus	alien	
Micropterus dolomieu	alien	
Micropterus salmoides	alien	

Spawning appears to occur over an extended four-month period from October to January. Van Rensburg (1966), analysing gonad development of 123 *B. capensis*, noted that gonad mass began to increase in August and September, peaked in October to December, and began to decline in January. During the project reported on here, four ripe-running males (34:39; 58:68; 44:54; 59:69 cm standard length to total length (SL:TL)) and one spent female (47:54 cm SL:TL) were recorded downstream of the spawn site in December 1993. All fish were measured live and released. The authors also collected eggs downstream of Clanwilliam Dam in late January 1992, Van Rensburg (1966) collected larvae (10-20 mm) a few kilometres upstream of the Clanwilliam reservoir in November and December, and Barnard (1943) found small juveniles in February.

The adults are most often found in clear, rocky pools, or deeper reaches, often with strong flow (pers. obs. by JAC). Juveniles mostly inhabit pools, sheltered backwaters and shallow, marginal areas of the mainstream

(Skelton 1987), but can also be active in the main channel, where they maintain position in the gentle summer flows (pers. obs. by JAC).

Bok (ECNC pers. comm.) reported that under hatchery conditions *B. capensis* eggs were negatively buoyant and slightly adhesive, until water-hardened after 45 minutes of immersion. The eggs were not adhesive once water-hardened, unlike those of some *Barbus* species which have strongly adhesive egg envelopes (Cambray 1983). All the wild-spawned eggs collected in this study were clear to pale yellow, non-adhesive and located in interstitial areas in cobble beds. None were attached to the cobbles.

From captive breeding experiments, Bok (ECNC pers. comm.) reported water-hardened eggs of *B. capensis* with diameters of 2.60 mm (n=20; S.D.=0.09 mm) in December 1987, 2.61 mm (n=11; S.D.=0.05 mm) in December 1989, and 2.67 mm (n=51; S.D.=0.2 mm) in November 1991. In the 1987 investigation, yolk diameter was recorded as 1.98 mm (n=20; S.D.=0.04 mm). Van Rensburg (1966) reported that in gonad stage four (active to ripe), the largest group of eggs had attained a diameter of 2.0 mm in the ovary, so the eggs appear to enlarge approximately 0.6 mm in diameter with water-hardening, to 2.6 mm. In an unpublished study, one of the authors (JAC) and Bok found that eggs kept at 22-23 °C took 63 hours before free embryos hatched which were 7.38 mm TL (n = 5; S.D. = 0.23) and 7.2 mm SL (S.D. = 0.27). These free embryos were photophobic. After ten to twelve days the larval fish began to feed. Thus, at water temperatures of 22-23 °C the embryos are predicted to spend at least nine to ten days on the spawning beds before swim-up occurs. They are then carried downstream by the current to shallow quieter waters (Figure 13.1), where they commence feeding and develop to larval fish. Throughout the remainder of this paper this study will be referred to as Cambray and Bok (unpubl. data).

13.4 METHODS

13.4.1 Water release strategy and associated hydraulic measurements

Department of Water Affairs and Forestry (DWAF) release records for Clanwilliam Dam prior to the discovery of the *B. capensis* eggs in January 1992 showed that discharges had been held relatively steady over the preceding weeks or, at least, any fluctuations had been less than 24 hours long. The collected eggs were viable and the developmental stage, embryo with tail free and moving, indicated that they had probably been spawned several days prior to collection (less than 2.3 days old at 22-23 °C; Cambray and Bok unpubl. data). Perhaps because of the lack of flow pulses, there were no signs of abundant spawning. Based on this knowledge, and that of the suspected spawning period and required water temperature for spawning, experimental releases of freshes from Clanwilliam Dam were planned for 1993 and 1994 to try and induce a migration run and spawning.

Staff of DWAF agreed to alter, within certain limits, the timing and magnitudes of water releases from the dam. Over the experimental periods, the same amount of water as usual had to be delivered to Bulshoek Weir, and so some water had to be held back prior to the release of the experimental high flows. With the pattern of releases changed from one of constant flow to a combination of a low flow followed by a pulse of high flow, careful co-ordination by DWAF was needed to accommodate downstream irrigators depending on Bulshoek Weir as well as to provide the experimental flows.

Different release strategies were followed in the two spring seasons reported on here (Table 13.2). In December 1993, three pulses of higher flow were released at different times and for different durations, to investigate if dusk or dawn spawning could be induced. In the six weeks preceding the exercise, releases were held within the range 7.6-8.1 $m^3 s^{-1}$, for supplying irrigators, and then reduced to 4.6 $m^3 s^{-1}$ on 10 December in order to store water for the experimental freshes. After initial inspection of the spawn site for eggs on 13 December, three experimental freshes were released over the next four days, the first two to try and trigger dusk spawnings and the third to try and trigger a dawn spawning. The first two releases were of four hours duration, and magnitude 8.0 $m^3 s^{-1}$ and 12.0 $m^3 s^{-1}$ respectively. Both decreased to the background level of 4.6 $m^3 s^{-1}$ about an hour before dusk, with this effect showing at the spawning site as dropping water levels over the next one to two hours. The third release was also of magnitude 12 $m^3 s^{-1}$, but of 12 hours duration and finished at dawn. After each experimental fresh, the release was dropped to 4.6 $m^3 s^{-1}$, and finally the usual release of 8.0 $m^3 s^{-1}$ was returned to on 18 December. The sizes of the freshes were restricted by the size of the dam outlets, but were deemed acceptable as they were within the magnitude expected of flow pulses at that dry time of the year.

Table 13.2 Release pattern of experimental freshes from Clanwilliam Dam wall, Olifants River system, in November to December 1993 and 1994. F - fresh.

DATE	TIME	DISCHARGE ($m^3 s^{-1}$)	FLOW DESCRIPTION
1993			
mid Nov to 9 Dec		7.6 - 8.1	Normal irrigation release
10 Dec		4.6	Storage
14 Dec	11h20	8.0	Release F1/93
	15h20	4.6	Low flow for study
15 Dec	11h00	12.0	Release F2/193
	15h00	4.6	Low flow for study
16 Dec	16h00	12.0	Release F3/93
17 Dec	04h30	4.6	Low flow for study
18 Dec		8.0	Normal irrigation release
1994			
mid Nov to 25 Nov		7.5 - 8.0	Normal irrigation release
25 - 29 Nov		5.0	Storage
29 Nov	17h00	2.0 - 3.0	Low flow for egg search
1 Dec	17h45	17.0	Release F1/94
2 Dec	06h00	3.0	Low flow for study
	18h00	9.0	Release F2/94
3 Dec	06h00	3.0	Low flow for study
4 Dec		7.5 - 8.0	Normal irrigation release

The passage of released water through the downstream river was recorded in two ways. At the spawning site (Figure 13.1), changes in water surface elevations were recorded at irregular intervals from a temporary gauge plate erected in the small channel where eggs were first found in 1992 (spawn bed B). There was no attempt to measure discharge there because of the complex, anastomosing nature of the river. Downstream, at the larval site, depth and velocity distributions were recorded on five occasions along a bank to bank cross-section, and

discharges calculated. Spot readings of velocity and depth were also taken where larval *B. capensis* were found.

Based on the results of this experiment, a fresh of 17 $m^3 s^{-1}$ was released for 12 hours overnight in December 1994. Prior to this, the release had been at 5.0 $m^3 s^{-1}$ until 29 November when it had been reduced to 2.0 $m^3 s^{-1}$ in order to store water for the experimental fresh (Table 13.2). The duration of the fresh was from 17h45 on 1 December to 06h00 on 2 December, and overnight on 2-3 December a further smaller fresh of 9.0 $m^3 s^{-1}$ was released to meet demand at Bulshoek Weir. No hydraulic measurements of the passage of water through the study site were taken during the 1994 visit, as the majority of the time was spent observing and filming fish movement from the vantage point of the bridge.

13.4.2 Water chemistry

During the 1993 experimental releases, conductivity and water temperature of the river were read at intervals from instruments established at spawn bed B. Water samples were also collected at intervals throughout the survey period, for analysis of Kjeldahl nitrogen, pH, ammonium (NH_4-N), nitrate and nitrite (NO_3+NO_2-N), fluoride (F), total alkalinity, TDS, dissolved organic carbon (DOC), sodium (Na), magnesium (Mg), silicon (Si), chloride (Cℓ), potassium (K) and calcium (Ca). Water samples were stored in bottles specially prepared and provided by the Institute for Water Quality Studies (DWAF), preserved with mercuric chloride in the field and analysed by the Institute (DWAF 1992b). During the 1994 experimental release, no water samples were collected, but spot readings of water temperature, pH and conductivity were taken at spawn bed B and at the larval site.

13.4.3 Biological data collection

Initially, there was no clear knowledge of the importance of the site to *B. capensis* as a spawning ground, and of what environmental variables might trigger spawning there. Thus, most of the time in December 1993 was spent in or adjacent to the river, collecting water samples and taking hydraulic measurements as described above, searching for eggs, free embryos, larvae and juvenile fish, and completing other activities not reported on here. After an initial search for and collection of the various life stages on arrival at the study site, additional collections were made before, after, and at intervals through, each experimental fresh. Cobbles were lifted whilst holding a fine-meshed scoop net downstream, with the current being sufficiently strong to carry the eggs and free embryos found at the spawn site into the net. The eggs were immediately examined and photographed microscopically on site to determine their stage of development, and to establish the approximate time when they were spawned. Drift nets were also placed downstream of the spawning site on several occasions, in order to collect any eggs washed downstream. Larval fish were collected at several localities at the downstream larval site using a 0.5 m-wide, fine-meshed D-net. Only limited qualitative sampling was carried out as intense quantitative sampling would not be ethical in that *B. capensis* is a Red Data Book species. After on-site examination, small samples of eggs, free embryos and larval fish were preserved in 4% buffered formalin for later examination by dissecting microscope and measurement using an ocular micrometer.

Because the above activities could have been a substantial disturbance to the fish, a different strategy was adopted in 1994. Activities centred on observations from the bridge of fish behaviour in and around the spawning site. Fish behaviour was recorded using a video camera, notes and tape recorder. No measurements

of water depths and velocities were taken at the spawning site during 1994, so the specific hydraulic conditions on the spawn beds remained unrecorded at that time. In follow-up studies, however, the hydraulics of the spawn beds were recorded, and are reported upon in Chapter 14.

In summary, then, the 1993 study concentrated primarily on: an assessment of the recorded hydrological conditions preceding the experimental freshes; gaining an understanding of the general pattern of attenuation of the freshes as they flowed through the study site and of the water quality conditions linked to them; and an investigation of whether or not a pattern of *B. capensis* embryonic development could be linked to any of these factors. The 1994 study concentrated on recording fish behaviour following two experimental freshes. In both years, a thorough search of the study site was made before any freshes were released, to collect early developmental stages of *B. capensis* and assess the extent to which spawning had occurred before the experiments.

13.4.4 Age determination of *Barbus capensis* embryos and larvae

B. capensis developmental times in the wild are based on a study by Cambray and Bok (unpubl. data). In this study, development of hatchery bred *B. capensis* was recorded under aquarium conditions at controlled temperatures of 22-23 °C. As is well documented, water temperature, oxygen conditions and biochemical resources supplied by the female parent can influence the rate of fish development (Mills 1991). The laboratory rates are therefore used with caution in the interpretation of developmental rates of embryonic *B. capensis* on the spawning beds, but are useful for estimating spawning frequency and timing.

13.4.5 Definitions of terms used

Fish developmental terminology follows that of Balon (1985). Due to differences in terminology in the literature the following terms have been defined for use within this report.

Multiple spawning: capacity to ripen successive batches of eggs within a season (Bagenal 1978).

Repeat spawning within days: is distinct from multiple spawning, as there must be a longer time period between two ovulations for multiple spawning. In repeat spawning, spawning may be spread over several days but draws on a single batch of matured eggs (Horvath 1986). A repeat spawner can also be a multiple spawner.

13.5 RESULTS

13.5.1 Local hydraulics

The local hydraulics of the river reach downstream of the dam are largely dictated by the anastomosing nature of the channel and the dense growth of *P. serratum* on the mid-channel islands. In 1993, changes in the volumes released at the dam showed 1.5 km downstream at spawn bed B as changes in water surface elevations with about a two hour time lag (Figure 13.2). Discharge measurements further downstream at the larval site showed that the increased volume of water reached there with very little additional lag time. This was probably because the character of the river changed shortly downstream of the spawn site, from vegetated multiple channels to a single-thread channel with no instream vegetation. Attenuation of the flood peaks, as shown by gauge heights, could not be well assessed from the measurements, but the hydrograph of the third

release fell quite sharply from its peak and had roughly the same duration as its corresponding release (Figure 13.2), suggesting that attenuation was not great. The baseflow outside the peaks gradually increased, however, indicating that some storage and slow release of water was occurring in the vegetated upstream areas.

13.5.2 Water quality

During the 1993 experiment, 62 sets of water samples were taken for analyses, together with accompanying readings of on-site instruments and the gauge plate (Table 13.3). Samples were taken at roughly half-hourly intervals during the releases, and less frequently at other times due to the demands of other tasks. Plots of water level versus any single water-quality variable (Figure 13.2) revealed no obvious trends of variation linked to the three experimental freshes. Linear regression analysis was used to assess further if the experimental freshes were affecting water chemistry. Because of the possibility of the concentration of chemical variables differing during a pulse (Britton 1990), the data were split into a group collected on the rising limb of the releases (n=40), and a group collected on the falling limb of releases (n=17). There were no significant relationships between water level and individual water chemistry variables in either group, with the exception of water temperature. This latter appeared to decrease slightly at the spawn beds during the experimental freshes and then recover, possibly reflecting overnight changes in air temperature. The few spot measurements of water quality taken at spawn bed B during the 1994 releases (Table 13.4) supported this finding.

It was concluded that water quality did not differ significantly in any way measured during the experimental releases, with the possible exception of temperature. It is not surprising that the chemistry did not change as the freshes were simply increases in the hypolimnetic releases already occurring. Thus, water chemistry probably did not play a major role in influencing any witnessed fish behaviour.

13.5.3 General overview of biological results

In 1993 two possible spawning beds were identified at the spawning site, through the presence of eggs. One (Figure 13.1: bed A) was in the fast-flowing water of the main channel and one (B) was in the quieter side channel where the eggs were found in 1992. Both had cobble beds, with no overhead vegetal cover but ample inundated marginal vegetation; bed A was larger than B and had noticeably faster and deeper flow (0.4-0.8 m and 0.2-0.3 m depth respectively, at a discharge of 4 $m^3 s^{-1}$).

No spawning was observed at these beds in 1993, possibly because of the activities of the survey team, and because only a limited time was spent on observations from the bridge. Much of the evidence of what happened that year is therefore pieced together from analysis of the eggs, free embryos and larvae collected. The general evidence pointed to sporadic spawning, some of which had happened before, and some during, the experimental period.

During the visit in December 1994 most time was spent viewing and filming fish activity from the bridge. Although the turbulent flow, submerged vegetation and light conditions made quantitative observations difficult, an unknown number of fish, but probably 100 or more, were observed moving upstream during the two days after the first experimental fresh and abundant spawning behaviour was observed. Two months later, "far more larvae and young juveniles than usual" (Impson pers. comm.) were found at the downstream larval site, and the evidence pointed toward a far more successful season of recruitment. Records of these

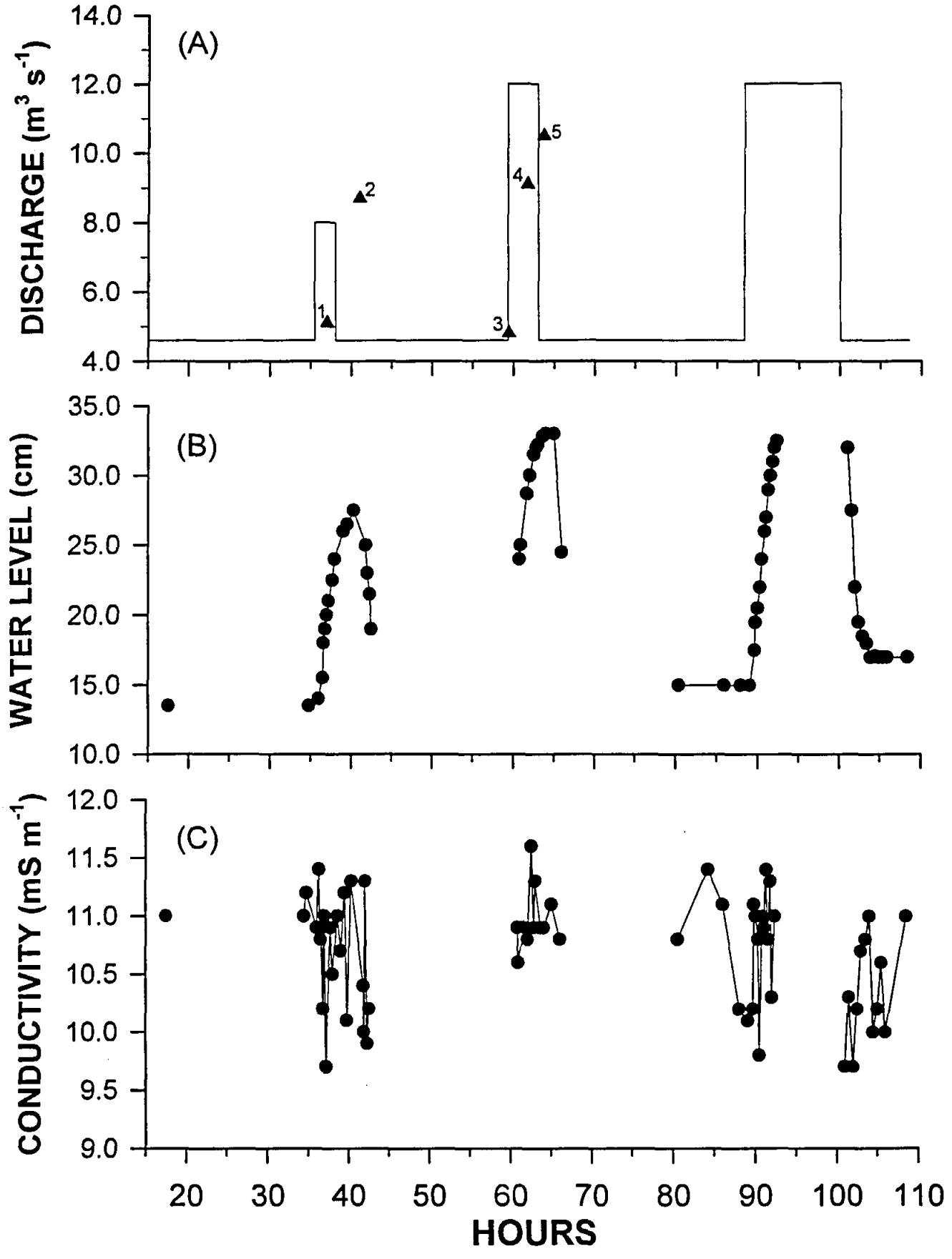


Figure 13.2 **(A) Movement of water through the study area below the dam in December 1993 is illustrated by the dam release pattern (—) and measured discharge levels at the larval site (▲); discharge ($m^3 s^{-1}$) readings at the larval site were taken as follows: 1) 14 Dec, 13h00 = 5.1; 2) 14 Dec, 17h00 = 8.7; 3) 15 Dec, 11h15 = 4.8; 4) 15 Dec, 13h45 = 9.1; and 5) 15 Dec, 15h45 = 10.5. (B) Water level (cm) at spawn bed B. (C) conductivity (mS m^{-1}) at spawn bed B.**

Table 13.3 **Ranges in water quality variables measured at spawn bed B in the Olifants River system (see Figure 13.1), before, during and after experimental freshes F1, F2 and F3 in December 1993.**

	BEFORE F1/93	DURING F1/93	BETWEEN F1/93 AND F2/93	DURING F2/93	BETWEEN F2/93 AND F3/93	DURING F3/93	AFTER F3/93
Water level (m)	0.135	0.140 - 0.260	0.265 - 0.190	0.240 - 0.3.22	0.328 - 0.150	0.150 - 0.325	0.320 - 0.130
Conductivity (mS m^{-1})	11.0 - 11.2	9.7 - 11.4	9.9 - 11.3	10.2 - 11.6	10.2 - 11.4	9.8 - 11.3	9.7 - 11.0
Temperature (°C)	21.8 - 22.9	22.0 - 22.3	22.3 - 22.5	22.7 - 23.0	22.1 - 22.9	22.5 - 23.1	22.1 - 22.2
Total alkalinity (mg ℓ^{-1})	7 - 9	7 - 9	7 - 10	7 - 11	7 - 10	7 - 12	7 - 12
Calcium (mg ℓ^{-1})	3	2 - 4	2 - 4	2 - 3	2 - 3	2 - 4	3 - 4
Fluoride (mg ℓ^{-1})	0.2	0.1 - 0.5	0.2 - 0.6	0.1 - 0.3	0.1 - 0.5	0.2 - 0.6	0.2 - 0.5
Nitrogen (mg ℓ^{-1})	0.26 - 0.43	0.24 - 0.46	0.21 - 0.30	0.22 - 0.40	0.25 - 0.37	0.23 - 0.34	0.21 - 0.29
Potassium (mg ℓ^{-1})	0.4 - 1.4	0.4 - 1.9	0.4 - 0.5	0.4 - 1.2	0.4 - 0.7	0.4 - 0.8	0.3 - 2.0
Magnesium (mg ℓ^{-1})	2 - 3	2 - 4	3 - 4	2 - 3	3 - 4	3 - 4	3 - 4
Sodium (mg ℓ^{-1})	11 - 12	11 - 16	11 - 12	11 - 13	11 - 13	11 - 12	11 - 12
Ammonium (mg ℓ^{-1})	0.04 - 0.07	0.04 - 0.07	0.04 - 0.10	0.04 - 0.05	0.04 - 0.06	0.04 - 0.06	0.04 - 0.07
Nitrate and Nitrite (mg ℓ^{-1})	0.14 - 0.18	0.14 - 0.20	0.15 - 0.27	0.16 - 0.19	0.14 - 0.21	0.15 - 0.21	0.14 - 0.19
Phosphate (mg ℓ^{-1})	0.009 - 0.120	0.010 - 0.015	0.009 - 0.017	0.008 - 0.026	0.006 - 0.017	0.011 - 0.019	0.007 - 0.023
Silicon (mg ℓ^{-1})	1.9 - 2.3	1.8 - 2.2	1.9 - 2.3	2.1 - 2.3	2.1 - 2.4	2.0 - 2.5	1.9 - 2.5
Sulphate (mg ℓ^{-1})	12 - 15	6 - 12	6 - 10	6 - 18	6 - 14	6 - 13	6 - 12
Total dissolved solids (mg ℓ^{-1})	62 - 64	53 - 69	56 - 63	58 - 61	53 - 69	55 - 68	57 - 64
Total phosphorus (mg ℓ^{-1})	0.014 - 0.016	0.015 - 0.020	0.014 - 0.020	0.012 - 0.029	0.012 - 0.034	0.013 - 0.033	0.016 - 0.024
DOC (mg ℓ^{-1})	2.4 - 2.9	2.0 - 4.3	2.4 - 3.4	1.8 - 3.2	2.0 - 5.3	2.2 - 4.0	2.1 - 2.9
Chloride (mg ℓ^{-1})	20 - 21	20 - 25	20 - 22	19 - 24	20 - 23	20 - 24	20 - 22
PH	7.3 - 7.4	6.4 - 7.4	6.9 - 7.2	6.8 - 7.2	6.4 - 7.2	6.8 - 7.3	6.8 - 7.5

observations of behaviour, together with measurements of the collected eggs, free embryos and larvae from both trips, are summarised below, with the relevant year indicated where appropriate.

Table 13.4 Spot readings of water conditions at spawn bed B in the Olifants River and at the larval site, linked to the 1994 experimental freshes F1/94 and F2/94.

DATE	TIME	COMMENT	PH	TEMPERATURE (°C)	CONDUCTIVITY (mS m^{-1})
Spawn Bed B					
1 Dec 94	17h40	just before F1/94		22.3	
		F1/94 released overnight			
2 Dec 94	05h00	just before end of F1/94		18.3	8.4
	12h15		5.94	20.7	9.3
	17h15		5.83	20.4	8.5
	18h10		6.73	20.2	8.4
	18h45		6.71	20.1	8.4
		F2/94 released overnight			
3 Dec 94	05h00	just before end of F2/94	5.55	19.1	7.4
	09h00		5.92		7.7
	11h10		5.99	19.7	7.9
Larval Site					
1 Dec 94	18h00			27.7	9.0

13.5.4 Fish movements observed (December 1994)

Observations were carried out from the bridge 06h00 - 13h00 and 17h00 - 18h00 on 2 December, and 05h00 - 08h00 and 09h20 - 11h10 on 3 December. With the 12 hour overnight release of 17 $m^3 s^{-1}$ reduced to 3 $m^3 s^{-1}$ at 06h00 on 2 December 1994, flow reduction at the spawn site combined with improving light conditions allowed good observations through the water from about 08h00 onwards, although fish movement could be detected earlier. Most activity and all spawning acts were recorded at spawn bed A (Figure 13.1). Fish moved upstream singly, or in small groups, throughout the day, using cover from partially-submerged bankside vegetation (Table 13.5). Many larger individuals moved out of cover to hold position at spawn bed A, and a few at spawn bed B, in the fast-flowing current. Smaller individuals tended to stay hidden under the vegetation, often in growing numbers as more groups arrived, with some occasionally darting upstream across areas with no overhead cover. Almost without exception, fish movement was upstream.

Over the two days of observations, large individuals were recorded holding position on spawn bed A at times throughout the day (Table 13.5). Once in position, neither full sunlight onto the bed or movements on the bridge appeared to affect them, although movement along the bank drove them from the site. Although it was difficult to keep track of individual animals, some appeared to hold station for up to one hour, occasionally shifting around the bed, perhaps pushed by currents, but always returning to the same place. Some eventually moved on upstream, but many stayed to spawn at A. Although a few individuals held station on spawn bed B for a while, all observed individuals eventually moved on without spawning.

Table 13.5 Movement of *Barbus capensis* at Spawn Bed A in the Olifants River in December 1994. Data from video recordings and on site tape recordings. (All fish measurements are estimated TL).

DATE	TIME	NO. OF FISH	BEHAVIOUR	COMMENT
2 Dec	0730	several groups of 3-8 fish	Moved upstream, crowding under overhanging vegetation.	Most were small 30-40 cm. Occasional large ones 50+ cm. Between 50-100 in total. Some moved upstream. Some moved out of cover onto gravel bed in current, then back under cover.
	0830	3	Large fish holding position on bed.	May have been there earlier, light and visibility poor. Bed in shade from bridge, but with no hydraulic or overhead cover.
	0900	2	Large fish holding position on bed.	Sun shining on bed. Several small fish under marginal vegetation.
	0940	4	1st spawn.	Two fish involved in spawn, others nearby.
	1000	5-6	2nd spawn.	
	1015	10-12	3rd spawn.	
	1025	3	4th spawn.	
	1040	2	5th spawn.	
	1050	1	Large fish moving downstream.	
	1115	3	6th spawn.	Large fish involved may have been same ones there since 0900 and involved in more than one spawn.
	1145	4	7th spawn.	
	1150		8th spawn.	
	1155			Eggs collected from bed.
	1300-1600			No observations.
	1700-1800		Fish still holding position. 5 more spawns seen.	Water depth 0.4-0.5 m. Water temp. 20 °C. Cond. 8.4 mS m^{-1}.
3 Dec	0500-0530		No fish seen.	Light poor. Water temp. 19.1 °C.
	0625	1	Small fish moving upstream.	
	0628	?	1st spawn.	
	0730	3	Medium-size fish moving upstream.	40-45 cm.
	0735-0800	2	Large fish holding position on bed.	50-60 cm.
	0800	5	Moved in, milled around, moved upstream.	Original large fish still there.
	0804	6	Large fish moved through site upstream.	
	0920	2	2nd spawn.	
	0950-1000	3	Holding position. Then all 3 involved in 3rd spawn. (Eggs collected).	Temp. 19.7 °C.

13.5.5 Spawning activity (December 1994)

Spawning acts were observed at bed A as early as 06h28 and as late as 18h00, with about 20 separate incidents recorded over the hours watched in the two days (Table 13.5). Spawning activity would often begin with a large individual that was maintaining position on the bed being joined by a smaller fish that darted out of the vegetal cover. The two would quickly line up facing upstream, parallel and close together, then rapid vibrations of both animals for 2-10 s would be followed by a cloud of milt drifting across the bed and downstream. Occasionally three fish would be involved in the spawning act with a smaller fish on each side of the large fish. On no occasion did any of the fish break the surface of the water during spawning, as has frequently been recorded for other cyprinids (Breder and Rosen 1966) and for *Labeo capensis* spawning in the Orange River (JAC pers. obs.). After the spawning act, the smaller fish moved away, usually upstream or returned to the edge cover, and the larger fish either moved on upstream, fell away in a tumbling manner downstream or, in many cases, maintained position on the bed suggesting a repeat spawner. At intervals through the two days, eggs were collected from spawn bed A for examination. A few were also found and collected at bed B.

13.5.6 Egg development (1993 and 1994)

Eggs were collected before, during and after the experimental releases in both 1993 and 1994. Most were collected at spawn bed B in 1993 and at spawn bed A in 1994. A total of 209 collected eggs were measured in 1993 and 127 in 1994 (Table 13.6). In 1993, the eggs had a mean total diameter of 3.55 mm (S.D. = 0.21, n = 209) with a yolk diameter of 2.93 mm (S.D. = 0.15, n = 75). In 1994, 127 eggs were measured and had a mean total diameter of 3.6 mm (S.D. = 0.16) with a yolk diameter of 3.00 mm (S.D. = 0.15, n = 118). In 1992 at spawn bed B, one of us (JAC) collected eggs which were considerably smaller (3.01 mm ± 0.12; yolk 2.4 mm ± 0.36, n = 7).

Developmental stages of the embryos indicated that in both 1993 and 1994 *B. capensis* had spawned prior to the releases of the experimental freshes (Table 13.6). However, some embryos collected were clearly from spawnings during the experimental periods. For instance, on 16 December 1993 at 08h00 embryos were collected that were one to two days old, and on 17 December 1993 at 08h00 8 to 32-celled embryos were collected that were less than 12 hours old.

In 1994 the embryonic stages collected prior to the experimental fresh indicated that there had already been at least three spawnings. However, the four-celled embryos collected on 2 December 1994 at 18h00 were spawned during the experimental period, probably resulting from the spawnings that were observed from the bridge. The number of developmental stages on the spawning beds at any one time, linked with the observations in 1994, suggested that *B. capensis* is a repeat spawner over several days.

13.5.7 Larval and juvenile habitat (1993 and 1994)

Larvae and juveniles of *B. capensis*, but no eggs or free embryos, were found at the larval site (Figure 13.1) during the visits in both December 1993 and December 1994. Several hundred larvae and about 50 juveniles were observed on 13 December 1993, prior to the first set of experimental releases, and numbers remained at this level throughout the visit (Table 13.7). Although viewing conditions at the larval site were poor in December 1994, there seemed to be far fewer larvae and juveniles present before and immediately after the two

experimental releases than in 1993. As stated above, however, larvae and young juveniles were unusually abundant some months later.

The wetted channel at the larval site was about 60 m wide, with a coarse sandy substratum, extensive shallows along the right bank, and no overhead or hydraulic cover. All larvae found were in very shallow, edge areas on the right bank, where there was little or no flow. The larvae moved backwards and forwards across the sandy expanse with the water edge as discharges changed. Random hydraulic measurements taken along the edge in December 1993 suggested that larval distributions were dictated both by current speed and by water depth (Figure 13.3). All larvae found were in water < 0.28 m deep, but many such areas contained no larvae. With two exceptions, however, the larvae were present in currents of velocity < 0.18 m s^{-1} but absent from higher flows. Froude numbers, which are ratios of inertial to gravity forces and have been described as Universal indicators of the state of affairs in free surface flow (Henderson 1966, cited in Wadeson 1995), revealed a similar picture, with larvae not present in areas with Froude numbers > 0.172. In the plot of depth versus velocity, a boundary line could be suggested, separating hydraulic conditions linked with larval presence from those linked with larval absence (Figure 13.3). On this plot, one of the two exceptions mentioned above now linked with the other records of absence, through being on the same side of this boundary line, possibly suggesting that low velocities are not usually selected if the water is deep.

The larvae did not school, but moved independently, often feeding from the substratum. During the visits in both 1993 and 1994, spot daytime water temperatures recorded in the vicinity of the larvae were 25.5-27.7 °C, and conductivities were in the range recorded for the spawn site (Tables 13.3 and 13.4).

13.5.8 Larval development (1993 and 1994)

The larval fish collected on the 13 December 1993 were *c.* 17 days old (Table 13.7). Assuming that all the larval fish collected in 1993 were from the same spawning, over the period from the 13 December at 16h00 until the 17 December at 13h00 the fish grew from a mean length of 13.12 mm TL (11.6 mm SL) to 14.38 mm TL (12.5 mm SL). The main developmental changes were increases in ray numbers, especially in the anal fin, and the formation of the distinguishing black vertical stripe on the caudal peduncle.

Juveniles were also collected in December 1993. These were about the same size (24 mm TL) as reached by those in the laboratory study 22-23 °C after approximately 50 days. If the same growth rate held true in the wild, the juveniles found in December would have been spawned in late October. However, temperatures at the larval site were around 27.7 °C, so the wild juveniles may have been developing faster than those in the laboratory study.

In 1994 the larval *B. capensis* collected prior to the experimental fresh were calculated to be 17-21 days old, which would indicate a spawning in early to mid-November.

13.5.9 Summary of developmental stages linked to dam releases

On-site microscopic examination of live eggs and free embryos proved invaluable for tentative allocation of the various developing embryonic stages into the periods, before, during and after experimental freshes. This information guided determination of the facts that the fish had spawned, in both 1993 and 1994, on a few occasions before the experimental releases and had also spawned throughout the experimental periods.

Table 13.6 Locality, dates and times of various *Barbus capensis* developmental stages collected at spawn beds A and B (see Figure 13.1) below Clanwilliam Dam wall, Olifants River system. *Age estimated from laboratory-raised embryos at 22-23 °C. NL – notochord length.

DATE	SPAWN BED	EGG SIZES		EMBRYOS	FREE EMBRYOS	TEMP	COMMENTS
		Diameter (mm ± SD)	Yolk Diameter (mm ± SD)	(± Age in Days*)	TL and NL (mm ± SD) (± Age in Days*)	(°C)	
				1992			
29 Jan. 1992 (12h00)	B	3.01 ± 0.12 (n = 7)	2.4 ± 0.36 (n = 3)	Near hatching, no eye pigment (2.4 d)	9.8 ± 0.62 TL (n = 32) 9.32 ± 0.56 NL (n = 32) (6 d) eye pigment (preserved specimen measurements)	25.0	Fish had spawned at least twice in past week. Larval fish stages downstream indicated two more spawnings.
				1993			
13 Dec. 1993 (16h00)	B	3.68 ± 0.2 (n = 29)	3.00 ± 0.2 (n = 26)	Head to tail of embryos cover 2/3 circumference of yolk (n=28); one embryo was ready to hatch, tail loose, revolved inside egg envelope (2.4 d)	9.43±0.38TL (n=16); 9.14±0.33NL (n=16) (5.3 d) light eye pigmentation, dorsal of head in line with body 12.4TL (11.0NL) (n = 1) (13.8 d) more advanced: caudal rays, 4 dorsal ray anlages, heavily pigmented eyes and body pigmentation	-	Fish had spawned at least twice in past week.
14 Dec. 1993 (11h00)	B	3.44 ± 0.14 (n = 25)	2.79 ± 0.15 (n = 17)	Earlier stage than for 13/12/93 (head not enlarged); 7 embryos ready to hatch (caudal area free from anterior yolk) (2.4 d)	9.0TL (8.7NL), 9.4TL (9.1NL) (5.3 d)- ant. yolk bulges, slight eye pigmentation, head partly free from ant. yolk sac, post. yolk tube still distinct	22.1	Free embryos collected with eggs after several cobbles lifted.
15 Dec. 1993 (09h00)	B	3.80 ± 0.13 (n = 39) (similar dev. stage)	-	Well developed, ready to hatch, revolving inside egg envelope (2.4 d)	8.2TL (8.0NL) (3.3 d) no eye pigment, ant. of yolk sac bulbous, head curved over yolk	22.3	Free embryos probably same spawn as embryos.
15 Dec. 1993 (10h00)	A	3.48 ± 0.09 (n = 78)	-	Well developed, ready to hatch, revolving inside egg envelope (2.4 d)	-	22.3	At A and B, eggs were at same developmental stage.
16 Dec. 1993 (08h00)	B	3.52 ± 0.06 (n = 20)	2.91 ± 0.17 (n = 20)	Embryo near mid-epiboly (<1.4 d)	10.4±0.95TL (9.93±0.9NL) (7.8 d). Heavily pigmented eye, mouth developed, anterior yolk bulge not pronounced	22.1	Mid-epiboly embryos possibly from dusk spawn on 15/12/93. First eggs spawned since manipulated flows. Three separate spawnings on bed.
		3.4 ± 0.08 (n = 4)	-	Well developed, tail loose, no eye pigment (2.3 d)			

Table 13.6 Continued.

DATE	SPAWN BED	EGG SIZES		EMBRYOS	FREE EMBRYOS	TEMP	COMMENTS
		Diameter (mm ± SD)	Yolk Diameter (mm ± SD)	(± Age in Days*)	TL and NL (mm ± SD) (± Age in Days*)	(°C)	
16 Dec. (11h45)	B	3.6 (n = 1)	3.4 (n = 1)	Embryo in mid-epiboly (<1.4 d)	1) 9.0TL (8.7 NL) slight eye pigmentation, ant. yolk sac bulge 2) 11.16±0.32TL (10.54±0.4NL) (n=5) Heavy eye pigmentation, no ant. yolk sac bulge	-	
16 Dec. 1993 (14h00)	B	3.6 (n = 1)	2.8 (n = 1)	Mid-epiboly (<1.4 d)	9.5TL (9.0 NL) (5.3 d) Slightly pigmented eyes ant. yolk bulbous	-	
		3.4 ± 0.06 (n = 3)	-	Well dev. embs and tails loose & revolving actively in egg envelope (<2.3 d)			
16 Dec. 1993 (16h00)	B	3.45 & 3.5 (n = 2)	3.3 & 3.2 (n = 2)	Mid-epiboly (<1.4 d)	10.9TL (10.4NL) (8.8 d) Eye well pigmented, head straight with body, mouth open and closes	-	Eggs at similar stage to those collected earlier same day.
17 Dec. 1993 (08h00)	B	3.13 ± 0.1 (n = 4)	2.63 ± 0.1 (n = 4)	8-32 cells (2-4 h)		22.1	Recently spawned.
17 Dec. 1993 (10h00)	B	3.0	2.4	64 cells (<6 h)	1) 10.08±0.21 TL (9.7±0.18NL) (n=4) (8.8 d). Eye well pigmented, head straight, pectoral fin buds, ant. yolk sac still bulbous, mouth forming, 3 gill arches formed 2) 11.13±0.78 TL (10.53±0.81 NL) (n=3) (9.8 d) 3-4 caudal fin ray anlages, eye well pigmented, mouth well developed, yolk tubular, pigment on dorsum of head, body and laterally	-	
17 Dec. 1993 (11h45)	B	3.1	2.6	Cytoplasmic streaming, late cleavage of blastodisc (<1 d)	1) 9.8TL(9.5NL) (6 d) eyes well pigmented head straight with body, ant. yolk sac bulged, pectoral fins paddle-shaped 2) 10.7TL(10.2NL) (8.8 d) 3 ray anlages, pigment on dorsum of head and body and lateral	22.3	
17 Dec. 1993 (13h22)		3.25	2.9	Epiboly of above embryos had commenced (<1 d)	-	22.5	

Table 13.6 Continued.

DATE	SPAWN BED	EGG SIZES		EMBRYOS	FREE EMBRYOS	TEMP	COMMENTS
		Diameter (mm ± SD)	Yolk Diameter (mm ± SD)	(± Age in Days*)	TL and NL (mm ± SD) (± Age in Days*)	(°C)	
				1994			
01 Dec. 1994 (11h00)	B	-	-	Neural ridge; optic cup, tail tip free; tail tip 3/4 way loose (1.4 d)	none	20.4	Three different embryonic stages indicate three spawnings.
01 Dec. 1994 *(12h00)*	A	-	-	4-celled embryo (2 h)	none	22.3	Possibly spawned early on 1-12-94.
02 Dec. 1994 (12h00)	A	-	-	Late high blastula; late gastrula; optic cup and 9 somites; 20-21 somites (all <1.4 d)	none	18.7	Three to four separate spawnings.
02 Dec. 1994 (18h00)	A	-	-	4, 8, 16, 32 and 64-celled (2-6 h) embryos; late gastrula	none	22.1	Earlier stages spawned after the water release and correlate to observed spawnings on morning of 2-12-94.

Table13.7 **Larval and early juvenile *Barbus capensis* developmental records, collected 2 km downstream of spawn sites A and B (see Figure 13.1) Olifants River system, December 1993 and 1994.**

DATE	TIME	NUMBER OF INDIVIDUALS	DEVELOPMENTAL STAGE	TL ± S.D. (mm)	SL ± S.D. (mm)	DESCRIPTION	APPROXIMATE AGE FROM EXPERIMENTAL WORK AT 22-23 °C
13 Dec 93	1600	14	larvae	13.12±0.55	11.60±0.41	In smallest fish: dorsal ray anlages just forming. In most developed fish: up to 9 dorsal rays but no anal rays and pelvic buds just forming.	17 days
14 Dec 93	1030	15	larvae	13.16±0.35	11.60±0.38	6-11 dorsal rays, no anal rays.	17 days
15 Dec 93		1	larvae	14.16 (12.90 NL)		Full caudal rays, 9 dorsal rays and fin with 5 ray anlages, caudal peduncle pigment spot.	21 days
		1	larvae	16.40 (14.40 NL)		As above, but pelvic fin buds also just formed.	>21 days
		1	juvenile	24.90	21.00	All fins fully formed, caudal peduncle vertical pigment stripe present.	53 days
		1	juvenile	27.70	22.70	As above.	>53 days
17 Dec 93	1300	47	larvae	14.38±0.54	12.50±0.43		21 days
		largest of these	larvae	15.33	12.90	Six anal fins, pelvic fin buds, vertical pigment stripe on caudal peduncle just forming.	
		smallest	larvae	13.50	12.00	No anal rays.	19 days
1 Dec 94		12	larvae	14.10±0.50		Range 13.2-15.0 mm TL. Swimbladder commencing , division into anterior and posterior chambers, and distinctive black caudal peduncle marking just forming.	17-21 days

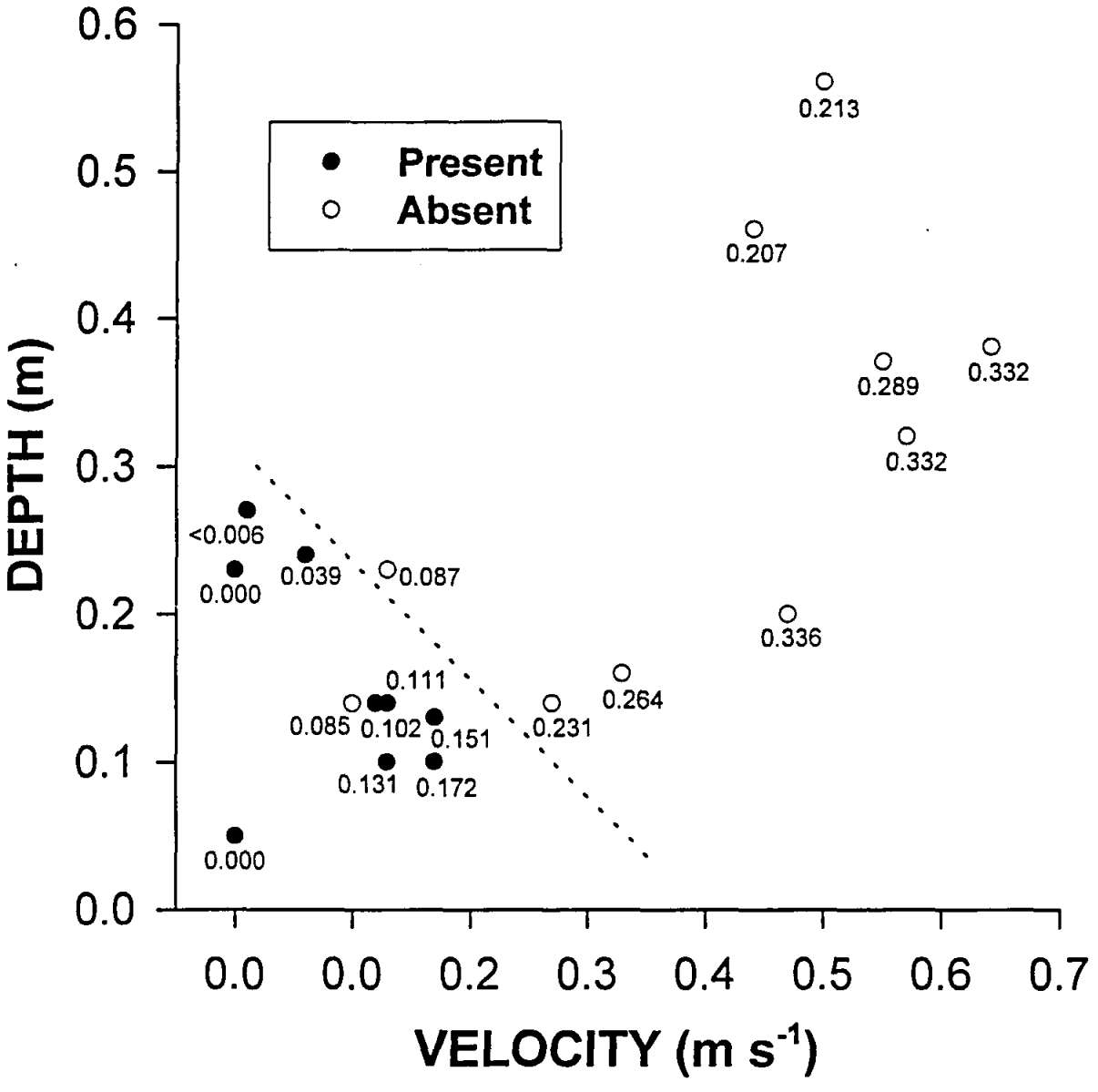


Figure 13.3 **Plot of hydraulic conditions at *Barbus capensis* larval fish site below Clanwilliam Dam. Froude numbers are given for each point measured. Larval fish present (O) and larval fish absent (●). The dotted line denotes a tentative boundary between presence and absence.**

Observations from the bridge in 1994, linked with the knowledge two months later of high larval numbers, indicated that 1994 had been an unusually successful year for recruitment. Although the evidence gathered so far is fragmented and many gaps in understanding and proof exist, it was tentatively concluded that the two experimental freshes in 1994 appeared to enhance spawning success of *B. capensis*. This enhanced spawning appeared to be linked to pulses of high flow but not necessarily to changes in water chemistry.

13.6 DISCUSSION

13.6.1 Spawning season of the genus *Barbus*

Annual spawning(s) are timed to ensure that optimal conditions are present and that the young hatch and commence feeding at the optimum season for survival (Nikolsky 1963; Bye 1984; Nestler *et al.* 1988). Wherever conditions permit, *Barbus* species also spread their reproductive effort in order to minimise egg predation (Kortmulder 1982). This translates into different spawning months depending on whether the species occurs in a tropical or temperate area; when the dry and wet seasons occur and whether they are in the Northern or Southern Hemisphere. Spawning seasons of *Barbus* species in general are very variable, from those that spawn in wet-season high flows to dry-season, low-flow spawners and those that can spawn throughout the year. Flow is a very important spawning trigger in tropical areas as most freshwater fish, which includes many *Barbus* species, spawn seasonally during the rainy periods (Lowe-McConnell 1975; Payne 1986).

In the summer rainfall areas in temperate areas many African *Barbus* species spawn after an increase in flow (Cambray and Bruton 1984; Tómasson *et al.* 1984; Cambray 1994). However, some *Barbus* species do not adhere to this general rule of spawning during the rainy season. *B. guildi* (West Africa) spawned only in the second rainy peak (Loiselle 1973) and *B. nigrofasciatus*, *B. titteya* and *B. dorsalis* in Sri Lanka (Silva *et al.* 1985), and *B. lorenzi* from southeastern Dahomey spawned throughout the year (Loiselle and Welcomme 1971). Some *Barbus* species have been recorded spawning in the dry season, similar to *B. capensis* in the Olifants River; two such species are an unspecified *Barbus* species from Togo (Loiselle and Welcomme 1971) and *B. melanampyx* from southern India (Harikumar *et al.* 1994). In the *B. melanampyx* population it was argued that the dry-season spawning was probably related to the more constant conditions provided then than during the more unpredictable monsoon season. Therefore the dry-season spawners, such as *B. melanampyx*, can afford to spread their reproductive effort over a number of days, whereas the wet-season spawners had "to make the most of each opportunity" (Harikumar *et al.* 1994). These authors also suggested that the timing of the reproductive season of *B. melanampyx* is not proximately influenced by flow rates but the fish are 'programmed' not to start breeding before November. In temperate conditions, such as in the western Cape, both winter photoperiod and low water temperatures would ensure that the Olifants River cyprinids would not spawn during the winter, wet season. In the Olifants River, minor increases in flow during the dry season may be the spawning trigger for large *Barbus* species, such as *B. capensis*, which spawn in the main channel.

13.6.2 Spawning and reproductive style of *Barbus capensis*

B. capensis occurs in a temperate area with a dry, summer season. It is suggested here that the species spreads its reproductive effort over the dry season and 'makes the most of each opportunity' of any increased flows, even minor ones, which occur during this period. *B. capensis* is a multiple spawning species per season

(Table 13.7) and within each spawning period lasting a day or two it appears to be a repeat spawner (Table 13.6). It may well be able to spawn in faster flows than observed in the present experimental flow releases, being a large powerful species, as witnessed by the way the large fish maintained their position for hours on the spawning beds.

B. capensis belongs in the following reproductive style as defined by Balon (1975): ethological section of non-guarders (A.); the ecological group of open substratum spawners (A.1.); and the guild lithophils (A.1.3.) which are rock and gravel spawners with benthic free embryos. When hatched, the free embryos are photophobic, and hide between and under cobbles. *B. capensis* eggs are non-adhesive whereas many other lithophilic cyprinids have adhesive eggs (Mills 1991). *B. capensis* has a suite of reproductive traits which include relatively late maturity, increasing reproductive effort with age and the potential to spawn in successive years. This suite of traits is typical of many of the larger temperate cyprinids except that many of them have a short spawning season (Mills 1991). Multiple spawning within a spawning season, which provides the capacity to have successive batches of eggs ripen within a season, and a fairly long spawning season (3-4 months) are several additional reproductive traits of *B. capensis*. These are shared with many other temperate African cyprinids (Cambray and Bruton 1984; Tómasson *et al.* 1984; Cambray 1991, 1994). Observations support the hypothesis that *B. capensis* is a repeat spawner, although final proof must wait until individual fish are tagged and watched. However, the high probability of the fish being repeat as well as multiple spawners is important when recommending suitable floods to trigger spawning below Clanwilliam Dam.

13.6.3 Linking dam releases with spawning of *B. capensis*

The eggs found in 1992 were about two days old. Records of releases from Clanwilliam Dam for the relevant period showed that only a modest change from 7.06-7.38 $m^3 s^{-1}$ had occurred. However, records are only kept for flow changes of greater than 24 hours duration, and shorter fluctuations, which have been shown here to trigger abundant spawning activity, may have occurred. Similarly, spawning had occurred before the experimental releases in both 1993 and 1994, but again, the record of dam releases was inconclusive as to why. Thus, spawning that occurred before the experimental freshes may have been triggered by small sustained increases in flow, or by unrecorded larger releases of less than 24 hours duration, or by conditions unrelated to flow changes.

To this confusing picture was added the problem that the success or extent of any of these spawning activities could not be gauged simply by the number of eggs on the spawn beds, as observation indicated that most seemed to be washed into downstream deep pools. In any case, no attempt was made to systematically search and collect eggs from the entire spawn beds because of the Red Data status of the species. Thus, the best available indicator of the success and magnitude of spawning was the survival of larvae and juveniles. Using this as a guide, spawning success did appear to be enhanced by the 1994 experimental freshes, as reflected in the unusually high abundance of larvae found two months later. The poorer success rate of the 1993 experimental freshes may have been due to the presence of researchers working in and along the river, or by other factors not yet understood.

It is not known what conditions triggered spawning before the experimental periods, but during the 1994 experiment there was extensive spawning activity on the receding limbs of the freshes. Evidence from the 1993 experiment, where conditions were as similar as possible to the 1994 experimental period, suggested that spawning did not seem to be linked to significant changes in water chemistry.

It was concluded that although much remains unclear regarding spawning triggers, spawning of *B.* capensis in the Olifants River is occurring through the fortuitous occasional occurrence of favourable conditions and can be enhanced by specifically designed and timed dam releases. Such releases appear to improve the chances of more individuals arriving at the spawning bed at the same time, and in this way probably increase overall spawning success.

13.6.4 River regulation and spawning in other southern African river systems

Cambray (1991) reviewed some of the studies that have included work on river regulation and spawning in southern Africa. In one of the better-studied areas, the flood plains below Pongolapoort Dam, it was recommended that a series of small floods and one large flood between November and March would maintain a natural, self-sustaining population of fishes (Merron and La Hausse de Lalouviere 1987). Many of these species are flood-dependent spawners that can be stimulated to spawn in the newly-flooded pans by the artificial flooding of this system. They are not main channel spawners as is the case with *B. capensis* in the Olifants River.

Two large *Barbus* species in the Orange River, *B. aeneus* and *B. kimberleyensis*, spawn on gravel/cobble beds within the main channel below Gariep Dam on the Orange River (Tómasson *et al.* 1984). It was recommended that manipulations such as releasing epilimnetic water in spring and early summer would lead to earlier breeding and higher recruitment for these species. There are no multi-level oulets in Clanwilliam Dam to aid mixing of released water to achieve a suitable downstream spawning temperature. If the dam wall is raised in future, as has been proposed, then urgent consideration should be given to providing such a facility.

The final phase of gonad maturation and the release of the gametes at a specific spawning site may require one or all of the following stimuli: increased current velocity, water quality changes, barometric pressure changes or pheromone releases (Lam 1983; Stacey 1984; Nestler *et al.* 1988). For riverine fish species, such as *B. capensis,* which occur in a winter rainfall area and spawn during the summer months, the risk of not being able to spawn can be reduced by the strategy of a prolonged potential spawning period over three to four months coupled with the ability to spawn on only a slight increase in flow. These minor increases in flow may be sufficient to prepare the spawning beds for spawning and cue local populations in the immediate area to congregate and spawn. Other southern Africa cyprinid fish species occurring in summer rainfall areas, such as *Pseudobarbus afer*, *P. asper* and *P. quathlambae* (Cambray and Meyer 1988; Cambray 1991, 1994), have been recorded as responding very quickly to increased flow. *Labeo umbratus* are also known to spawn after localised rainfall events (Tómasson *et al.* 1984).

13.6.5 Management of flows in the Olifants River for conservation purposes

The ever increasing negative impacts of catchment development are having a deleterious effect on the remaining populations of the potamodromous (freshwater migrator) *B. capensis*. The species spawns during the summer months of low flow, which is the time of maximum water abstraction for agricultural purposes (King and Tharme 1994). At times in summer, the middle reaches of the river may cease to flow or even become dry. Additionally, construction of Clanwilliam and Bulshoek Dams has eradicated from the middle and lower reaches of the Olifants River the small flow increases due to dry-season local rainfalls that probably

triggered spawning. With such extreme flow modifications, a species reliant on small flow pulses in the dry season is extremely vulnerable at spawning time.

From the results of these preliminary studies it would appear that *B. capensis* can be triggered to spawn by managed small flow peaks below Clanwilliam Dam. It is now important to conserve this spawning area and to further study and manage the water releases to increase the recruitment of this once abundant large yellowfish in this section of the Olifants River. Studies earlier in the spawning season are now required. A water release strategy during November to trigger spawning should be considered. Studies should also commence on the larval and juvenile habitats below the spawning site. It would appear that there is a heavy mortality once the larval fish develop and move into the deeper, main channel. A concerted effort to remove alien fish from this area is urgently required. It is clear from this study that an understanding of the spawning ecology of *B. capensis* will provide river managers with data on the instream flows for the recovery of this species. These are the same objectives used for the endangered Colorado squawfish, *Ptychocheilus lucius* (Nestler *et al.* 1988).

13.6.6 Management of spawn beds at Clanwilliam

Loss of spawning grounds

In a review of cyprinid reproduction, Mills (1991) noted that it is important to establish the extent of upstream movements and to ensure that the adult fish remain free to migrate upstream to spawn. A migrating species such as *B. capensis* not only loses large sections of the river when dam walls such as at Clanwilliam and Bulshoek are erected but also loses spawning grounds. There were probably few natural spawning beds in the area between and below the two impassable dam walls of Bulshoek and Clanwilliam anyway, although " above the latter there is an uninterrupted passage to the upper reaches in the gorge," Harrison (1950). Recruitment of *B. capensis* in this gorge area still occurs (Impson pers. comm. 1994). The population between the dams, however, exists in what is essentially a sand-bed river, and the few metres of spawn bed observed during this study appear to be all that is available. Many eggs must get washed from this area rather than sinking between the gravel and cobbles. Expanding the area of spawn beds, guided by those with knowledge of how they are used, would be a practical way of enhancing recruitment of the population in the future. Such modifications should be linked to water release patterns designed to maintain the condition of the spawn beds, ensure that appropriate spawning triggers occur and safeguard the drifting larval *B. capensis*. A final key to enhanced recruitment of the species would be implementation of control measures for the most important alien fish species, namely bass, tilapia and bluegill sunfish.

13.6.7 Future work

Balon (1985) illustrated that in controlled aquarium conditions more complex series of spawning behaviours can be observed than under natural conditions. Current speeds, group activity during vigorous spawning activity and lighting conditions will restrict observations in nature. More time spent filming *B. capensis* under slow-flowing water in the wild could yield the finer details of spawning. An underwater video camera at spawning site A, remotely operated, would have provided important information on spawning behaviour. There is also an urgent need for further studies of the larval site, with special regard to food resources and predation by alien fish species. Rowntree and Wadeson (in press) studied the effect of the December 1993 flow releases on biotope characteristics at a similar site several kilometres downstream of the larval site. They recommended that there is a need to assess the impacts of flow releases for both irrigation and fish spawning.

Depending on their magnitude, duration and frequency, releases from Clanwilliam Dam may have a serious impact on aquatic life in the sand bed channel of the Olifants River that needs to be assessed.

13.7 CONCLUSIONS

Collections and immediate microscopic examination of live eggs, free embryos, larval and juvenile fish, combined with direct observations and video recordings of spawning behaviour helped to interpret the reproductive biology of *B. capensis* in the Olifants River below Clanwilliam Dam. *B. capensis* is a mid-channel spawner, and probably a repeat spawner, over several days and a multiple spawner over the four months, October to January. The spawning beds studied were a mixture of gravel, cobble and boulders with no cover. During and after freshes released from Clanwilliam Dam, large individuals (> 60 cm) maintained position for several hours on the spawning bed and were occasionally joined by smaller individuals which aligned beside the larger one, vibrated, released milt and then returned to edge cover or moved upstream. The large individuals usually remained in position on the spawning beds and awaited another spawning event. Spawning episodes occurred throughout the day following the fresh. The eggs and the photophobic free embryos appear to remain in the spawn bed for 10-12 days. Thereafter there is a swim-up stage when the fish are carried by currents to the larval fish site several kilometres downstream. Larval *B. capensis* favour areas characterised by shallow (< 30 cm), warm water and low velocities (< 0.2 m s^{-1}) with a sand substratum.

It is suggested that the release of water from Clanwilliam Dam can be used as a management tool to help conserve the population of *B. capensis* downstream of the dams on the Olifants River. There is an urgent need for a programme to remove alien fish species, to declare the area downstream of Clanwilliam Dam a conservation area for this magnificent, large, rare cyprinid and to study the impacts of dam releases on the downstream biota.

14. LINKED EFFECTS OF DAM-RELEASED FRESHES AND WATER TEMPERATURE ON SPAWNING OF THE CLANWILLIAM YELLOWFISH *BARBUS CAPENSIS*

Written by J. King, J.A. Cambray (Albany Museum) and D. Impson (Western Cape Nature Conservation)

14.1 INTRODUCTION

14.2 STUDY SITE

14.3 METHODS
14.3.1 Water temperature and associated physical variables
14.3.2 Fish surveys
14.3.3 Larval counts and bridge watch
14.3.4 Water release strategy
14.3.5 Spawn bed hydraulics
14.3.6 Fish response to experimental freshes

14.4 RESULTS
14.4.1 Water temperature and associated physical variables
14.4.2 Fish surveys
14.4.3 Larval counts and bridge watch
14.4.4 Water release strategy and related fish activity
14.4.5 Presence of eggs and larvae
14.4.6 Hydraulics and related characteristics of the spawn beds

14.5 DISCUSSION
14.5.1 Upstream migrations
14.5.2 Water temperature and flow as triggers for spawning
14.5.3 Hydraulic conditions related to spawning
14.5.4 Reproductive style
14.5.5 Conservation considerations

14.6 CONCLUSIONS

This chapter describes a short follow-on research project after that described in Chapter 13. The purpose of the study was to investigate if the successful spawning experiment of the previous year could be repeated.

14.1 INTRODUCTION

Chapter 13 described experimental freshes released from Clanwilliam Dam on the Olifants River, at the beginning of the dry season in 1993 and 1994. This correlated with the suspected spawning season of a large cyprinid endemic to the system, the Clanwilliam yellowfish *B. capensis*. During those research trips, two spawn beds used by *B. capensis* were located downstream of the dam and spawning of the species was witnessed. Eggs, free embryos and larvae were collected and their stages of development described.

During that study, a picture began to emerge of multiple and possibly repeat spawning in late spring and early summer (November-January). This seemed to be linked to rising water temperatures in the river downstream of the dam, which brought the fish into breeding condition. In captive breeding experiments, temperatures at or above 20-22 °C were linked to achieving this condition. Once in breeding condition, fluctuating releases of water from the dam appeared to trigger spawning. In recent years, appropriate fluctuations might have occurred fortuitously due to changes in releases for downstream irrigators. In 1994, however, extensive spawning followed the release of two experimental freshes, with subsequent recruitment levels that were considered unusually high. No obvious changes in water quality and temperature occurred during these freshes, which merely increased the magnitude of the hypolimnetic releases.

A link between freshes and spawning seemed to be emerging, but could the results of the experiment be repeated? To answer this, a further experiment was planned for the spring of 1995. This chapter reports on the 1995 study of experimental releases and the responses to these of *B. capensis*. The study focused on the following questions:

- what were the water temperatures in the reach downstream of Clanwilliam Dam in the weeks leading up to the experimental period;
- did these temperatures correlate with the distribution and movement of *B. capensis* in the river downstream of the dam;
- could the success of the 1994 experimental freshes be repeated, using the same pattern and magnitude of releases at the same time of year?

14.2 STUDY SITE

The study site was described in Chapter 13.

14.3 METHODS

14.3.1 Water temperature and associated physical variables

A maximum-minimum thermometer was placed in the river at the farm Hoopvol 7 km downstream from Clanwilliam Dam. From 3 September 1995, it was read by the farmer daily and the results faxed through to the University of Cape Town weekly. Occasional readings were also taken at the spawn site by the DWAF dam operator, when flows were sufficiently low to maintain the thermometer in the river. These records informed researchers of when the river's temperature was reaching the assumed critical level of 20-22 °C.

Once the experimental releases began, temperature readings at one or both spawn beds were taken at approximately hourly intervals during daylight hours, along with readings of air temperature, water level and dissolved oxygen.

14.3.2 Fish surveys

One of the authors (DI) and other divers completed the fish counts using snorkel or scuba gear. Four sites between Bulshoek Weir and Clanwilliam Dam were visited on three occasions before the experimental freshes and one occasion afterwards. The counts were designed to inform on whether or not *B. capensis* were distributed throughout the reach in winter and moved upstream in spring as water temperatures increased. Counts were limited in terms of visits and sites due to a shortage of experienced divers and the inaccessibility of the area. The sites were all pools and were located as follows:

- Pool 1: surrounded by palmiet, downstream of the dam wall and upstream of the spawn site;
- Pool 2: surrounded by palmiet, immediately downstream of the spawn site;
- Pool 3: in the open channel at the larval site, 2 km downstream of the spawn site;
- Pool 4: in the open channel, 0.5 km upstream of the full-supply level of Bulshoek reservoir.

It was intended to establish at least one site between pools 3 and 4. However, this reach had no known fish refugia in the form of deep pools, at least partly because extensive stretches had been bulldozed to raise the banks and decrease flooding of adjacent cropland.

Fish surveys were completed by the divers counting fish species as they moved downstream through a pool in line abreast, and then repeating the count in an upstream direction. Each diver was allocated a segment of water to his front and partially to each side, and encompassing the whole water column.

14.3.3 Larval counts and bridge watch

The authors could not maintain a constant presence at the spawn site. However, the decision to begin experimental releases could not be made without some knowledge of how fish behaviour was changing as temperatures increased. Thus, two activities were initiated, both carried out by local inhabitants of Clanwilliam. As sporadic spawning does occur at the site (Chapter 13), an employee of Western Cape Nature Conservation (WCNC) visited the larval site daily from mid November to watch for the first appearance of fish larvae.

Additionally, one of two local people carried out bridge watches as their time allowed, to record signs of fish activity on or near the spawn beds.

14.3.4 Water release strategy

The same strategy for experimental releases as in 1994 (Section 13.4.1) was planned. Once temperatures were at the critical level and some larval fish had been found, releases from the dam were to be reduced to store water for the experimental freshes. This operation was managed by DWAF with the knowledge and co-operation of downstream irrigation farmers. The planned schedule of storage and releases was the same as in 1994: from a normal irrigation release of 7.5-8.0 $m^3 s^{-1}$, discharge was to be dropped to about 3.5 $m^3 s^{-1}$ for four days to store water. Two experimental freshes were then planned, one of 17.0 $m^3 s^{-1}$ for 12 hours overnight, and one of 9.0 $m^3 s^{-1}$ for 12 hours the following night. Releases would drop to 3.5 $m^3 s^{-1}$ between freshes, and return to the background level of about 7.0 $m^3 s^{-1}$ a few hours after the second fresh because of irrigation demands. Developments during the experimental period led to changes in this release pattern, as described in Section 14.4.4.

Temporary gauge plates were erected at the edge of the spawn beds to record the water surface elevations (WSEs) associated with fluctuating discharges. The plates were not surveyed in to a common datum, and so the same reading on both did not necessarily reflect the same WSE. The gauge plates showed that freshes lagged about two hours behind the time of release at the dam, as they flowed through the upstream braided channel and dense palmiet beds (Chapter 13).

14.3.5 Spawn bed hydraulics

Before the experimental freshes were initiated, hydraulic measurements were taken at bed A (Figure 14.1) where spawning exclusively occurred during the 1994-observation period. No attempt had been made to measure hydraulic conditions at the bed in 1994, so as not to disturb the spawning fish. However, in the water-storage phase of the 1995 experiment, releases were at almost the same level as during spawning in 1994 (3.4 $m^3 s^{-1}$ in 1995 as opposed to 3.2 $m^3 s^{-1}$ in 1994). The opportunity was thus taken to obtain data on the approximate hydraulic conditions prevailing at bed A during the 1994 spawning. Immediately after the 1995 experimental freshes, hydraulic measurements were also taken at the other spawn bed (bed B) at a higher discharge, for reasons explained in Section 14.4.

For each bed, a grid was delineated using tapes. This covered the whole wetted area around and including the spawn bed. For the large bed A, measurements were taken every metre from bank to bank, at cross-sections laid out every two metres along the river length. For the smaller bed B, cross-sections were one metre apart. At every measurement point the following were recorded: distance from wetted edge, dominant and sub-dominant substrata, presence and nature of aquatic vegetation, overhead cover, water depth, visual appearance of flow (flow type) and velocity at six-tenths depth (average velocity). Velocities were recorded with a Scientific Instruments AA flow meter. Particle sizes of substrata (after King and Tharme 1994) were sand: < 2 mm; gravel 2-32 mm; small cobble 32-128 mm; medium cobble 128-256 mm; large cobble 256-512 mm; and boulder > 512 mm.

14.3.6 Fish response to experimental freshes

Before, during and after the experimental freshes, observations of fish activity at and near the spawn beds were recorded from the bridge. Four locations were observed (Figure 14.1): the two spawn beds and two nearby pools. Pool D was also pool 2 of the fish survey. For bed A, the observations usually lasted for 10-15 minutes in each daylight hour, whilst observations at the other locations were sporadic depending on time availability. An observation record consisted of a count of any fish movement seen through the area within the observation time. Movement could be upstream or downstream, and the count could be of one fish several times or of many different ones as it was impossible to distinguish between the two.

Eggs, free embryos and larval fish were collected before and after the experimental freshes, as described in Chapter 13. They were also measured and photographed as described in that chapter.

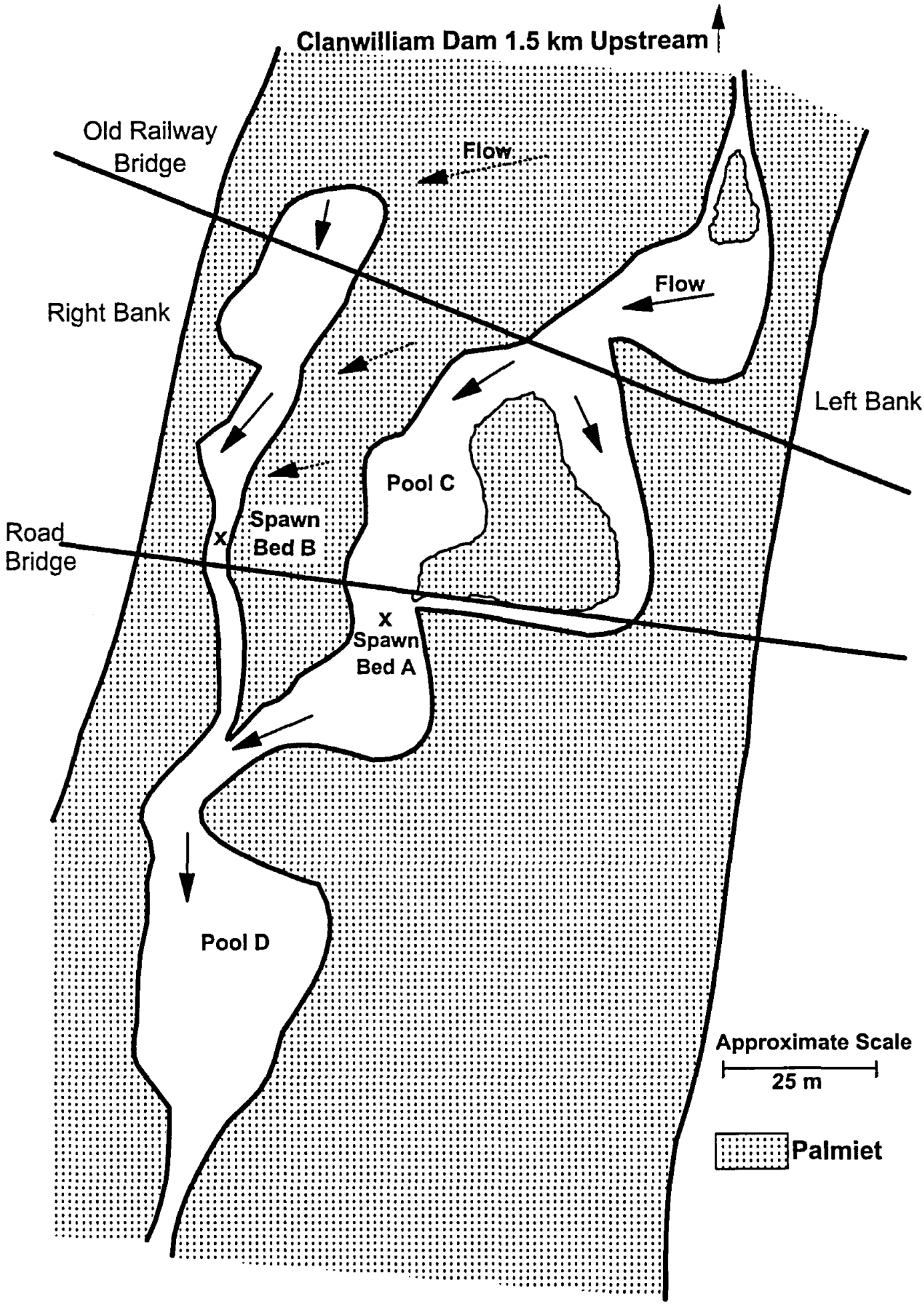


Figure 14.1 Sketch map of the spawn site for *Barbus capensis* in the middle reaches of the Olifants River, South Africa.

14.4 RESULTS

14.4.1 Water temperature and associated physical variables

Water temperature records for Hoopvol, 7 km downstream of the dam, revealed daily maxima of 16-21 °C and minima of 12-19 °C in September and October (Figure 14.2). At this time the dam was spilling, and some fluctuations may have been due to a mixture of hypolimnetic releases and surface spillage. Two large floods in October increased discharges downstream of the dam to 80 $m^3 s^{-1}$ at times, with the second one also noticeably reducing temperatures for two to three days.

Water temperatures at Hoopvol started to climb in early November until surface spillage stopped on 15 November (Figure 14.3). At this time, water temperatures at the spawn site were 22-23 °C and at Hoopvol were 22-28 °C. Hypolimnetic releases then became the only source of downstream water, causing temperatures to drop to 15-16 °C at the spawn site. Daily maxima at Hoopvol fell by about 8 °C and minima by about 5 °C. Although daily maxima rose again at Hoopvol within about two weeks, to close to the temperatures occurring before surface spillage stopped, daily minima did not approach a sustained recovery until toward the end of December. Thus, the cold water released during the day was being warmed by high air temperatures, but water temperatures at Hoopvol at night were little above those of the hypolimnetic release. Occasional temperature readings at the spawn site until 17 December reflected this sluggish recovery (Figure 14.3).

Water temperatures during the experimental freshes, and data on associated physical variables, are dealt with in Section 14.4.4.

14.4.2 Fish surveys

Late rains and flooding in the river up to the end of October delayed diving trips, whilst the irregular releases of hypolimnetic water from the dam produced cold turbid waters that made counts difficult. Surveys were completed on 30 September and 15 November, and then on the days before and after the first experimental fresh (28 and 29 November).

The pools used for the fish surveys reflect the general character of that stretch of river (Table 14.1). They were not identical in size, but by repeating the sampling effort at each pool on each visit, results over time for any one pool could be compared. Pools 1 and 2 were in vegetated braided channels, whilst 3 and 4 were in open sandy areas. A single upstream or downstream count took from 2-5 minutes in pools 1 and 2, and 5-9 minutes in pool 3. No *B. capensis* were seen in the most downstream pool (pool 4) on either of the first two surveys, and it was not re-visited. Pool 4 contained alien smallmouth bass *Micropterus dolomieu* and banded tilapia *Tilapia sparrmanii*.

Water chemistry variables measured reflected the pure, slightly acidic character of the water (Table 14.1). Further details of the water chemistry of this stretch were given in King and Tharme (1994) and in Chapter 13.

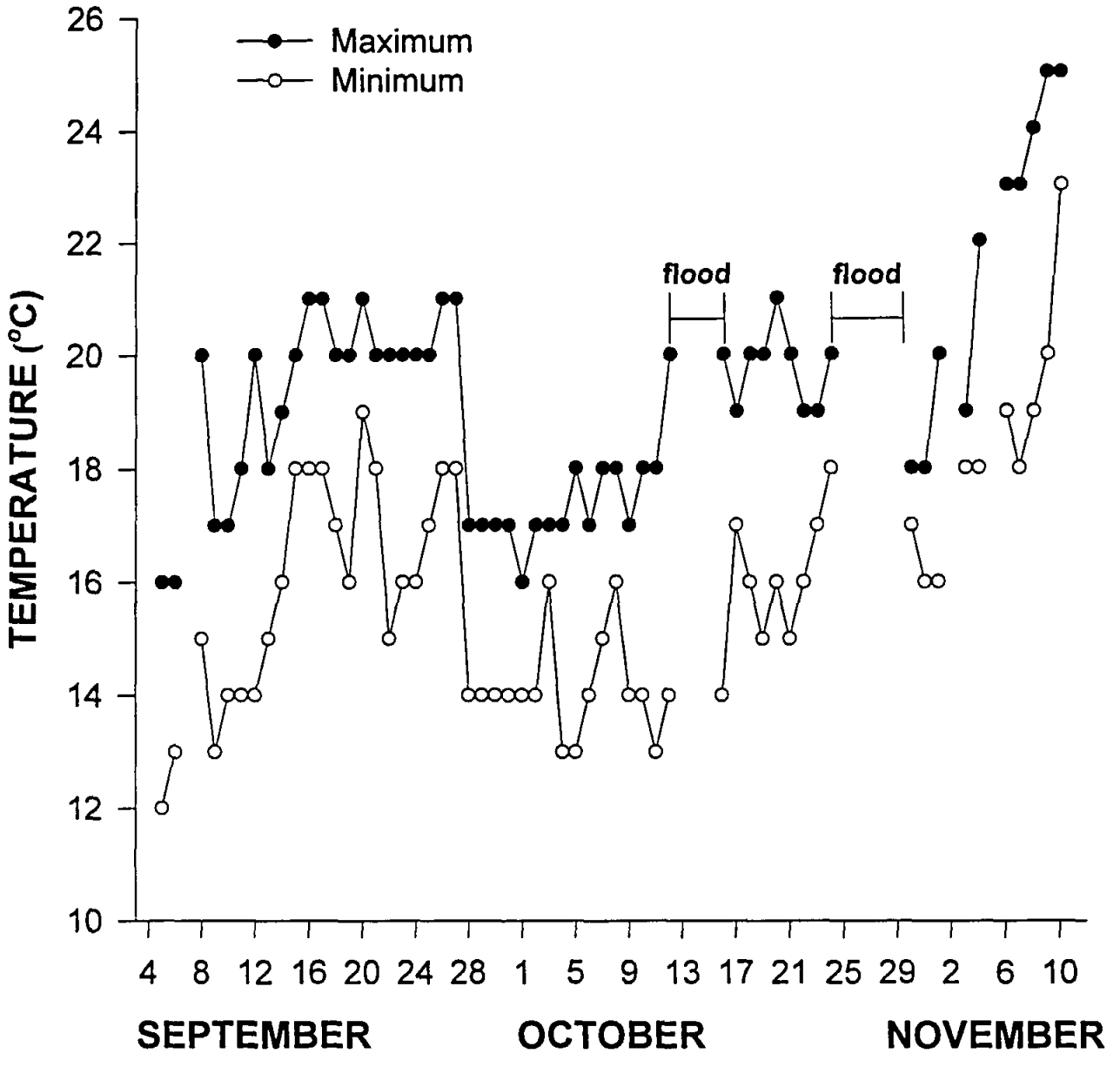


Figure 14.2 **Maximum and minimum water temperatures at the farm Hoopvol, 7 km downstream of Clanwilliam Dam on the Olifants River, from 4 September to 10 November 1995. Gaps in data are due to removal of thermometers during high flows.**

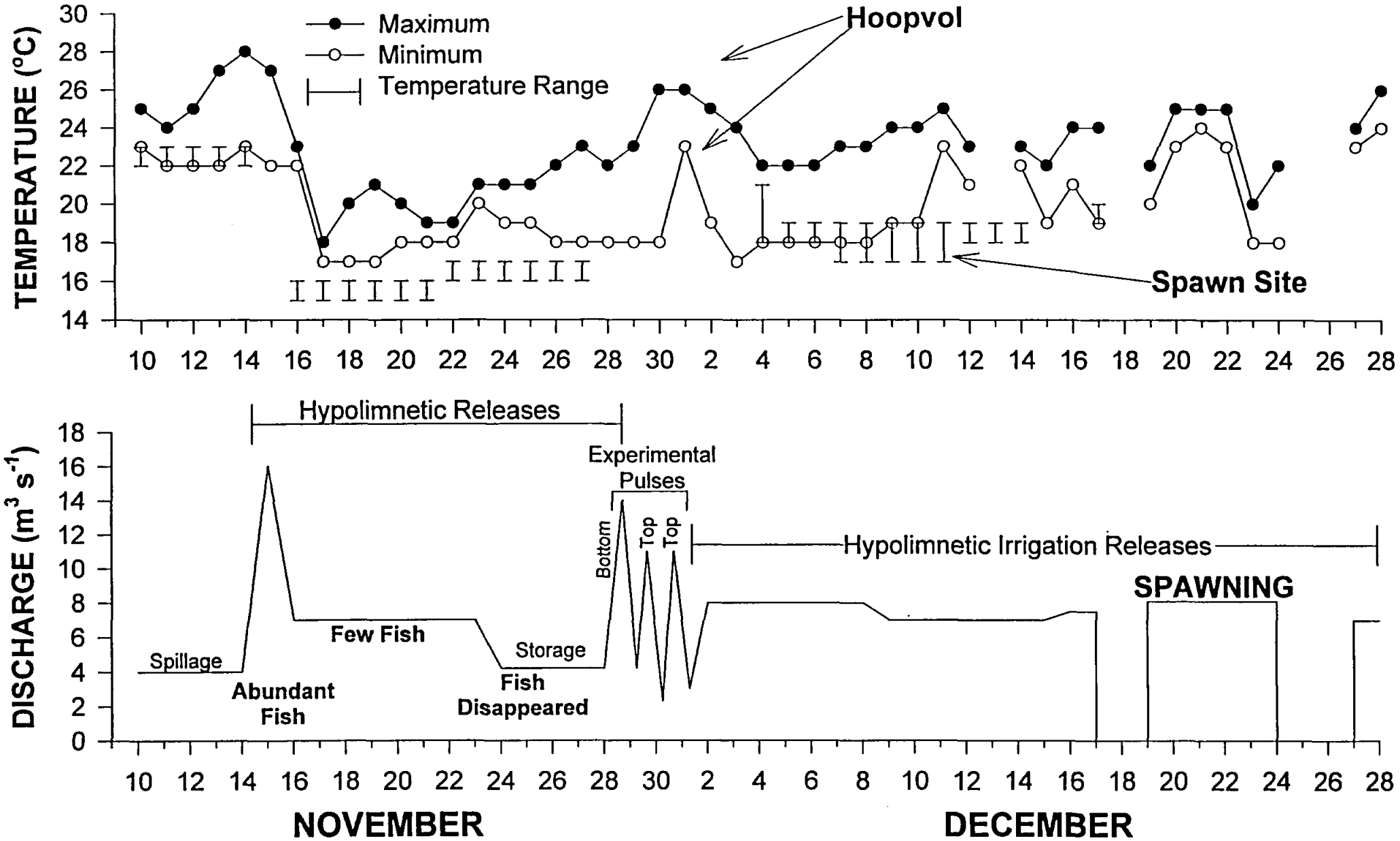


Figure 14.3 Water temperatures at the spawn site and at Hoopvol, and water release and spillage patterns at Clanwilliam Dam, from 10 November to 28 December 1995. Temperatures shown for Hoopvol are daily maxima and minima. The sequence of days with similar values shown for the spawn site is the result of occasional spot readings, which indicated the temperature range recorded by max-min thermometers during the preceding few days. Releases from the dam fell to zero on two occasions in late December, for operational reasons.

Table 14.1 Characteristics of fish survey pools.

	POOL 1				POOL 2			
LATITUDE / LONGITUDE	32° 11' 00" / 18° 52' 30"				32° 10' 20" / 18°52' 18"			
Location	100 m downstream of dam				30 m downstream of spawn site			
Approximate Length (m)	30				15			
Approximate Width (m)	10				6			
Average Depth (m)	1.5 – 2.0				1.0 –2.0			
General Appearance	Cobble, bedrock and some boulders in current with sandy beds in backwaters. Moderate presence of palmiet and *Salix*.				Bedrock and some boulders in current with vegetated sandy beds in backwaters. Profuse marginal plant cover.			
Date of visit	20 Sep	15 Nov	28 Nov	29 Nov	20 Sep	15 Nov	28 Nov	29 Nov
Visibility (m)	2.0 – 3.0	2.0 – 3.0	1.5 –2.0	1.0 – 1.5	2.0 – 3.0	3.0 – 4.0	3.0 – 4.0	3.0
pH	6.4	6.3	6.2	6.4	6.4	6.3	5.9	6.3
Water Temperature (°C)	16.7	23.0	17.5	18.0	16.6	23.2	18.5	17.2
Conductivity (mS m^{-1})	22.0	15.5	7.1	8.1	13.9	15.5	6.5	7.1

	POOL 3				POOL 4			
LATITUDE / LONGITUDE	32° 10' 04" / 18° 52' 25"				32° 03' 50" / 18° 49' 30"			
Location	2 km downstream of spawn site				0.5 km upstream of Bulshoek reservoir			
Approximate Length (m)	100 (60 m surveyed)				150 (60 m surveyed)			
Approximate Width (m)	10				30			
Average Depth (m)	2.0 – 3.0				2.5 – 3.5			
General Appearance	Bedrock and some boulders in current with vegetated sandy beds in backwaters. Profuse marginal plant cover.				Bedrock and some boulders in current with vegetated sandy beds in backwaters. Moderate presence of palmiet at margins.			
Date of visit	20 Sep	15 Nov	28 Nov	29 Nov	20 Sep	15 Nov	28 Nov	29 Nov
Visibility (m)	3.0	1.5	4.0	3.0	3.0	-	4.0	1.0 – 1.5
pH	6.6	6.1	6.7	-	6.6	-	6.3	6.1
Water Temperature (°C)	17.3	20.6	19.4	-	19.0	-	24.4	17.4
Conductivity (mS m^{-1})	8.8	6.6	8.0	-	13.6	-	10.9	9.8

Pools 1-3 contained *B. capensis* on all visits (Table 14.2), but numbers varied. The counts are inadequate for statistical analysis, but some general comments on fish movements can be made, albeit at low confidence level. The two pools closest to the dam showed similar trends, with the number of counted *B. capensis* generally being higher on 15 November than on 30 September. This possibly reflected the upstream movement of fish, encouraged by the warm surface waters from the dam. The counts were considerably lower again on 28 November, perhaps reflecting a downstream movement of fish in response to the cold, hypolimnetic releases that started on 15 November. Numbers were generally higher again after the first (hypolimnetic) experimental fresh, possibly reflecting upstream movement in response to the higher flows.

Pool 3, some 2 km downstream, on sandy warm flats, showed an opposite pattern, but not clearly. Counts on 28 November were higher than before the switch to hypolimnetic releases, and it seems possible that fish were moving downstream from pools 1 and 2 into the somewhat warmer waters at pool 3.

14.4.3 Larval counts and bridge watch

Larval counts and the bridge watch were designed to aid the authors to decide, from a distance, when the fish were ready to spawn, and thus when water storage for the experimental freshes should begin.

Table 14.2 **Fish counts at pools 1, 2, and 3 on three occasions before and one occasion after the first experimental fresh. YF - Clanwilliam yellowfish, Alien - alien fish species. Pool 2 was also pool D as shown in Figure 14.1.**

	DIVER	EQUIPMENT	DIRECTION	20 SEPT YF	20 SEPT ALIEN	15 NOV YF	15 NOV ALIEN	28 NOV YF	28 NOV ALIEN	29 NOV YF	29 NOV ALIEN
POOL 1	1	snorkel	downstream	6	0	2	1	4	0	13	0
			upstream	3	0	1	0	3	0	7	0
	2	scuba	downstream	1	0	-	-	3	0	-	-
			upstream	0	0	-	-	1	0	-	-
	3	scuba	downstream	14	1	13*	2	2	0	4	0
			upstream	8	0	6	1	0	0	3	1
	4	snorkel	downstream	4	0	-	-	5	1	-	-
			upstream	10	1	-	-	6	0	-	-
			Total downstream	25		15		14		17	
			Average per diver	6.3		7.5		3.5		8.5	
			Total upstream	21		7		10		10	
			Average per diver	5.3		3.5		2.5		5.0	
POOL 2	1	snorkel	downstream	7	0	16	0	0	0	6	0
			upstream	3	0	15	0	0	4	2	0
	2	snorkel	downstream	0	0	-	-	0	0	-	-
			upstream	11	0	-	-	0	0	-	-
	3	scuba	downstream	6	0	17	0	0	0	13	0
			upstream	7	0	6**	0	0	0	13	0
	4	scuba	downstream	2	0	-	-	11	0	-	-
			upstream	0	0	-	-	3	0	-	-
			Total downstream	15		33		11		19	
			Average per diver	3.8		16.5		2.8		9.5	
			Total upstream	21		21		3		15	
			Average per diver	5.35		10.5		0.5		7.5	
POOL 3	1	scuba	downstream	14	0	3	1	11	0	-	-
			upstream	18	0	5	3	21	0	-	-
	2	snorkel	downstream	8	3	2	1	13	0	5	0
			upstream	20	0	4	2	3	0	9	0
	3	scuba	downstream	14	1	-	-	7	0	7	2
			upstream	7	0	-	-	9	0	15	2
	4	snorkel	downstream	14	0	5	1	24	0	-	-
			upstream	25	0	20	2	9	0	-	-
			Total downstream	50		10		55		12	
			Average per diver	12.5		3.3		13.8		6.0	
			Total upstream	70		29		42		24	
			Average per diver	4.5		9.7		10.5		12.0	

*may be overcount

**probably undercount

Quantitative counts of the larvae in the sheltered edge areas of the larval site were attempted, with the hope of ascertaining from these the approximate dates and degree of success of earlier upstream spawning events. These proved unsuccessful, as fluctuating releases from the dam and the vagaries of weather meant that larval numbers differed widely between days and areas. When non-quantified larval counts began on 13 November, however, six larval *B. capensis* were found, followed by approximately 40, 50, 20, 60 and 100-120 larvae on 17-21 November. The presence of the larvae indicated that it was time to begin storage for the experiment.

Bridge watches did not develop into a structured programme. Anecdotal information for the two weeks up to the experiment revealed that there had been many fish in the area and around the two spawn beds up to about 18 November, but none after that. Their disappearance correlated well with the switch to hypolimnetic releases.

14.4.4 Water release strategy and related fish activity

The fish survey on 15 November and the presence of larval fish on 13 November showed that *B. capensis* was in spawning condition. Thus, water for the experiment was stored from 25-28 November (Figure 14.2) through reducing the releases to meet downstream agricultural demand. At the same time, and coincidentally, wet-season spillage over Clanwilliam Dam became insufficient for downstream needs and top (epilimnetic) releases were halted altogether. Thus, as the experiment began, the switch had to be made from warm surface spillage to cold hypolimnetic releases. As a result, water temperatures at the spawn site at the beginning of the experiment were 3-4 °C cooler than during the successful spawning in 1994. This situation may well have accounted for the disappearance of the fish from the immediate vicinity of the spawn beds (Sections 14.4.3-14.4.4). However, it was decided to proceed with the experimental freshes in the hope that these would trigger an upstream spawning run. The first fresh was released on 28 November, three days earlier than in 1994.

Records of fish activity, linked to discharge, air and water temperatures and DO concentrations, were maintained throughout the daylight hours of the day before the first release and for the next four days (Figure 14.4). By that time all three releases had been completed. The values of physical and chemical variables mentioned refer to bed B unless otherwise stated. The pattern of percent saturation of DO shadowed that for its concentration, ranging between 73-99% saturation, and is not shown in Figure 14.4. The weather was sunny and hot apart from a short period of rain (Figure 14.4).

The first experimental fresh

The first experimental fresh, of magnitude 14.0 $m^3 s^{-1}$, was released hypolimnetically for 13 hours overnight on 28-29 November. It was reduced to a hypolimnetic release of 4.2 $m^3 s^{-1}$ at 06h00.

Water temperatures showed a modest increase during the day before the first release, but only within the range 16.9-17.5 °C (Figure 14.4). Judging by those pertaining at daybreak next day, they did not increase with the first fresh. DO levels rose by up to 2 mg ℓ^{-1} during the fresh, presumably mainly due to aeration from the highly-pressured bottom release. As the fresh declined at dawn to a lower flow, DO levels fell again, while water temperatures held steady.

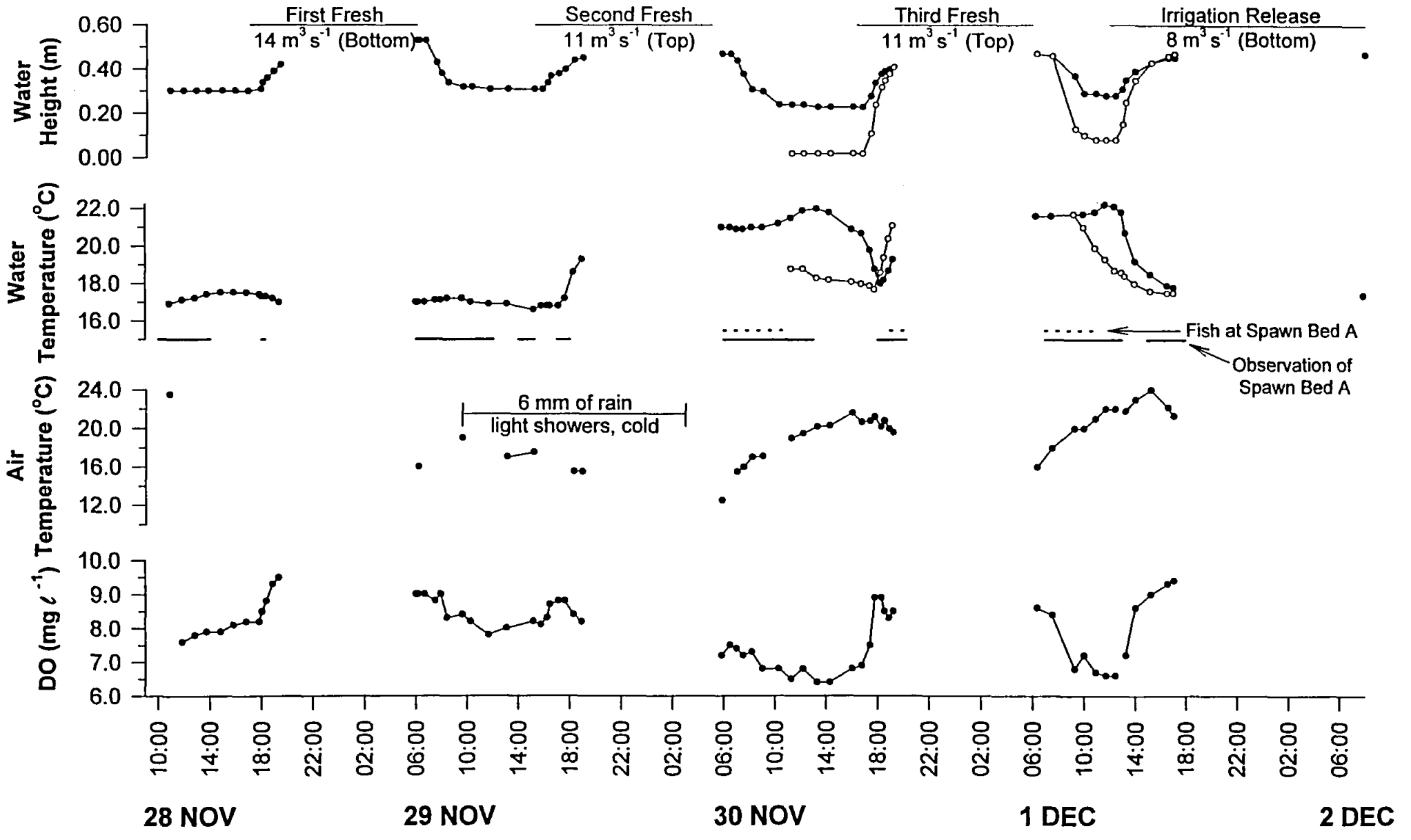


Figure 14.4 **Water level, water temperature, air temperature and dissolved oxygen levels at the spawn site during the experimental hypolimnetic and epilimnetic releases from Clanwilliam Dam, related to the presence of *Barbus capensis* on spawn bed A. Sunrise at 06h15, dark at 20h00. Open circles represent spawn bed A and solid black circles spawn bed B. Solid horizontal lines show times of observation at Bed A, and dashed lines, the presence of the fish on bed A. A final spot reading of water height and temperature was made before departure on 2 December.**

No fish were seen on bed A before the release (Table 14.3), although there were many present immediately downstream in pool D (Table 14.2). Nor did the fish arrive on bed A after the experimental fresh. By mid afternoon it was clear that the fresh had not triggered the same spawning response from the fish as the previous year. Low water temperatures were the suspected cause, although light rain during the afternoon introduced an unexpected confounding factor. The planned release pattern was then changed, to allow a release of warm water over the top of the dam wall. Its magnitude and duration were dictated by dam design and water-resource considerations, but it did run for sufficiently long to change the temperature conditions at the spawn site.

The second fresh

The second experimental fresh, of magnitude 11.0 $m^3 s^{-1}$, was released from the top of the dam wall for 15 hours overnight on 29-30 November. It was reduced to a hypolimnetic release of 2.3 $m^3 s^{-1}$ at 06h08.

The water temperature rose slightly during the day after the first fresh, then fell slightly with the rain (Figure 14.4). It then began to increase noticeably in mid-afternoon, two hours after the top release began, and by daybreak the next morning was 4 °C higher than before the release (21 °C). It did not decline as the fresh gave way to a low hypolimnetic release, but rather showed a further modest increase during the hot, sunny morning. It eventually began to decrease in late afternoon, some ten hours after the top release ceased.

By this time it had been realised that bed A was responding differently to B in terms of WSE and temperature. Readings of both thus were initiated at bed A during the day of 30 November. These showed that water was arriving at bed A faster than at B, where it had to filter through the dense palmiet beds (Figure 14.1). Water levels and water temperatures at bed A showed a much faster response to different kinds of releases from the dam than did those at bed B. Whereas bed B took ten hours for water temperatures to decrease after the top release stopped, those at bed A took less than five hours. Both had reached the same low of 17.7-18.0 °C by 12 hours after the top release ceased.

In contrast to conditions with the first fresh, DO concentrations at bed B did not increase with the second fresh. Instead they showed a decline at dusk, as the water temperatures rose (Figure 14.4). The DO levels were still lower at daybreak and did not increase until late in the afternoon, coinciding with the decline in water temperature. Thus, the highly pressured bottom releases appeared to oxygenate the downstream water more than did top spillage.

Fish were on bed A at daybreak on 30 November (Table 14.3 and Figure 14.4), sometimes in groups of up to 15 individuals, presumably in response to the presence of warm epilimnetic water. There was continuous activity at bed A until about 11h00, five hours after the top release stopped. Fish biologists felt the fish were "preparing to spawn" (Dr K. Hamman, WCNC pers. comm.). All activity then ceased and the fish disappeared. Shortly after that the water temperature at bed A was measured for the first time, at 18.8 °C and falling. In addition to those fish present at bed A, there was a moderate amount of activity in pool C, upstream of bed A, and some activity in pool D, downstream of bed A (Table 14.3).

Table 14.3 **Fish counts on spawn beds A and B per 10 minutes and observed movements in two pools per 10 minutes. * denotes a spot record, ✓ denotes fish movement, and ✓✓ more than 10 movements observed in 10 minutes.**

DATE	TIME	SPAWN BED A	SPAWN BED B	POOL C	POOL D	COMMENTS
27 Nov	12:00-13:00	0				Bottom release continuous at 4.2 $m^3 s^{-1}$
	13:00-14:00	0				
	14:00-15:00	0				
	15:00-16:00	0				
	16:00-17:00	0				
	17:00-18:00	0				
	18:00-19:00	0				
	19:00-20:00	0				
28 Nov	06:00-07:00	0	0			
	07:00-08:00	0	0			
	08:00-09:00	0				
	09:00-10:00	0				
	10:00-11:00	0				
	11:00-12:00	0				
	12:00-13:00	0				
	13:00-14:00	0				
	14:00-15:00					
	15:00-16:00					Fish-count survey
	16:00-17:00					
	17:00-18:00					Bottom release increased to 14.0 $m^3 s^{-1}$ at 17:00
	18:00-19:00	0	1*			
29 Nov	06:00-07:00	0				Bottom release reduced to 4.2 $m^3 s^{-1}$ at 06:00
	07:00-08:00	0				
	08:00-09:00	0				
	09:00-10:00	0				Scattered rain
	10:00-11:00	0				Scattered rain
	11:00-12:00	0				Scattered rain
	12:00-13:00					
	13:00-14:00	0				
	14:00-15:00	0				Light rain
	15:00-16:00					Bottom release stopped. Changed to top release of 11.0 $m^3 s^{-1}$ at 15:15
	16:00-17:00					
	17:00-18:00	0				Heavy rain
30 Nov	05:00-06:00	2.0				
	06:00-07:00	1.3				Top release stopped. Changed to bottom release of 2.3 $m^3 s^{-1}$ at 06:08
	07:00-08:00	0				
	08:00-09:00	6.2		✓		
	09:00-10:00	1.3				
	10:00-11:00	0.3		✓		
	11:00-12:00	0		✓	✓	
	12:00-13:00	1.0		✓		
	13:00-14:00				✓	Sporadic viewing only, as searching for eggs and photographing larvae
	14:00-15:00	0		✓		
	15:00-16:00			✓		
	16:00-17:00			0		Bottom release stopped. Changed to top release of 11.0 $m^3 s^{-1}$ at 16:20
	17:00-18:00	0		✓		
	18:00-19:00	0.2				
	19:00-20:00	0.1				

Table 14.3 Continued.

DATE	TIME	SPAWN BED A	SPAWN BED B	POOL C	POOL D	COMMENTS
1 Dec	06:00-07:00	0.2		✓		
	07:00-08:00	0	1*	✓	✓	Top release stopped. Changed to bottom release of 3.0 $m^3 s^{-1}$ at 07:00
	08:00-09:00	1.7				
	09:00-10:00	2.0		✓	✓	
	10:00-11:00	0.7		✓✓	✓	
	11:00-12:00	0		✓✓	✓✓	
	12:00-13:00	0			✓✓	Bottom release increased to 8.0 $m^3 s^{-1}$ at 12:00 to meet irrigation demand
	13:00-14:00					Searched for eggs and embryos
	14:00-15:00	0	0	0	✓✓	
	15:00-16:00	0.7	0	0	✓✓	
	16:00-17:00	0	0	✓	✓✓	
	17:00-18:00	0.8	0	✓	✓	
2 Dec	08:00-09:00	0	0	✓	✓	
	09:00-10:00			0		
	10:00-11:00	0		0	✓	
	11:00-12:00			✓		
	12:00-13:00	0	0	✓	✓	

As a result of these observations, another fresh of top water was arranged. As before, there were restrictions to its duration and magnitude.

The third fresh

The third experimental fresh, of magnitude 11.0 $m^3 s^{-1}$, was released from the top of the dam wall for almost 15 hours overnight on 30 November-1 December. It was reduced to a hypolimnetic release of 3.0 $m^3 s^{-1}$ at 07h00.

Water temperatures began to rise at the spawn site two hours after the top release began, with the rise beginning earlier and more sharply at bed A (Figure 14.4). By next morning both beds were at 21.6-21.7 °C. The temperature began to fall at bed A about two and a half hours, and at bed B five hours, after the top release stopped.

The DO level began to increase in late afternoon of 30 November, mirroring the falling water temperature at the end of the second fresh (Figure 14.4). It then showed a slight decline till dusk, as the effect of the third fresh manifested as increasing water temperatures. By the following morning DO was still considerably higher than at the same time after the second fresh, for reasons not understood. The level began to rise again in the afternoon, again mirroring the falling water temperature at the end of the third fresh.

Fish were again moving around bed A at daybreak of 1 December, though in smaller numbers and mostly as single individuals or pairs. Again, the fish disappeared at about 11h00, some four hours after the top release stopped. Water temperature at bed A at that time was 19.3-19.9 °C and falling. More activity than on the previous day was noticed at pools C and D, with both pools showing much movement as the fish left bed A. Within an hour of leaving bed A, however, most fish seemed to have moved downstream to pool D.

After the freshes

The hypolimnetic release was increased to 8.0 m^3 s^{-1} at 12h00 on 1 December, in order to meet irrigation demand. Water temperatures remained low and, apart from two brief sightings, no more fish were seen at bed A that day or until observations ceased the following midday.

Daily observations by a Clanwilliam resident continued at the bridge during December, with 15 minutes spent making counts of fish movement at each of the two spawn beds and each of the two pools. Although the high irrigation releases reduced visibility through the water, fish appeared to be absent from these areas until 17 December. At that time, in a confusing sequence of events, light rain occurred, fish were seen at the study site and the dam release was stopped completely overnight for operational reasons. When the release began again next day, spawning occurred prolifically throughout the day (Figure 14.4), with up to 15 fish on spawn bed A at any one time (Mr D. du Toit pers. comm.). The discharge at the time of spawning is not known as releases, and thus hydraulic conditions, fluctuated for some time after the shutdown. Up to 14 December, water temperature immediately downstream of the two spawn beds had ranged between 17 °C and 19 °C. At the time of spawning, it was between 18 °C and 20 °C.

14.4.5 Presence of eggs and larvae

Prior to the first experimental release, spawn beds A and B were inspected for eggs or free embryos. At bed B at 16h30 on the 27 November 1995, no eggs were collected but there were six free embryos ranging in size from 11.0(10.4)-12.0(11.4) mm total length or TL (notochord length-NL). Only one free embryo was collected at bed A, but this may have been because the swift currents made collecting difficult. At 18h00 on the same day another free embryo was collected (11.2 (10.65) mm TL (NL)) at bed B. If these embryos had been developing at 22 °C (see Cambray *et al.* 1997) then they would have been approximately 17-21 days old when collected.

On the 28 November 1995 at 11h20, five larval *B. capensis* were collected from the larval site. These ranged in size from 12.3 (11.5)-15.5 (13.4) mm TL (standard length - SL). The swimbladder had started to divide and their approximate age was 17-21 days. At 13h00 on the same day six free embryos were collected at bed B, ranging in size from 11.2 (10.6) to 12.4 (11.5) mm TL (NL). The largest had just inflated its swimbladder and actively attempted to avoid capture.

On the 30 November 1995 at 14h00, that is, after the first two experimental freshes, a total of 28 larval fish were collected ranging in size from 13.6 (12.0) to 17.0 (14.4) mm fork length or FL (SL). Later that day at 15h40-16h45, both beds A and B were searched before the next experimental release, but no eggs or free embryos were found.

On 1 December 1995 between 12h35 and 12h55, that is, after the third experimental fresh, no eggs or free embryos were found at bed A. At 12h58 to 13h20 no eggs, but several free embryos, were found at bed B. The free embryos would have originated from a spawning prior to the experimental releases.

The presence of free embryos and larvae indicated that some spawning occurred at least three to four weeks prior to the experimental releases, when water temperatures had been above 20 °C. The early life-history

collections also suggested that no spawning occurred from when the cold, hypolimnetic releases began to the end of the experimental period.

14.4.6 Hydraulics and related characteristics of the spawn beds

Hydraulic conditions in the channel encompassing bed A were investigated using data collected from 97 points along six cross-sections. The readings were taken when water was being stored for the first experimental fresh and thus discharge was similar to that occurring when spawning was observed in 1994. Vegetation and substrata maps were drawn of the whole channel at that point (Figure 14.5), as well as contour maps of water depth, average water-column velocity and Froude number (Figures 14.5 and 14.6). Froude number is defined as $v/(gd)^{0.5}$, where v = mean water velocity (m s^{-1}); d = water depth (m); and g = the acceleration due to gravity (m s^{-2}). Gordon *et al.* (1992) stated that it reflects the interaction between flow depth and velocity at a given point. Wadeson (1995) found that it distinguished well between different kinds of hydraulic habitats in rivers. The spawning area, or bed A, was located on the maps from study of the video footage.

Beds of submerged aquatic vegetation, *Aponogeton,* occurred along the edges of the wetted channel, extending well into the channel along the left bank. Dense stands of *Typha* sp. flanked the outside edge of the *Aponogeton* on the left bank, whilst overhead cover on the right bank was provided by trees such as the native willow *Salix mucronata*. Fine sediments, up to the size of gravel, were confined to the vegetated edge areas, with larger particles in the open channel. Boulders and large cobble dominated bed A, giving way downstream to medium and small cobble. In the open channel, the substrata were clean and not embedded in fines. Water depths at bed A were mostly in the range 0.3-0.5 m, and mean column velocities 0.8-1.3 m s^{-1}. Froude numbers were in the range 0.4-0.6.

The hydraulic conditions in the channel encompassing bed B were also investigated. The first eggs of *B. capensis* found in this study were at this bed (Chapter 13), and fish visited it during the successful spawning period at bed A in 1994. However, individuals never held position there for long, and always left without spawning. Possibly, when the release was then at a low 3.0 m^3 s^{-1}, water depths were too shallow at bed B. In 1993, when the dam release was 4.6 m^3 s^{-1}, 11 spot measurements of hydraulic conditions at bed B had shown water depths of no more than 0.26 m and velocities of 0.2-1.0 m s^{-1} (JMK unpubl. data). Thus, at a higher discharge than was present during spawning in 1994, the velocities at bed B encompassed the range present during spawning at bed A, but the depths were lower.

Fish do spawn at bed B, however, as eggs have been found there on several occasions (Chapter 13). It might be that the bed becomes more suitable for spawning when the higher irrigation releases are occurring. Thus, at the end of the 1995 experimental freshes, when the release was increased to 8.0 m^3 s^{-1} for irrigation purposes, the hydraulics of bed B were further investigated. Addressing the same variables as for bed A, 48 readings were taken from eight cross-sections (Figures 14.7 and 14.8).

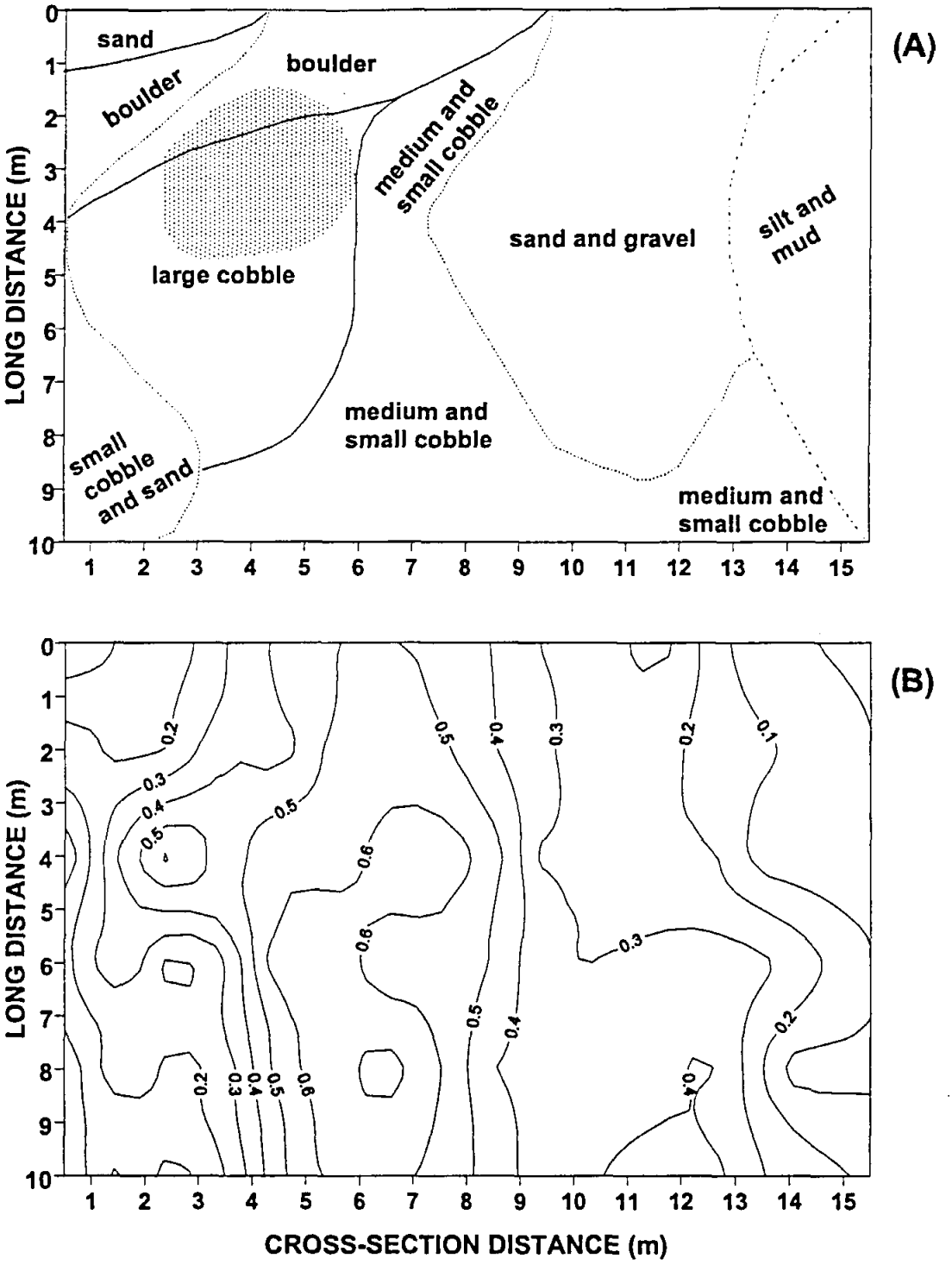


Figure 14.5 Vegetation and substrata maps (A), and contour maps of Froude numbers (B), at a dam release of 3.4 m^3 s^{-1}, for the channel area encompassing the designated spawn bed A. Solid lines demarcate categories of substrata; dotted lines the limit of the macrophyte *Aponogeton*; and dashed lines the limit of the *Typha* stand. All observed spawning of *Barbus capensis* in 1994 (Cambray *et al.* 1997) occurred in the shaded area. Flow direction is from the top of the map.

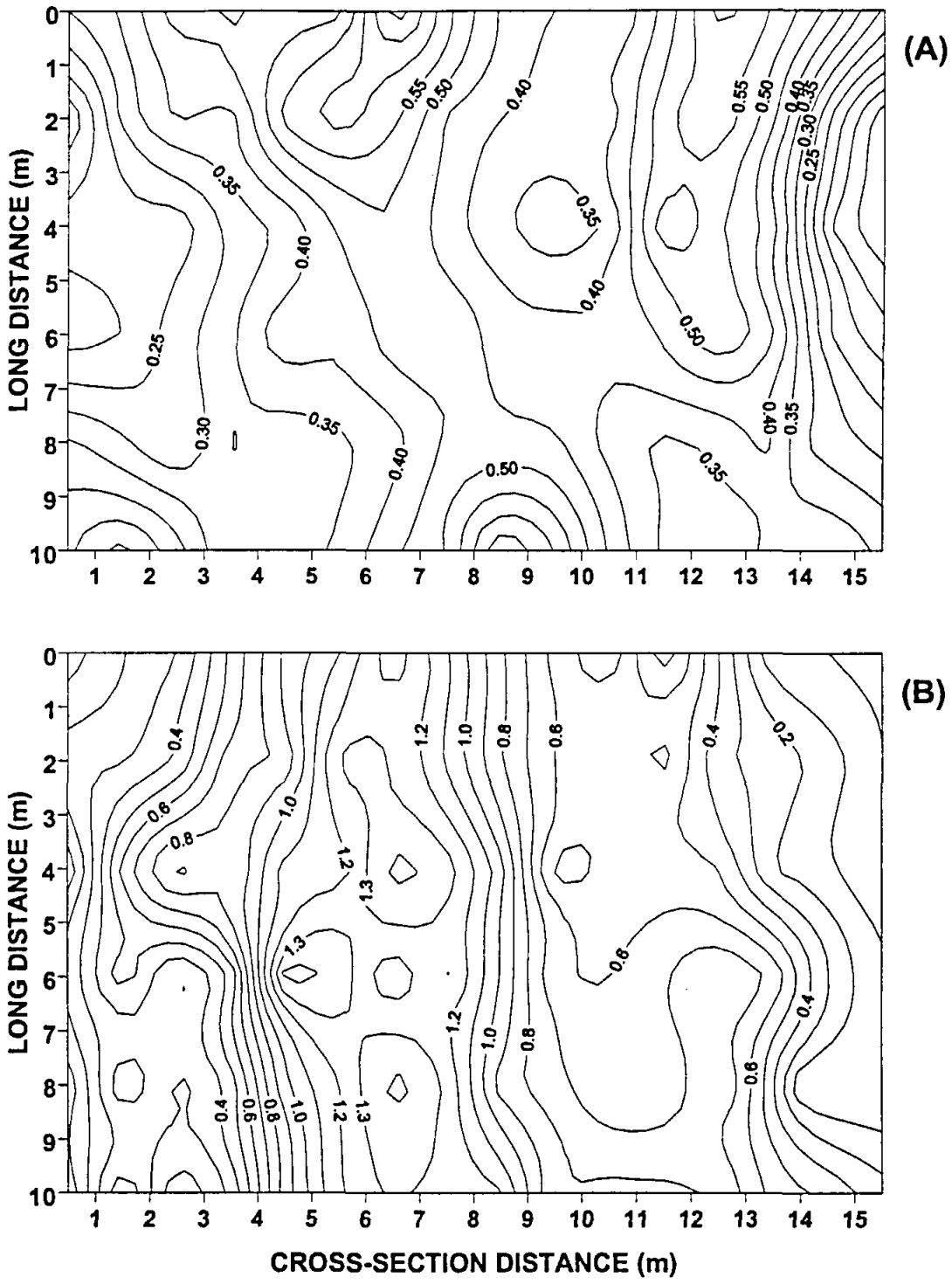


Figure 14.6 Contour maps of water depths (A) and average column velocities (B) at a dam release of 3.4 $m^3 s^{-1}$, for the channel area encompassing the designated spawn bed A.

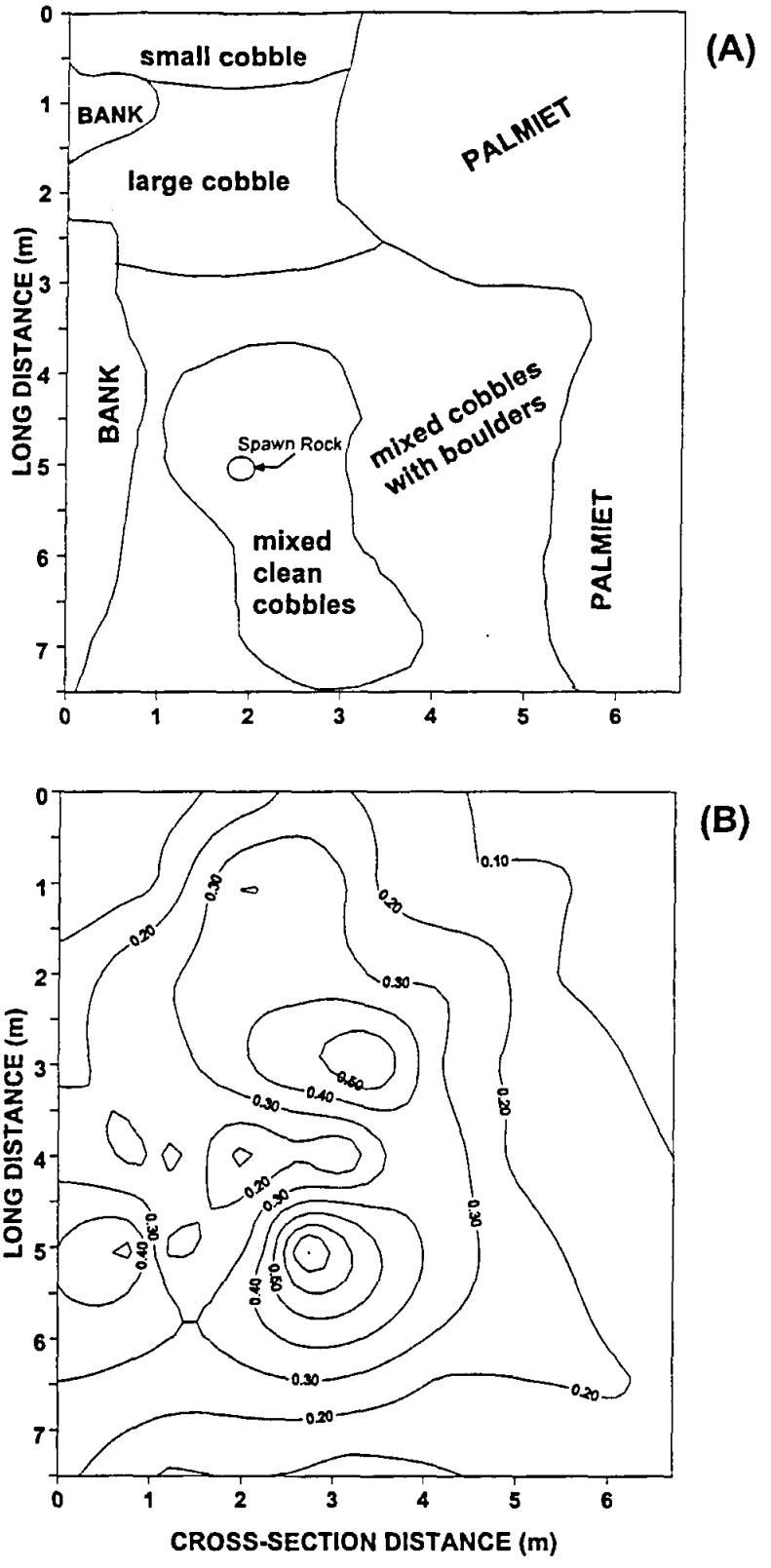


Figure 14.7 Vegetation and substrata maps (A), and contour maps of Froude numbers (B) at a dam release of 8.0 m^3 s^{-1}, for the channel area encompassing the designated spawn bed B. No spawning was observed at bed B on any of the field trips, but live eggs and free embryos were found downstream of "Spawn Rock", each summer from 1992-95. Flow direction is from the top of the map.

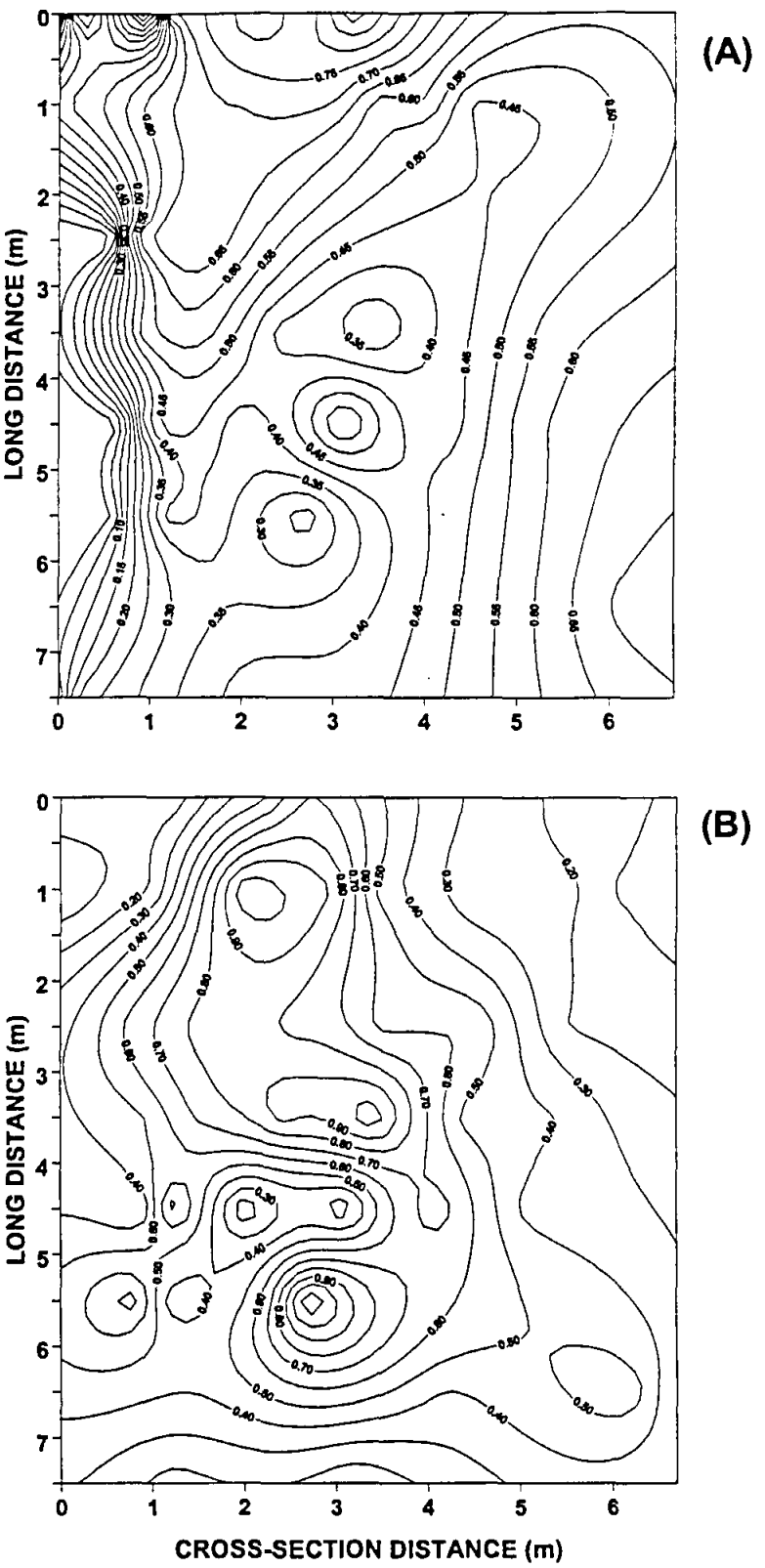


Figure 14.8 **Contour maps of water depths (A) and average column velocities (B) at a dam release of 8.0 $m^3 s^{-1}$, for the channel area encompassing the designated spawn bed B.**

Dense beds of palmiet lined the left bank, with the right bank being low and grassy. An upstream deep pool (Figure 14.1) gave way to a confined fast-flowing reach encompassing bed B. The bed consisted of mixed large, medium and small cobbles with areas of submerged boulders. In most of the reach all of these had a slippery epilithic coating, but were clean in the area of bed B and the site where free embryos were found (Figure 14.7). A boulder in the middle of the site provided downstream hydraulic cover to the area where the eggs were found (Chapter 13). Water depths at bed B were in the range 0.3-0.6 m and excluding the areas in hydraulic cover, mean column velocities were 0.6-1.2 m s^{-1}. Froude numbers were in the range 0.4-0.8.

Thus, it seems possible that the hydraulic characteristics at bed B at high irrigation releases are more similar to those pertaining at bed A during the 1994 spawning than to those at bed B when discharges are lower. This suggests that, despite its small size, the smaller channel might be an important spawning area for much of the summer when releases are driven by irrigation demands. At these higher discharges, hydraulic conditions at bed A might be less suitable for spawning.

14.5 DISCUSSION

14.5.1 Upstream migrations

There seems to be a hierarchy of environmental factors that stimulate fish migration in rivers (Northcote 1984). Jonsson (1991) stated that in some cases initiation of migrations appears to be associated with temperature or flow or, for some species, with an increase in flow within a certain temperature range. Northcote (1984) felt that increasing and decreasing photoperiod was the predictive, proximate factor in temperate areas, enabling the fish to sense spawning and non-spawning seasons. In most parts of the world, there are obvious links between photoperiod, temperature and river flow, with the natural variability in the two latter factors at least partly accounting for the annual variation in spawning migrations (Jonsson 1991). In the Olifants River, *B. capensis* gonad mass begins to increase as early as August (Van Rensburg 1966), whilst water temperatures are still low and flow is high. Photoperiod may well be implicated in these early signs of reproductive activity but, as pointed out by Tyus (1990), there is still much to learn about the causal mechanisms of fish migration. As with a North American cyprinid, the Colorado squawfish *Ptychocheilius lucius* (Tyus 1990), the influence of environmental cues in initiating spawning migrations of *B. capensis* is not yet clear.

Whatever triggers migration, when it does occur in the Olifants River in spring, the population of *B. capensis* inhabiting the stretch of river between Clanwilliam Dam and Bulshoek Weir appears to show some co-ordinated movement upstream toward the dam wall in an attempted migration run to headwater spawning grounds. The dam prevents accessibility to these spawning grounds, however, and potential spawning grounds are further limited by the virtual non-existence of cobble beds in the middle reaches of the river. Nevertheless, some spawning does occur at the two areas downstream of the dam wall identified in this study. Understanding of the complex relations between environmental and biological variables linked to such activities (Tyus and Karp 1989) is slowly increasing.

14.5.2 Water temperature and flow as triggers for spawning

At the spawning grounds, water temperature is clearly one of the major factors influencing fish spawning (Wootton 1990). Most authors report temperature in terms of the general condition of the river at the spawning season. Baras *et al.* (1996), for instance, noted that in the River Ourthe in Belgium, the common barbel *Barbus barbus* started spawning when the daily minimum water temperature was 13.5 °C but suspended breeding whenever temperatures fell below this threshold. It is less common for temperature to be reported in relation to the effects of dams on spawning. Such effects are important, because for some distance downstream of dams, normal river temperatures may no longer pertain. Indeed, water temperatures may be providing environmental cues at variance with other possible ones such as photoperiod.

In this study in 1994, dam releases of freshes in spring did correlate with successful spawning. It seemed probable that *B. capensis* had been brought into spawning condition by the general spring increase in river temperatures, possibly linked to factors such as photoperiod. Once in condition, spawning then appeared to be triggered by a small increase in discharge.

The 1995 experiment, designed to repeat the previous year's success, did not result in any acts of spawning. The extension of winter conditions into spring, characterised by snow on the mountains of the headwaters and floods, resulted in heavy flows of cold water arriving in Clanwilliam reservoir later than usual. This was reflected in spillage from the dam wall, which continued until 15 November 1995, compared to 1994, when it stopped in August. When spillage stopped in November 1995, coincidentally just before the experimental period, relatively warm surface water pouring over the dam wall was replaced by hypolimnetic releases of deep, cold water. As a result, daytime water temperatures during the experimental period in 1995 (outside of the period linked to the top experimental releases) were 16.6-17.5 °C at both beds A and B, compared to 19.1-20.0 °C at bed A and 21.8-23.1 °C at bed B in 1994.

The fish had been in the spawning area in November 1995 prior to the switch to hypolimnetic releases, and divers completing fish counts immediately before the experimental releases reported that the fish downstream of the spawning area were in breeding condition. Male *B. capensis* exhibited nuptial golden coloration and the females had extended abdomens. Fish were scarce in the spawning area after hypolimnetic releases began, but arrived on bed A, once warm epilimnetic releases began. All arrived from a downstream direction, exhibiting what was interpreted as pre-spawning behaviour. The behaviour never culminated in spawning, and the fish left bed A again, after the first top release but before the first temperature reading was taken there (18.8 °C). They appeared on bed A again with the second top release, but left it again after the top release stopped and when the temperature was dropping 19.9-19.3 °C. In comparison, as noted above, the temperature at bed A during spawning in 1994 was 19.1-20.0 °C. Thus, a hypothesis for future testing is that water temperature on the spawn beds for successful spawning of *B. capensis* has to be at least 19 °C and stable or rising. It may be further hypothesised that if the fish are in breeding condition and these temperature conditions are met, then experimental freshes of the size described in these studies will trigger spawning.

Although no spawning was induced during the 1995 experiment, *B. capensis* free embryos of approximately two to three weeks of age (based on development at a constant 22 °C, see Chapter 13) were found on the spawn beds during the study. Conditions prior to the experiments, that is, in mid-November or earlier, thus

must have been suitable for spawning. In early November, water temperatures thought to be suitable for spawning (22-23 °C) were recorded downstream of the dam (Figure 14.3), with temperatures of about 20 °C recorded since early October (Figure 14.2). If these temperatures had brought the fish into breeding condition, then the floods in late October (Figure 14.2) would have been the most likely trigger for spawning. After the fish had spawned in late October or early November, the embryos would then have experienced temperatures below 18 °C after 15 November (Figure 14.3), which probably considerably slowed their development. It is thus possible that the free embryos collected during the experiment were over four weeks old. That some of the collected embryos were dead or deformed may have been due to these temperature fluctuations.

14.5.3 Hydraulic conditions related to spawning

Conditions other than those discussed above are important for successful spawning. Variables such as availability of spawning grounds and the use of such features as flooded vegetation or clean cobbles are mentioned in the literature (Wootton 1990). Hontela and Stacey (1990) give water level, water quality, nutrients, temperature, breeding substrate and vegetation as the factors which may determine spawning success of cyprinids. All of these factors may fluctuate on a seasonal basis. More recently, authors have reported in detail on the substrata and local hydraulics of spawning grounds. The main features usually mentioned are water depth and velocity, area, and particle size of the substratum. For instance, Baras *et al.* (1996) described spawning pits for the common barbel *Barbus barbus* that had a gravel substratum (20-50 mm), water depth of 0.10-0.26 m and water velocity of 0.26-0.67 m s^{-1}. The Colorado squawfish spawned at 22 °C over a rubble substratum at a water depth of 0.9-2.1 m and a velocity of 0-0.1 m s^{-1} (Tyus and McAda 1984). These are lower velocities and deeper water than recorded here for the Clanwilliam yellowfish (0.3-0.6 m; 0.6-1.3 m s^{-1} Figs 14.5-14.8).

Moir and Soulsby (1996) included Froude number in their report, describing spawning areas, or redds, for Atlantic salmon *Salmo salar,* with the following mean characteristics: utilised depth of 0.25 m, velocity of 0.54 m s^{-1}, mean Froude number of 0.34 (Froude numbers for *B. capensis*: 0.4-0.8) and substratum particle size of close to 15 mm (*B. capensis*: substrata > 256 mm). They, and Wadeson (1995), pointed out that the Froude number is useful because it is dimensionless and can therefore be applied to a range of sizes of river and fish. Wadeson (1995) also found that Froude number was a useful index distinguishing hydraulic biotopes such as riffle, rapids and cascades on the one hand, and pools or runs. The Froude numbers for *B. capensis* spawning grounds are in the range that Wadeson identified as "riffles". The use of coarse substrata and high velocities by *B. capensis* for spawning reflects its powerful build and evolution in what is essentially a large and powerfully flowing river.

14.5.4 Reproductive style

Many species with extended spawning seasons are multiple spawners with individual females producing several clutches of eggs, as does *B. capensis*. In many oviparous teleosts, multiple clutches of eggs are spawned periodically in a reproductive season and ova undergo group synchronous development (Wallace and Selman 1981; Heins and Robito 1986). The ovaries of these species cycle through maturing, mature and ripe stages as noted for *Notropis leedsi* (Heins and Robito 1986) and suggested for *B. anoplus* (Cambray 1983) and *Pseudobarbus asper* (Cambray 1991). This implies for species such as *B. capensis*, which have

the ability to spawn more than once per season, that it is important to provide spawning and recruitment flows more than once per spawning season in order to ensure survival of a year group.

14.5.5 Conservation considerations

Cueing to the natural hydrographic and thermal regimes in riverine environments may congregate fish at spawning sites and increase spawning success. Modde and Irving (in press) suggested, for the razorback sucker in North America, that increases in discharge initiate movement and aggregation of reproducing adults, and thus maximize the reproductive success of small genetically isolated populations. Tyus and Karp (1990) suggested several other factors for consideration in the recovery plans for this species. These included habitat alteration and loss in the form of curtailment of spring flooding, loss of historic flow and temperature regimens, and reduced flooding of bottomlands. A further important factor requiring active management was predation by non-native fishes. Tyus (1990) pointed out that migration and other reproductive behaviour traits of the Colorado squawfish are attuned to a variety of stimuli, including intrinsic mechanisms and dependable environmental clues. He suggested that a better understanding of these mechanisms and cues could assist management and recovery efforts for this endangered species.

Most of these factors are relevant to the continued presence of *B. capensis* in the middle reaches of the Olifants River. Suitable flows and temperatures for spawning have been discussed, and the need to synchronise aggregation in the spawning areas noted. Additionally, bed A appears to be very sensitive to changes in dam releases. The water level there, drops rapidly after the release is reduced (Figure 14.4). If the release were halted, this area, being a high point in the channel, would be particularly vulnerable to drying out. Until this vulnerability has been investigated, it is recommended that special caution be taken to avoid total shut down of releases during the summer breeding season. A further factor that may be affecting successful spawning and recruitment is the small size of the spawning area described in this study. It appears to be the only such area available to the *B. capensis* population between the two dams, and when spawning was witnessed there, most eggs may well have been swept from the cobbles to downstream pools by the strong current. The matter is discussed in Chapter 13.

A question that remains unanswered is whether or not the two to three month delay in the onset of summer temperatures downstream of the dam allows sufficient time for the species to complete early life stages before winter.

14.6 CONCLUSIONS

Dry-season freshes are recommended in most IFR studies using the BBM (Chapter 3), mostly to promote fish migration and spawning. Providing suitable hydraulic conditions at the appropriate time, however, will not trigger spawning if the water temperatures are not suitable. Temperatures of released water can be manipulated from dams with multiple release structures. This strategy may have limited success, however, unless the downstream temperatures are then maintained at levels that are suitable for the developing young.

The spawning activities reported on in this chapter and Chapter 13 reveal that modest increases in flow appear to play a role in successful spawning. Discharges between three and eight times the baseflow occurring immediately beforehand were linked (via the descending limbs of their hydrographs) with

successful spawning (Chapter 13) or "preparation to spawn" (Chapter 14). Most of the experimental freshes were barely larger than the irrigation releases already occurring outside the experimental period, and appeared as increased discharges merely because releases had been reduced earlier to store water for the experiments. At this stage it is not known if such irrigation releases, occurring as a constant discharge, would adequately provide cues for spawning, or if some fluctuation in discharge is necessary. Some spawning does occur during the weeks of irrigation releases, but the dam records do not reveal if short-term fluctuations in releases occur. It is also not known how much smaller the freshes could be and still result in spawning. Finally, it is not known if the important feature of the freshes is simply their magnitude *relative* to the previous baseflows (and thus the change in hydraulic conditions), or if they provide some other cue for spawning, such as inundation of dry areas.

15. CRITICAL HYDRAULIC CONDITIONS FOR THE MOBILISATION OF SEDIMENT IN A CAPE COBBLE-BED RIVER

Written by R. Simpson (then at the University of Stellenbosch)

15.1 INTRODUCTION

15.2 OBJECTIVE OF THIS RESEARCH

15.3 APPLICATION OF A DISCHARGE-SCOUR RELATIONSHIP

15.4 A MATHEMATICAL MODEL DESCRIBING THE SCOUR OF SEDIMENT IN A RIVER

15.5 A SUMMARY OF THE CONSERVATION OF POWER LAW AND THE POWER APPROACH

15.6 DERIVATION OF THE MATHEMATICAL MODEL

15.7 THE RIVER SELECTED TO DETERMINE THE DISCHARGE-SCOUR RELATIONSHIP

15.8 FIELD TRIPS TO THE MOLENAARS RIVER

15.9 MEASUREMENTS TAKEN ON THE FIELD TRIPS

15.10 IDENTIFICATION OF THE SIGNIFICANT DISCHARGE

15.11 IDENTIFICATION OF POSSIBLE SIGNIFICANT DISCHARGE SCENARIOS

15.12 IMPLICATIONS OF THE SIGNIFICANT DISCHARGES

15.13 THE MOLENAARS RIVER SEDIMENT SCOUR MODEL

15.14 STEPS IN THE APPLICATION OF THE MOLENAARS RIVER SEDIMENT SCOUR MODEL TO OTHER RIVERS

15.15 COMMENTS ON THE DEVELOPMENT OF THE MODEL

15.16 CONCLUSIONS

This Chapter summarises a one-year M. Eng. project done by Mr Richard Simpson (Simpson 1997). The work was funded by this project and registered at the Department of Civil Engineering, University of Stellenbosch. The supervisors were Prof. Rooseboom of the University of Stellenbosch and Dr King (UCT).

15.1 INTRODUCTION

During the normal annual cycle of river flow in a Cape river, it has been observed that sediment tends to accumulate under and around the cobbles on the river bed during the low summer discharges, and to be removed from these areas during the higher winter discharges. This is known as scouring and is an important process in the river ecosystem, as many organisms are specific to the physical habitats under and

around the cobbles (Hynes 1970). These organisms form the base of the food chain and in the event of their physical habitats being obliterated by continued settling of fine sediments, they will disappear from the affected reaches. This can result in a dramatic drop in abundance and diversity of animals and plants living on the bed of the stream. A failure in the part of the food chain that they represent thus may have negative consequences for other parts leading, *inter alia*, to a reduction in the capacity of the river to "assimilate" effluents and cleanse poor-quality water flowing in it. It is thus desirable that accumulation of sediments on the bed of a river should be contained to natural levels through regular scouring, in order to maintain the health of the river.

The BBM as discussed in Chapters 3-6, was designed to allow development of a description of a flow regime that should occur in a river in order to maintain the ecosystem at some pre-determined level of health. Such a regime is known as the IFR for the river. The IFR changes along a river's length as the ecosystem character changes. Scouring flows should, in accordance with the philosophy behind the BBM, be incorporated into the IFR of each characteristic river section.

In order to be able to include a scouring flow in an IFR, it is important *inter alia* to first of all be able to determine what size sediment is required to be scoured from the bed of a river, and then based on this requirement, what discharge is required for its scouring. Sampling and sieving accumulated bed sediments will quickly indicate the size of the sediment to be scoured. Research was therefore required to determine the discharge that would cause scouring of a sediment of a given composition of particle sizes in a given section of a river. As the scouring and the flow requirement for scouring varies considerably depending on the characteristics of the bed of the river, and the time available for this research was somewhat limited, the research focused only on scouring in the upper reaches of a Cape river which is characterised by a cobble bed.

A general understanding existed that the higher the high flows in the winter, the greater the extent to which sediment was scoured in the river. Preliminary observations indicated that after a high flow, the sediment on the bed of the river would be reduced in volume and increased in average particle size. The larger the high flow experienced, the smaller the quantity of sediment left on the bed of the river, and the coarser it was. This suggested that a relationship exists linking the discharge in the river to the extent of scour that takes place in the river.

At the beginning of the project, it was hypothesised that a flow for scouring could be identified by determining the size of the sediment scoured. Using this information, a relationship could be developed that linked the discharge in a river to the extent to which scour occurred. This would allow determination of the required discharge for removing fine sediments, thus achieving the objective of the research.

15.2 OBJECTIVE OF THIS RESEARCH

The objective of this research was, therefore, to establish a relationship linking the extent to which scour takes place within the river to discharges that have occurred in the river. The degree of scour is measured by the average diameter (also known as the d_{50}) of the sediment left on the river bed after a given discharge has taken place.

The results can be used to guide on the required magnitude of a discharge for moving sediments of a given particle size. Elements of floods that are of direct ecological significance, such as their timing or duration, are not dealt with here.

15.3 APPLICATION OF A DISCHARGE-SCOUR RELATIONSHIP

In a natural river environment, the annual cycle of deposition and scour is maintained. Dammed rivers, however, tend to have smaller and fewer scouring flows downstream of the dam wall, because many floods that would naturally flow down the river are reduced considerably in size or are harnessed completely by the dam (McCully 1996). Thus, natural scouring will not take place. If a relationship existed linking the extent to which scour would take place for any given discharge, the required scouring could be determined and the appropriate discharge could be released from the dam to perform the scouring. This would facilitate maintenance of a healthy ecosystem in the downstream river.

Relationships describing sediment scour at cross-sections in a river could be obtained directly from data surveyed at those cross sections. However, if this relationship is to be valid for other areas along the surveyed river, and areas along other rivers with similar river bed characteristics, it needs to be shown to conform to some kind of mathematical model.

15.4 A MATHEMATICAL MODEL DESCRIBING THE SCOUR OF SEDIMENT IN A RIVER

Although very little work has been done on sediment scour in cobble-bed rivers, extensive work has been done on sediment scour in sand-bed rivers. A mathematical model describing scour on the bed of a duned sand-bed river has been developed (Rooseboom 1992). It was expected that a model could be developed from Rooseboom's model, to describe scour of sediment from under and around cobbles on the bed of a cobble-bed river. This is because the general shapes of sand-bed and cobble-bed rivers are similar, as shown in Figure 15.1.

The model is based on the conservation of power law and the power approach. These concepts are treated comprehensively in Rooseboom (1992) and are described briefly below.

15.5 A SUMMARY OF THE CONSERVATION OF POWER LAW AND THE POWER APPROACH

As water flows from a high point to a lower point, it loses potential energy. The conservation of power law states that this energy is used in the water flow at the same rate at which it is released. In an ideal uniform flow, all this energy is used to overcome the resistance to flow provided by the environment through which the water is flowing. The energy released per second by the water is known as the available power. The energy required per second to overcome the resistance is known as the applied power. The applied power is therefore equal to the available power. In the real world, the applied power is always less than the available

power as some energy is always lost in other ways, such as in the form of heat through internal friction, besides overcoming resistance to movement.

Sediment is scoured and carried downstream when less power is required to lift it off the river bed and into the flow, than is required to flow over and around non-moving sediment. In a stream experiencing low flow there may be no movement of sediment along the river bed. As discharge increases, a point will be reached where sediment will begin to be scoured. At this point, the conditions which exist in the river are called **critical conditions**. The model links these critical conditions occurring in the river to scour taking place, through comparing the power applied at the bed of the river by the water moving downstream, to the power required to lift the sediment from the bed of the river. The conditions in turn can be related to the average velocity of the water above the area of scour, using the Chézy equation. The discharge down the river is proportional to the average velocity at a cross-section. In this way a mathematical model can link the sediment scour in the river to the discharge in the river. The derivation of the model and the concepts behind the derivation are described briefly in the next section.

15.6 DERIVATION OF THE MATHEMATICAL MODEL

The conditions that occur in a uniform section of a river at a cross section can be described by three variables as shown in Figure 15.2:

- The average depth of flow, D;
- The energy slope, s;
- The average roughness of the bed of the river, k.

The Chézy equation (Chow 1964) states:

$$\bar{v} = 18\log(12R/k) * \sqrt{Rs} \qquad \textbf{Equation 15.1}$$

Where:

$\bar{v}$ = the average velocity of the water flowing through the river cross-section

R = the hydraulic radius of the section in the river and is equal to A/P which is the cross sectional area divided by the wetted perimeter ($R=D$ for a 1 m wide section of the river bed)

k = the average roughness of the river bed, which is specifically defined in this case as the difference in the average height of the tops of the cobbles and the average heights of the surfaces of the pockets of sediment between the cobbles

s = the energy slope of the river, which is equal to the surface and the bed slope in the case of uniform flow

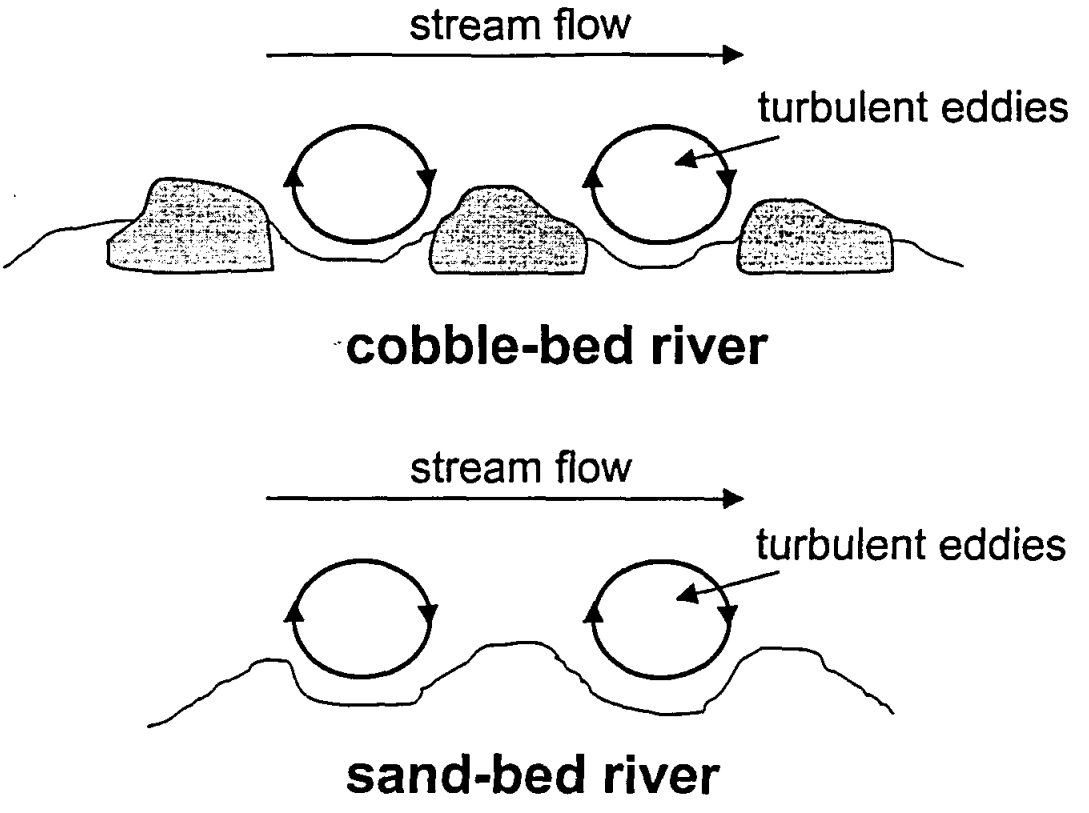


Figure 15.1 **Diagrammatic representation of the similarity between cobble-bed and duned sand-bed rivers.**

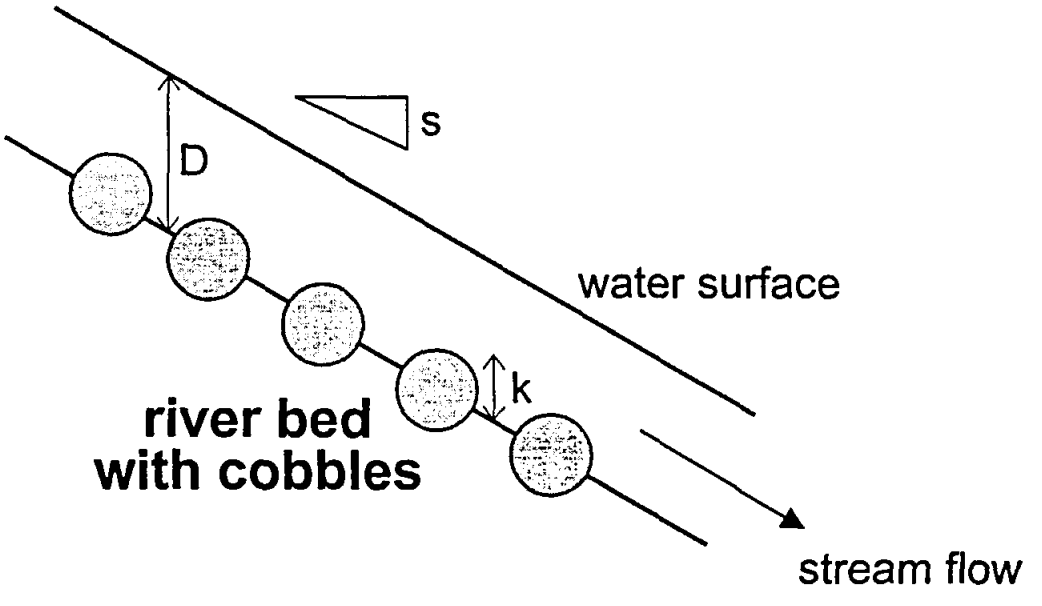


Figure 15.2 **Schematic view of a long section of a river showing the variables used to describe the conditions in the river.**

For turbulent flow, which is the dominant flow in rivers in all but the smallest of discharges, a second equation (Rooseboom 1992) describes the power applied at the bed of the river per unit volume of water as being proportional to:

$$\rho g D s \frac{\sqrt{gDs}}{d}$$ **Equation 15.2**

With all the variables as described above or as for Equation.15.3.. A further equation (Ruby 1933) describes the power required to lift sediment from the bed of the river as being equal to:

$$\sqrt{\frac{4(\rho_\sigma - \rho)gd}{3\rho c_d}}(\rho_\sigma - \rho)g$$ **Equation 15.3**

Where:

ρ_σ = density of the sediment particles

ρ = density of water

g = gravitational acceleration

c_d = drag coefficient of the sediment particle

d = diameter of the sediment particle

The settling velocity of the sediment particles which is included in Equation 15.3 is equal to:

$$V_{ss} = \sqrt{\frac{4(\rho_\sigma - \rho)gd}{3\rho c_d}}$$ **Equation 15.4**

It is expected that when the power applied at the bed of the river multiplied by some efficiency factor is equal to the power required to lift the sediment from the bed of the river, scouring will begin to take place. Therefore Equation 15.2 will be equal to some constant multiplied by Equation 15.3.

Manipulating the resulting equation shows (Yang 1973) that:

$$\frac{\sqrt{gDs}}{V_{ss}} = \text{constant 1}$$ **Equation 15.5**

Since $\sqrt{gDs}$ is proportional to the average velocity, $\bar{v}$, above the section, if C in the Chézy equation is considered constant, and V_{ss} is proportional to the square root of the sediment diameter, as shown in Equation 15.4, the following equation is derived:

$$\frac{\bar{v}}{\sqrt{d}} = \text{constant 2} \qquad \textbf{Equation 15.6}$$

It should therefore be possible to show a linear relationship between the square root of the average sediment diameter on the bed of the river and the average velocity of the flow above that section of the bed.

As the development of this relationship needs to be done in both the field and the laboratory and there was limited time available, it was decided to leave laboratory work for later research and concentrate on the field work. A river in which to perform the fieldwork had to be selected.

15.7 THE RIVER SELECTED TO DETERMINE THE DISCHARGE-SCOUR RELATIONSHIP

The river selected to calibrate this relationship was the Molenaars River in the Du Toit's Kloof. This river was selected because it displays classic cobble-bed characteristics, was suffering from sediment build-up due to erosion occurring in the river valley, and was easily accessible to the survey team who resided in Stellenbosch. It also has a continuous discharge-monitoring weir (H1H018) operated by DWAF, approximately 3 km downstream of the selected study area, and so average daily discharge data were available for the study.

15.8 FIELD TRIPS TO THE MOLENAARS RIVER

During the period May to October 1995, field trips were carried out approximately every 10 days, with 17 trips completed in total. Typically, on each field trip, measurements were taken at three of six transects at the study site, giving a total of around 50 sets of transect data.

15.9 MEASUREMENTS TAKEN ON THE FIELD TRIPS

Six reaches were chosen in areas of different sloping, long sections within the study site. A typical transect was surveyed in each reach. An example is shown in Figure 15.3. From these transects, stage-wetted area, wetted perimeter, and hydraulic radius relationships could be obtained as shown in Figure 15.4. As the transects were chosen in areas of near uniform flow, the energy slope of the river could be considered to be equal to the water surface slope of the river. The water surface slope of the river, the water depth, and the weir discharge were measured on each transect survey on the survey trips. Bed roughness was also measured, by surveying in the tops of the cobbles and the surface level of the sand between the cobbles. Thirty to 50 measurements of bed roughness were taken per cross-section survey. These data, together with

the Chézy equation, were used to develop stage-discharge graphs for each transect as shown in Figure 15.5. The stage-discharge graph was a combination of the measured data at lower flows, and the theoretical data of higher flows.

The roughness was measured as a check for the roughness used in the stage-discharge graphs. It was found to vary, but as it is included in a logarithmic term and the variation is small, for the purposes of the stage-discharge graphs it was assumed constant.

At each measurement, sediment samples were taken. A plastic scoop was used to collect the sediment from the spaces between the cobbles on the bed of the river. After collecting the sediment, it was covered and brought to the surface of the water. Samples were bagged and transported back to the laboratory for air-drying and measurement. Average diameters of the sediment samples were determined by laboratory sieve analysis, thus supplying the one set of values for the Molenaars River Sediment Scour Model. Sieve sizes used ranged from 0.6-26.5 mm. Sediment larger than 26.5 mm was not considered for the purposes of this research as it does not constitute a threat to organism habitats on the bed of the river due to the relatively large habitable spaces which exist between settled particles.

In order to calibrate the model, the discharges that caused each measured average sediment diameter to be as measures needed to be identified. These discharges are referred to as the **significant discharges**.

Continuous hydrographs from the Molenaars River weir were available as shown in Figure 15.6, allowing the average daily discharge at any point in time to be determined.

15.10 IDENTIFICATION OF THE SIGNIFICANT DISCHARGE

It was initially theorised that the sediment on the bed of the river was always at critical conditions. In explanation, as discharge increased, scouring of the sediment would begin instantly, but would cease instantly if discharge stabilised. Sediment would be deposited as the discharge in the river decreased and once again if the discharge stabilised, sediment deposition would cease. The defining feature about this theory is that it is assumed that changes in sediment character and volume only occur when the discharge is changing, and the moment the discharge stabilises, any change in sediment on the bed of the river ceases because all the sediment that could have been removed in this given discharge was removed instantaneously as the discharge first occurred. In the real world the definition of 'instantaneous' changes in this case to meaning within an hour or two.

Using the discharge data, sample data on sediment diameter and the stage-discharge graphs for each cross-section sampled (as shown in the thesis appendices, Simpson 1997), it was possible to link a flow velocity at the time of sampling with the average sediment diameter. Analyses of these data showed a poor correlation between sediment size and discharge at the time of sampling. Thus, this theory appeared to be an oversimplification. It was therefore assumed that critical conditions did not exist on a continuous basis and some other explanation had to be sought.

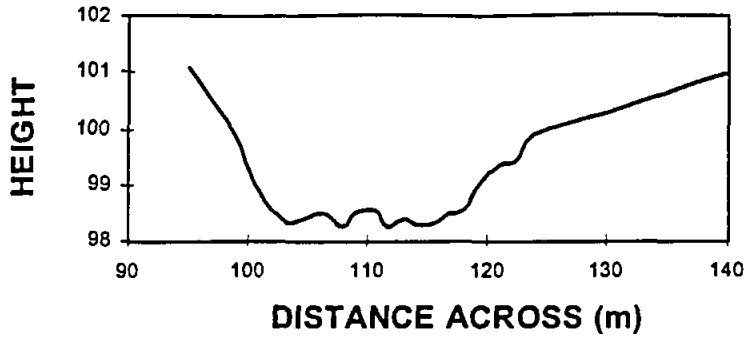


Figure 15.3 The profile of transect 4, as surveyed in May 1995.

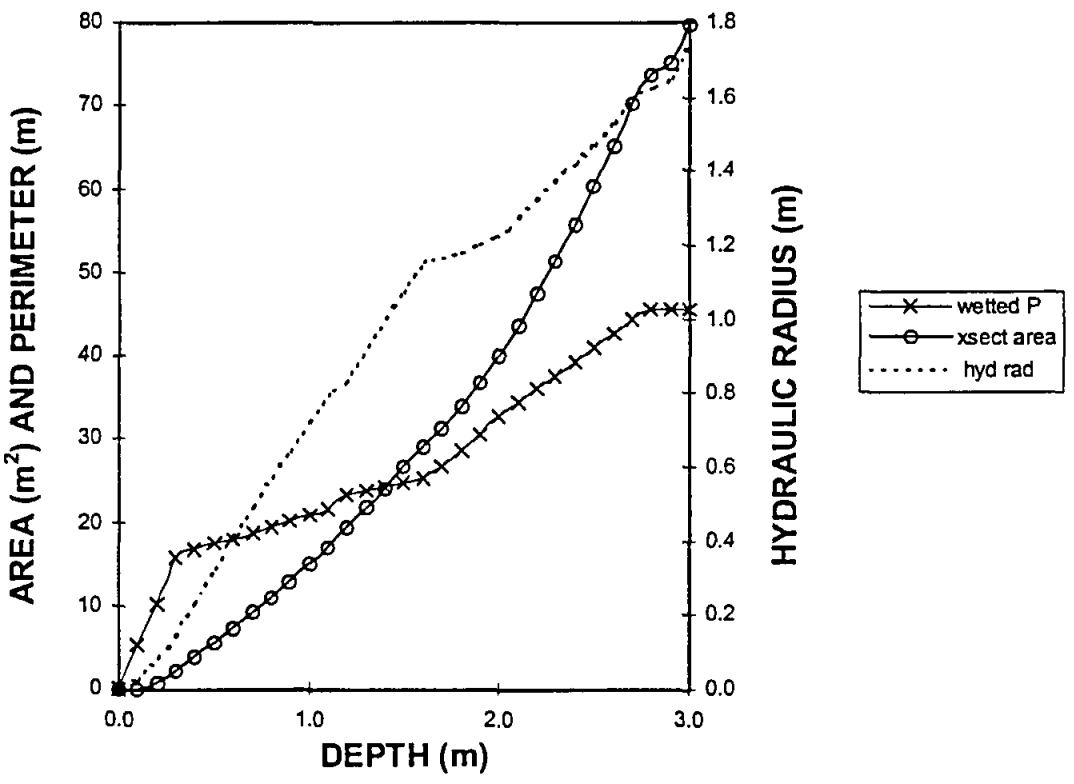


Figure 15.4 The relationship of wetted perimeter (wetted P), wetted area (xsect area), and hydraulic radius (hyd rad) to depth, for transect 4.

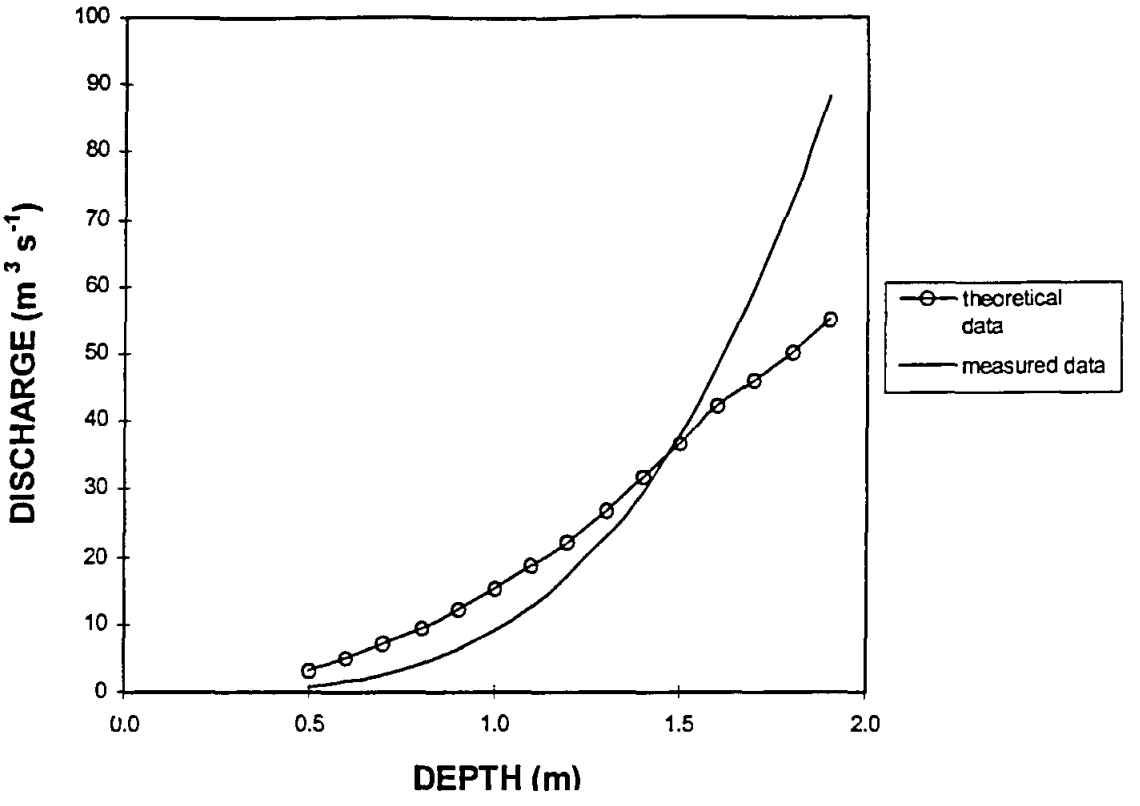


Figure 15.5 Stage-discharge graph for transect 4 using measured data, and theoretical data predicted using the Chézy equation and the transect.

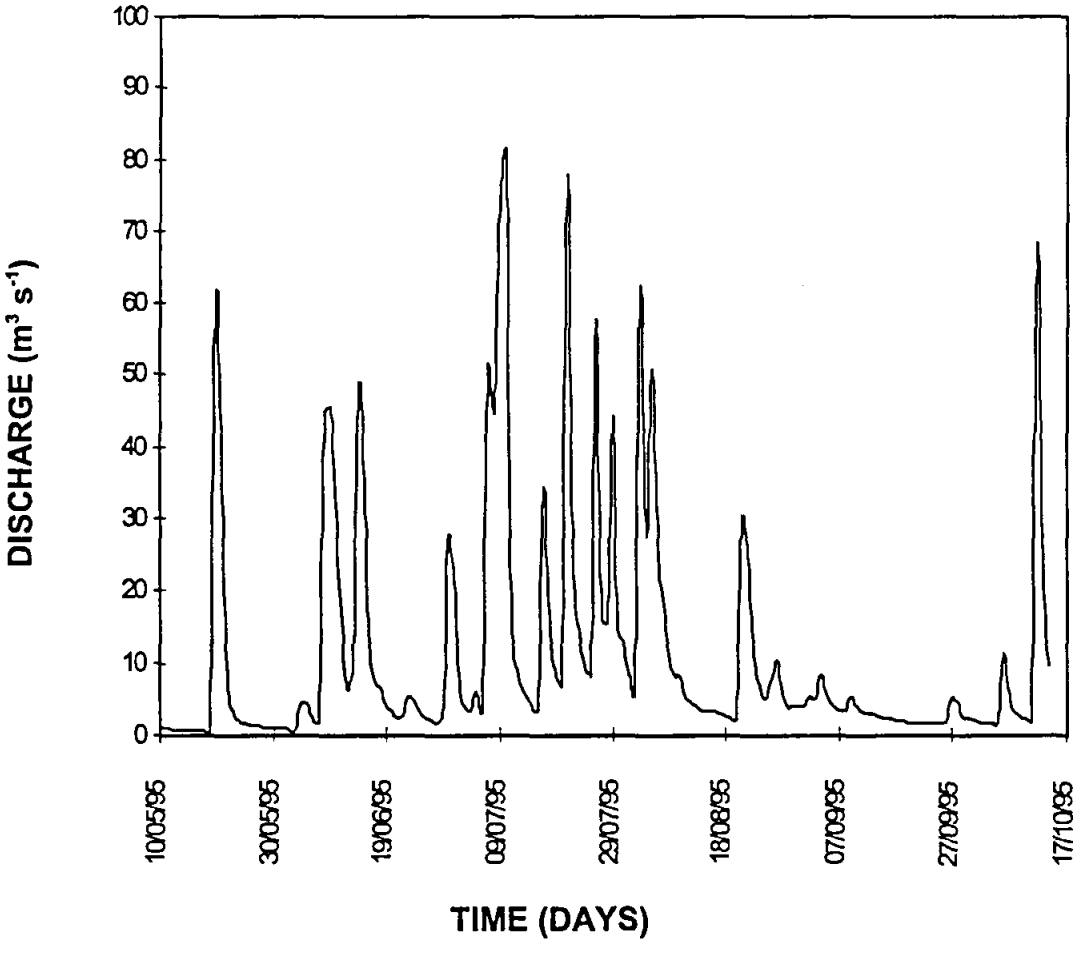


Figure 15.6 Hydrograph of weir H1H018 on the Molenaars River over the period 10/05/95-15/10/95.

It was then hypothesised that on the rising limb of a flood, the amount of power applied at the bed would initially be insufficient to lift sediment from the river bed, but with increased discharge, critical conditions would be reached , and scouring would begin. As the discharge continued to increase, equilibrium scour would occur almost instantaneously until the flood peaked. Here, for a brief moment, the sediment would not be scoured, but would be in equilibrium with the power applied at the bed of the river. On the decreasing limb of the flood, the deposition of sediment would not be fast enough to maintain critical conditions. The slow deposition rate would mean that re-establishment of critical conditions could take days, weeks, or even months. As the time for critical condition re-establishment was unknown, the average sediment diameter had to be compared to the largest power applied on the river bed in time periods ranging from a day to several weeks before the time the sediment sample was taken. Because there is not an instant re-establishment of critical conditions it is also possible that the sediment diameters were the result of the average power applied over a time period before the sample was taken. It was therefore necessary to test relationships linking the square route of the sediment sample diameter with the maximum $\bar{v}$, and the average $\bar{v}$ over different time periods before sampling as per Equation 15.6.

Table 15.1 The relationships tested between the square roots of the sediment diameters and the average velocities above the test section. The resulting functions are listed along with the R^2 value indicating goodness of fit.

AVERAGE SEDIMENT DIAMETER, $\sqrt{d}$ VS:	AVERAGE VELOCITY, $\bar{v}$	REGRESSION FN.	R^2
	ave. 1 day before	$\bar{v} = 0.919\sqrt{d}$	0.61
	ave. 2 days before	$\bar{v} = 0.938\sqrt{d}$	0.61
	ave. 5 days before	$\bar{v} = 0.966\sqrt{d}$	0.59
	ave. 10 days before	$\bar{v} = 0.976\sqrt{d}$	0.55
	ave. 15 days before	$\bar{v} = 0.970\sqrt{d}$	0.54
	ave. 20 days before	$\bar{v} = 0.980\sqrt{d}$	0.55
	max. on day of collection	$\bar{v} = 0.957\sqrt{d}$	0.62
	max. in 2 days before	$\bar{v} = 1.026\sqrt{d}$	0.62
	max. in 5 days before	$\bar{v} = 1.211\sqrt{d}$	0.59
	max. in 10 days before	$\bar{v} = 1.347\sqrt{d}$	0.58
	max. in 20 days before	$\bar{v} = 1.420\sqrt{d}$	0.62
	max. in 30 days before	$\bar{v} = 1.495\sqrt{d}$	0.62

Regression analyses were used to determine the best-fit model function for each relationship listed in Table 15.1. The analyses also provided a measure of how well the function fits the data for each relationship in the form of an R^2 value. (R^2 = -1 indicates a perfect negative correlation, R^2 = 1 indicates a perfect positive correlation, and R^2 = 0 indicates no correlation). The data and the resulting regression

functions are also shown in Table 15.1. Figure 15.7 shows average $\bar{v}$ over a time period of three days before the sample was collected versus the square root of the average sediment diameter of the sample.

15.11 IDENTIFICATION OF POSSIBLE SIGNIFICANT DISCHARGE SCENARIOS

The regression analyses do not show any single relationship as having a variance with a significantly better fit than any other. The analyses do, however, indicate that the relationships using the maximum discharges consistently had a marginally better fit than those involving average discharge. Although the differences are small, they are based on all the information that is available at this time. It therefore must be concluded that it is the maximum $\bar{v}$ in time periods before sampling which is the dominant discharge event in determining the average sediment particle size of the samples collected.

The regression analyses functions for the maximum $\bar{v}$ shown in Table 15.1 are the calibrated equations that comprise the Molenaars River Sediment Scour Model. The model is illustrated in Figure 15.8. The implications of the results are discussed in the next section.

15.12 IMPLICATIONS OF THE SIGNIFICANT DISCHARGES

Sediment on the bed of a river is logically lifted more easily from the bed of the river when it lies between small cobbles than when it lies between larger cobbles. The lower limit of the model is therefore a sandy river bed where the 'cobbles' are the same size as the sediment particles. This situation is described by the following calibration of the model function which is Equation 15.6 (Rooseboom 1992):

$$\bar{v} = 0.220\sqrt{d} \qquad \textbf{Equation 15.7}$$

The value of d is given in mm, $\bar{v}$ is measured in m s^{-1}, and the calibration constant (0.220) in $\sqrt{m}/s$. The assumption is also made that the value for the Chézy constant is indeed constant for varying flows. This lower limit is labelled as 'uniform sand bed equation' in Figure 15.8. Based on this figure, it is possible to describe a chain of events that led to the average sediment diameter to be as it was measured on any one sampling occasion.

Initially the river would have a low discharge, possibly running at baseflow, and may be at critical conditions. In the event that it is not at critical conditions (Section 15.5), deposition will be taking place. In the event that it is at critical conditions, and the discharge is constant, the character and volume of the sediment on the river bed will not be changing. In the first instance, as discharge increases, critical conditions will be reached, and a scouring of fine sediment will begin. In the second instance, as discharge increases, scouring of fine sediment will begin immediately.

In both instances, as the discharge continues to increase, scouring of increasingly larger sediment will occur. At the peak of the discharge, equilibrium between deposition and scour will exist. The discharge will then

begin to decrease, critical conditions will cease, and deposition will begin. The rate of deposition is relatively slow in comparison to scour, so a sample taken one or two days after the high discharge event would be very close in average diameter to that of the sediment on the bed of the river at peak discharge.

A week or so later, some deposition of fine sediments would have taken place and the average diameter of the sediments on the bed of the river would have decreased. The decrease in average diameter of settled sediments would continue, and a sample taken two weeks after the high discharge event would contain, on average, smaller particles than one taken after one week. Similarly, a sample taken after four weeks would contain, on average, smaller particles than one taken after two weeks.

The rate of decrease over time of the average sediment diameter would be dependent on the amount of sediment available in the river and catchment. It can however be assumed that the sediment available over a winter is constant to some extent. This is because the vegetation cover will not change significantly over that time period and therefore the availability of the sediment will not change significantly either.

The converse of this hypothesis is that given an average diameter required on a given day at a cross-section, a certain maximum discharge would have been required to have scoured that cross-section on the previous day. The maximum discharge could have occurred, for instance, ten days before the day of sampling, but in that case the deposition that would take place between the time of the maximum discharge and the time of sampling would have to be taken into account. In other words, to achieve the same end particle size, the maximum discharge ten days previously would have had to be bigger than the maximum discharge one day previously. Thus the graphs fit the hypothesis, with the required maximum discharge for a given sample size increasing with increasing time before the sample was taken.

15.13 THE MOLENAARS RIVER SEDIMENT SCOUR MODEL

Based on the previous sections, the Molenaars River Sediment Scour Model equation, Equation 15.6, is calibrated for a one-day maximum discharge as:

$$\bar{v}/\sqrt{d} = 0.9569$$

Equation 15.8

Where the diameter is measured in mm, and the average velocity in m s^{-1}. When the diameter is measured in m and the average velocity in m s^{-1}, the equation becomes:

$$\bar{v}/\sqrt{d} = 30.26$$

Equation 15.9

This equation links the extent to which sediment is scoured, shown by the average diameter of the sediment sampled on the bed of the Molenaars River, to the maximum velocity of the discharge over the area of scouring in the 24 hours before the sample was taken. Due to the fact that limited deposition occurs after the maximum discharge in 24 hours, the sample average diameter can be said to approximate closely that of

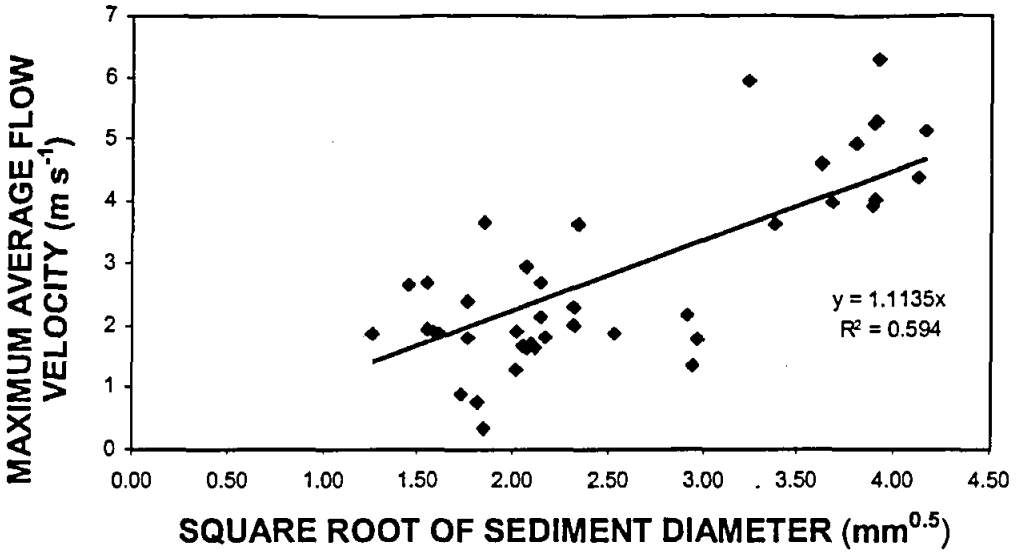


Figure 15.7 Graph comparing average sediment sample diameter with maximum flow velocity above sample area over 3 days before sample collection. Regression analysis results are shown on the graph.

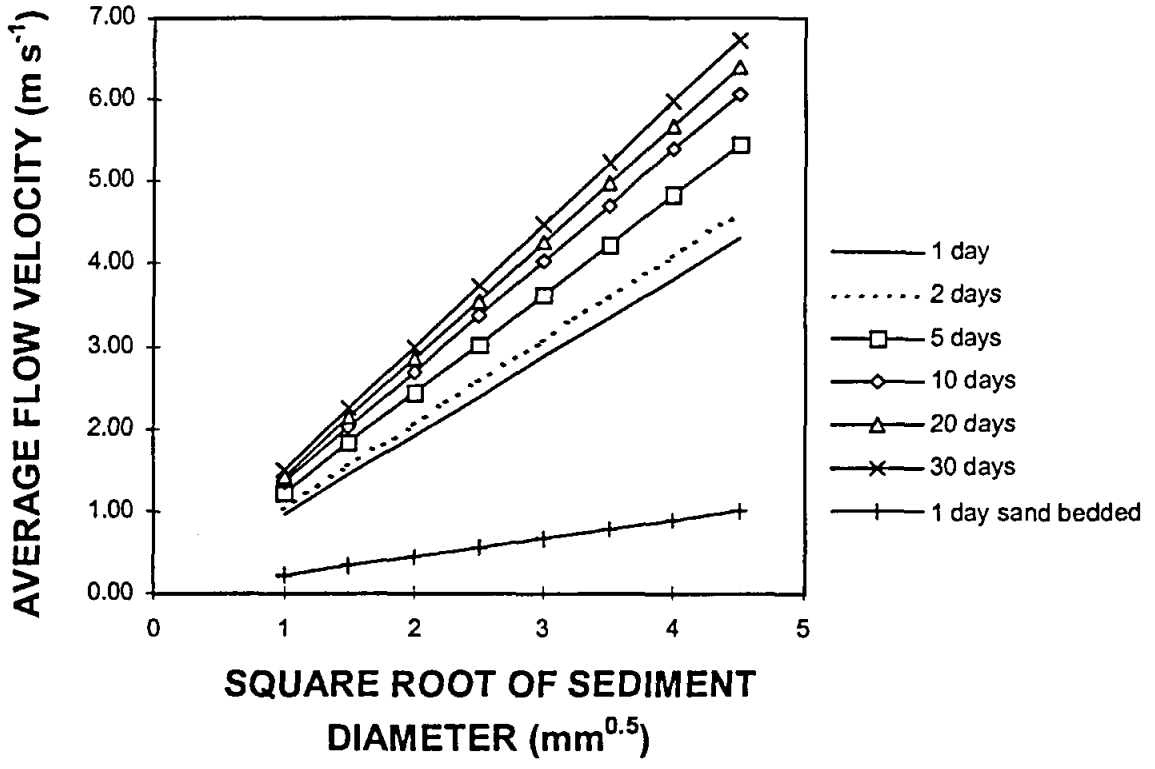


Figure 15.8 The Molenaars River Sediment Scour Model and the uniform sand bedded river equation. The graph indicates the flow velocities at critical conditions for given sediment diameters.

the sediment on the bed of the river at the time of the maximum discharge. Therefore, for each given sediment diameter, the one-day graph series gives a good conservative approximation of the velocity at which sediment will begin to be lifted into the stream flow and carried downstream.

The model achieves the objective of this research by establishing a link between the extent to which scour takes place within a river and the discharge which occurs in the river. The key equation in the Molenaars River Sediment Scour Model is described by Equation 15.9 and reveals that using the known constant of 30.26 and knowing the average diameter size of the particles to be moved, the velocity needed to move the sediments can be identified. It therefore defines the "Critical Hydraulic Conditions for Mobilisation of Sediments in a Cape river".

The value of the remaining series on the graphs is limited for application to the Molenaars River in subsequent winters, as they are very much dependent on the rate of deposition that was occurring in the river over the winter of 1995. Subsequent winters are unlikely to have the same deposition rate and therefore the series cannot be applied. However, the small amount of deposition occurring within the 24 hours after the maximum discharge event does allow Equation 15.9 to be applied to subsequent years and also to other rivers with similar bed characteristics to the Molenaars River.

15.14 STEPS IN THE APPLICATION OF THE MOLENAARS RIVER SEDIMENT SCOUR MODEL TO OTHER RIVERS

Within the limits described above, the model is also applicable to other cobble-bed rivers. The bed roughness values measured on the field trips ranged from 0.05 m to around 0.25 m. The model should therefore be capable of describing scouring in river beds with roughness values within this range.

In order to determine the discharge required to scour a similar river bed to a given extent, the following steps should be taken:

1) Identify the stretch of the river where the fine sediment is required to be scoured and choose a transect from within this stretch where the bed has a near uniform gradient. A river bed can be considered to have a near uniform gradient if the gradient over a 60m section is within 10% of the gradients across shorter distances within the same stretch. Shorter distances would be 10-20 m.

2) Survey the transect. From this, calculate the wetted perimeter, P, the wetted area, A, and the hydraulic radius, R, for any depth of flow, D, using a program or macro.

3) Measure the bed slope, s and the roughness, k. Using the Chézy equation (Equation 15.1), draw up a stage-discharge graph for the cross-section. Although the Chézy equation may be inaccurate at low discharges, the scouring of the river takes place predominantly at higher discharges, and this is the relevant part of the graph.

4) The sediment on the bed of the river should be sampled and analysed through the same range of sieves as described in Section 15.9, to produce the average sediment diameter, d.

5) The amount of sediment to be scoured from the bed of the river needs to be described in terms of the required average sediment diameter after scouring. This diameter is chosen using the sampled d as a guide. The chosen diameter is used to calculate the required average velocity, $\bar{v}$, using Equation 15.9. This value can be substituted into the Chézy equation (Equation 15.1) together with other measured variables, to obtain a depth of flow above the area sampled.

6) Using the level of the sampled area, the level of the lowest point on the cross-section, the depth obtained from step 5, a depth of flow from the lowest point on the cross section can be determined. This depth can be used to read off a discharge from the stage-discharge graph. This is the discharge required to produce the depth of flow above the sample area as determined in step 5, and thus the required average velocity for scouring.

7) In the event of a controlled release dam upstream, the required discharge can be released for a period of a day to scour the bed of the river. Releases of shorter duration may be adequate, but at this time a more precise release duration cannot be identified based on the research completed.

A flood frequency return graph will show how often any required discharge occurs under natural conditions, and will provide an indication of how soon a sediment deposit should be removed naturally.

Using the Molenaars River Sediment Scour Model, and the stage-discharge graphs, it is possible to construct a graph as shown in Figure 15.9. If sediment build-up occurs in the river, a sediment sample may be taken at any one of the cross-sections, the average diameter measured, the required average diameter determined and the minimum discharge required to obtain that diameter identified.

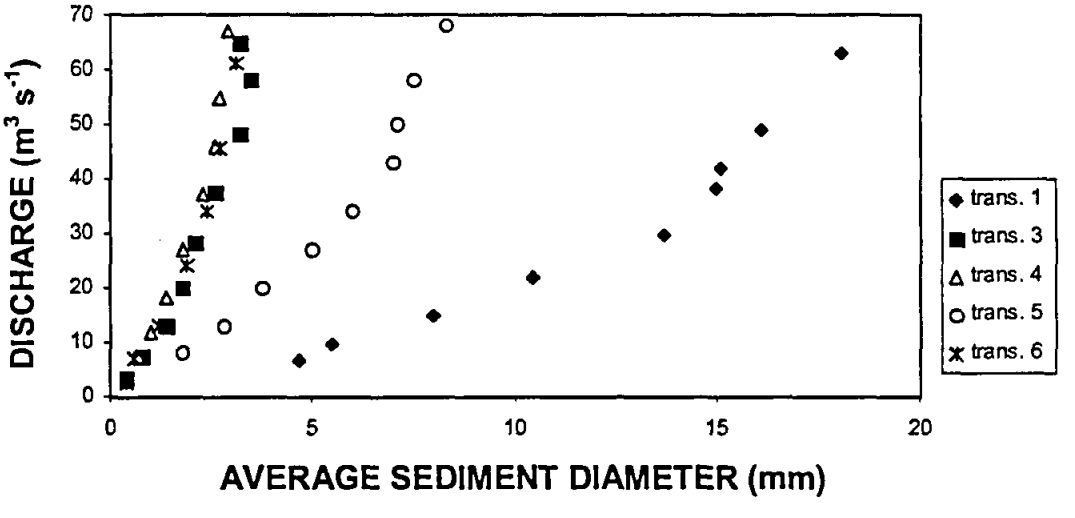


Figure 15.9 Critical discharges at transects 1, 3 to 6 in the Molenaars River.

15.15 COMMENTS ON THE DEVELOPMENT OF THE MODEL

The relationships listed in Table 15.1 show a significant variation in average sediment diameter with time, which indicates a rapid re-establishment of sediment on the bed of the river. This appears contrary to some observations made by on of the authors of this report (JMK), who felt that sediment levels after high flows take several days or even weeks to be re-established. A possible explanation is that when the observations were made by JMK in 1994, the banks of the river were fully vegetated, but during the winter of 1995 when the data for this research were collected, the banks were close to completely denuded of vegetation. This was as a result of a programme of clearing alien vegetation that took place in the river valley during the autumn of 1995. The cutting down and burning of the alien vegetation left large areas of soil completely exposed to erosion adjacent to the river. Heavy erosion had also taken place in the Posstroom, a tributary of the Molenaars River (pers. obs.). Sediment was therefore very much more readily available for transport into and down the river during the winter of 1995 than during the winter of 1994 and, therefore, the sediment could be re-established more quickly.

The measurement of fit, the variance, for the relationship which is finally defined as the Molenaars River Sediment Scour Model, is around 0.63. This value indicates that the data do not fit the equation resulting from the linear regression particularly well. This is as a result of using data collected in the field. Probably the most difficult variable to measure accurately was the slope of the river bed. Unfortunately, this was also the variable to which the results of calculations of discharges down the river were most sensitive, as it carries a power value of 3/2 in the Chézy equation. Despite this less than 'good' fit, the trend of increasing average sediment diameter with increasing flow velocity over the sampled areas is definite.

15.16 CONCLUSIONS

A method for determining a minimum discharge required to scour fine sediments from a cobble-bed river to a given extent is described, where the extent of scour is measured by the average diameter of the sediment in the river. "Critical Hydraulic Conditions for the Mobilisation of Sediment in a Cape river" are therefore defined.

16. CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

Written by J. King and R. Tharme

16.1 THE BUILDING BLOCK METHODOLOGY
16.1.1 Conclusions
16.1.2 Key role of the Building Block Methodology in South African water-resource management
16.1.3 Recommendations for further development of the Building Block Methodology
16.2 EFFECTS OF DIFFERENT MAGNITUDE FLOWS ON RIVERINE ECOSYSTEMS
16.2.1 Conclusions on dry-season baseflows
16.2.2 Conclusions on the relationship between dry-season baseflow and common hydrological indices
16.2.3 Conclusions on dry-season freshes
16.2.4 Conclusions on wet-season floods
16.2.5 Recommendations for flow-related research, with particular reference to its application in instream flow methodologies

16.1 THE BUILDING BLOCK METHODOLOGY

The Terms of Reference for the project include two main requirements. These are to continue development of the BBM, and to study the effects of different magnitude flows on riverine ecosystems issues are addressed below. The current status of the BBM is summarised in this section.

16.1.1 Conclusions

Due to the extensive collaboration between, and contributions of, the South African DWAF, the WRC and the national body of river scientists between 1991 and 1996, the BBM was developed to what could be considered the end of Phase One of its development. Attainment of this stage could be measured by the following achievements:

- incorporation of multidisciplinary data and expert knowledge into a structured process for assessing instream flow requirements for rivers;
- development of this process, through application in approximately 20 real-life water projects, to the point where its concepts and assumptions could be stated and standard procedures described (Chapters 3-6);
- institutional acceptance of the BBM by DWAF, the Department of Environment Affairs and Tourism, regional Nature Conservation bodies, leading Non-Governmental Organisations (NGOs), and assessors within the World Bank;
- use of the output of the BBM in South Africa to specify water for environmental maintenance in permits for water-resource developments;

- growing international interest in, and queries about, the BBM (from Australia, England, Taiwan, Lesotho, Uganda, Italy and Portugal) and its inclusion as currently the most comprehensive holistic methodology, in a review of international instream flow methodologies (Tharme 1996, 1997c);
- writing of a practical manual for application of the BBM by the national body of scientists responsible for its development. This manual is in preparation and due for completion by December 1998.

Few would question the need for the BBM to develop further. Some suggestions for its further development are given in Section 16.1.3.

16.1.2 Key role of the Building Block Methodology in South African water-resource management

Development of the BBM paralleled increasing public awareness and government action regarding the deteriorating condition of the country's rivers. The new, post-apartheid government that came into power in 1994 identified many new priorities for the country, including a more equitable distribution of the nation's water resources. This latter priority is reflected in the new Water Bill, presently before Parliament.

Among the 28 Principles of the proposed new Water Law are four which outline a new, sustainable approach to water-resource management.

Principle 7

The objective of managing the quantity, quality and reliability of the nation's water resources is to achieve optimum, long term, environmentally sustainable social and economic benefit for society from their use.

Principle 8

The water required to ensure that all people have access to sufficient water shall be reserved.

Principle 9

The quantity, quality and reliability of water required to maintain the ecological functions on which humans depend shall be reserved so that human use of water does not individually or cumulatively compromise the long term sustainability of aquatic and associated ecosystems.

Principle 10

The water required to meet the basic human needs referred to in Principle 8 and the needs for the environment shall be identified as "the Reserve" and shall enjoy priority of use by right. The use of water for all other purposes shall be subject to authorisation.

The BBM played a vital role in establishing in law the concept of an "environmental reserve", as it illustrated that such amounts of water could be quantified. With this concept established, the challenge now is to set in place a range of methodologies, one of which will presumably be a (continually improving) BBM, to allow the environmental reserve to be quantified. Tharme (1997c) envisages an hierarchy of methodologies for this purpose. At the bottom tier could be rapid, coarse hydrologically based methods for use in initial reconnaissance-level planning at the regional level. At the second tier, could be holistic methodologies such as the BBM, for fairly detailed instream flow assessments for any river. The top tier could be reserved for

rivers of special conservation importance. This tier could consist of an holistic methodology, linked to sophisticated habitat-modelling techniques for assessing the flow-requirements of species of concern.

DWAF is presently driving development of the rapid, first-tier approach to be used for regional planning. It is widely recognised that a nation-wide, coordinated research and management programme must be established, to further the defensible quantification of the environmental reserve. Discussions have started among research and government bodies on how this could best be achieved.

16.1.3 Recommendations for further development of the Building Block Methodology

Desired State versus Current Environmental Values

One of the concepts central to the BBM has been that the flow regime described for the river – the IFR – is designed to achieve and maintain some pre-determined Desired State of the river. In practice, this has been difficult to work with, as the DS has tended to be described in a somewhat general way. This has not guided the IFR team well in determining exactly what they are describing flows for. Nor has it provided the post-development monitoring team with specific targets for assessing the success of the IFR in maintaining that DS. Recent production of a protocol for defining the desired state of riverine systems (Rogers and Bestbier 1997) will help improve this situation.

Australian colleagues (Dr Jane Doolan, Department of Conservation and Natural Resources, Melbourne pers. comm. and Prof. Barry Hart, Monash University, Melbourne pers. comm.) have employed a somewhat different approach, using the concept of Current Environmental Values (CEV). They have suggested that an IFR team could describe a flow regime for maintaining pre-identified CEVs, with each specialist thus having a clear objective for recommending certain flows. Other conditions of the river linked to other possible flow-release patterns could be addressed at the Scenario Meetings. Using this approach, targets for monitoring maintenance of the CEVs would also be clarified.

An alternative scenario-based approach has recently begun development in a collaborative effort between South African, Australian, North American, New Zealand and Lesotho scientists. This removes all aspects related to stakeholders value judgements from the IFR workshop deliberations, and places them in a different part of the process. The IFR workshop then concentrates specifically on the consequences to the river ecosystem of a range of possible flow regimes.

The implications of all these approaches, and the extent of their compatibility, need to be researched.

Habitat mapping and physical-habitat modelling

Scientists working on IFR assessments have expressed the need for information that places in context the surveyed cross-sections forming the basic information on an IFR site – a kind of bird's eye view of the site. There is also a need to be able to visualise how physical habitat changes across the entire site with flow and how this might affect biotic distributions. Both of these facilities exist in the Instream Flow Incremental Methodology (IFIM) (Bovee 1982). However, there are many drawbacks to applying IFIM, including its complexity and the uncertainty regarding the meaning of some of its central outputs (King and Tharme 1994).

Different kinds of approaches which could meet the scientists' requirements are being researched as part of the Kruger National Park Rivers Research Programme and at the University of Cape Town, in what have been named Abiotic-Biotic Link projects. These approaches, and any other potential ones, need assessment and further development. As the BBM is essentially a methodology for organising data and understanding, it has the potential to be able to use data from any such approach.

Water-quality link-up

Most emphasis in the BBM is on water-quantity requirements for the river ecosystem, and the water-quality component is weak. Strengthening the quantity-quality link could take several forms, but should probably include a modelling component. This should be able to inform on probable future chemical conditions in the river at a resolution relevant to ecological studies. There would need to be a parallel comprehensive programme of data collection on the range of tolerances of riverine biotas, in order to predict the biological responses linked to predicted future chemical changes. A current WRC project is examining ways of predicting changes in water chemistry resulting from changes in stream flow (Dr J.A. Day, FRU, UCT pers. comm.).

Taking the Building Block Methodology into the Feasibility and later Phases of developments

After the IFR workshop, the water manager uses the recommendations on flows in yield analyses. These reveal the viability of the water-resource scheme with IFR flows in place. If a project of dam building is to proceed, the IFR is also considered in dam design and in any Scenario Meetings where flow-release options other than the IFR are considered. The ecological, economic and social consequences of each option are conveyed to interested and affected parties through a Public Participation Process. Environmental management plans may be drawn up for the construction phase and monitoring programmes set in place.

All of these activities take place at present, but no recognised set of procedures exists. The BBM urgently needs further development in these areas.

16.2 EFFECTS OF DIFFERENT MAGNITUDE FLOWS ON RIVERINE ECOSYSTEMS

Three components of the flow regime, as recognised in the BBM, were concentrated on, namely dry-season baseflows, dry-season freshes and wet-season floods. These are discussed in the following sections.

16.2.1 Conclusions on dry-season baseflows

The results of a low-flow study of the effects of natural and unnaturally low (experimental) dry-season baseflows on four upper-river ecosystems (detailed in Chapters 10 and 11) lead to the following general conclusions:

To assess the potential impacts of abnormally low flows, the natural baseflow regimes at the time of the study and the experimental low flows were placed in the context of the natural, long-term hydrological history of each river. This revealed that the impact locations on the three experimental rivers were subjected to unnaturally low discharges during the experiment, whilst their control locations and the fourth river, used as a control, were not. The experimental reductions represented marked hydrological disturbances to the

three riverine ecosystems, being well beyond most that they had been subjected to historically over a known period of almost 30 years.

The magnitude of experimental flow reduction was shown to be an inadequate index of the degree of hydrological disturbance the rivers had experienced. Rather, this reduction needed to be placed into perspective by being expressed as a percentage of the natural low flow pertaining at the time. Even so, it became clear that it would be inappropriate to use a single low-flow index to identify suitable low flows for an entire dry season, and for all rivers. Each of the experimental rivers in each month of the dry season naturally exhibited an individual hydrological character. Only the peak dry month, February, showed a highly stable, low range of baseflows at all sites.

On a month-by-month basis, three hydrological indices routinely used internationally to identify low flows were shown to have merit for delineating the approximate discharge range below which flows probably have an unnatural impact. These were the Q_{90} and Q_{95} values from 1-day annual FDCs, and the median 7-day consecutive low flow. Each of these showed merit in that the natural flows measured during the study lay near their values, whilst the experimental low flows were closer to other indices representing lower flows, such as the absolute minima on record. However, there was no clear correlation between any one of these indices and the natural measured discharge of the rivers over the dry season. Thus, in the selection of appropriate low flow indices for instream flow studies, cognisance should be taken of which hydrological indices best represent the flows that a particular river usually experiences.

The natural and experimental dry-season baseflow regimes were assessed in terms of their effects on the following ecosystem components: water chemistry; biotope proportions and hydraulic character; and the composition and distribution of BMI assemblages.

Under natural baseflows, the four studied rivers exhibited considerable between-river, inter-reach and temporal variability in measured components. Differences in values of many chemical variables between summer months were often fairly low, indicating high overall stability during a dry season with no clear relationships with baseflow magnitude. The main, natural changes in chemistry appeared to occur at the very end of the dry season, with the initiation of a natural shift to autumn climatic conditions. The proportions and hydraulic character of physical biotopes clearly changed in response to increases or decreases in discharge magnitude, although the dominant biotopes were always riffles and runs, each of which supported a distinct BMI assemblage. Changes in the distribution of BMIs were seldom consistent during the study period, however, suggesting that other factors in addition to discharge, particularly biotope hydraulics and proportions, and season, contributed to observed trends. An understanding of the extent of such variability is essential in order to differentiate between a river's natural functioning and responses directly associated with unnaturally low flows. It is therefore suggested that the emphasis in a low flow assessment should be especially on the presence of the overall combination of biotopes and their associated BMI assemblages, rather than only on those of riffles as has often occurred historically. Such knowledge would be a critical point of departure in the development of a recommended flow regime.

When assessing the effects of experimental low flows, quantification of the impacts was ecologically more meaningful when related to some combination of the components than to any one component. For instance,

an increase in a species of BMI with reduced flows might be related to animals packing into a reduced area of suitable habitat rather than to an increase in abundance *per se*.

Experimental low flows had limited secondary impacts on water chemistry. With flow cessation over much of the location, as well as extremely low velocities and the creation of isolated standing waters in other areas, there was some deterioration in water quality, mostly in conductivity, TDS, TA and pH. The most pronounced effects were recorded in isolated pools in the Riviersonderend River, where the reduction in flow resulted in increased conductivities. A free-flowing main channel would undoubtedly have limited the extent of such changes. In slow moving or still waters, tertiary effects of flow reduction included the proliferation of algae and desiccation of instream/marginal vegetation, the biological implications of which have yet to be explored. Freshes occurring late in the dry season flushed the poor-quality water from the affected locations, rapidly restoring natural conditions. It should be noted that the study sites were near-natural and that there might be additional or more significant impacts of unnaturally low flows on the water chemistry of polluted reaches.

With reference to biotope dynamics, experimental low flows mostly resulted in highly exaggerated forms of the changes observed with natural low flows. However, there were some major shifts in biotope dominance and hydraulic character. There seemed to be a redistribution of invertebrates between runs and riffles in response to such biotope changes, with the BMI composition of run assemblages thus growing to more closely resemble that of riffles. The responses were difficult to detect, probably due to various, poorly understood contributing factors. These could have included the extent to which the hyporheos acted as a refuge area; facultative versus obligate utilisation of biotopes other than riffles and runs; downstream drift of insect species; and species' life-cycle generation times and emergence patterns. Factors such as the inherent resistance and resilience of the BMI assemblages in relation to flow-related physical disturbance may also have played a role. For instance, a lesser response by BMIs to extreme low flows was evident in the Riviersonderend than the Du Toits impact location, despite similar experimental reductions in discharge in both rivers. This could perhaps be attributed to factors such as the naturally, very low midsummer baseflows and highly heterogeneous composition of physical biotopes in the Riviersonderend, relative to the other sites. Finally, BMI response might have been occurring within families, which would not have been detected in the analyses done for this report. Extreme low flows might also be resulting in longer-term, cumulative stresses on the biota, not necessarily detectable using the types of data collected in this study or within its timeframe. Overall, the changes were variable and not necessarily proportional to the degree of flow reduction. It is worth noting that the above analyses were based only on data from the first impact month. Data for the second impact month, collected but not yet analysed (Tharme in prep.), might reveal different levels of response. In particular, the intensity of biotic interactions might well increase during the longer period of extreme low flow.

Biotic variables such as diversity, the number of families per biotope, and the total number of individuals, showed merit as suitable measures of invertebrate response to low flows. However, due to the complexity of observed responses, it is likely that a suite of biological indices would need to be used to assess impacts of low flow, in conjunction with further, more sensitive multivariate analyses. Apparent trends in responses could also be due to factors other than, or in addition to, changes in the magnitude of baseflow, some of which might not be flow-related. As yet, the observed biotic responses to low flows and the implications thereof for the ecosystem remain poorly understood.

In conclusion, although various changes were observed in all ecosystem components at low flows, there were few clear trends. Furthermore, ecosystem responses to experimental low flows were not simply directly related to the graded percentage decrease in flow from natural magnitudes. Between-river differences in BMI assemblages were typically more pronounced than differences related specifically to natural or experimental dry-season baseflow regimes. River zone, season, biotope type and hydraulic character, and substratum component appeared to have at least as much influence on the composition of the BMIs as baseflow magnitude. This precluded the identification of a series of well-substantiated low flows, each strongly linked to a set of consequences for the river ecosystems. Hence, although at this stage the study has not revealed that experimental flow reduction had major impacts on the BMI assemblages of the rivers, broad-scale generalisations in instream flow assessments regarding suitable low flows would be simplistic.

Comparison of the results of the study with other international or local literature was limited by the dearth of similar published studies on the effects of abnormally low flows on perennial rivers. Additional similar research would be required before significant further development could occur of the scientific basis of the baseflow component of instream flow methodologies like the BBM (Section 16.2). Some of the required research, which follows on from the preliminary results of this study, is underway (Tharme in prep.).

16.2.2 Conclusions on the relationship between dry-season baseflow and common hydrological indices

The investigation of relationships between wetted perimeter and discharge in 11 Western Cape streams and rivers revealed that the graphical inflexion points, reflecting the discharge at which the wetted bed began to shrink in from the banks, that is the Wetted Baseflow (WBF), were correlated with one or more common hydrological indices (Chapter 12). The most promising indices in terms of consistency across the full sample of rivers assessed were linked to:

- the MAR;
- the 1-day annual flow duration curve;
- the 1:2 Year Maximum Average Daily Flow (MADF).

The WBF lay between 29% and 42% of the MAR (expressed as an instantaneous flow), between the 40th and 53rd percentile on the annual flow duration curves; and between 2.0% and 2.9% of the 1:2 year flood (expressed as a maximum daily average). The relative consistency of the WBF links with these three indices suggested that the WBF in south-western Cape streams could be calculated from some coarser hydrological statistic.

It had been expected that the WBF would be a very low flow, tentatively named the critical minimum, below which, for ecological purposes, discharges should never be reduced. It had been expected that some higher flow would be more acceptable as the low flow necessary for ecosystem maintenance, with the WBF or critical minimum being the level that flow could drop to only in drought years. However, the research showed that the WBF approximated the median flow of each river, or on average the 47th percentile of the annual flow duration curve. Hence, relatively undisturbed rivers would experience flows lower than the WBF for about half the year, probably during the dry season. They must therefore experience wetted-bed shrinkage in the normal course of events and the aquatic species must be adapted for such seemingly adverse conditions.

It was concluded that the lower limit of flows for the dry season should be based not on the WBF, but on biological considerations. Nevertheless, the WBF was found to be a useful first point when considering ecologically-acceptable low flows, as it provides an order-of-magnitude indicator of a low flow that is within the normal baseflow regime of a non-gauged river.

16.2.3 Conclusions on dry-season freshes

At the beginning of the project, freshes were described in a general sense as *a small increase in discharge over a short period during the dry season.* A preliminary hydrological analysis of dry-season data (Appendix 1.1) generated an hypothesis that a fresh is "any increase in flow that is large enough to ensure that the average flow of the next two days is 1.7 or more times that of the preceding two days". Although such a small increase in flow may have little significance during the rainy season, when large floods would mask it, such increases increase vastly in importance during the dry season. At this time, when little rain may fall for up to 4-5 months, small increases in flow can briefly induce quite substantial changes in hydraulic conditions, water temperature and water chemistry. The plant and animal species may have evolved in many ways to cue life-cycle activities to these changes.

A good example of such cues is the apparent link between freshes and migration and spawning of the Clanwilliam yellowfish, *Barbus capensis*. Experimental releases of freshes from Clanwilliam Dam on the Olifants River (Chapters 13 and 14) led to several hypotheses regarding these links, for further testing. They are, that upstream migration toward spawning grounds may be influenced by factors other than freshes, including photoperiod and temperature. These factors also appear to bring the fish into breeding condition. Once in this condition, freshes appear to trigger spawning, but will not do so if water temperatures are below a threshold value of 19 °C. Manipulation of dam release levels may allow temperature requirements to be met, but further development of the embryonic fish could be jeopardised if such temperatures are not maintained.

In summary, it is suggested that freshes incorporated into IFRs should have more complex specifications than simply a magnitude, duration and timing. Where they are requested for the spawning and recruitment of fish species, freshes should be released when the fish are in breeding condition after ensuring that suitable spawn sites exist. Such releases should be of the magnitude and duration normal for that time of year. They should provide at the spawn site water of a temperature known, through observation and experimentation, to allow successful spawning of the fish species. These temperatures should be maintained after spawning, to allow survival of the early life-stages. If such conditions are not met, the water released as freshes to enhance fish recruitment may not achieve its purpose.

16.2.4 Conclusions on wet-season floods

Fine sediments entering high-gradient, cobble-bed reaches of rivers are normally scoured out during wet-season floods. Damming of rivers and other manipulations of flow regimes often lead to a loss of floods. This in turn can result in a reduction in the scouring of fines. Many small aquatic animals, particularly the rare and endemic species, live on the underside of cobbles or in the substrata beneath the cobbles (Chapters 10-11 and Appendix 1.2). Sedimentation of these reaches can destroy their refuge areas, often seriously affecting the normal functioning of such ecosystems and possibly eradicating species from the river. In

recognition of this and associated problems, flushing flows are usually included in recommendations for flow regimes for rivers targeted for water-resource development. This is done without clear understanding of whether such flows would scour fines from cobble beds.

Research was therefore undertaken to establish a relationship between the extent to which scour takes place within a cobble-bed river, and the magnitude of discharges that have occurred in the river (Chapter 15). Using field data on cross-section river size and shape, the position of the top of cobbles and of the sand surface, sediment size and discharge, collected in the Molenaars River in Du Toit's Kloof, a stage-discharge relationship was obtained for the study reach and a Sediment Scour Model developed. The model links the extent to which sediment is scoured (as revealed by the average diameter of the sampled sediments), with the maximum velocity of the water flowing over the area of scouring in time-spans from 24 hours to 20 days before the sediment was sampled. Thus, for instance, for any average-size substratum, the velocity that needed to occur the previous day to achieve that size could be identified. The higher discharges that would be needed at time-spans ranging from two to 20 days earlier, to achieve the same result, could also be identified.

At present, use of the full Sediment Scour Model is limited to applications to the Molenaars River in subsequent winters in which the same catchment conditions apply, as the relationship derived is strongly dependent on the rate of supply and deposition of sediments during the study. With changing conditions, the further removed the sampling day is from the day of maximum discharge, the more uncertain the derived relationship between flow and sediment size becomes. However, relatively little deposition usually occurs in the 24 hours after maximum flood discharge. Thus, the part of the Model that links sediment size with a required maximum flow 24 hours previously can be applied more confidently to the Molenaars for subsequent years, as well as to other rivers with similar bed and sediment input characteristics.

16.2.5 Recommendations for flow-related research, with particular reference to its application in instream flow methodologies

Recommendations for further research on dry-season baseflows

A considerable body of further work is required to define low flows on an ecological basis and to identify ecological-relevant low flow indices. This work needs to include the development of a conceptual model of the possible effects of low flows on perennial riverine ecosystems, and the establishment of low flow modelling techniques, linking hydrology with habitat quality and quantity and the instream biota. These sets of information are required for the further development of the baseflow component of the BBM and other holistic instream flow methodologies. In this regard, apart from the continuing focused work on dry-season baseflows (Tharme in prep.) based on the findings of the current report, several main areas of research were identified that require further effort.

Linking present hydrological events with aspects of the historical flow record

Currently, links between present-day hydrological events and the longer-term flow record are seldom made, and usually only on a case-specific basis within instream flow assessments. Although low flow regionalisation is now well underway (Smakhtin *et al.* 1995; Smakhtin and Watkins 1997; Smakhtin and Toulouse 1998; *inter alia*), there is considerable further scope for the inclusion of more ecologically meaningful hydrological indices, specifically those describing flow variability and predictability, in the

hydrological component of the BBM. As part of this process, analyses of flow percentiles are required to identify which of them (or other low flow indices) represent baseflows that are commonly experienced in different river zones, and various sizes and types of river during the different months of the dry season. Additionally, research should be continued on the implications of flows derived on the basis of various low flow indices, for riverine ecosystems. From a field perspective, as a result of the natural variability in discharge as measured within a reach, many instantaneous measurements of dry-season baseflow are required before a relationship between present-day discharge and various flow indices derived from the long-term record can be adequately established. Thus, there is a need to develop a better understanding of the relationship of instantaneous baseflow to average daily discharge and other discharge variables for the same time period. In relation to such records of instantaneous discharge, accurate measurement at low flows is extremely difficult, and there is a high experimental error associated with the results. Such a margin of error could mean that small fluctuations in discharge magnitude that are potentially important from an ecological perspective might be masked. More accurate ways of measuring discharge and associated hydraulic variables at very low flows, therefore, need to be established to avoid this potential problem.

The importance of antecedent hydrological events also needs to be considered when assessing the impacts of low flows on rivers or developing instream flow recommendations for their baseflow component. For example, antecedent flows may allow the development of certain conditions favouring a particular state of the ecosystem. Although subsequent flow events may alter conditions of individual ecosystem components, it may be to a lesser degree or take a different amount of time for changes to be reflected, due to previous flow conditions. Research is required into ways in which antecedent flow events can be adequately incorporated or considered when developing instream flow recommendations. These various issues have been recognised as aspects of the BBM that require further research effort, with more intensive collaboration between freshwater ecologists and hydrologists (Tharme 1997c; this study).

Development of broad scale relationships between baseflow and ecosystem variables

The relationships between dry-season baseflow magnitude and concentrations of various chemical variables need to be established on the basis of different river zones, types of rivers and regions of the country, particularly with reference to observed natural ranges of variability. Work is also required to establish similar regional trends in wetted perimeter-discharge relationships, as well as patterns of biotope dominance, biotope diversity, and other relationships between hydraulic variables and dry-season discharge.

The importance of differences in physical biotopes at low flows

Further work is required to objectively classify riffles, runs and other biotopes, and to describe the extent to which various transitional biotopes differ from them, both hydraulically and in terms of their invertebrate assemblages, at low flows. Relationships between various combinations of substratum and flow type, which comprise physical biotopes, and the assemblages inhabiting them, are the focus of a current WRC project (King pers. comm.). Such knowledge might further assist with the identification of similar areas of river habitat for the biota.

It was evident from this study that additional work would be required to establish at what proportions, and with what ranges in hydraulics, various biotopes become critically limiting for the biota, as a result of flow reduction. Such an assessment would require knowledge of the specific biotope requirements of the various taxa comprising each invertebrate assemblage, and hence an indication of preferred and marginal biotopes

for each. For instance, the relative suitability of fully immersed cobbles compared with partially exposed cobbles and entirely exposed substratum, as invertebrate habitat has yet to be examined.

Direct influences of low flows on instream biota

Research is required to explore the probability, as suggested in the international literature, that there is an increase in the importance of biotic interactions at very low flows. Additional related studies are required to address the implications of unnaturally low flows for aspects of the life cycles of individual species, such as generation times and growth rates. Currently, there is no research on these topics, yet it is urgently required if progress in understanding the fundamental requirements of instream biota for dry-season baseflows is to be achieved.

Comparative studies of the similarities and differences between biotic (and other) responses to low flows in perennial and temporary rivers, for different baseflow regimes, and under undisturbed and anthropogenically-altered conditions, are required. Such work would assist with establishing the degrees of resistance to disturbance and resilience in terms of recovery from impacts, associated with different types of river.

Recommendations for further research on dry-season freshes

Present knowledge is inadequate regarding the fundamental nature of freshes, and of their role in river ecosystem functioning.

Freshes are loosely defined as small pulses of higher flow during the dry season. There is still no clearer definition, or recognition of what such a definition should be based on. It seems probable that the most meaningful definition will eventually emerge from an enhanced understanding of the relevance of freshes to river ecosystem functioning. For instance, they appear to be extremely important from the perspectives of fish breeding and water quality. It is therefore recommended that basic research be designed to further understanding of how these and other lesser-researched components are affected by natural freshes. How important are freshes in terms of bringing relief to fish or other instream aquatic species, from extreme dry-season chemical or thermal conditions? What level of hydrological variability is encompassed in freshes, and what are the consequences of their loss in regulated rivers?

In parallel, research should continue to address management issues, such as how the operating rules for dams could help mitigate their effects on downstream reaches. In this respect, it is recommended that the experiments to trigger spawning of *Barbus capensis* in the Olifants River, reported on in Chapters 13 and 14, be repeated, in order to test the hypotheses developed. These hypotheses are that:

- the water temperature on the spawn beds for successful spawning of *B. capensis* should be at least 19 °C and stable or rising;
- if the fish are in breeding condition and these temperature conditions are met, then experimental releases from Clanwilliam Dam of the size used in this study will trigger spawning.

The research reported on in Chapters 13 and 14 has already stimulated plans for similar studies elsewhere in South Africa and in Australia. An opportunity exists to co-ordinate efforts and enhance collaborative endeavours.

Recommendations for further research on wet-season floods

The role of floods in maintaining silt-free cobble beds is poorly documented. With increasing levels of soil erosion in the country, and consequent increasing sedimentation of the rivers, river habitat for aquatic species must be deteriorating in both area and quality. Research is needed on the effects of siltation on river ecosystems, and the extent to which floods reverse or exacerbate this situation.

At the level of a student project, the Molenaars River Sediment Scour Model could be used to identify the discharge that would remove sand from the cobble bed. There was not time in the project reported on here to use the river's hydrological record to ascertain the return period of this discharge. This further step is essential, in order to compare it with the magnitude of floods normally recommended in instream flow assessments.

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Appendix 1.1 Analysis of the effect of freshes in river flow regimes.

This Appendix presents the abstract of a fourth-year thesis completed as part of the requirements for the degree of B. Eng. (Civil) at the University of Stellenbosch (De Kock 1995). The thesis topic was inspired by this project, but after preliminary discussions with Dr J.M. King, was undertaken as an independent study by M. de Kock, under the supervision of Prof. A. Görgens. It was not funded from this project. The research took as its starting point the fact that flow duration curves (FDCs) are commonly used by hydrologists to demonstrate how water-resource developments alter river flow. They are, however, difficult for ecologists to use, as they do not clearly illustrate which components of the flow regime have been altered. The purpose of the work was thus to investigate the effect on FDCs of removing freshes from selected hydrological data sets.

Written by M. de Kock

The importance of the maintenance of the natural flow regimes of rivers for the sustaining of river ecology recently came to the attention of DWAF. The department appealed to ecologists to characterise the preferred flow patterns for all the rivers involved in water-resource developments.

The ecologists needed a reliable way to analyse daily flow data for many years. They showed specific interest in the occurrence and effect of freshes. A fresh is a sudden increase in the baseflow, not large enough to be a flood. These freshes fulfill important ecological functions.

The purpose of this thesis was to create a spreadsheet within which daily flow data could be analysed. The specific objectives were:

- to quantify a fresh;
- to find a method to identify a fresh;
- to replace the identified fresh with baseflows;
- to observe the effect the replacement had on FDCs;
- to find the average, minimum and maximum number of freshes per season;
- to indicate the average, minimum and maximum sizes of the peaks of freshes;
- to indicate the average, minimum and maximum volumes of freshes;
- to perform the analysis for summer and spring.

The most important findings were:

- a fresh is any increase in flow that is large enough to ensure that the average flow of the next two days is 1.7 or more times that of the preceding two days;
- the size of freshes is dependent only on the preceding flow and they do not have a minimum magnitude;
- the same definition for a fresh is applicable for all seasons;
- the replacement of freshes by baseflows causes the middle part of a FDC to decline, whilst the end points stay the same.

Appendix 1.2 A preliminary study of the effect of spates on the species composition and abundance of benthic macroinvertebrate assemblages on rocks in two biotopes in a Cape mountain stream.

This Appendix presents the abstract of a thesis completed by A. Emery as part of the requirements for the degree of BSc. (Hons.) at the University of Cape Town (Emery 1994). The research was funded from this project and Dr King acted as supervisor. The purpose of the research was to ascertain where benthic macroinvertebrates normally live in a cobble-bed river (top or bottom of cobbles, or in the underlying substrata), and if they migrate vertically down into the river bed during floods. The research was designed to contribute to an understanding of the need for flushing flows, by illustrating how macroinvertebrates move around within a sediment-free cobble bed.

Written by A. Emery

Differences between riffles and runs in the Molenaars River, in terms of their hydraulic characteristics and their rock-associated macroinvertebrate assemblages, were determined under conditions of winter baseflow and winter flood. The effects of a spate (flood) on the hydraulic characteristics, assemblages and individual taxa of a riffle and a run were investigated using a sequence of samples taken before, during and after the spate.

Classification procedures (CLUSTER analysis and ordination by multi-dimensional scaling) were used to search for changes in the assemblages and to compare these with changes in the hydraulic characteristics of the aquatic habitat.

There was evidence of distinct hydraulic and biological differences between the riffle and the run under conditions of winter baseflow. The hydraulic differences, in terms of Froude number and depth, but not in terms of velocity, remained distinct for the two biotopes through the course of the spate. The differences in terms of the species composition and abundance were destroyed by the spate and there was then a process of recovery over the next seven days that was detectable by the partial return of the invertebrate species to their original biotope.

There did not appear to be a mass migration of invertebrates into the cobble bed during the spate. Some taxa appeared to move to the tops of the rocks, and there were also signs that some were washed downstream. However, other taxa did show a movement into the cobble-gravel substrata under the bed. Thus, for some species at least, the areas between and under the cobbles were essential refuges during spates. These areas would be unavailable in sediment-choked cobble-bed rivers.

Appendix 1.3 The effects of anthropogenic perturbations on the aquatic invertebrate fauna of the Olifants River system.

During the course of the WRC project, one of two student theses (see also Appendix 1.4) was written that was inspired by the project although not directly linked to its objectives. It was written by P. Kuun, Zoology Department, University of Cape Town, and submitted in partial fulfillment of the Bsc (Hons.) degree (Kuun 1994). The research was funded from this project, and Dr J.M. King acted as supervisor. The original objective of the research was to investigate if meta-analysis (the analyses of biotic communities at resolutions from species to phylum) could be used to assess the severity, in terms of invertebrate communities, of different kinds of physical disturbance to rivers. This objective had to be modified somewhat for the purposes of a student project. The thesis abstract is presented below.

Written by P. Kuun

In order to ascertain possible anthropogenic perturbations on the tributaries of the Olifants River in the southwestern Cape, benthic macroinvertebrate communities were surveyed in early May 1994. Surveys were done at 13 sites on tributaries of the mainstream Olifants River, as well as one site on the mainstream itself. Environmental variables were also measured at these sites in order to identify any patterns between biotic and abiotic variables. Six environmental variables were found to best explain the observed distribution pattern of macroinvertebrate communities, although this correlation was not strong.

Two criteria, *viz.* extensive water abstraction from, and bulldozing of, rivers were used to *a priori* identify four out of the 14 sites as 'disturbed'. These sites were compared to less disturbed sites using non-parametric multivariate analysis of similarity of community data, and the two groups of sites were found to be remarkably different. The 'disturbed' sites correlated well with subsequent analysis of biotic data using cluster analysis and multi-dimensional scaling. This suggested that the major impacts on these rivers could easily be detected without complicated analysis. Three taxa were found to be potentially indicative of less-disturbed conditions.

The rivers that were disturbed by bulldozing and extensive water abstraction are identified as being in need of corrective management. Maintenance of riparian vegetation is identified as probably being important in the management of the Olifants River tributaries, and research on the effects of riparian vegetation in alleviating the impacts of heavily farmed areas on rivers is consequently recommended.

Appendix 1.4 Researching the implications of pre-determined release policies and dam sizes on the natural flow regimes of Western Cape rivers.

During the course of the WRC project, one of two student theses (see also Appendix 1.3) was written that was inspired by the project although not directly linked to its objectives. The thesis was written by M. de Villiers, a fourth-year student in the Department of Civil Engineering, University of Stellenbosch, and submitted in partial fulfillment of the degree of B. Eng (Civil) (De Villiers 1995). The research was not funded from the project and, after preliminary discussions with Dr King, proceeded as an independent study under the supervision of Prof. A. Gorgens. The abstract is presented below.

Written by M. de Villiers

This study was undertaken to assess the impact that water development, and especially irrigation-farming practices, of the Western Cape has on the flow regimes of the rivers. It was designed to determine if the flow regime linked to a certain Desired State (DS) for a river could be provided by following a pre-determined release policy for reservoirs. Eight rivers were used in the study, of which four were used to obtain the effect of dam sizes on flow regimes and the remaining four were used to find the balance between DS and sustainable irrigation supplies. Hypothetical dams were placed in the rivers, to simulate the accumulative effect of farm dams in the catchment areas. These dams accounted for losses due to evaporation, the absorption and delaying of floods and also the demands from the rivers and from farmers for water to fulfill their needs.

First, the desired flow regime was determined for each river, linked to one or more pre-determined and hypothetical DSs. This was done using the Bulk Water Estimate approach (King pers. comm.), and by studying aspects of the flow regimes for each river. The desired flow regime was then superimposed onto the relevant river system, by replacing the normal dam releases with the Bulk Water Estimate.

It was discovered that one MAR dams are more suitable for sustaining demands from the river and the irrigation farmers than smaller half-MAR dams. Furthermore, it was concluded that very small rivers should not be dammed because it will be very difficult to supply enough water at the right time to sustain a release policy for a certain DS.

Rivers with a DS of "Wild and Scenic" are not suitable for use by irrigation farmers, but rivers declared as "Working" rivers are. This study concluded that there is a middle way, and that irrigation farmers and river scientists can work together for the benefit of both irrigated farmland and the rivers that supply the water.

Appendix 2.1 **Specialists who responded to the questionnaire on flows for river maintenance in 1991 (see Section 2.4.1). Some specialists provided responses for more than one area.**

Name	Institution
W. Alexander	University of Pretoria
J. Alletson	Natal Parks Board
P. Ashton	Watertek, CSIR
B. Benade	Northern Cape Nature Conservation
A. Bok	Eastern Cape Nature Conservation
J. Bosch	Forestek, CSIR
Chief Executive	Rand Water Board
M. Chutter	Watertek, CSIR
A. Coetzer	Cape Nature Conservation
B. Davies	University of Cape Town
J. Day	University of Cape Town
F. De Moor	Albany Museum
I. Gaigher	University of Venda
J. Grobbelaar	University of the Orange Free State
D. Hughes	Rhodes University
J. King	University of Cape Town
C. Kleynhans	Transvaal Nature Conservation
M. Mayekiso	University of Fort Hare
R. Scholes	University of the Witwatersrand
K. Tarboten	University of Natal
R. Tharme	University of Cape Town
F. Venter	National Parks Board

Appendix 2.2 **Delegates attending the workshop "Toward a methodology for instream flow assessments for South African rivers" at Rand Afrikaans University on 7 July 1993 (see Section 2.4.2).**

Name	Institution
M. Claassen	CSIR
C. Dickens	Umgeni Water
W. Ellery	University of the Witwatersrand
B. Gale	Cape Nature Conservation
R. Heath	CSIR
M. Henshall-Howard	University of Cape Town
J. King	University of Cape Town
J. Lee	Ninham Shand Consulting Engineers Inc.
M. Mokone	University of the North
C. Palmer	Rhodes University
G. Ractliffe	University of Cape Town
W. Rowlston	Department of Water Affairs and Forestry
M. Silberbauer	Department of Water Affairs and Forestry
S. Taljaard	CSIR
L. Taylor	University of the North
R. Tharme	University of Cape Town
M. Uys	Rhodes University
B. van der Waal	University of Venda
H. van Linge	University of Cape Town

Appendix 9.1 Explanation of the codes used when surveying the cross-sections and biotope maps in the control and impact locations at each site (see Section 9.5.1).

CODE	DESCRIPTION
EL	Elands site
MO	*Molenaars site*
DU	Du Toits site
RI	*Riviersonderend site*
C	Control location at site
I	*Impact location at site*
CM1, CM10, ..., CM60	Control location map, beacon indicating transect line at 1 m, 10 m, ..., 60 m
IM1 - IM60	*Impact location map, beacon indicating transect line at 1 m, 10 m, ..., 60 m*
IC5, IC15, ..., IC55	Control map, intermediate pegs indicating transects at 5 m, 15 m, ..., 55 m intervals
I5, I15, ..., I55	Impact map, intermediate pegs indicating transects at 5 m, 15 m, ..., 55 m intervals
BM	Survey benchmark(s) for the location
ELR1, ELL1, ...ELL4	For each site (e.g. EL), cross-section headstakes on right (R) and left (L) bank, for cross-sections 1 - 4; cross-sections 1 and 2 represent the control location, and cross-sections 3 and 4 represent the impact location
EW1, EW2, ..., EWn	Right bank and left bank water's edges (EW – edge of water) of the main channel, at all 10 m transects
WSE1, WSE2, ..., WSE5	Water surface elevations 1 - 5, for each cross-section
WSEM1, WSEM2, ..., WSEMn	Water surface elevations at each of the 10 m transect intervals, for each location
1.1, 1.2, ..., 1.n	Dry points or water's edge spotshots, e.g. bank, in sequence across cross-section 1
2.1, 2.2, ..., 2.n	Dry points or water's edge spotshots, in sequence across cross-section 2
3.1, 3.2, ..., 3.n	Dry points or water's edge spotshots, in sequence across cross-section 3
4.1, 4.2, ..., 4.n	Dry point or water's edge spotshots, in sequence across cross-section 4
MA1, MA2, ..., MAn	Instream spotshots, in sequence across each cross-section; instream spotshots are linked with other point microhabitat data (depth, velocity, etc.) using the MA (Marker) numbers

Appendix 9.1 Continued.

CODE	DESCRIPTION
0	General code for benchmark, beacon, headstake, WSE or EW spotshots
1	Bank
2	Water's edge of main channel
3	Water's edge of secondary channel or wetted area
4	Water trickling between substratum elements
5	Exposed/dry instream cobble/boulder bar
6	Run
7	Riffle, as a patch
8	Riffle, as a cobble/boulder bar
9	Pool, instream or marginal
10	Pool-run transition
11	Riffle-run transition
12	Bedrock rapid (not encountered)
13	Location of temporary flow diversion weir
14	Exposed/dry boulder
15	Submerged boulder
16	Backwater pool
17	Undercut bank
18	*Prionium serratum*, instream or marginal
19	Bedrock outcrop on channel edge
20	Inflow from side stream or seep
21	Tree roots/branches/woody debris
22	Water's edge on exposed boulder/bedrock
23	*Isolepis*, instream or marginal
24	Standing or stagnant water
25	Exposed/dry bedrock
26	Submerged bedrock
27	Drying side channel

Appendix 9.2 **Benthic macroinvertebrate samples collected from the wetted channel edges of the four impact locations, during the flow reduction phase of the low-flow study. Samples were not collected from the control locations, but samples from the Elands impact location represented the control data.**

SAMPLE TYPE	ELANDS	MOLENAARS	DU TOITS	RIVIERSONDEREND
FEBRUARY TRIP				
GRAVEL				
No. of samples	3	3	3	3
Sampling dates	28/02/95	24/02/95	20/02/95	15/02/95
DAMP FULLY EXPOSED				
No. of samples	3 x T, B and U	3 x T, B and U	3 x T, B and U	3 x T, B and U
Sampling dates	28/02/95	24/02/95	20/02/95	22/02/95
PARTIALLY EXPOSED				
No. of samples	3 x T, B and U	3 x T, B and U	3 x T, B and U	3 x T, B and U
Sampling dates	28/02/95	24/02/95	20/02/95	22/02/95
MARCH TRIP				
GRAVEL				
No. of samples	3	3	3	3
Sampling dates	16/03/95	24/03/95	13-14/03/95	11/03/95
DAMP FULLY EXPOSED				
No. of samples	3 x T, B and U	3 x T, B and U	3 x T, B and U	3 x T, B and U
Sampling dates	16/03/95	24/03/95	13/03/95	11/03/95
PARTIALLY EXPOSED				
No. of samples	3 x T, B and U	3 x T, B and U	3 x T, B and U	3 x T, B and U
Sampling dates	16/03/95	24/03/95	14/03/95	11/03/95