DEVELOPMENT OF NUMERICAL METHODS FOR PREDICTING RELATIONSHIPS BETWEEN STREAM FLOW, WATER QUALITY AND BIOTIC RESPONSES IN RIVERS

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PREFACE

This document is one of a series of reports arising from the Water Research Commission project K5/956 "Development of numerical methods for assessing water quality in rivers, with particular reference to the environmental flow requirements process". The reports are:

1. *Malan, H.L. and Day, J.A. (2002) Development of numerical methods for predicting relationships between stream flow, water quality and biotic responses in rivers. WRC Report no. 956/1/02.*
This volume details the development of the models and tools produced during the project and outlines the results of several applications of these tools to rivers in South Africa.

2. *Malan, H.L. and Day, J.A. (2002) Linking discharge, water quality and biotic response in rivers: a literature review. WRC Report no. 956/2/02.*
This volume presents a review of literature pertinent to the project in the fields of *inter alia* hydrogeochemistry, water quality modelling, environmental flow assessments and biomonitoring.

3. *Malan, H.L. and Day, J.A. (in prep) Predicting water quality and biotic response in ecological Reserve determinations. WRC Report no. TT XX.*
This volume will be a technical guide allowing water resource managers and consultants to use the tools described in (1) above.

Aspects of a previous WRC project (K5/626 "Water quality requirements for riverine biotas"), in particular the Biological-chemical database (Biobase), were continued in the current project (K9/956) and are reported in:

4. Dallas, H.F. and Janssens, M.P. (1998) Biological and chemical database: User manual. WRC Report No. TT 100/98.

EXECUTIVE SUMMARY

Introduction

This document presents the results of Water Research Commission project K5/956 "Development of numerical methods for assessing water quality in rivers, with particular reference to the Instream Flow Requirement process". The numerical methods that have been developed, or investigated in the course of the study enable predictions of stream flow-concentration relationships to be made for some key water quality variables in rivers. In addition, techniques have also been developed to predict the effect that changes in water quality may potentially have on the aquatic biota. These methods have been refined through application to actual Reserve Assessments. Another major product of this project is a survey of the literature pertinent to the study which is titled "Linking discharge, water quality and biotic response in rivers: a literature review" (Malan and Day 2002). In addition, a training manual is being written which can be used to teach some of the techniques developed in the project to managers and consultants in the field of water resource management.

Aims and objectives of the project

The aims of the project were partly to follow up on aspects of previous projects on water quality (aims A and B) but mainly to develop a new research area (aim C).

A) Pursue the use and curation of the Biobase (Biological-Chemical database) developed in project no. K5/626:

- Curate the database and capture additional data, as and when the need arises.
- Investigate future maintenance and accessibility of the Biobase, including the potential to provide an Internet facility for extracting and adding new data.

- Continue to interrogate the database with a view to:
  - verifying the scores recommended for the biomonitoring system SASS.
  - identifying water quality tolerances of key taxa of invertebrates for use in water quality assessments for the Instream Flow Requirement (IFR) process.
  -verifying the methodology for establishing regional water quality guidelines for selected constituents as proposed in project K5/626.

B) To continue to examine, in a low-key manner, the extent to which water quality determines the distribution of south-western Cape endemic invertebrates.

C) To address the question of the relationship between water quality and water quantity (particularly stream flow), with particular reference to the assessment of Instream Flow Requirements:

- To provide a review of the kinds and forms of data needed and of methods developed elsewhere for providing these data.
- To develop methods for processing water quality data to derive time and flow dependencies for key water quality constituents.
- To identify and assess methods for modelling selected water quality constituents at the feasibility level of the IFR process.
- To investigate the extent to which available data (e.g. in the Biobase) can be used for setting environmental water quality requirements as part of IFR assessments.
- Based on the above, produce a framework for the assessment of water quality as part of IFR studies.

Because of the urgent need for the development of tools to implement determination of the water quality Reserve, the attention of the project team was mainly directed towards Aim C). As a result, all the objectives in that section of the project were attained. In addition, through curation of the Biobase, as well as development of the "Biotic protocol" most of the objectives in Aim A) have been achieved.

Rationale for the project

For many rivers in South Africa, discharge (i.e. the volume of water flowing past a given point per unit time) varies substantially from season to season and water quality is profoundly influenced by flow. Where water quality refers to the concentration of chemical constituents and values of physical variables in the river. Typically, the instream concentration of most chemical constituents increase as discharge is reduced, a consequence of the increasing proportion of discharge resulting from groundwater (which is higher in concentration of mineral ions), and often, to diminished dilution of pollutants. The situation is not always so simple, however, and some constituents, largely due to wash-off from the surrounding land, increase in concentration with increasing discharge (at least in the low-flow portion of the discharge range). Discharge-concentration relationships are site-specific and depend on the particular chemical constituent under consideration. Although the methods used in South Africa to determine the quantity of water required for efficient ecological functioning are well advanced, so far the effect of alterations in river discharge on water quality is not adequately addressed in this process. It is important that predictions of the water quality that will result from a changed discharge regime be made and that these be taken into account when setting environmental flows, since the aquatic biota is profoundly influenced by the water quality to which they are exposed. In short, it is necessary to include water quality modelling (i.e. quantitative predictions of instream concentrations and values of physical variables) within Reserve Assessments to ensure that in meeting the ecological Reserve with regard to quantity, the water quality component of the Reserve is also attained. This will ensure adequate protection of the aquatic ecosystem, which forms the basis of the resource. The methods developed during this project were designed for use in determination of the ecological Reserve, although not specifically investigated, it would appear that they may be useful in other areas of water resource management.

The approach used

The project is composed of two major sections:

a) development of methods to predict the concentrations of given chemical components and values of physical variables that will result in a given river reach from the flow

regime that is recommended by the specialists at the Instream Flow Requirement (IFR) workshop;
and,
b) development of methods to predict the effect such changes in water quality may have on the abundance (or other measures) of key invertebrate species and thus on the ecological functioning of the aquatic resource.

Part a) of the project was carried out by means of the steps listed below:

- i) A review was undertaken of water quality modelling methods (both complex computer programs as well as simple spreadsheet methods) that are available internationally.
- ii) The completeness, availability and form of discharge and water quality data available in South Africa were assessed.
- iii) Consideration was given to the attributes the predictive method should possess.
- iv) From consideration of the steps above, the modelling method(s) most suitable for the project were chosen.
- v) The chosen modelling methods were then used and refined in actual Instream Flow Assessments.

Part b) of the project is concerned with linking the water quality tolerance ranges of selected biota with predicted constituent concentrations and values of physical variables. Thus, after using numerical modelling methods to predict the level of a water quality variable in a reach under the proposed discharge regime, the effect on the biota must be predicted. The following steps were taken in order to address this part of the project:

- i) The methods used elsewhere to assess the implications of altered water quality for the biota were reviewed.
- ii) An assessment of the availability of data and information regarding tolerance ranges of indigenous aquatic biota was undertaken.
- iii) A predictive tool was then developed.
- iv) The predictive method was applied to the results of Instream Flow Assessments and further refined.

Summary of the chapters in this document

This document is divided into the following chapters:

- **Chapter 1:** General background information on several relevant topics is summarised in Chapter one. These topics include, the new South African Water Law and the ecological Reserve; the variability of South African rivers with regard to discharge and water quality; and the need for the incorporation of quantitative predictions of water quality into Environmental Flow Assessments. The aims of the project are also listed in this section, as well as a summary of methods used elsewhere in the world to predict water quality, and to assess the effect of changed water quality on the biota. The attributes that are required in the chosen modelling method are listed. Finally, the elected models are presented.
- **Chapter 2:** A detailed description of the Discharge-concentration (Q-C) modelling method, developed for predicting chemical concentrations is reported. Results from application of the method in various Instream Flow Determinations; information that can be obtained using Q-C modelling; and most importantly, the assumptions and limitations in the method are also discussed.
- **Chapter 3:** This chapter reports the results from a countrywide survey of the discharge-concentration patterns for different DWAF primary drainage regions, as well as for different ecoregions. An attempt is made to predict likely patterns for certain water quality variables in a given area.
- **Chapter 4:** In this section of the report, a description of the method used to obtain time-series of chemical components (notably salinity) and time-series of water quality-induced stress that will be experienced by the aquatic biota is presented. An application to the Olifants and Breede Rivers, as well as the assumptions and limitations in the method, are reported. Finally, the usefulness and ecological interpretation of such time-series is discussed.
- **Chapter 5:** This chapter describes an application of the water quality model QUAL2E to the Lower Olifants River. Included, is a detailed discussion of the process and the problems that were encountered during the application. In addition, the use (including the advantages and disadvantages) of this modelling technique within the Reserve determination process is discussed.

- **Chapter 6:** This chapter presents the Biotic Protocol, which is a series of steps that enables predictions to be made of the effects that alterations in water quality may have on the biota. The protocol is described in detail, followed by the results from applications to the Middle and to the Lower Olifants River. Assumptions and limitations in the method are discussed, as well as future developments that are required.
- **Chapter 7:** Integration of water quality modelling and predictions of the relationships between discharge, water quality and the aquatic biota in the Reserve determination process, is the major consideration of chapter 7. In particular, the role and use of the tools developed in this project are discussed. Finally a framework for incorporation of predictions of water quality changes, and the implications for the biota, into the Reserve determination process is presented.
- **Chapter 8:** The final chapter summarises the major results and conclusions of the project. Potential developments and gaps in the methodology are pointed out and recommendations arising from the project are documented.

Products arising from the project

As a result of this project four major tools have been developed:

- a technique (Q-C modelling) for predicting the resulting concentration of a chemical component for a given discharge;
- a procedure (the "Biotic protocol") for predicting the likely effects of changed water quality on the biota;
- a framework and protocol for incorporating water quality predictions and implications for aquatic biota into the Reserve process;
- a technique (concentration and stress time-series modelling), developed in conjunction with Prof. Palmer at IWR, Rhodes University, for obtaining time-series of chemical concentration and stress, which can be used for comparing the water quality consequences and the implications for the biota, of differing complex flow scenarios.

Other products arising from this project are:

- A literature survey reviewing the links between river flow, concentration and biotic response.
- A training manual for teaching the Q-C modelling method and the Biotic Protocol to water resource managers.
- Spreadsheet templates for Q-C modelling.
- The curated Biological and chemical database (largely a product of the previous project, K5/626 "Water quality requirements for riverine biotas"). A CD of the database ("Biobase") and a User manual are now available from the WRC.
- Reports and conference papers.

Major conclusions arising from this project

The major conclusions arising from the project are detailed below:

- Some form of water quality modelling within Instream Flow Assessments is essential to ensure that in setting the ecological Reserve with regard to quantity, the Reserve for water quality will also be attained.
- In order to make realistic predictions of water quality it is necessary to take into account the various sources of water as well as their pollutant loads that can, or do, contribute to the total flow at a particular site on a river. Thus a consideration of different water quality management (or system operation) scenarios, in addition to flow scenarios, is important in Reserve assessments. At the time of writing this report, this is not a routine step in Reserve determinations. Although, incorporation of management scenarios would greatly increase the complexity of the entire process, it is an essential step if a balance is to be maintained between the use of a water resource on one hand and protection of the aquatic ecosystem on the other.
- No single, water quality modelling method posses all the attributes that are required. Furthermore, the attributes exhibited by the models would not necessarily be required in all situations. Therefore a hierarchy of methods should be employed, with three tiers of modelling complexity:
  - A simple discharge-concentration (Q-C) method.
  - Mass-balance modelling or QUAL2E.

- A catchment runoff model.

- Discharge-concentration (Q-C) modelling is a useful method for estimating the concentration of chemical constituents that will arise from the implementation of a particular discharge regime. It can be used as a screening tool to identify IFR (Instream Flow Requirement) sites where the water quality Reserve is not likely to be attained under the recommended flow regime. There are major limitations in the method, however, and these should be clearly recognised when applying the model.
- Mass balance models (either spreadsheet or QUAL2E) can be used to examine the results of different water quality management scenarios.
- QUAL2E can be useful for providing a more mechanistic description of the processes affecting water quality in a given reach compared to the empirical approach used in Q-C modelling. It has fairly extensive data requirements, however, and frequently additional data will need to be collected. Thus in the context of Reserve determinations it is suitable for use as part of a Comprehensive Reserve Assessment for key IFR sites. It is not suitable for Rapid or Intermediate levels of Reserve determination. This model can be most usefully employed in situations where: there is a complex situation of pollutant loading; when nutrients, dissolved oxygen or temperature need to be modelled; and when different management scenarios need to be assessed.
- Despite the approximations inherent in the derivation of concentration time-series, these can be used for comparing and ranking different flow scenarios as generated by the IFR or yield model (WRYM), with regard to their potential water quality consequences.
- The use of stress time-series shows potential for ranking flow scenarios with regard to likely impacts on the aquatic biota. More research is required, however, to link the concentration of chemical constituents and values of physical variables with biotic response.
- The Biotic protocol can be useful for assessing the likely implications of a proposed water quality scenario for benthic macroinvertebrates. It can be used to identify taxa that may be lost from or regained in a system under a recommended flow regime and to make predictions of the theoretical SASS score and Assessment class likely to be attained.

- From the point of view of water quality, DRIFT (Downstream Response to Instream Flow Transformation) appears to be more flexible than the BBM (Building Block Methodology). The use of a database means that the results from modelling of different water quality management scenarios can be recorded and the optimum scenario chosen. In addition, using DRIFT, the discharge regime can be made available before the IFR workshop so that the water quality consequences (obtained from Q-C modelling), as well as the consequences for the biota (obtained from the Biotic protocol) can be derived. These results can then be used in the workshop to further refine the recommended flow regime.
- The tools developed in the course of this project, although not specifically examined for this purpose, should be useful in a wider field than Environmental Flow and Reserve determinations. Simple Mass balance modelling can be used to predict the instream concentrations of contaminants that would result from the loading of different point sources. The Biotic Protocol could be used to assess the likely impacts of a particular water quality scenario (not necessarily arising from implementation of the Reserve) on aquatic invertebrates, and possibly on other biotic components.

Major recommendations and future research areas

The major recommendations and future research areas arising from the project are detailed below:

- In order to arrive at realistic predictions of water quality, attention must be directed towards incorporation of water quality management scenarios into the Reserve process. In other words it is essential that how the system will be operated, and thus the relative proportions of the different water sources (and constituent loading), be known. Currently the normal practice is to examine the consequences of different flow scenarios at a given point on the river irrespective of considerations of the origins of that water (e.g. from a relatively unimpacted upstream dam or from a polluted tributary). This is adequate if only water quantity is considered but is simplistic if realistic predictions of water quality are required. Simple Mass balance modelling could be used to predict the water quality consequences of each management scenario. Attention should be given to the use of the DRIFT database to store and analyse the results from such a modelling exercise so that the optimum management scenario can be chosen.

- There should be extensive co-ordination between monitoring networks, in particular, between that of water quality and biomonitoring. Collecting measurements of water quality as well as biological data (presence/absence of macroinvertebrate taxa and SASS scores) at the same time and same place, maximises the usefulness of both. This enhanced usefulness is primarily because links between water quality and the presence or absence of specific taxa can then be made.
- In many of the Reserve Assessments encountered in this study, under the recommended IFR flow regime, whilst occasions of very high concentrations of salinity (or other constituent) were avoided, periods of very good quality water would also be removed. This is a consequence of the fact that it is normally during the period of high flow that water would be harvested for other users. The overall result is an attenuation of the current salinity profile so that the extremes of very high and very low concentration would no longer occur. Whilst the removal of episodes of poor water quality is obviously advantageous, the effect of removing periods of "fresh" water is not clear. This matter requires further research and should form part of a detailed study of the long-term effects of implementation of the IFR for key rivers. It should be noted that this is an urgent research need, not only to test the predictions of water quality modelling, and of the Biotic protocol, but to assess the effectiveness of the entire Reserve determination process.
- At present, the form in which water quality data are available in South Africa is not optimal. Water quality simulations as developed in this project (and also for the Reserve) require examination and comparison of segments of the time series for given variables. The relevant data are available in several different forms. Despite this, there is no completely reliable, easily accessible manner that will enable a user to view the entire time series and that will also calculate summary statistics for different time segments. Manipulation of data to obtain the required information can be time consuming.
- Linked water quality and discharge data should be directly available from DWAF. This would avoid errors due to the use of the incorrect rating curve etc.

- *Some statistical aspects of the Q-C method may need refinement. This includes the use of r^2, correlation coefficients and testing the significance of non-linear relationships.*

Conclusion

Several tools have been developed in this project that can be used to integrate water quality and quantity within Reserve assessments. They are complementary and given the assumptions and limitations in the methods, can yield useful information. Prescribing environmental flows for a river, especially systems that are large and complex, is not an easy task. Methods for determining the quantity and timing of flows that are required have evolved in South Africa over several years. The field of water quality lags considerably behind that of quantity and this is the first major attempt to integrate the two fields within Reserve assessments. Development of predictive tools and integration of these into Reserve determinations has been brought about by participation in the process and accompanied by liaison with practitioners in the field of environmental flow assessments. It is likely, however, that much development of the method is still needs to be carried out and feedback is required from water resource managers in order to make the tools as useful and relevant as possible. This is especially as recommended flow regimes are implemented in South African rivers and Resource Quality Objectives need to be met. Despite the approximations and limitations that are inherent in the tools that have been developed, these methods represent an attempt to predict the water quality that will be experienced under a proposed IFR regime, and to assess the implications for the aquatic biota. It is the opinion and hope of the project team that the use of these tools will aid in the integration of water quality and quantity and will result in a more balanced approach to the use of water resources and protection of aquatic ecosystems.

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CONTENTS

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ABBREVIATIONS

ASPT – average score per taxon
BBM - Building Block Methodology
Biobase – Biological and chemical database
C – concentration
CAT-IWR - Centre for Aquatic Toxicology-Institute for Water Research
CD – compact disk
CSIR – Council for Scientific and Industrial Research
DO – dissolved oxygen
DRIFT – Downstream Response to Imposed Flow Transformations
DSS – decision support system
DWAF – department of water affairs and forestry
EC – electrical conductivity
EFA – Environmental Flow Assessment
ERC – Ecological Reserve Category (formerly called ecological management class)
EFR – Environmental Flow Requirement
FRU – Freshwater Research Unit
HSPF – Hydrological Simulation Program Fortran
IBT – inter-basin transfer
IFA – instream flow assessment
IFR – instream flow requirement
LC_{50} – the lethal concentration that corresponds to a cumulative probability of 50% for death of the test population (DWAF 1996)
OREWRA – Olifants River Ecological Water Requirement Assessment
PES – Present Ecological State

Q – flow
Q-C – flow-concentration
PHABSIM – Physical Habitat Simulation system
RC – Reference condition
RDM – resource directed methods
RQO – resource quality objective
SASS – South African scoring system
TIN – total inorganic nitrogen
TP – total phosphorus
TDS – total dissolved solids
TSS – total suspended solids
TWQR – target water quality range
US EPA – United States Environmental Protection Agency
WQOD – Water quality on disk
WRC – Water Research Commission
WRYM – water resource yeild model

CHAPTER 1

INTRODUCTION

1.1 Background

This report documents the results of a three-year study, the aim of which was to develop numerical methods for assessing water quality in rivers, this was with particular reference to the Instream Flow Requirement (IFR) process. Another product of the project is a survey of the relevant literature pertinent to the study which is titled "Linking discharge, water quality and biotic response in rivers: a literature review" (Malan and Day 2002). Key findings and conclusions from that work are included in the present report but the reader is referred to the literature review for details. A training manual for teaching some of the tools developed during this project is also being written.

This project was initiated shortly before promulgation of the National Water Act (1998) and has been carried out during the period of the inceptive Reserve determinations and associated development of methodology for assessing such determinations. The senior author was fortunate to be able to attend Reserve assessments in the initial stages of the project and in the later stages to be able to take part and use the predictive tools developed. Thus, the Reserve Determination for the Mhlatuze River (1998) was attended by one of the project team (HLM) as an observer. Water quality predictions were made using a tool (Discharge-concentration modelling) developed in the project for the Palmiet River (1998) and the Pienaars River (1999). This work was carried out after the Instream Flow Requirement (IFR) workshop however and the results were not utilised in setting the Reserve. The results from Discharge-concentration modelling and the preparation of salinity time-series were utilised and recorded in the four IFR workshops for the Olifants River Ecological Water Requirement Assessment (OREWRA) held in 2000, and the former also for the Breede River Reserve determination (2001). Finally, tools developed as part of this project will be utilised in the IFR determination for the Thukela Water Project to be held in 2002.

This document is divided into the following chapters:

- **Chapter 1:** This chapter presents general background information on several relevant topics. These include; the new South African Water Law and the ecological Reserve, the variability of South African rivers with regard to discharge and water quality and the need for the incorporation of quantitative predictions of water quality into Environmental Flow Assessments. The objectives of the project are also listed in this section, as well as a summary of methods used elsewhere in the world to predict water quality, and to assess the effect of changed water quality on the biota. The attributes that are required in the chosen modelling method are listed in this chapter. Finally, the elected models are presented.
- **Chapter 2:** A detailed description of the Discharge-concentration (Q-C) modelling method developed in the project for predicting chemical concentrations is reported in this chapter. Results from application of the method in various IFR determinations; information that can be obtained using Q-C modelling and most importantly, assumptions and limitations in the method are also discussed.
- **Chapter 3:** This chapter reports the results from a countrywide survey of the discharge-concentration patterns for different DWAF primary drainage regions, as well as for different ecoregions. An attempt is made to predict likely patterns for certain water quality variables in a given area.
- **Chapter 4:** In this section of the report, a description of the method used to obtain time-series of chemical components (notably salinity) and time-series of water quality-induced stress that will be experienced by the aquatic biota is presented. An application to the Olifants and Breede Rivers, as well as the assumptions and limitations in the method, are reported. Finally, the usefulness and ecological interpretation of such time-series is discussed.
- **Chapter 5:** This chapter describes an application of the water quality model QUAL2E to the Lower Olifants River. Included, is a detailed discussion of the process and problems encountered. In addition, the use (including the advantages and disadvantages) of this modelling technique within the Reserve determination process is discussed.
- **Chapter 6:** This chapter presents the Biotic Protocol. This is a process that enables predictions of the effects that alterations in water quality may have on the biota to be made. The protocol is described in detail, followed by the results from an application

to the Middle and Lower Olifants River. Assumptions and limitations in the method are discussed, as well as future developments that are required.

- **Chapter 7:** Integration of water quality modelling and predictions of the effect of changes in discharge, on water quality and hence on the aquatic biota into the Reserve determination process, is the major consideration of chapter 7. In particular the role and use of the tools developed in this project are discussed. Finally a framework for incorporation of predictions of water quality changes, and the implications for the biota, into the Reserve determination process is presented.
- **Chapter 8:** The final chapter summarises the major results and conclusions of the project. Potential developments and gaps in the methodology are pointed out and recommendations arising from the project are documented.

1.2 Aims and objectives

The aims of the project are partly to follow up on aspects of previous projects on water quality (aims A and B) **but mainly to develop a new research area (aim C).** The extent to which each of the objectives has been achieved is discussed at the end of this chapter.

A) Pursue the use and curation of the Biobase (Biological-Chemical database) developed in project no. K5/626:

- Curate the database and capture additional data, as and when the need arises.
- Investigate future maintenance and accessibility of the Biobase, including the potential to provide an Internet facility for extracting and adding new data.
- Continue to interrogate the database with a view to:
  - verifying the scores recommended for the biomonitoring system SASS.
  - identifying water quality tolerances of key taxa of invertebrates for use in water quality assessments for the IFR process.
  -verifying the methodology for establishing regional water quality guidelines for selected constituents as proposed in project K5/626.

B) To continue to examine, in a low-key manner, the extent to which water quality determines the distribution of south-western Cape endemic invertebrates.

C) To address the question of the relationship between water quality and water quantity (particularly discharge), with particular reference to the assessment of Instream Flow Requirements:

- To provide a review of the kinds and forms of data needed and of methods developed elsewhere for providing these data.
- To develop methods for processing water quality data to derive time and flow dependencies for key water quality constituents.
- To identify and assess methods for modelling selected water quality constituents at the feasibility level of the IFR process.
- To investigate the extent to which available data (e.g. in the Biobase) can be used for setting environmental water quality requirements as part of IFR assessments.
- Based on the above, produce a framework for the assessment of water quality as part of IFR studies.

1.3 Rationale for the project

Water is a limited resource in South Africa and protection of our aquatic resources is of prime importance. Rivers provide many "goods and services", can perform a self-cleansing function and, if protected, can replenish the resource (Dobbs and Zabel 1994; Davies and Day 1998). It is perhaps not yet sufficiently widely realised that, if rivers are subjected to environmental impacts such as, the discharge of pollutants, invasion of riparian and aquatic habitats by alien plant species, or excessive water abstraction, impaired ecosystem functioning will result in a loss of some desirable attributes. Water quality (i.e. the instream concentration of chemical constituents and values of physical variables) is then often impaired and the potential uses of the water resource become increasingly limited. This is the pragmatic reasoning behind the emphasis on resource protection in the new South African Water Act (Act No. 36, August 1998). A fundamental principle of the Act is that, the quality, quantity and reliability of water required to maintain ecological functioning shall be maintained, in order that human use of water does not compromise the long term sustainability of aquatic ecosystems.

The act also states that "as soon as reasonable after the act comes into force, the Minister must determine the Reserve for all, or part of each significant water resource". The Reserve is defined as:

- The quantity and quality of water required to supply the basic needs of people who are, or may in the future be, reliant on a particular water resource (the "basic human needs Reserve").
- It is also the water *quantity* and *quality* required to protect ecosystems in order to secure ecologically sustainable development and use of the relevant water resource (the "ecological Reserve").

In the rest of this document, unless otherwise stated, the term "Reserve" refers to the "ecological Reserve". Although the new act recognises that water quantity and quality are interdependent and should be managed in a co-ordinated manner (Rabie 1996), how the two are to be integrated is not detailed. It should be noted that the water representing the Reserve has priority, and the quantity and quality of this water must first be calculated before the quotas for other water users may be allocated. It is therefore imperative that the Reserve be determined as quickly as possible. Different levels of accuracy of the Reserve (including water quality) can be determined, in which a varying degree of confidence can be placed. These levels range from a Rapid determination to full Comprehensive Reserve. A Rapid determination involves a desktop assessment of water quality, largely using existing data, and a quick field assessment of the present status. Such an estimate can be used for licensing purposes in unstressed catchments, or where the impact is likely to be small, but has to be reviewed after a certain time period. A Comprehensive Reserve determination, on the other hand (in which relatively high confidence can be placed), involves a site-specific assessment of the instream flow requirement determined by specialised riverine ecologists at a workshop. Additional water quality data may be collected if required for arriving at suitably reliable results (DWAF 1999).

The ecological component of the Reserve is considered to consist of four aspects, namely; water quantity, water quality, habitat (both instream and riparian) and the aquatic biota, all of which need to be assessed. Water quality is assessed for each significant resource in terms of each of three categories of variables, namely toxic substances, system variables and nutrients. For each resource, the levels of each water quality variable in the un-impacted condition (Reference condition) are derived. These values are compared with the Present Ecological State values in order to assess whether the river is degraded with regard to water quality. These values are also compared to those associated with the future Ecological Category (where an "A category river" represents one in a pristine condition and a "D category river" one which is highly modified) in order to identify what management actions are required. The Reserve process and incorporation of water quality therein is discussed fully in the manual titled "Resource directed measures for protection of water resources" (DWAF 1999). A useful overview of the process is to be found in O'Keeffe (2000).

South African rivers are naturally variable in their patterns of discharge and frequently stream flow varies substantially from season to season and from year to year depending on the balance of evaporation and precipitation. In addition, abstraction of water, as well as regulation by means of impoundment or weirs, can lead to alterations in the natural hydrological regime. It is well known that changes in discharge can have a profound effect on water quality (Edwards 1973; Walling and Foster 1975; Sidle 1988). Such changes in water quality are often complex and difficult to predict, however. Because fresh water is a limited resource in this country, pollutants cannot always be dealt with by dilution, as is the case in many more mesic or well-watered lands. Due to high rates of evaporation, pollutants can become concentrated in rivers during periods of low flow. For this reason it is especially important that the water quality of aquatic resources is monitored and assessed. Despite this, there is a paucity of documented information concerning the resultant effect of changes in discharge (either natural or anthropogenic) on water chemistry and on the values of system variables in this country. It is also well known that water quality can exert a profound effect on the composition, abundance and functioning of the aquatic biota (Dallas and Day 1993). Once again there are few data on the tolerance ranges of aquatic biota in South Africa to individual water quality variables and the effect of such variables on biotic functioning.

Methods have been developed to calculate the quantity and timing of discharge required for adequate functioning of riverine ecosystems, currently known in South Africa as the "Instream flow requirement" of a river or IFR (King and Louw 1998). This method is widely used in the planning and operation of large water supply projects. The question of water quality at the beginning of this project was inadequately addressed within the IFR process, however, and for a proposed change in discharge, only descriptive predictions of the resulting water quality were usually made. According to King and Louw (1998), provision of a suitable discharge regime will not result in optimum ecosystem health if the question of water quality is not also considered. Thus it is essential that quantitative predictive methods of water quality be incorporated into the IFR process. This will ensure that in setting the quantity Reserve for a particular water resource, the water quality of the Reserve will also be attained.

1.4 Terminology

In this document, the term "discharge" is used to describe the volume of river flow passing a given point per unit time (e.g. $m^3.sec^{-1}$). A note should also be made at this point to clarify the terms "Instream Flow Requirement (IFR)", "Environmental Flow Requirement ([1]EFR)" and determination of the "ecological Reserve", which are all conceptually similar, but differ slightly in meaning. An assessment of the IFR or EFR for a river involves determining the discharge regime required to maintain some aspect of the riverine environment. In line with the trend worldwide it is more appropriate to use the latter term (R. Tharme, FRU, UCT, *pers. comm.*), because a wider area, such as the river banks, than merely within the stream is considered. Such discharges however, are still referred to as "instream flows" in South Africa. Thus, when referring to the general methodology or philosophy for assessing such discharges, the term "EFR" is employed in this document, but if the reference pertains to the process in South Africa, "IFR" is used. A more thorough discussion of environmental flows and the terminology used to describe them is given in Chapter 5 of the literature review (Malan and Day 2002). In a similar vein, it is recognised by the authors that the term "Ecological Reserve determination" rather than "Instream Flow Requirement" may possibly be more correct in the project title. Where IFR refers to determination of the discharge regime required, but the ecological Reserve refers to the far wider process of setting of Resource Quality Objectives, management classes, stakeholder participation etc, of which setting the IFR is a smaller, but vital part. The authors have chosen to retain the original title however,

[1] EFR: This is not to be confused with Estuarine Flow Requirement, not referred to in this report, but also frequently abbreviated as EFR.

and in this document, the terms IFR and Reserve will be used in the sense just described.

The term "water quality modelling" in the context of resource management, often (but not always) entails studying water quality data in relation to discharge and deriving relationships which enable a realistic water quality value to be simulated corresponding to each chosen discharge value. In this document the term "*water quality modelling*" is used to describe techniques employed to obtain *quantitative predictions* of what the concentration of chemical constituents in a given river reach would be under given conditions of discharge (e.g. a proposed IFR discharge regime).

1.5 The water quality variables considered

A description of the most important water quality variables, as well as the main interactions between them, is given in the literature review. Only the salient points are presented here.

For the purpose of this study, definitions of the terms *water quality* and *water quality constituent* are taken from the South African Water Quality Guidelines (DWAF 1996). *Water quality* refers to the "physical, chemical, biological and aesthetic properties of water that determine its fitness for a variety of uses and for the protection of the health and integrity of aquatic ecosystems." The range of optimum values for the various water quality constituents will depend on the proposed use of the water. In the present project we are largely (although not exclusively) concerned with assessment of water quality necessary for protection of aquatic ecosystems. The term *water quality constituent* is defined as "any of the properties of water and/or the substances suspended or dissolved in it."

The following water quality constituents were considered in this project:

- **System variables** - temperature, **pH,** dissolved oxygen (DO).
- **Non-toxic inorganic constituents** - total suspended solids (TSS), **total dissolved solids (TDS), individual ions such as K^+, Mg^{+2}.**
- **Nutrients - phosphates (ortho-phosphate and total phosphate), inorganic nitrogen (nitrate, nitrite, ammonia/ammonium).**

- Toxins and pollutants – e.g. metal pollutants, organic substances including pesticides, whole effluents.

The water quality constituents shown in bold are those that are deemed to be most important in this project and largely include those considered in the water quality assessment component of the Reserve. Whilst the other variables are also important, there is usually a lack of data concerning them, making predictions of their behaviour under conditions of altered discharge problematic. Note that in the Reserve methodology, TDS and TSS are included as system variables rather than as non-toxic inorganic constituents (DWAF 1999) and that ammonia can also be considered to be a toxin under certain conditions.

The category "toxic substances" represents a very diverse group of substances. These chemicals frequently enter water bodies as point-sources and are best managed at the site of entry into the water resource. Toxic substances such as agricultural pesticides, which originate from diffuse sources, are of special concern since they are difficult to exclude from aquatic ecosystems. This project is largely concerned with predicting concentrations of water quality constituents as a result of changes in discharge. In order to do this data are required concerning the variable. There are limited long-term data concerning toxic substances in South Africa (Pegram, Quibell and Görgens 1997) and for that reason this topic will not be given the attention in the current work that it deserves. Other issues (for example TDS and nutrients) are more pressing and will be dealt with here. Nevertheless, pollution due to metals and pesticides is becoming an increasingly serious problem in our rivers and will need to be modelled at a later date.

From the point of view of water quality modelling, it is necessary to distinguish between conservative and non-conservative constituents. Conservative chemical constituents (e.g. TDS, Cl^-) do not normally undergo chemical transformation in their progress along a watercourse. Non-conservative constituents on the other hand (e.g. nutrients, organic carbon, dissolved oxygen), are taken up by living organisms, and may undergo a wide range of chemical reactions. A modelling exercise aimed at accurately simulating a non-conservative constituent must account for all the processes that will affect the concentration of that constituent. Modelling non-conservative chemical constituents is therefore more challenging than modelling conservative constituents (Dortch and Martin 1989).

1.6 Factors determining water quality

1.6.1 Some common discharge-concentration (Q-C) patterns

Concentrations of different chemical constituents frequently vary with discharge in a characteristic manner. The mathematical relationship between discharge and solute or sediment concentration is commonly called a rating curve and is site-specific (Sidle 1988). Predictions of such a relationship for individual ions may be made for a given catchment from a consideration of the climatic region and catchment land-use, as well as from the underlying geochemistry. Discharge-concentration (Q-C) relationships are complex, however, and influenced by a wide range of factors, whilst some water quality constituents may show little apparent variation with discharge (McDiffett, Beidler, Dominick *et al.* 1989). Factors influencing these responses include antecedent rainfall patterns; the geology and climate of the surrounding catchment; the hydrological regime; season; and anthropogenic effects such as pollution, land-use and construction of impoundments and weirs (Johnson and East 1982; Davis and Keller 1983; Williams 1989). A more detailed discussion of the factors affecting discharge-concentration relationships is given in the literature review (Malan and Day 2002).

An attempt was made (Malan and Day 2002) to summarise, as reported in the literature, the various trends exhibited by chemical constituents and physical variables in response to increased discharge. The expected complexity and variability in these trends was confirmed and the authors show that clear patterns were not always easy to discern. A summary of typical discharge-concentration trends derived from these papers is given below. Nevertheless, because of the variability in response, predictions of stream chemistry resulting from changes in discharge should be made with caution and require verification with field data.

- **Suspended sediments** generally increase with discharge but the rate of increase may level off at high discharge as the amount of erodable substrate becomes limited. Thus, due to limitation in the supply of the transported material, storms occurring early during the wet season are likely to carry heavier loads of sediments compared to storms occurring later in the season.

- The concentrations of **inorganic ions** derived from the underlying substrate are likely to decrease as discharge increases (Fig. 1.1 a). This decrease is a result of the fact that river water during baseflow is usually relatively rich in solutes, since it is derived from groundwater. During spates, this groundwater is diluted by water originating from interflow and surface flow. Such water resides for a shorter length of time in the catchment and hence the period in which dissolution can take place is reduced (Edwards 1973).
- The effect of increased discharge on **total dissolved solids (TDS)** is difficult to predict, reflecting as it does the sum of effects on the minor ions as well as the major ions. Due to the high rate of evaporation in this country, in addition to the trends in inorganic ions noted above, in non-impacted catchments TDS is likely to be maximal during periods of low flow and minimal during periods of high flow (Fig. 1.2 b). In urban, or polluted, areas however, or where surface wash-off of material is likely to be substantial, such a response may be obscured. For example, salinised catchments in the winter-rainfall region (and possibly in the summer-rainfall parts of the country as well) seem to represent an important exception to the general dilution effect (section 2.11). In such catchments at the beginning of winter, TDS may increase with increased discharge due to the wash-off of accumulated salts from the soils of the surrounding catchment.
- In some circumstances **pH** is likely to decrease during storm events. This has been attributed to the flushing of organic acids from the soil of the surrounding catchment (Lawrence and Driscoll (1990) citing the work of Johnson (1969) and Bishop, Grip and O'Neill; 1990) and the dilution of alkaline components. Such an effect is especially prevalent in the south-western Cape where the dominant vegetation in non-impacted catchments is fynbos. This variable is likely to decrease in Cape rivers in autumn (due to a build up of organic acids in the soil during the long dry summer) but may increase during the latter part of winter due to a dilution effect (resulting from depletion of acids in the soil). Increased acidity during high flow events is also to be expected in other parts of South Africa although the effect may not be as pronounced as in the south-western Cape.
- **Total phosphorus** often increases with discharge (Fig. 1.2 c). A large portion of the total load of inorganic phosphorus is bound to sediments and a smaller fraction dissolved in the water column (Holtan, Kamp-Nielsen and Stuanes 1988). During high-flow events, elevated loads of sediment and consequently, of phosphates, enter

the watercourse due to erosion of the surrounding catchment. Even in the absence of rain, an increase in discharge, due to releases from an upstream impoundment for example, may result in increased phosphate loads. This is as a result of the churning of benthic sediment with the concomitant release of phosphate from the banks and river-bed (Verhoff *et al.* 1982). The Q-C plot in Figure 1.1 shows the concentration of total phosphorus first increasing rapidly with increased discharge and then reaching a plateau phase. At even higher discharges, unless the loading of this constituent is excessive, the concentration of total phosphorus will eventually decrease as dilution becomes the dominant process.

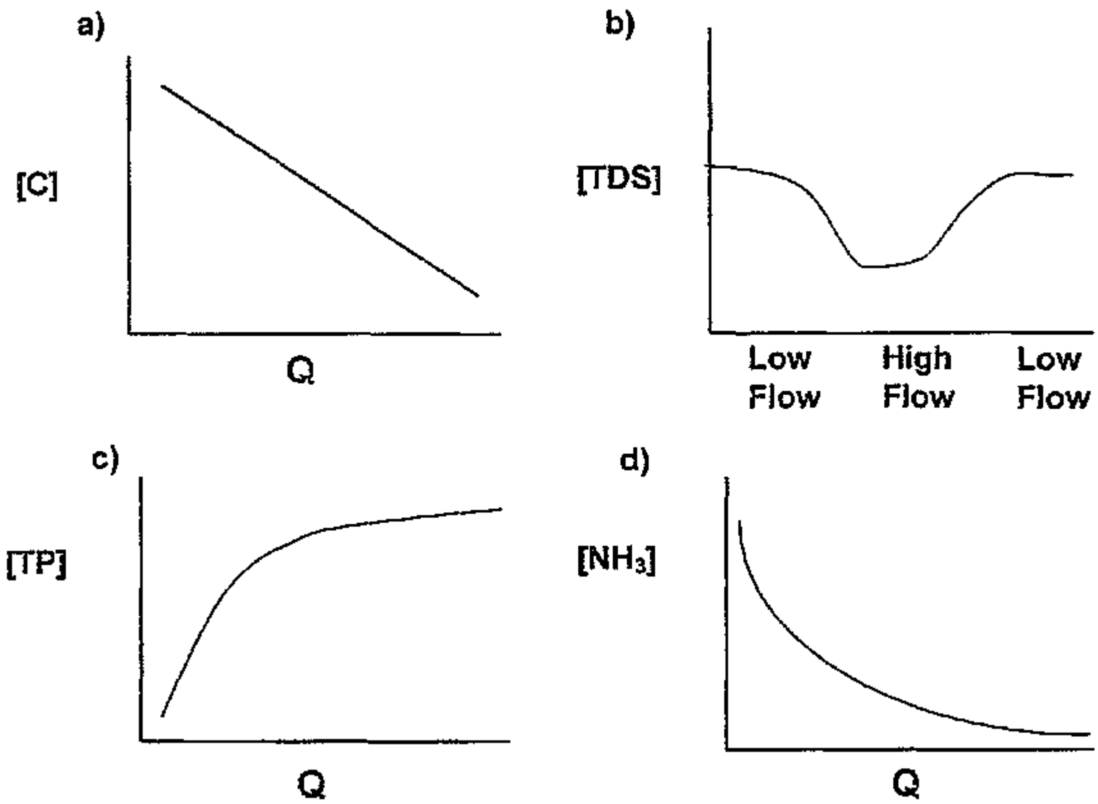


Figure 1.1 *Characteristic patterns of change in the concentration of selected water quality variables with increasing discharge (Q).* ***a)*** *A conservative inorganic ion (C).* ***b)*** *Typical changes in TDS concentration throughout the hydrological year.* ***c)*** *Total phosphorus.* ***d)*** *Un-ionised ammonia.*

- The amount of phosphates entering the stream will, to a large extent, be influenced by the nutrient status of the soil. Intensive land use and agricultural activity lead to enhanced phosphate runoff (Houston and Brooker 1981). In the absence of point sources of pollutants, as discharge is increased, **dissolved phosphate (ortho-phosphate)** is likely to decrease or remain constant in oligotrophic systems due to rapid uptake of this nutrient by the aquatic biota (Dallas, Day and Reynolds 1994). In urban areas, or regions of intense farming activity, however, this trend may well be reversed due to wash-off effects of pollutants or phosphate fertilisers. If point sources of phosphate are actively discharged, the overall trend will depend on the proportional contributions from each source. Dissolved phosphate levels may well increase during low flow periods as the proportion of effluent to river water increases.
- Due to its high degree of mobility in the soil, **nitrate** is likely to increase during storm events, or during the initial part of the rainy season. Depending on the nutrient status of the soils in the surrounding catchment, such a flushing effect may be sustained in urban areas, or in regions of intense agricultural activity. In nutrient-poor soils such as fynbos, the flushing effect may be short-lived being followed by rapid assimilation of nitrates by aquatic organisms.
- **Un-ionised ammonia** also frequently exhibits a rapid decrease with enhanced discharge (Fig. 1.2 d), a consequence of increased dilution. In addition, ammonia is rapidly taken up by the biota as the riverbanks become inundated and is sequestered within these organisms or is transformed to other forms of nitrogen. Such processes lead to a reduction in the ammonia content of stream water.

To conclude, water quality is dependent on processes taking place in the entire catchment, which are often poorly understood. These processes are often conflicting and frequently result in responses of water quality variables that are difficult to rationalise or to predict. In addition, as river catchments become increasingly developed, the effects of point and diffuse sources of pollution are likely to mask the natural cyclic patterns in aquatic ecosystems to a greater extent.

1.6.2 The effect of impoundment on water chemistry

In South Africa, the discharge from a large proportion of our rivers is regulated, and an increasing proportion of the mean annual runoff (MAR) is stored in impoundments. Consequently discharge is controlled by means of these structures. The chemistry of the water flowing out of an impoundment is to a large extent dependent on processes occurring within the reservoir and the interaction of such processes and driving forces which determine water quality is complex (Bath, de Smidt, Görgens *et al.* 1997). As each reservoir has its own combination of biological, physical, chemical and hydrological characteristics, the effects of reservoirs on the downstream reaches of a receiving river are variable (Webb and Walling 1997). The combined effect of South African climate and morphology results in many impoundments that are stratified for almost nine months of the year (Bath *et al.* 1997). The volume and depth of the body of water will determine thermal and chemical stratification. Water may be released from the bottom (hypolimnetic discharges), or top (epilimnetic discharges) or in the case of more modern reservoirs, from several strata by means of multi-level offtake towers (Petts 1989). Thus, the release mechanism and management of the impoundment is pivotal in determining the effect of reservoir releases on downstream water quality. Because of thermal stratification, water released via hypolimnetic discharge is often cool, deoxygenated and laden with nutrients (especially ammonia) compared to water from epilimnetic or top-releases (Hart and Allanson 1984; O'Keeffe *et al.* 1996). One of the most important effects of impoundment on river chemistry is the reduction in transport of suspended sediments, which may be especially marked in the absence of hypolimnetic releases. The fact that impoundments may act as "settling ponds", trapping sediments, also profoundly affects nutrient transport. In particular, the transportation of phosphorus, especially that portion associated with sediments, is usually also curtailed by these structures (Pedrozo and Bonetto 1989). Impoundment of rivers also tends to result in a reduction in seasonal amplitudes not only of temperature but also of other water quality variables (Hart and Allanson 1984). For example, frequently there is an attenuating effect on natural salinity ranges so that the peak values of natural discharges are reduced. On the other hand the natural flushing effect of flood flows that once "cleaned" out the system are now replaced with flows of higher salinity. The resulting effect downstream is provision of water with a smaller annual range but generally higher average electrical conductivity value.

It should be noted that in regard to the objectives of this project, in many ways the presence of impoundments on a river course complicates an already complex picture. This is largely because the effect of an impoundment on a given water variable is difficult to predict. In highly regulated systems, stream water chemistry is frequently dominated by that of upstream reservoirs so that it may be difficult to discern trends in Q-C relationships. A further consideration is that in catchments where development and environmental degradation have resulted in moderately to severely impacted rivers, regulation by means of impoundments may have an ameliorating effect on downstream water quality, due to the retention of sediments and nutrients. The effect of impoundment on water quality is discussed more fully in Chapter 3 of the literature review (Malan and Day 2002).

1.6.3 The effect of inter-basin transfers (IBTs) on water quality

Inter-basin transfer schemes (IBTs) can be defined as "..the transfer of water from one geographically distinct river catchment or basin to another, or from one river reach to another....." (Davies, Thoms and Meador 1992). In essence, the effect of water transfer from a donor water body to a recipient system, on the water chemistry of the latter, depends on the magnitude of the difference between the two. This is both in terms of the concentrations of chemical constituents (e.g. TDS), as well as with regard to the given physical variables (e.g. temperature). The amount of water transferred and the proportion this represents of the total discharge in the recipient system is also important. The effect of IBTs on water quality is discussed in Chapter 3 of the literature review (Malan and Day 2002), and more comprehensively in Snaddon, Davies and Wishart (2000).

1.7 The general approach used

The relationships between discharge (Q), water quality (WQ) and the aquatic biota are summarised in Fig. 1.2. The scope of the study is also indicated, as well as the tools employed or developed during the project to quantify each relationship.

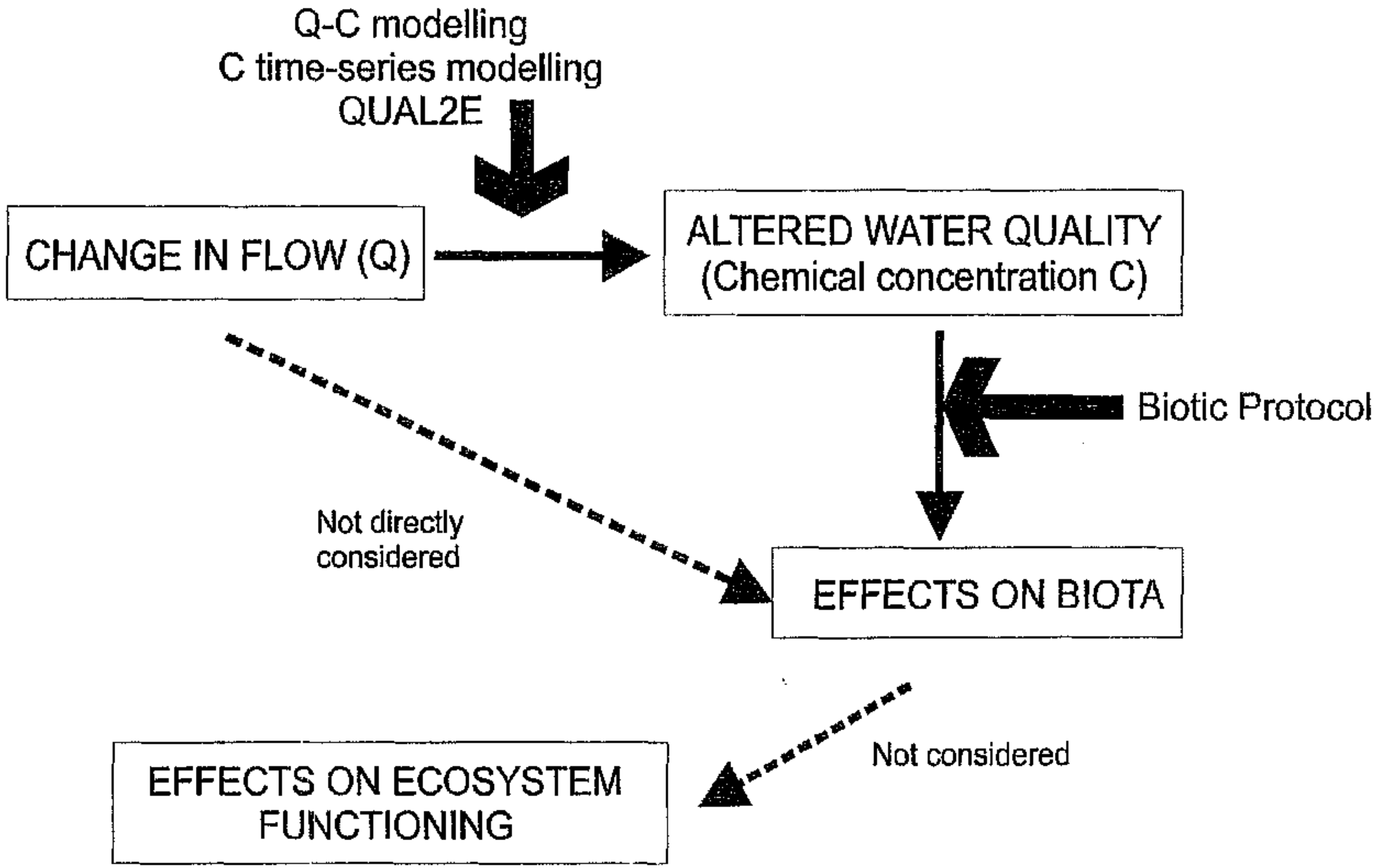


Figure 1.2 *A summary diagram of the most important relationships addressed in the study and some of the tools used, or developed, to predict them.*

As can be seen from the diagram, this project is composed of two major sections: a) development of methods to predict the concentrations of given chemical components and values of physical variables, for proposed discharge reduction (or augmentation) scenarios in a given river reach and b) predictions of the implications such changes in water quality may have on the abundance of key invertebrate species and thus on the ecological functioning of the aquatic resource. The direct effect of change in discharge on the biota was not considered in this project to any great extent, neither was the effect of reduced or altered species composition and abundance on ecosystem functioning. Another aspect that was not covered in this project was the effect of altered water quality on faunal groups, other than invertebrates. There is scope however for the "Biotic Protocol" to be adapted to other sectors of the biota (Chapter 6).

Part One of the project, which involves predicting water quality for a given (proposed) discharge regime, was carried out using the steps listed below:

- i) A review was undertaken of water quality modelling methods (both complex computer programs as well as simple mathematical methods) that are available internationally. This is discussed in section 1.8, as well as, more comprehensively in Chapter 4 of the literature review.
- ii) The completeness, availability and form of discharge and water quality data available in South Africa were assessed (section 1.8.1).
- iii) Consideration of the attributes the predictive method should possess (section 1.8.2).
- iv) From consideration of the steps above, the modelling method(s) most suitable for the project were chosen, and are discussed in section 1.8.3.
- v) The chosen modelling methods were then used in actual Instream Flow Assessments in South Africa and were continually refined. These applications are presented in Chapters 2 and 5 of this report.

Part Two of the project is concerned with linking the water quality tolerance ranges of selected biota with predicted constituent concentrations and values of physical variables. Thus, after using numerical modelling methods to predict the level of a water quality variable in a reach under a given discharge regime, the effect on the biota must be predicted. The following steps were taken in order to address this part of the project:

- i) The methods used elsewhere to assess the implications of altered water quality for the biota were reviewed. This is discussed in section 1.9, as well as more comprehensively in Chapter 6 of the literature review.
- ii) An assessment of the availability of data and information regarding tolerance ranges of indigenous aquatic biota was undertaken. Discussed in section 1.9.4.
- iii) A predictive tool was then developed.
- iv) Application of the predictive method to Instream Flow Assessments (IFAs). Both steps iii) and iv) are discussed in Chapter 6 of this report.

During the entire project, integration of the tools, or predictive methods into the Reserve determination process was considered. Specialists in other countries were consulted with regard to the extent and method of incorporation of water quality considerations into environmental flow determination assessments in their respective areas. On-going liaison with the relevant specialists in this country was undertaken in order to establish exactly what was required. Finally a framework for incorporation of water quality considerations into the process was produced. These aspects are presented in Chapter 7.

1.8 Methods used to derive Q-C relationships

A full description of the methods that have been used elsewhere in the world to predict the relationship between water quality and discharge is given in the literature review (Malan and Day 2002). Also included in the review are definitions of some of the terms used in water quality modelling, the types of models available and how to chose an appropriate one. An outline of the generalised modelling method is given, as well as examples of how to model specific water quality variables. Therefore only a summary of the type of modelling approaches available and some of their advantages and disadvantages is given here.

Many water quality models have been developed to assess the influence of changes in discharge on water chemistry and a fairly wide range of modelling methods have been used, from simple methods (rating curves, simple mass balance modelling) to complex computer models (Gregory and Walling 1973; Dortch and Martin 1989). Modelling approaches include rule-based, deterministic and stochastic methods. Under each of these generic headings however, there is a range of models from those that are simple, and can be applied using spreadsheets to those that are complex and employ sophisticated programming routines. Computer models include empirical and mechanistic models, as well as those that simulate steady-state and dynamic conditions of discharge and the values of water quality variables. Computer models also extend from those that simulate instream changes in various water quality constituents, to sophisticated catchment runoff models. This second type of model (also termed watershed models) simulates the hydrologic processes by which precipitation is converted to streamflow and the resultant water quality (Wurbs 1995). In the case of instream models, only processes occurring in the river are considered, whereas in catchment runoff models, factors arising in the catchment are also taken into account. Catchment runoff models are especially useful in assessing the impact of non-point sources on water quality. A comprehensive review of methods for assessing the effects of non-point sources is given in Pegram and Görgens (2001). In addition, this type of model is useful for assessing the effects of change in land-use on water quality. (Wurbs 1995).

The various modelling methods have associated advantages and disadvantages (Crockett 1994). Simple mathematical techniques, such as the rating curves mentioned above, or mass balance modelling, are quick to set up, simple to use and do not require extensive data. They suffer from the disadvantage, however, of being not particularly accurate, nor suitable for simulation of temperature or DO. Commercially available software on the other hand, whilst tending to be more time, labour and data intensive, as well as sometimes requiring specialist knowledge to set up and run, are frequently more accurate and allow complex systems to be simulated. Steady-state models are usually simpler than dynamic models, and instream models are less demanding in data and time than catchment run-off models. From the above, it is apparent that in choosing a modelling method, a balance between the accuracy of the output, and the data and time required to set it up, often needs to be struck. This is considered further in section 1.8.3, which describes the choice of the modelling methods used in this project. A list of essential, or useful, attributes that the modelling method should possess was drawn up by the project team, and is presented in section 1.8.2.

1.8.1 Hydrological and water quality data available in South Africa

The South African Department of Water Affairs and Forestry (DWAF) has an extensive database (the Hydrological Information System (HIS)) which includes data concerning discharge as well as many water quality variables. Such data are available in several different forms (see below) which were assessed for their completeness and suitability of use in this project. In addition, in some catchments, data are collected by individual water boards (e.g. Umgeni Water, Rand Water). Such sources of hydrological and water quality data were not examined in this project, however. The reason for this was either because the Instream flow assessment under consideration was not undertaken in such a catchment, because data from DWAF were found to be adequate or, in the case of discharge data, could be simulated using appropriate hydrological models.

Because of the scarcity of water in areas of South Africa, monitoring of water quantity has been undertaken for several decades and extensive hydrological records are available for over a thousand gauging stations. In addition, rainfall has been measured in some areas of the country since the early part of the last century and a reasonably comprehensive monitoring network is in place (Midgley, Pitman and Middleton 1994). Rainfall data can be used as input in catchment runoff models to generate estimates of river flow. The discharge data in the Surface Water Resources of South Africa 1990 (Midgley *et al.* 1994) have been generated in such a manner. According to Hughes (2000) the above work represents the most complete reference source for natural streamflow characteristics in this country. Observed discharge data are available in several forms from DWAF including as daily or monthly averages. In addition, rating curves for each gauging weir are available. From this, water height at a given point in time at the gauge can be converted to discharge.

Water quality samples, on the other hand, have been routinely collected and analysed only since the late 1970's and even more recently in some catchments. The frequency of data collection varies extensively from site to site, and also sometimes at a site, being weekly, fortnightly, monthly or even less frequent. There is often the situation where adequate discharge data are available but insufficient water quality data. At a limited number of sites in this country, continuous data (pH, DO, temperature and conductivity) are available (F. Mouski, DWAF Western Cape, *pers. com.*) although such data were not examined in this study.

The data in the HIS database are also limited with regard to which water quality variables are covered. Measurements of DO, temperature and toxic substances are generally lacking. The availability of data on sediments and turbidity (in addition to pathogens, metals and other constituents) is highly uneven and where they are available there are often missing values (Pegram and Görgens 2001). One of the reasons why monitoring of TSS is not carried out on a routine basis in South Africa is, that turbid rivers are common in this country, and are not considered to be problematic (Dallas and Day 1993). In addition, high TSS levels are frequently associated with high run-off events and are therefore episodic and not easily monitored. Nevertheless, suspended sediments can have a profound both on water quality as well as on the aquatic biota (Sweeting 1994). It is therefore important that this variable should be measured in order to be able to predict changes in water quality and ecological functioning in a comprehensive manner.

Depending on how recently the database has been updated, values of all water quality variables for the past year may not be available. Samples of water for chemical analysis are usually taken only when water is flowing in a river. Impacts on water quality may be most severe during periods of interrupted flow, or when a river changes from a perennial to a seasonal system (Pollard, Weeks and Fourie 1996).

Water quality data and discharge data can be requested directly from DWAF via email (see "Useful web site addresses" at the end of this document). Such information usually arrives within a few days. The Institute for Water Quality Studies (IWQS), a directorate of DWAF, also maintains a web site in which the data are presented in the form of graphs and as statistical summaries. This was found during the project to be useful for examining trends in data (either seasonal or over the time-period of data collection). This data source suffers from the disadvantage that data can be viewed but not downloaded. It also suffers from the disadvantage that web sites can be unavailable due to malfunction, or slow due to too much traffic. At the time of writing this report the future of this web site was uncertain (M. Silberbauer, IWQS, DWAF, Pretoria, *pers. comm.* 2001). DWAF water quality data are also available on a CD titled "Water Quality on Disk"

(WQOD) available from the CSIR (Contact person Derek Hohls; see "Useful web sites"). After initial problems, this data source is now available and can be ordered from the CSIR web site at a reasonable price and will be updated regularly (Hohls, D. CSIR *pers. comm.* 2001). It was found that water quality data could be fairly easily down-loaded from the CD into a commercial spread-sheet or statistical package (Microsoft Excel and Statistica were the ones used in this project) and further manipulated from there. Care must be taken, however, to ensure that missing data (i.e. sampling not undertaken for a particular date) are not recorded as a concentration value of zero. Because the data are immediately available and because of the "user-friendliness" of the software, WQOD was the data source most used in this project. As well as the usual water quality variables, WQOD also includes gauge height. Using this value and the appropriate rating curve from DWAF (see above), river discharge at the time of collection of a particular water quality sample can be derived, although it can be somewhat laborious to do so. Moves are afoot within DWAF to make linked water quality and discharge data accessible (M. Silberbauer, IWQS, DWAF, Pretoria, *pers. comm.* 2001), but at the time of writing this report such data were not available.

Finally, DWAF water quality data are available in another form, namely as the software package "Watermarque". This is written in Unix and is available on Sun workstations. Since one of the aims of this project was to develop a Q-C prediction method that was user-friendly and widely available, this avenue of information was not pursued further. Effluent discharge data are stored by DWAF as part of the POLMON database. Such data were however unavailable during this project.

1.8.2 Attributes required of the modelling method

A list of the attributes that the modelling method(s) chosen for this project should exhibit is given below. The list is divided into important attributes that should be available and useful additional attributes.

Important attributes

The modelling method should have most of the following attributes.

- It should be able to predict the concentration or value of most of the following water quality components with regard to change in discharge: TDS, temperature, nutrients (nitrates, ammonia, soluble phosphate) and possibly TSS, and DO.
- It should be flexible and be able to handle a variety of "flow-modifiers" (i.e. causes of changed discharge, such as abstraction, inter-basin transfers, weirs and impoundments). Different reservoir management options should be handled either by the model itself or by being compatible with or interfacing with other models that can for instance handle the problem of multi-level outflow from stratified impoundments. Thus the modelling method should be applicable to both regulated and unregulated rivers.
- It should have a proven track record in a climatically similar country, if not in South Africa itself. Expertise and support to run the model should be readily available and the model should be well documented.
- It should be compatible/readily incorporated into the Building Block Method for IFR determinations and thus should have appropriate spatial and temporal detail. It should also ideally be compatible with DRIFT, another developing Instream Flow Assessment method (Chapter 7).
- The data input requirement (both format and amount) should be compatible with the data that are likely to be available.
- It should be able to work on a PC and not require a specialised platform such as a UNIX work-station.
- It should be relatively cost effective (both to run and to obtain) and readily available.
- It should be user friendly and the effort required to a) learn how to use model, **b)** prepare data for and **c)** run the model, should not be prohibitive.

Useful attributes

- The chosen method should be able to compute the discharge required to obtain a predetermined concentration/value of a given water quality constituent. For example, QUAL2E can compute the discharge that is required to obtain a pre-determined concentration of DO.

1.8.3 Choice of the modelling method

It was requested by the Steering Committee that the scope of this project be restricted to instream models and not extended to a consideration of catchment runoff models. Another request was that the method should avoid complex modelling systems and be practical. Furthermore, the limitations in data availability in this country also had to be kept in mind. A survey was undertaken to assess exactly how, and to what extent, water quality modelling is incorporated into determinations of environmental flow requirements elsewhere in the world. In summary, it would appear that although extensive water quality modelling has, and is, been carried out in many countries and on many river systems, water quality modelling is not formally and routinely incorporated into instream flow assessments; or if it is, this is not explicit. Thus, there was no other system of integration of water quality and environmental flows that could be copied.

It became apparent to the project team that no single modelling approach possessed all the attributes that are required. In addition, not all attributes would necessarily be required in all situations. It is proposed, therefore, that rather than using a single modelling method for predicting water quality, a hierarchy of methods should be employed.

The three proposed tiers of modelling complexity are:

- A simple discharge-concentration (Q-C) method.
- Mass-balance modelling or QUAL2E.
- A catchment runoff model.

The modelling methods range from the simplest method (Q-C modelling in which a regression equation is derived for concentration against discharge) to more complex mass balance modelling which is carried out either using spreadsheets or the instream computer model (QUAL2E). Compared to simple statistical methods QUAL2E (Brown and Barnwell 1987) provides more advanced simulation capacity, including calibrated numeric routines for all of the main generic constituents. In addition, it provides heat exchange calculations and other capabilities so that temperature can be predicted. The most sophisticated level of water quality modelling proposed is a catchment runoff model, possibly HSPF (Bicknell, Imhoff, Kittle *et al.* 1993) or the South African model ACRU, which has more limited water quality capabilities (Kienzle, Lorentz and Schulze 1997). In some river systems, such as for example one that is not particularly ecologically sensitive or important, it may only be necessary to carry out the Q-C regression method. Indeed in many catchments, lack of data may preclude use of more sophisticated modelling methods. Financial and time constraints are additional factors likely to influence the choice of modelling method. In some sensitive catchments, especially if considerable changes in land-use (e.g. urban development) are anticipated in the near future, a full catchment runoff model may need to be set up (e.g. HSPF). The simulations from such models could be used in the IFR process. Note however that such models are data, time and labour intensive. It is unlikely that such models would be set up especially for an IFA, because of budgetary restraints (unless the catchment was of particular importance). Because of the decision of the Steering Committee not to consider catchment runoff models in this project, this avenue of research was not pursued further.

The extent to which the chosen modelling methods satisfy the required attributes is discussed in this document after a description of each modelling method (Chapters 2 and 5). Application of Q-C modelling to various river systems is described in Chapter 2. An application of the water quality model QUAL2E to the Lower Olifants River is described in Chapter 5.

1.9 Methods used to predict the effect of water quality on the biota

Having predicted (by whatever modelling method) the effect of change in discharge on the levels of the different water quality variables, it is important to extend this to an assessment of the potential implications for the aquatic biota. The term "aquatic biota" is used to describe all the biotic communities in an aquatic ecosystem and includes: fish, macroinvertebrates, plants in the form of riparian vegetation, macrophytes, algae, and micro-organisms. Especial attention has been given in this project to the use of macroinvertebrates (insects, crustaceans, worms and molluscs) as predictive tools. This is because macroinvertebrates occupy a key role in the food chain (Petts and Maddock 1994). They also possess several features that make them useful in biomonitoring. As a consequence they are used in the South African Scoring System (SASS) a biomonitoring technique that has been developed specifically for South African rivers. Furthermore the "Biobase" has been developed as part of a previous WRC-funded project. This is a database containing historical records of invertebrate taxa and the results of sampling water quality (section 1.9.4).

Several approaches to assessing the effects of water quality on biota have been used in the literature. These can be loosely grouped into three types, namely toxicity testing; multivariate analyses of field-derived data; and complex (computer-based) ecological models. The division is rather arbitrary however and there are considerable overlaps between the approaches. For example, the mathematical equations used in complex ecological models are usually derived from empirical studies in the field. Each of these approaches is discussed below.

1.9.1 Toxicity testing

Ecotoxicology can be defined as the "science of how chemicals at toxic concentrations influence basic ecological relationships and processes" (Chapman 1995, citing Brown 1986). Typically, toxicity testing is carried out by exposing organisms to known concentrations of a chemical substance for a given length of time. Thus, if the probable concentration of a chemical substance at a field site is known, by examining the results from toxicity tests some idea can be gleaned of the likely effect on the biota. The majority of ecotoxicological studies are carried out under laboratory conditions, the advantage being better control of environmental variables that might influence the toxic response. Environmental factors known to affect toxicity, other than pollutant concentration, are temperature, pH and water hardness (Mason 1991). The presence of other chemical constituents that may exert additive, synergistic or antagonistic toxic effects can also be controlled (e.g. Musibono and Day 1999). The major disadvantage of laboratory-based studies is that it is difficult to extrapolate results to toxic effects in the field. On the other hand, evaluation of instream impacts are difficult to interpret, due to environmental variability (Eagleson, Lenat, Ausley *et al.* 1990).

From the above discussion, it can be seen that the derivation of ecotoxicological parameters can be useful in predicting potential impacts on a water resource. If the predicted concentration of a pollutant in a water body is likely to be relatively high, for example greater than the LC_{50} for that chemical and for sensitive species, there is a risk of serious implications for the biota. The lower the likely concentration of the pollutant, however, the more difficult it is to predict what the consequences will be for an aquatic ecosystem.

1.9.2 Multivariate studies in the field

Because the results of toxicological experiments are difficult to extrapolate to the field, many researchers have concentrated on studies of natural systems. Cause-effect relationships between water quality and biotic response have been studied by examining the correlation between the concentration of chemical constituents or values of physical variables and species presence/absence or abundance (Rutt, Weatherley and Ormerod 1990; O'Keeffe 1995). Alternatively, in a few cases, water quality has been manipulated in the field and the responses of the biota examined (Rundle, Weatherley and Ormerod 1995). A major problem in studying cause-effect relationships in the field is the multitude of factors that can influence biotic response. This necessitates the use of some form of multivariate statistics, such as Multivariate Discriminant Analysis (MDA) to identify the most important variables.

1.9.3 Complex ecological models

In a manner similar to water quality models, ecological models can also range from a simple equation linking, for example, invertebrate score and the value of an environmental variable, to complex, computer-based models simulating and requiring data for many variables. Furthermore, ecological models also range from those based on purely empirical relationships to those in which the process is conceptualised (mechanistic models). Only limited progress has been made in the second type of model, however, due to the fact that detailed understanding of the processes linking abiotic factors and biological response is lacking. As a result, most ecological models are based on a description of events rather than on the processes which control those events Kovalak (1981).

Although the methods described above show potential for assessing the implications of shifts in water quality on aquatic organisms, making such predictions is unlikely to be straightforward. Aquatic ecosytems are complex. There are many interactions between biotic and abiotic components, some of which are known and can perhaps be quantified, and others of which are totally unknown. Part of the reason for a lack of understanding of these interactions in South Africa is the paucity of research on indigenous organisms.

Furthermore, natural genetic variability in populations gives rise to differing responses of individuals. Consequently, aquatic ecosystems are highly variable, and different responses to stressors are possible under different circumstances. As a result, most predictions relating to ecosystem behaviour carry a high degree of uncertainty (Hohls 1996). In addition, there are several water quality variables, many of which interact and consequently exhibit synergistic and antagonistic effects. Furthermore, change in discharge itself, over and above the effect on water quality, is likely to have a profound effect on the hydraulic habitat of the aquatic biota. Thus making predictions with regard to the effect of proposed changes in discharge and hence water quality on the aquatic biota is a challenging problem. Although several approaches are available, the complexity and variability of ecological systems necessitates the compilation of extensive field data sets. Large databases are required in order to infer statistically robust biotic responses to changes in water quality impacts, and to make them valid for different regions of the country. Although this is costly and time-consuming, it would appear from the literature (Armitage 1994) that it is essential for making reliable predictions of the implication of changes in water quality for aquatic biota.

1.9.4 Availability of water quality tolerance data

Several forms of data linking water quality (concentration of chemical constituents and values of physical variables) with the abundance, composition (i.e. presence or absence of taxa) or functioning of aquatic macroinvertebrates (both indigenous and exotic) are available and are discussed below.

i) *South African Water Quality Guidelines*

The South African Water Quality Guidelines (DWAF 1996) for aquatic ecosystems give recommended ranges of water quality variables (termed the "target water quality range" (TWQR)). These ranges were derived from consideration of the limited data on indigenous organisms and from examination of the guidelines from other countries. With regard to toxins, nearly all of the TWQRs are derived from laboratory-based toxicity data. These data have been gathered from all over the world for a wide variety of taxa. Safety factors have been incorporated into the values of the recommended ranges. These ranges represent a very useful tool for initial screening of predicted water quality values.

Values that fall within the TWQR, can be considered to pose little risk to the biota. Values of water quality constituents that fall outside of the TWQR, on the other hand, may pose a risk and would merit further consideration.

ii) *The Rivers database*

The South African Scoring System (SASS) is a biomonitoring technique that has been developed specifically for South African rivers. It is a scoring system for assessing the chemical quality of stream and river water from the familial composition of the benthic macroinvertebrate fauna (Chutter 1994) and is one of the major components of the River Health Programme. Results from this biomonitoring programme will be stored in the Rivers database (Fowler, Dallas and Janssens 2000). Although the main focus is on the collection of biological indicators, water chemistry and habitat data will also be collected at the same time as biomonitoring data. It should be noted that macroinvertebrates are identified only down to family level in this resource.

iii) *The Biobase*

The Chemical-Biological database (Biobase), is a database comprising biological (macroinvertebrate) and water quality data derived from documented studies of riverine ecosystems in South Africa (Dallas, Janssens and Day 1999). It has been developed at the Freshwater Research Unit, UCT, as part of a previous WRC project. Measures of physical attributes and concentrations of chemical constituents are incorporated with associated species assemblages of invertebrates from samples collected over many years throughout the country. These data are of mixed taxonomic resolution since some taxa are identified to family level but most are at species level. Since the natural distribution of aquatic organisms is associated with geographic patterns of water quality, this screening tool can be interrogated to evaluate, or predict, the expected or desired macroinvertebrate assemblages associated with natural background water chemistry (Dallas, Day, Musibono and Day 1998). The Biobase on CD, in conjunction with a User manual (Dallas and Janssens 1998), is available from the Water Research Commission.

iv) *Ecotoxicological data*

Ecotoxicological data for aquatic organisms are available in the form of the ECOTOX database developed by the US EPA. This database was found to be of limited use in the project however, since it was difficult to extrapolate to the effects on indigenous organisms and does not include system variables (e.g. temperature, DO). Nevertheless, for assessing the impact of toxic substances it is a useful resource (see "Useful web sites" for the web site address).

Values of some ecotoxicological parameters for chemical constituents and indigenous organisms are given in the South African Water Quality Guidelines (DWAF 1996) as well as in Dallas and Day (1993) and elsewhere (*inter alia* Goetsch and Palmer 1997; Musibono and Day 1999; Palmer and Scherman 1999). Currently, few empirical tolerance data exist for southern African organisms or for standard test organisms exposed under local conditions. An emerging, potentially useful resource is a database under development at the Centre for Aquatic Toxicology (CAT), IWR, Rhodes University. This database records the results of toxicity experiments using indigenous macroinvertebrates gathered from rivers all over South Africa. Toxicity with regards to the major components of TDS (namely sodium chloride and sodium sulphate), some metals, as well as well-defined effluents, has been determined.

A method (termed the "Biotic Protocol") has been developed during the course of this project to assess the implications of a predicted water quality scenario for the biota. It makes use of all available relevant data linking water quality and macroinvertebrate abundance and community composition. The method makes especial use of the Biobase and Rivers database. This protocol is presented in Chapter 6.

1.10 Attainment of the project objectives

The aims of the project as given in the original project proposal and the extent to which they have been attained (indicated by a tick or a cross), are listed below:

To follow up on aspects of previous projects on water quality (aims A and B) **but mainly to develop a new research area (aim C).**

A) Pursue the use and curation of the Biobase (Biological-Chemical database) developed in project no. K5/626:

- a) Curate the database and capture additional data, as and when the need arises. ✓

- b) Investigate future maintenance and accessibility of the Biobase. ✓
- c) Continue to interrogate the database with a view to:

  i) Verifying the scores recommended for the biomonitoring system SASS. x

  ii) Identifying water quality tolerances of key taxa of invertebrates for use in water quality assessments for the IFR process. ✓

  iii) Verifying the methodology for establishing regional water quality guidelines for selected constituents as proposed in project K5/626. x

B) To continue to examine, in a low-key manner, the extent to which water quality determines the distribution of south-western Cape endemic invertebrates. x

C) To address the question of the relationship between water quality and water quantity (particularly discharge), with particular reference to IFR assessments:

- **a)** To provide a review of the kinds and forms of data needed and of methods developed elsewhere for providing these data. ✓
- **b)** To develop methods for processing water quality data to derive time and flow dependencies for key water quality constituents. ✓
- **c)** To identify and assess methods for modelling selected water quality constituents at the feasibility level of the IFR process. ✓
- **d)** To investigate the extent to which available data (e.g. in the Biobase) can be used for setting environmental water quality requirements as part of IFR assessments. ✓
- **e)** Based on the above, produce a framework for the assessment of water quality as part of IFR studies. ✓

Due to the importance of the development of tools to implement the Reserve methodology, the major emphasis of the project was aim **C)**. Consequently all the points under this section were attained and several tools have been developed (see below). In addition, most of aim **A)** has been achieved. The exceptions are part c) i), namely, verifying the scores recommended for the biomonitoring system SASS, and part c) iii) verification of the methodology for establishing regional water quality guidelines for selected constituents as proposed in project K5/626. These were not carried out because Helen Dallas left the project team and no one with the requisite expertise or time was available. As part of her Ph.D., examining the factors affecting SASS4 scores and the implications for setting Reference conditions, the temporal and spatial variation of SASS4 scores will be examined. In addition to forming part of a Ph.D. thesis, this work will also form part of a WRC-funded project (K8/404) titled "Spatial and temporal heterogeneity in lotic systems: implications for defining reference conditions for macroinvertebrates" (H. Dallas, FRU, UCT, *pers. comm.*). Aim B was not addressed due to the lack of a suitable post-graduate student to under-take the work.

1.11 Products arising from the study

As a result of this project, four major tools have been developed:

- A technique (Q-C modelling) for predicting the resulting concentration of a chemical component for a given discharge.
- A procedure (the "Biotic protocol") for predicting the likely effects of changed water quality on the biota.
- A framework and protocol for incorporating water quality predictions and implications for aquatic biota into the Reserve process.
- A technique (concentration and stress time-series modelling), developed in conjunction with Prof. Palmer, IWR, Rhodes University for obtaining time-series of chemical concentration and stress, which can be used in comparing the water quality consequences and the implications for the biota, of differing complex flow scenarios.

Other products arising from this project are:

- A literature survey reviewing the links between flow, concentration and biotic response.
- Spreadsheet templates for Q-C modelling.
- The curated Biological and chemical database (BIOBASE) and User manual (largely a product of the previous project (K5/626 Water quality requirements for riverine biotas), is now available from the WRC.
- Reports and papers presented at conferences.

CHAPTER 2

LINKING WATER QUALITY AND DISCHARGE USING THE DISCHARGE-CONCENTRATION (Q-C) MODELLING METHOD

2.1 Introduction

It was explained in Chapter 1, that water quality can be altered by changes in discharge. Furthermore, the Reserve as specified in the National Water Act is comprised of a water *quantity* component (i.e. the volume and timing of streamflow). It is also comprised of a water *quality* component, which prescribes, for a given ecological Reserve class (previously called ecological management class), the range of concentrations of chemical constituents and values of physical variables that should not be exceeded in a water resource as a result of human activities. In setting the ecological Reserve for a site on a river, therefore, water quantity and quality need to be linked. It was also explained in Chapter 1 that a hierarchy of modelling methods should be employed for this task. In the present chapter, the simplest method, discharge-concentration (Q-C) modelling is discussed. Also included is the extent to which the attributes required in the modelling method are satisfied by this technique as well as the inherent assumptions and limitations in the method. A refinement of the Q-C modelling method is then presented which addresses some of the above limitations. Finally, the major findings arising from application of the modelling method to actual IFR determinations are reported.

2.2 Description of the Q-C modelling method

The steps that comprise the basic Q-C modelling method are discussed below and are summarised in Figure 2.1. A refinement of this method is presented in section 2.7.1. Application of the modelling method to specific rivers is discussed in section 2.8

(Pienaars River); section 2.9 (Palmiet River); section 2.10 (Olifants River) and section 2.11 (Breede River).

The modelling method is carried out at each IFR site for which appropriate discharge and water quality data are available. Each water quality variable is modelled separately, although in the case of nutrients, the final assessment category is determined from a ratio of two separate variables (DWAF 1999). A spreadsheet template (in Microsoft Excel) has been developed as part of the project into which the discharge and water quality data are entered. The required calculations to determine the regression line and confidence limits for the Q-C plots are built into the spreadsheets. A computer disk with a copy of this template, is to be found at the back of the training manual (Malan and Day *in prep.*).

The steps of the Q-C modelling method are given below:

- i) *Collate all available data on water quality, point-sources of pollution, hydrological structures, hydrology, land-use, topography etc. Identify the locations of the IFR sites relative to water quality monitoring stations and discharge gauging sites, as well as any other significant hydrological features. Produce maps indicating the above.*

- ii) *Identify the different ecoregions through which the river flows according to the method of Kleynhans (1999). Using this information, as well as the location of dams and significant tributaries (hydrological features), derive reaches within which water quality would be expected to be uniform.*

- iii) *For each IFR site, using the discharge and water quality data from the nearest appropriate gauging and water quality monitoring station, correlate mean monthly discharge values as well as median monthly concentration values for each water quality variable (C). This is done:*

  i) For the Reference Condition (i.e. the least impacted state)

  ii) For the Present Ecological State (possibly impacted).

In this step, tables are prepared recording monthly mean discharge (natural and present day), as well as median monthly values of each water quality variable (Reference Condition and present day). An example of such a table (Table 2.1) for a site on the Pienaars River is shown in section 2.8.1.

- iv) *Examine the relationship between discharge and the concentration of C.*

This is performed separately for both the Reference Condition and for the Present Ecological State, which may or may not be impacted. Graphs are drawn of concentration versus discharge. A regression line is then drawn through the data points using the trendline function in the spreadsheet package (Microsoft Excel). This provides a choice of five regression equations (linear, exponential, power, logarithmic and polynomial). The "best fit" is chosen by using the relationship that yields the highest value of the coefficient r^2, in addition to expert judgement. Where there is little difference between r^2 values the simplest relationship (i.e. linear or logarithmic) is used. In this document, the regression equation for the Reference Condition is termed function "N" and that for the Present Ecological State is termed "M".

- v) *Estimate the simulated concentrations of variable C for the Reference Condition and Present Ecological State.*

The simulated values of the variable (C) in the Reference Condition are estimated using the regression equation "N" and plotted. Likewise, the simulated values of C for the present state are derived from the regression equation "M".

In summary:

"N" = a function relating the Reference Condition concentration of C ($[C]_{RC}$) to discharge (Q)

e.g. $[C]_{RC} = b(Q) + c$ ....equation 1

"M" = a function relating the present state concentration of C ($[C]_{pes}$) to discharge (Q)

e.g. $[C]_{ps} = b(\text{Log}Q) + c$ ....equation 2

where: b, c are constants (the slope and y-intercept respectively).

The 95% confidence interval for each predicted $[C]_{RC}$ or $[C]_{pes}$ is calculated using formulae (Appendix A) incorporated into the spreadsheet templates. Standard statistical methods have been used to calculate the confidence interval (A. Joubert, Department of Statistical Sciences, UCT, *pers. com.*). Depending on the mathematical form of the relationship linking discharge and concentration (function "M"), the data may first need to be transformed. Linear functions use values of X and Y (i.e. discharge and corresponding concentration) that are untransformed. Logarithmic functions use x = ln X, exponential functions use y values transformed to ln Y and power functions use discharge values transformed to ln X, as well as concentration values transformed to ln Y. These transformations are performed automatically in the spreadsheet templates.

- *vi) Predict the concentration of C for the prescribed IFR discharge regime and estimate the 95% confidence interval around this prediction.*

The prescribed flow regime, as derived during an IFR workshop using the Building Block Method (Chapter 7), is given in the form of values of the baseflow for each calendar month. This is for both normal hydrological years (maintenance baseflow) and drought years (drought baseflow). Using the function "M", the predicted concentration of each water quality constituent for each month can be calculated in addition to the 95% confidence interval. An example of such calculations for the Pienaars River is shown in Table 2.2.

- **vii)** *Calculate the extent of deviation of the present state values of C and predicted concentrations of C from the Reference Condition, or in the case of nutrients, from a pre-defined value.*

The predicted concentrations can be compared with the concentration of C in the Reference Condition and the difference calculated using the following:

$$\text{Deviation from RC} = \frac{(\text{Predicted } [C] - [C]_{RC})}{[C]_{RC}}$$

....................equation 3

These values are then tabulated as set out in Table 2.2 for the Pienaars River.

- **viii)** *Assign the Assessment Category for each month.*

The assessment category that each water quality reach would fall into (with respect to a particular variable) is then derived for each month under maintenance baseflow and drought baseflow (Table 2.2). In addition, the mean category for the entire year is also calculated. Assessment categories and their derivation are discussed in section 2.3.

Figure 2.1 Summary of the simple discharge-concentration (Q-C) modelling method

i) Collate available data on water quality, point-sources of pollution, hydrological structures, hydrology, land use, topography etc. Locate IFR sites.

ii) Identify the ecoregions through which the river flows. Derive reaches expected to exhibit uniform water quality.

iii) Correlate mean monthly discharge values as well as median monthly concentration values for each variable (C), at each IFR site.

a) For the Reference Condition (least impacted state)

b) For the Present Ecological State (possibly impacted).

iv) Examine the relationship between discharge and the concentration of C.

v) Estimate the simulated concentrations of C for the Reference Condition and Present Ecological State.

vi) Predict the concentration of C for the recommended discharge regime and estimate the 95% confidence around this prediction.

vii) Calculate the extent of deviation of the Present Ecological State values of C as well as the predicted concentration values from the Reference Condition.

viii) Assign the Assessment class for each month for each recommended flow scenario. If required, calculate the minimum flow that is required to result in a "D" (or higher) assessment class for each month.

2.3 General considerations

- i) The modelling method has been developed so as to be compatible with the Building Block Methodology (King and Louw 1998) which is the procedure currently used for instream flow assessments used as part of the Reserve determination process. Thus steps 1 and 2 as well as some of step 3, would automatically be carried out as part of an Intermediate or Comprehensive Reserve Determination (DWAF 1999). It is envisaged that steps 1-5 would be carried out prior to the IFR workshop. At the workshop, the specialists first specify the recommended flow regime, which is designed to keep the river in a predefined Ecological Reserve Class. The model is then used to predict the monthly concentrations of each water quality variable. These are referred to as the "water quality consequences" of the proposed discharge regime. The method is also compatible with DRIFT (Downstream Response to Instream Flow Transformations) a newly emerging methodology for assessing environmental flow requirements (Chapter 7). This methodology assesses the implications of successive reductions of the flow regime (although augmentation can also be considered). For each reduction the water quality consequences can be calculated using Q-C modelling.
- ii) Conversely, the modelling method can also be used to estimate the discharge that would be required to attain a given assessment category (see point viii) with regard to a particular water quality variable under the current pollution loading - for example the discharge required to attain a "D" ecological category, which is the lowest Ecological Reserve Category that is considered to be sustainable (DWAF 1999).
- iii) The method can be used for TDS, sulphate and other conservative constituents. Nutrients can also be modelled in this manner, although this is not usually as successful as for conservative constituents (see results from Pienaar's River, section 2.8.1). Nutrients often exhibit considerable scatter when concentration is plotted against discharge. This is likely to be a consequence of the various processes that influence instream concentration (e.g. microbial conversion between chemical forms, adsorption/desorption from sediment particles, uptake by the biota).
- iv) Median monthly values of water quality variables both for the Reference and Present State are derived according to the rules set out in DWAF (1999).
- v) Monthly *mean* discharge values were correlated with *median* monthly concentration values for each variable (C). Mean discharge values were used

because this is the convention in the field of hydrology. Median water quality values were employed, on the other hand, since concentrations can range widely and a single extreme event can alter the mean significantly. It is thus statistically more correct to use median values.

- vi) Scenarios can be envisaged in which the proposed discharge is higher than in the natural state. This might occur if the river under consideration is the recipient of an inter-basin transfer (IBT) scheme, and the IFR site is downstream of the point of entry of the donor water. Alternatively, if additional water needs to be released from an upstream impoundment during the dry season to satisfy irrigation demands downstream (termed "capping flows"). The Q-C method should be applicable in such cases as long as the provisos in sections 2.5 and 2.6 are kept in mind.
- vii) The spreadsheet template also has the facility for producing seasonal graphs of concentration and discharge for each month of the year. A consideration of the seasonal patterns can be useful in understanding the dynamics between water quality and quantity at a given site (see for example Figure 2.4, section 2.11).
- viii) Assessment categories are assigned according to the criteria outlined in DWAF (1999), by using a method developed by Palmer and Rossouw (2000), or by using other criteria still to be introduced. The initial guidelines that were included as part of the Resource Directed Measures (DWAF 1999), make use of tables for system variables, nutrients and toxic substances. For each type of water quality variable, the concentration or value range that is allowed for each category (A - E) is defined. In the case of nutrients, the class limits are defined as ratios, for example as TIN: dissolved P. The system variables pH, temperature and DO are expressed in terms of the percentage deviation from Reference Condition that is allowable for each category. In DWAF (1999) the system variable TDS was also treated in a similar manner. For example, in systems judged to be category "A" with regard to this variable, it was stipulated that TDS concentration should not deviate more than 15% from that of the RC. During the assessment of water quality for the OREWRA Palmer and Rossouw (2000), it was found that the system of defining categories for TDS on the basis of deviation from RC was impracticable. Using this system nearly all reaches of the Olifants River were found to be in an E or F category for TDS, yet this was not substantiated by the results from biomonitoring. The SASS4 scores collected for the river indicated that that water quality was significantly better (most reaches C – D). A new system for deriving the TDS concentration ranges that define

each category was developed based on ecotoxicological parameters. This system is discussed in detail in Palmer, Rossouw, Malan *et al.* (*in prep.*). At the time of writing this report the entire question of how to delimit Assessment categories, particularly for salinity is under review.

2.4 Attributes of the Q-C method

The extent to which the Q-C method satisfies the attributes required in the modelling technique (section 1.8.2) is discussed below. Attributes possessed by the model are indicated by a tick and those that are not satisfied, are marked with a cross. Cases where the requirements are partially satisfied are indicated by both.

The modelling method should:

- *be able to predict the concentration or value of most of the following water quality components with regard to change in discharge: TDS, temperature, nutrients (nitrates, ammonia, soluble phosphate) and possibly TSS, DO.* ✓ x

  As discussed in section 2.3 above, the Q-C method is most suited to the modelling of conservative constituents (e.g. TDS, TSS, sulphate, individual inorganic ions) and is less suited to model non-conservative constituents such as nutrients. Unlike QUAL2E it is unable to simulate DO or temperature.

- *be flexible and be able to handle a variety of "flow-modifiers" (i.e. causes for altered discharge, such as abstraction, inter-basin transfers, weirs and impoundments). Different reservoir management options should be simulated either by the model itself or by being compatible/interfacing with other models that can e.g. handle the problem of multi-level outflow from stratified impoundments. Thus the modelling method should be applicable to both regulated and unregulated rivers.* ✓ x

  This issue is discussed more fully in sections 2.5 and 2.6. The modelling method is applicable to regulated and unregulated rivers, although, the water quality predictions arising from this method are valid only if the system is operated in the same manner as when the Q-C dependencies were determined. Thus, if discharge is reduced due to the construction of an impoundment upstream for example, the same concentration of a constituent for a given discharge can no longer be expected.

- *have a proven track record in a country climatically similar to South Africa, if not in South Africa itself. Expertise and support to run the model should be readily available and the method should be well documented.* ✓x

  As described in the literature review to this project (Malan and Day 2002), this simple numerical technique (rating curves) was one of the first forms of water quality modelling. It has been used in various forms, and in numerous studies.

- *be compatible/readily incorporated into the Building Block Method for IFR determinations and thus should have appropriate spatial and temporal detail. It should also ideally be compatible with DRIFT, another IFR method currently under development (Chapter 7).* ✓

  The Building Block Methodology (BBM) generates monthly values for discharge. It is felt, therefore, that using concentration and discharge data for water quality modelling at the relatively course scale of a month is appropriate. In addition, at some monitoring sites, water quality samples are collected only monthly or even less frequently. Furthermore, both the BBM and the DRIFT method utilise points on the river (IFR sites) which are considered to be representative of that section of the system. When considering water quality, the river is divided up into reaches that are assumed to be homogeneous (for a comprehensive description of delineating water quality reaches, see DWAF (1999)). Typically, there are usually many more water quality reaches than IFR sites. Q-C modelling can be carried out for each water quality reach, providing that there are appropriate discharge and concentration data available. Thus the spatial scale of Q-C modelling is also compatible with the BBM and with DRIFT. The method should be compatible with other holistic enviromental flow methodologies that are developed in the future as long as they are also at a similar spatial and temporal scale.

The modelling method should:

- *be able to work on a PC and not require a specialised platform e.g. UNIX work-station.* ✓

- *should be relatively cost effective (both to run and to obtain) and fairly readily available.* ✓

  It is specified in the Reserve methodology that the values of the water quality variables be calculated and documented as monthly median values. These values are then used directly in the Q-C model, thus saving time and effort.

- *be user friendly and the effort required to a) learn how to use model, b) prepare data and c) run model should not be prohibitive.* ✓

  To make the Q-C method as user-friendly as possible standardised, pre-prepared spreadsheets have been developed. Steps such as, fitting the "best" regression equation ("trendline"), or calculating the discharge required to attain a predetermined water quality assessment category, can be carried out reasonably easily by using the built-in features of the software package. This project has so far mostly been carried out using Microsoft Excel spreadsheets, but can be performed using any spreadsheet software.

- *The chosen method should be able to compute the discharge required to obtain a predetermined concentration/value of a given water quality constituent.* ✓

- *The data input requirement (in terms of both format and amount) should be compatible with the data likely to be available.* ✓

2.5 Assumptions and limitations in the method

The following assumptions and limitations in the method have been identified.

- Many factors, apart from discharge, influence water quality. Instream concentrations of chemical constituents resulting from a given discharge can vary depending on (*inter alia*) season, antecedent rainfall, temperature, and the operation of an upstream impoundment, which are not taken directly into account in this modelling method. As such, discharge-concentration modelling is a very simple approach and is aimed at providing an estimate of predicted water quality.

- Use is made of monthly median values of concentration and monthly average discharge, through which a trend-line is fitted. An alternative approach would be to use all data points for concentration linked to the discharge at the time of sampling. The latter can be determined by using the gauge height and the appropriate rating curve. Pionke and Nicks (1970) found that several widely divergent salinity values corresponded to the same discharge on an instantaneously-determined basis, whereas averaging the data on a monthly basis removed much of the variability. In this project, comparisons were made of the trend patterns, as well as the correlation coefficients obtained, between measured and simulated data using either all data points or monthly data. It was found that Q-C trends could be more clearly discerned using monthly data, than if all data points were used. This confirms the findings of Pionke and Nicks (1970) above, in that less variability is exhibited by the data if monthly values rather than instantaneous values are used. Plots of all data points can be however useful for confirming the trend found using monthly values, to examine the variability in response to discharge, as well as to view the Q-C relationship over the full range of discharges found at a given site. This last aspect is required when drawing up the Q-C matrix for time-series modelling (Chapter 4).
- Unless measured water quality data are available for very low flows and very high flows, extrapolation to these conditions (as required when converting discharge- to salinity time-series, see Chapter 4) is likely to be inaccurate.
- The modelling method is severely constrained by the availability of data. It should perhaps be noted that this is not unique to Q-C modelling and that the absence of appropriate data makes the use of any water quality modelling method difficult, if not impossible (see Chapter 5). The degree of confidence in the predictions is influenced by the accuracy and completeness of the data used, as well as the proximity of the water quality monitoring stations to the IFR site in question. The greater the distance between the site and the monitoring station, the less it can be assumed that the water quality predictions are representative of that IFR site. In addition, if a major tributary or other hydrological feature (e.g. a weir) is situated between the monitoring station and the IFR site, confidence in the results is also diminished. Indeed, *if the tributary is known to be significantly different in water quality from the mainstem river, and/or contributes substantially to the total discharge, and if no other data are available, Q-C modelling should not be undertaken for that IFR site.*

- From the above, it can be seen that expert judgement is important in the use of this method. It is required especially for understanding the limitations and assumptions that are inherent in the modelling method, but also in choosing the monitoring and gauging weirs that are used to provide data for an IFR site, and in understanding Q-C relationships obtained for each water quality variable. In addition, derivation of monthly median water quality concentrations, particularly for the Reference Condition, is not always straightforward. Frequently no pre-impact data are available. Although guidelines are given in the RDM manual (DWAF 1999) for such situations as this, experience of the natural water quality to be expected in different ecoregions is essential.
- It is assumed in the method that, if discharge is altered, apart from the concentration of the water quality variable under concern, all other parameters (e.g. the pollution load) will remain constant. In practice, if discharge is altered drastically it is likely that the source of the water will be altered (for example by means of the impoundment of tributaries). In addition, changes in operation of upstream impoundments may also be involved. In other words, this method does not take into account changes in management scenarios. This question is discussed further in section 2.6.
- The Q-C modelling method is not suitable for making predictions of temperature or dissolved oxygen, since the former is a physical variable, rather than a chemical entity. The concentration of dissolved oxygen in the water column is complicated by many factors including hydraulic turbulence, temperature, presence of aquatic plants and is not amenable to the simple approach used in this method.
- The method makes use of the standard "trendline" regression functions available in commercial spreadsheet packages. Considering the inaccuracy inherent in the modelling method, it is considered that this simplification will not detract from the overall accuracy of the results.
- Dortch and Martin (1989) list several other simplifications that are frequently made when modelling water quality in streams. They also apply to Q-C modelling and are;

  - the assumption that there is complete lateral and vertical homogeneity in water quality (i.e. as in one-dimensional water quality models)

  - the assumption that any water from point sources or tributaries will mix instantaneously with the water in the mainstream.

  - the assumption that discharge and water quality are uniform within a reach.

2.6 Consideration of instream versus catchment effects

At a given point (e.g. an IFR site) on an unregulated, unimpacted river, the characteristic discharge-concentration patterns exhibited by each water quality constituent are reflections of the fluctuation of seasonal precipitation and of processes in the catchment, upstream of that point. Such patterns will depend on the geology and the topography, as well as the climate, of the catchment area. In the case of catchments where water quality is impaired, the discharge-concentration patterns will also be influenced by point- and diffuse sources of pollutants. Point-sources are discharged directly into a river and are largely independent of catchment processes. Diffuse sources, on the other hand, enter the river *via* surface or sub-surface flow over a wide area. They are therefore very dependent on processes, such as precipitation and runoff, occurring in the catchment. The Q-C modelling method correlates monthly median concentration of a given water quality constituent against monthly mean discharge. These values are calculated (in the case of the present-state conditions) from the data-set for the previous five years. Thus, the derived values reflect seasonal discharge-concentration trends that are characteristic of that site, and that result from the upstream processes in the catchment.

In the case of IFR sites that are downstream of impoundments or weirs (and are therefore situated on regulated rivers), depending on the distance of the site from the hydrological control, plots of concentration against corresponding discharge may not necessarily reflect catchment effects. The closer the site is to the impoundment the more likely it will be that the discharge-concentration trend will be dominated by impoundment effects rather than factors arising from the surrounding catchment. "Impoundment effects" include the operation strategy of the impoundment (i.e. when water is released and how much), processes taking place within the impoundment that affect water quality, the off-take depth, and the resultant water quality of water issuing from that source. For example, it was noted in section 1.6.1 that nitrates in agricultural areas are often washed off from the surrounding land into the stream during the onset of the rainy season. Plots of nitrate concentration versus discharge frequently show marked increase with increasing discharge. It was also noted that impoundments, depending on the construction type, can have a marked effect on water quality, altering stream temperature and most importantly, from the point of view of Q-C modelling, retaining nutrients and sediments. Thus an IFR site just downstream of an impoundment may not show a characteristic increase in nitrates with increased discharge. If the site is situated further downstream, however, so that a significant proportion of discharge is derived from runoff from the land, rather than from impoundment releases, instream concentration of nitrates may well be found to increase with discharge. This is especially if the runoff is from agricultural lands. In addition, at some sites downstream of an impoundment, sporadic releases may result in a poorly defined relationship between concentration and discharge.

The amount of water and the chemical composition of that water at a site, for a given point in time, is the cumulative result of all the different sources. For example, 60% of the water may originate from the mainstem river and 40% from a tributary entering just upstream of the site. Each source of water may have very different water quality. The characteristic discharge-concentration relationship for each water quality variable is dependent on the relative proportions of the two water sources. Should the proportions be significantly changed (for example by the construction of an impoundment on the tributary so that in the new scenario, only 10% of the total discharge is now supplied from this source), or if the pollutant loading is changed, the Q-C relationship at the IFR site may also be altered. Alternatively, should the concentration of a water quality constituent be significantly altered (for example by the introduction of an additional point-source) the Q-C relationship is again likely to be changed. The Q-C modelling method correlates monthly median concentration of a given water quality constituent against monthly mean discharge and this relationship is used in the method to *predict* the concentration for a given discharge. From the above it follows that the reason for the change in discharge (the "flow-modifier") is important and should be taken into account when interpreting the Q-C patterns obtained. It should be clear from the discussion that these predictions will be valid only if the system is operated in the same way as used to derive the discharge-concentration relationships. In other words, the pollutant loading from the various sources must remain the same. One of the major limitations of the IFR methodology as it now stands is that the sources and contributions of different streams or sources of water is not usually known or defined at the time of the workshop. This is not a limitation if water quantity only is considered, but it is obviously simplistic if realistic predictions of water quality are to be made. This aspect is discussed further in Chapter 7.

2.7 Simple mass-balance equations

That predictions of water quality using Q-C modelling are valid only if the system is operated in the same way as used to derive the discharge concentration relationships is a major limitation in the method. In simple systems (two or three point-sources or tributaries), where only conservative constituents are of concern, an estimate of resultant concentration can be obtained by simple mass balance calculation. An example, might be a situation in which an IFR site is immediately downstream of the confluence between a tributary and the mainstem river. The Q-C relationship for the IFR site is a reflection of the relative contributions from the two sources (which may not be constant throughout the year). For a given point in time, an estimate of the resultant concentration at the IFR site can be calculated by adding the loads from the respective sources and dividing by total discharge:

$$C_{IFR} = \frac{C_m Q_m + C_t Q_t}{Q_{IFR}}$$

....................equation 4

where:

C_{IFR} = Concentration of constituent C at the IFR site
C_m = Concentration of constituent C in the mainstem channel
C_t = Concentration of constituent C in the tributary
Q_{IFR} = Total discharge at the IFR site
Q_m = Discharge from mainstem channel
Q_t = Discharge from tributary

The water quality model QUAL2E is able to calculate the resultant water quality arising from multiple instream sources (e.g. tributaries, point-sources) for a wide range of water quality variables including both conservative and non-conservative constituents. In the case of conservative variables, QUAL2E uses simple mass-balance equations such as equation 4, to calculate the instream concentrations of a given variable for each segment of river. This water quality model however also has limitations and is discussed further in Chapter 5. Mass balance modelling can be used to calculate the instream concentrations of contaminants that would result from the loading of different point sources. It can also be used to estimate the maximum loading of contaminants that would be consistent with a given Reserve category. QUAL2E is currently frequently used in the USA to estimate Total Maximum Daily Loads for significant water bodies (US EPA 1995).

2.7.1 A refinement of the Q-C method

Consideration of point-source and non-point source loading of rivers, as discussed in section 2.7 above, has led to the development of a prototype refinement of the Q-C method in which mass-balance equations are set up at each site for each constituent that is to be modelled. The basic Q-C method and the refinement differ only in step v) of the modelling protocol ("Calculate the Reference Condition and present state simulated concentrations of variable C"). In the simple Q-C method this is carried out using the regression equations "N" and "M" which describe how the concentration of C in the Reference Condition and Present Ecological State, respectively, vary with discharge. In the refinement described below the present day concentration of C is described in terms of the Reference Condition load and the present impact load. A mass-balance equation is set up in this step and is used to calculate the predicted concentration for a given discharge.

The revised Step v) of the modelling protocol when using mass-balance modelling is given below:

v) *Describe the Present State concentration of C in terms of the Reference Condition load and the present impact load.*

Equation 4 can also be used to describe the Present State concentration of variable C, where the Present State concentration is given by the Reference State concentration and an additional load (the "impact load") resulting from return flows or point sources of pollutants.

$$[\,C\,]_{PS} = \frac{L_{RC} + L_P}{Q}$$

.....................equation 5

Where for a given discharge (Q):

$[C]_{PS}$ = the concentration of C in the Present State

L_{RC} = the reference condition load of C (i.e. the load under "natural" conditions)

L_P = the impact load of C (i.e. the load of C due to pollution)

The term "L_P" may be sub-divided into individual impact (pollution) loads if this information is available.

For example:

$L_P = P_1 + P_2 + P_3 \ldots\ldots\ldots\ldots\ldots$

....................equation 6

Where: P_1, P_2, P_3, etc. are individual sources of pollutants

Note that in equation 5, the "natural" and "impact" loads of water quality constituent C are divided by total discharge in order to obtain a concentration value for $[C]_{PES}$.

Equation 5 can then be expanded to the following generic model, derived to account for the concentration of C as a result of the reference condition concentration ($[C]_{RC}$), withdrawal of water from the river (P_{ABS}), return of water from point or non-point sources (P_{RF}) and a loading function to describe the input from diffuse sources etc.

$$[C]_{PS}\,\mathrm{SIM} = \frac{((Q_{RC} \times (1-P_{ABS}) \times \text{"N"}) + ((Q_{RC} \times P_{ABS} \times P_{RF}) \times (\text{"M"})) + P_x}{((Q_{RC} \times (1-P_{ABS})) + (Q_{RC} \times P_{ABS} \times P_{RF})}$$

....................equation 7

Where:

$[C]_{PS}$ SIM = the simulated concentration of C in the Present State.

Q_{RC} = Discharge under reference conditions.

P_{ABS} = Proportion of discharge that is removed due to abstraction

P_{RF} = Proportion of P_{ABS} that re-enters the system as return flow.

P_X = additional loads of C

"N" = a function relating the reference condition concentration of C $[C]_{RC}$ to discharge (Q)

e.g. $[C]_{RC} = a(Q) + b$

"M" = a function relating the pollution concentration of C ($[C]_p$)to discharge (Q)

e.g. $[C]_p = c\ (\text{LogQ}) + d$

and: a, b, c and d are empirically derived constants (section 2.2).

The functions "N" and "M" above were derived in step iv) of the modelling method. Using the generic model (equation 7), the simulated, Present State concentration of C can be calculated over a range of discharge values, and plotted. Values for P_{ABS} and P_{RF} can be derived from a comparison between natural runoff (Pitman runoff estimates as given in Midgley *et al.* 1994) and present day discharge as well as from consultation with hydrologists involved with the instream flow assessment. By adjusting the parameters a and b, a simulation curve is obtained that described the variation of $[C]_{PES}$ with discharge (Q). The model is calibrated by empirically adjusting the values of the parameters a, b, c and d so as to obtain the "best fit" between the simulated values of $[C]_{PES}$ ($[C]_{PES}$ SIM) and the observed data i.e. to maximise the coefficient describing the correlation between simulated and observed concentrations.

The generic model is an attempt to describe the river system in a more mechanistic manner than the purely empirical Q-C modelling method. Estimates of the proportion of discharge abstracted and the proportion of that discharge returning to the system as diffuse flow, are incorporated into the model. In addition, the accuracy of the model can be improved by inclusion of the concentrations and discharges of significant point sources. Therefore, if these parameters are changed due to the system being operated in a different way, it should be necessary to alter only the appropriate value in the generic equation. For example if an additional source of effluent is to be discharged just upstream of the site, by incorporating the concentration and discharge into the model, the resulting water quality can be predicted. Further work needs to be carried out to test this modelling method.

2.8 Application to the Pienaars River

This Q-C modelling exercise was carried out in retrospect (i.e. after the Reserve workshop had been held). Water quality in the Pienaars River, division of the river into ecoregions and into three water quality reaches (WQ 1, WQ 2 and WQ 3), identification of monitoring sites and potential sources of pollution are described in the Water Quality

Report for the Pienaars River Intermediate Reserve Determination (Rossouw, Hohls and Jooste 1999). Unless otherwise specified, the data used in this exercise were supplied by the specialists involved in the Reserve Determination for the Pienaars River.

A schematic representation of the Pienaars River system (adapted from Rossouw *et al.* 1999) is shown in Figure 2.2. Also indicated are the important DWAF water quality monitoring and gauging sites, as well as the water quality reaches.

2.8.1 <u>Results</u>

In order to explain the modelling method and to highlight some of the associated issues, the results for modelling of TDS in Water Quality reach one (WQ 1, i.e. Pienaars River, above Roodeplaat Dam) are discussed in detail below. This is presented in the form of the sequential steps of the modelling method, followed by the results for each step. Selected results for other water quality variables and for the other two reaches are discussed in section 2.8.2. A summary of all the results for the Pienaars River is given in Table 2.3 in the same section. The Q-C plots for all the variables modelled in the Pienaars River, as well as for the other systems modelled during the course of the project are given in Appendix B.

2.8.1.1 TDS in Water Quality Reach 1.

The steps of the Q-C modelling method are as follows:

- **i)** *Collate all available data on water quality, point-sources of pollution, hydrological structures, hydrology, land-use, topography etc. Identify the locations of the IFR sites relative to water quality monitoring stations and discharge gauging sites, as well as any other significant hydrological features. Produce maps indicating the above.*
- **ii)** *Identify the different ecoregions through which the river flows according to the method of Kleynhans (1998). Using this information, as well as the location of dams and significant tributaries (hydrological features), derive reaches within which water quality would be expected to be uniform.*

The results from the above two steps are discussed in Rossouw *et al.* (1999).

- **iii)** *For each IFR site, using the discharge and water quality data from the nearest appropriate gauging and water quality monitoring station, correlate monthly mean*

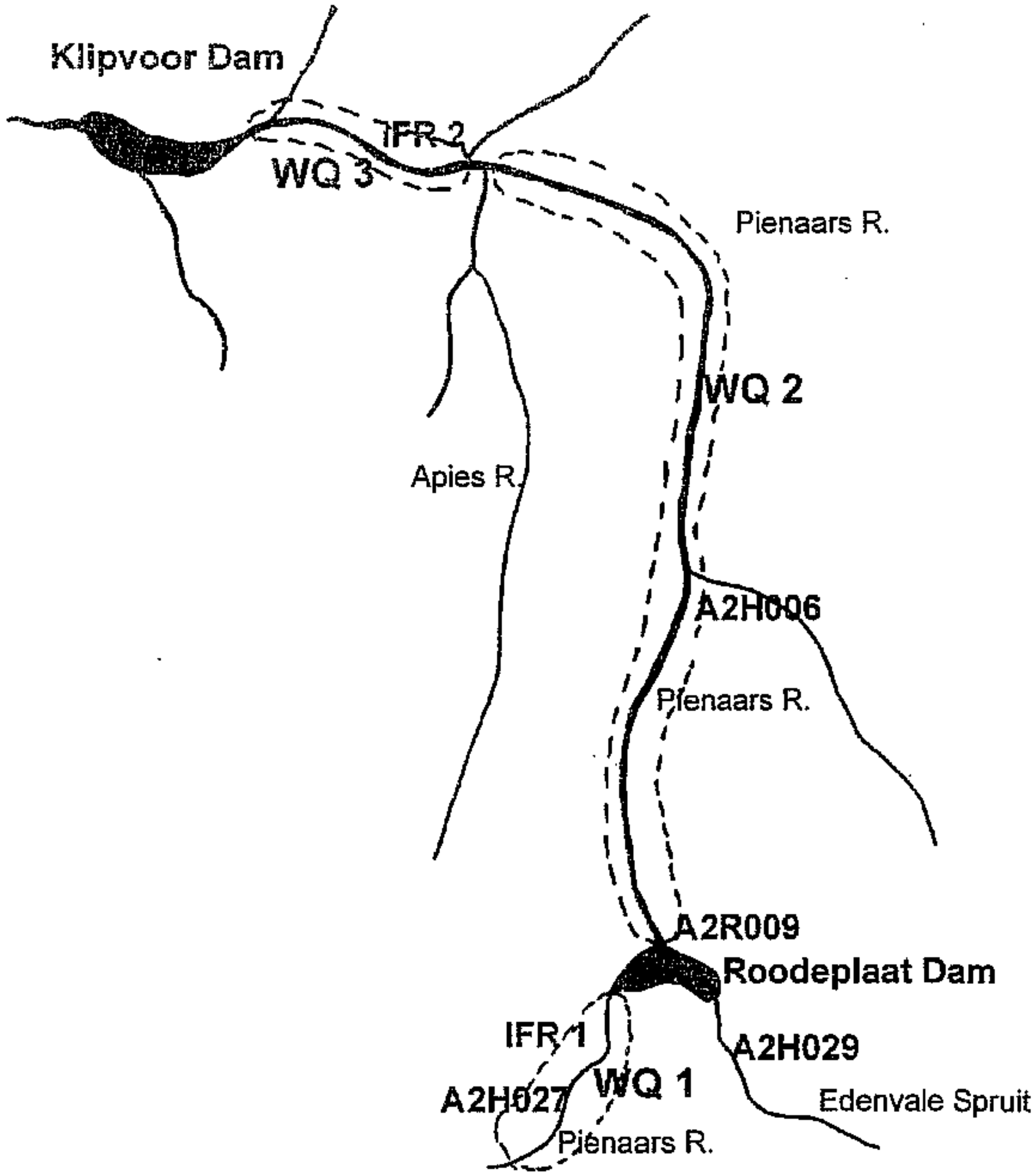


Figure 2.2 *Schematic representation of the Pienaars River system showing the water quality reaches (WQ 1, 2 and 3), significant hydrological features, IFR sites and water quality monitoring stations.*

discharge values as well as median monthly concentration values for each water quality variable (C).

The Reference Condition values for discharge were obtained from WR90, naturalised simulated data (Midgeley, Pitman and Middleton 1994) for the gauging site A2H027 (Pienaars River at Baviaanspoort) since natural observed discharge data for the site were not available. Present day discharge was taken from DWAF observed data at the same site (time period1988-1998). Water quality data for the Reference Condition was derived from a monitoring site on a nearby tributary (A2H029Q01 Edenvale Spruit at Leeuwfontein), since no pre-impact water quality data were available for Baviaanspoort. The time period used was 1993-1998. Present-day monthly TDS median values were derived from Baviaanspoort (time period 1993-1998). The location of the monitoring stations relative to the water quality reach modelled is shown in Figure 2.2. The data are presented in Table 2.1. It can be seen from the table that present day water quality is considerably impacted compared to the natural state.

- **iv)** *Examine the relationship between discharge and the concentration of C.*

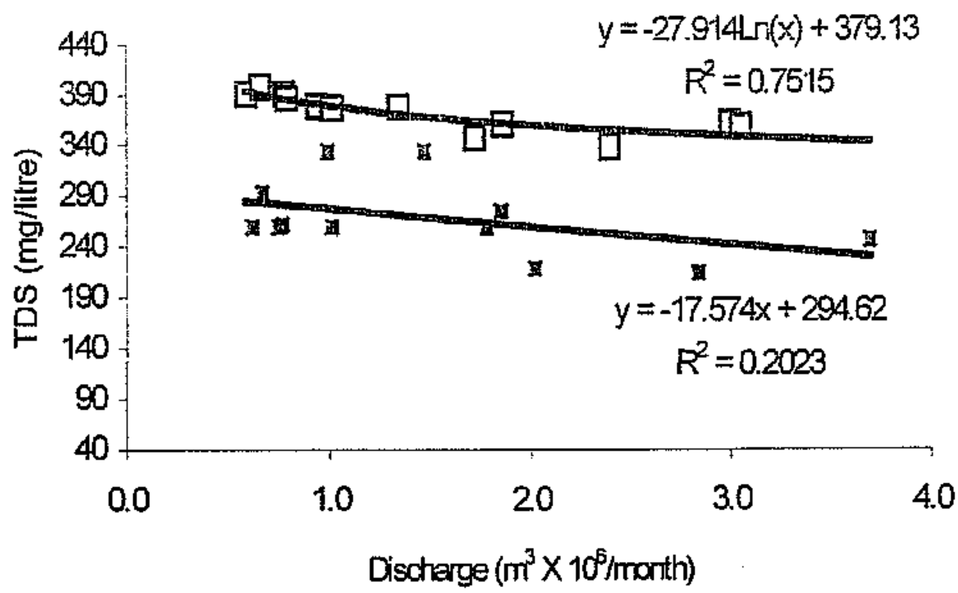
TDS in this reach was found to decrease with increasing flow both in the Reference Condition and in the Present State (Figure 2.3 a). The data for TDS concentration in the Reference Condition were found to exhibit considerable scatter, and was best described by a linear relationship which was used to simulate the reference condition concentration of TDS for a given discharge (Q). The measured data for TDS in the Present State were described most closely using a logarithmic relationship.

- **v)** *Calculate the simulated concentrations of variable C for the Reference Condition and Present Ecological State.*

The simulated values of the variable (C) in the Reference Condition were calculated using the regression equation "N" ($[C]_{RC}$=-18(Q)+295). The values are shown in Table 2.1 as well as in Figure 2.3b (line $[TDS]_{RC}$ simulated). Likewise, the simulated values of C in the Present State were calculated using the regression equation "M" ($[C]_{PES}$= -28(LnQ)+379). These values are also shown in Table 2.1 and in Figure 2.3b (as line $[TDS]_{PES}$ simulated). A reasonably good correlation (correlation coefficient = 0.867) was obtained between the simulated and measured TDS concentration values. The 95% confidence interval for the Present Ecological State model is also shown.

Table 2.1 *Measured discharge and TDS data, as well as simulated values for water quality reach 1 (WQ1), of the Pienaars River. Natural (Reference Condition) and current (Present Ecological State) discharge given as mean monthly values (m^3x10^6), TDS concentrations (mg/litre) reported as monthly median values for the reference condition ([TDS] RC) as well as the present state ([TDS] PES). Simulated Reference and Present state concentrations of TDS derived from the functions $y=-18x+294$ ("N") and $y=-28Lnx+379$ ("M") respectively.*

Month	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Mean
RC discharge	0.99	1.47	1.85	3.69	2.84	2.02	1.78	1.01	0.77	0.75	0.67	0.62	**1.54**
[TDS] RC	333	334	275	246	214	219	259	259	261	259	293	259	**268**
[TDS] RC simulated	277	269	262	230	245	259	263	277	281	281	283	284	**268**
PES discharge	0.94	1.34	1.86	3.1	3.05	2.4	1.72	1.01	0.79	0.78	0.66	0.59	**1.51**
[TDS] PES	378	378	361	361	358	338	347	377	386	391	398	391	**372**
[TDS] PES simulated	381	371	362	348	348	355	364	379	386	386	391	394	**372**



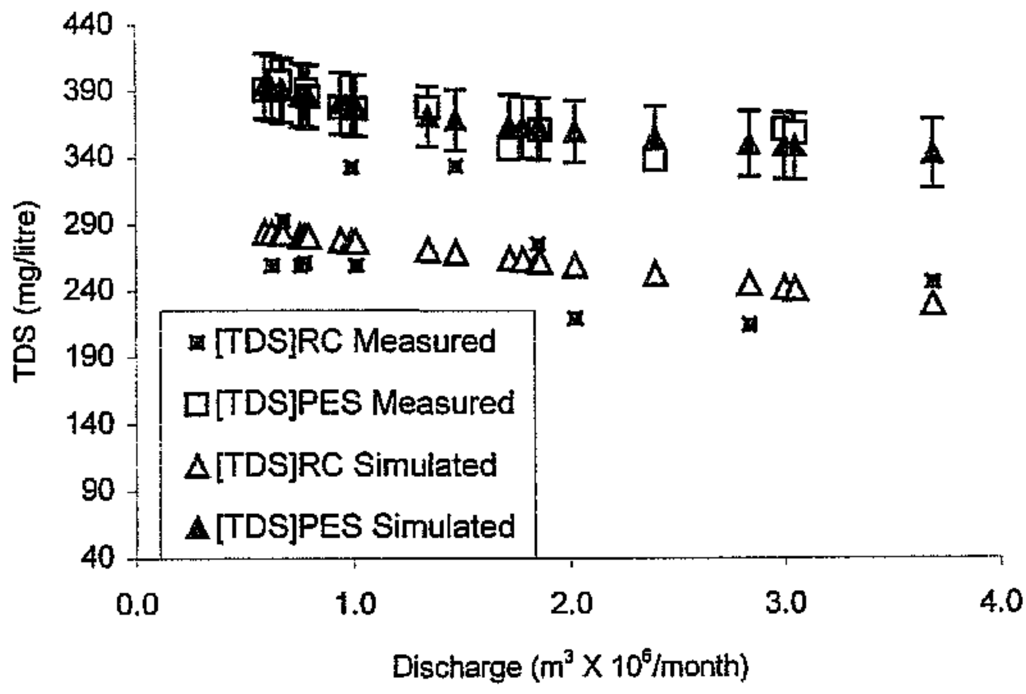


Figure 2.3 *Q-C plot for TDS in Water Quality Reach 1 on the Pienaars River.* ***(Above)*** *Measured Reference Condition and Present Ecological State concentrations of TDS showing the optimum regression (trendline) through each.* ***(Below)*** *The same plot, showing the regression lines and 95% confidence interval for the Present Ecological State.*

- **vi)** *Predict the concentration of C for the prescribed IFR discharge regime.*

The monthly discharge values for maintenance and drought baseflows and the corresponding TDS concentrations are presented in Table 2.2. It can be seen, by comparing Table 2.1 and 2.2, that under the recommended maintenance baseflow regime, TDS will remain more or less the same, with a mean TDS concentration of 375 mg/litre compared to 372 mg/litre at present. Under drought baseflow conditions however, mean TDS levels will be elevated to 402 mg/litre.

- **vii)** *Calculate the extent of deviation of the present state values of C and the predicted concentrations of C from the Reference Condition, or in the case of nutrients, from a pre-defined value.*

The percentage deviation of the Present Ecological State from the Reference Condition, for each month is recorded in Table 2.2. Also shown is the deviation from Reference Conditions of the predicted concentrations for each month, under both maintenance and drought baseflows. As is to be expected, under the recommended maintenance baseflow, percentage deviation from the Reference Condition is the same as at present in the system (approximately 39-40%). This will increase to 50% under drought baseflow conditions.

- **viii)** *Assign the Assessment category for each month.*

The assessment category that the predicted TDS concentration would fall into for each month under either maintenance baseflow or drought baseflow is also shown in Table 2.2. In the case of the Pienaars River Reserve determination, the assessment category was determined using the method detailed in the manual for Resource Directed Measures (DWAF 1999). This method is based on the percentage deviation from the reference condition concentration for each month. Where the predicted TDS concentration is more than 20% higher than the concentration for the same month in the Reference Condition, a category of "C" is assigned, if greater than 30% a "D" is assigned, and if greater than 40% an E/F category is assigned. From Table 2.2 it can be seen that the mean assessment category for TDS under the current conditions in the system in Water Quality reach one is a "D". This would not be expected to change under the recommended maintenance baseflow, but would drop a class (to "E/F") under drought conditions, if there were no amelioration of the current pollution loading.

Table 2.2. *Monthly discharge values (Million m^3/month) and TDS values for a recommended flow regime (under both maintenance and drought baseflow) for Water Quality reach 1 (WQ1) of the Pienaars River. The corresponding predicted monthly % deviation from Reference Condition concentrations of TDS (mg/litre) and Assessment Category are also shown. Assessment Category derived as described in the text. In the final column the mean annual value is given. Measured and simulated RC and Present Ecological State (PES) values of TDS given in Table 2.1. The values in boxes are the predicted critical values for the recommended maintenance and drought flow (referred to in Chapter 6).*

Month		**Oct**	**Nov**	**Dec**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**June**	**July**	**Aug**	**Sept**	**Mean**
Maintenance baseflow	Discharge ($m^3 \times 10^6$)	**1.0**	**1.3**	**1.5**	**3.0**	**2.2**	**1.8**	**1.5**	**1.0**	**0.7**	**0.7**	**0.6**	**0.5**	**1.3**
Drought baseflow		0.5	0.6	0.8	1.0	0.9	0.8	0.7	0.5	0.3	0.2	0.1	0.2	**0.5**
Maintenance baseflow	Concentration (mg/litre)	**379**	**372**	**368**	**348**	**357**	**363**	**367**	**379**	**389**	**389**	**394**	$\boxed{397}$	**375**
Drought baseflow		**405**	**393**	**385**	**379**	**382**	**385**	**389**	**398**	**413**	**424**	$\boxed{443}$	**424**	**402**
Present ecological state	%Deviation of the PES and predicted C from RC	37	38	38	52	42	37	36	37	37	37	38	39	**39**
Maintenance baseflow		37	38	40	52	46	40	39	37	38	38	39	40	**40**
Drought baseflow		46	46	47	65	56	49	48	44	47	51	57	49	**50**
Present ecological state	Assessment category	**D**	**D**	**D**	**E/F**	**E/F**	**D**	**D**	**D**	**D**	**D**	**D**	**D**	**D**
Maintenance baseflow		**D**	**D**	**E/F**	**E/F**	**E/F**	**E/F**	**D**	**D**	**D**	**D**	**D**	**D**	**D**
Drought baseflow		**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**	**E/F**

RC = Reference condition, PES = Present ecological state

2.8.2 Points arising from this application

Table 2.3 shows a summary of the modelling results for the Pienaars River. For each of the three water quality reaches, the table indicates which DWAF station was used as data source, as well as the time-period of the discharge and concentration data used in the model. The table also shows, for each variable modelled, the trend in concentration with increasing discharge, and in addition, the accuracy of the simulation (given by the correlation coefficient between simulated and measured data).

- In the case of the Pienaars River, observed discharge data from the nearest DWAF gauging weir were used to represent discharge for the entire reach. In the case of many of the other studies, the discharge data used were provided by the specialist hydrologist for the Reserve determination. The data were usually simulated data (from rainfall-runoff, or from some other type of hydrological modelling) and represented the discharge at a given IFR site.
- Although the Pienaars River was divided into three water quality reaches, no modelling could be undertaken for the lowest reach because of the lack of water quality data. This was compounded by the fact that the system is fairly complex in this area, with several tributaries, an impoundment (Klipvoor dam), and associated wetlands and floodplains.
- Two IFR sites had been identified for the Reserve determination process (see Figure 2.2), namely IFR site I, situated in Water Quality reach 1 (WQ 1) and IFR site 2, situated in WQ 3 (for which modelling could not be undertaken). Modelling was therefore undertaken for WQ 1 (and would be applicable to IFR site 1) and WQ 2 (for which there was no designated IFR site).
- The present day water quality of the Pienaars River is relatively impacted, a consequence of extensive development within the catchment (Rossouw *et al.* 1999). In comparison with the Q-C plot for the Present Ecological State, that for the Reference Condition for many of the variables showed only slight change with discharge; on occasion, an almost horizontal line was obtained. Consequently the value of r^2 was very low, indicating that a very small proportion of the variation in concentration was due to X-axis variable, namely discharge (Zar 1984).

Table 2.3 Summary of the results obtained using Q-C modelling for all water quality variables and reaches modelled for the ***Pienaars River****. The source and time-period used for RC and PES discharge and water quality data is shown. For each variable, the trend in concentration with increasing discharge is shown as well as the regression coefficient (r^2), for the RC and PES. The accuracy of the simulation is given by the correlation coefficient between simulated PES and measured PES values. All concentrations expressed as mg/litre.*

WQ Reach	Water quality data		Discharge data		WQ Var.	RC Q-C relationship		PES Q-C relationship			Comments
	RC	PES	RC	PES		Q-C trend	r^2	Q-C trend	r^2	corr.coeff.	
1	A2H029 -Q01 '93-'98	A2H027 -Q01 '93-'98	WR90 data for A2H027	A2H027 observed data '88-'98	TDS	↓(linear)	0.202	↓(log)	0.752	0.867	
					PO_4^{-3}	v.slight ↑ (linear)	0.631	↓(power)	0.820	0.877	PES PO_4^{-3} levels very high compared to RC.
					Tot P	no change (linear)	-	↓(power)	0.680	0.782	
					NO_3^- + NO_2^-	v.slight ↑ (linear)	0.319	↓(log)	0.464	0.680	
					NH_3	scatter	-	scatter	-	-	Data points scattered. Not modelled
					pH	slight ↓ (linear)	0.318	no change (linear)	0.054	0.827	pH is largely independent of Q.
2	A2H006 -Q01 '76-'81	A2H006 -Q01 '93-'98	WR90 data for A2H006	A2H006 observed data '90-'98	TDS	↓(linear)	0.086	↓(power)	0.705	0.835	Good simulation
					PO_4^{-3}	↓(log)	0.508	slight ↑	0.826	-	↓ at high discharges in PES.
					Tot P	No data		No data			
					NO_3^- + NO_2^-	scatter	-	scatter	-	-	Not modelled
					NH_3	scatter	-	scatter	-	-	Not modelled
					pH	no change (linear)	-	no change (linear)	-	-	pH is largely independent of Q.
3	No data				Modelling not carried out						

Q = discharge, C = concentration, RC= Reference condition, PES = Present ecological state, ERC = Ecological Reserve Class (previously termed Ecological Management Class).
Lin = linear, log = natural logarithm, exp = exponential, pow = power.

- Current levels of ortho-phosphate in WQ 1 were considerably higher than in the Reference Condition and showed a marked decrease with increasing discharge. This is likely to be due to the discharge of sewerage into this reach of the river (Rossouw *et al.* 1999). A good correspondence between measured and simulated values of ortho-phosphate concentration was obtained (correlation coefficient = 0.877). Ortho-phosphate in WQ 2, on the other hand, was found to increase with discharge at low flows and then level off at higher flows. A discussion of typical Q-C trends for ortho-phosphate was presented in section 1.6.1. The increase with discharge under lower flows may well be the result of wash-off from surrounding lands, or due to churning of benthic sediments and the concomitant release of ortho-phosphate into the water column. The levelling off at higher flows is likely to be a result of substrate becoming limiting.
- Nitrogen (combined nitrate and nitrite, as well as ammonia), were modelled in Water Quality reaches 1 and 2 with varying degrees of success (Table 2.3). Simulations of ammonia in both water quality reaches were not possible since there was a poorly defined relationship between discharge and concentration for this chemical constituent in both sections of the river. Concentrations of un-ionised ammonia (NH_3) were derived from ionised ammonia concentrations (NH_4^+), taking into account ambient water temperature and pH, and using the method prescribed in DWAF (1996). Monthly mean water temperature data were calculated from daytime air temperature taken from the nearest weather station from which a value of 2 degrees Celsius was subtracted (DWAF 1999).
- It should be noted that in WQ 2, nutrients were modelled in two different ways. Firstly, by using the same approach as Rossouw *et al.* (1999) and assuming that the impoundment of water in Roodeplaat dam resets water quality. Thus data from A2R009 (Roodeplaat dam) were used to set the Reference Condition. Data from a monitoring station downstream (A2H006, Klipdrift) were then used to describe the present state. Ortho-phosphate levels were found to be higher in the Reference Condition than for the Present Ecological State. Combined nitrate and nitrite were also slightly higher in the former than the latter, using this approach. In section 2.6 allusion was made to the fact that often nutrients, and in particular phosphorus, become bound to sediments within impoundments. As sediments settle to the bottom, this effectively removes them from the water column. If no sluice gates are present (as is the situation at Roodeplaat dam) nutrient-laden sediments are retained

within the system and water released from the impoundment tends to be low in nutrients (compared to water flowing into the system). The Pienaars River is highly impacted and Roodeplaat dam tends to be eutrophic, however. Consequently, the sediments are likely to be saturated with nutrients (although nutrient dynamics within impoundments can be complex and this cannot be stated with complete certainty without specialised research). In any case, the resultant effect is that nutrients are not retained within the impoundment and the assumption that the dam resets water quality is not valid when trying to model nutrients in this system.

In the second approach that was used to model nutrients, the water quality data set from A2H006 (Klipdrift) was divided into RC (1976-1981) and PES (1993-1998). Note that Roodeplaat dam was constructed in 1957 and that no pre-construction water quality data were available. This approach was more successful in that reference condition concentrations were lower than those representing the Present Ecological State for all variables. Despite this, there was limited success in modelling nutrients due to the considerable scatter of data points.

2.9 Application to the Palmiet River (Western Cape)

In the case of the Palmiet River, as for the Pienaars River, water quality modelling was carried out after the IFR workshop had taken place. The focus was therefore on further development of the modelling method and the accuracy of the simulations, rather than the results in terms of assessment class and concentrations. Current and reference condition water quality in the Palmiet River are described in Dallas (1998).

Initial attempts had been made during an early trial Reserve determination on the Crocodile River (Mpumalanga) to describe the variation of Present Ecological State water quality with discharge using a mechanistic, mass balance modelling method (Bath, A., Water corporation, Perth, Australia, *pers. comm.*). The above modelling method was developed further during the study on the Palmiet River and resulted in the "generic model" which was described in section 2.7.1. The generic model is given by the following equation:

$$[C]_{PES}\ SIM = \frac{((Q_{RC} \times (1-P_{ABS}) \times \text{"N"}) + ((Q_{RC} \times P_{ABS} \times P_{RF}) \times (\text{"M"})) + P_x}{((Q_{RC} \times (1-P_{ABS})) + (Q_{RC} \times P_{ABS} \times P_{RF})}$$

....................equation 7

where:

$[C]_{PES}$ SIM = the simulated concentration of C in the Present Ecological State.

Q_{RC} = Discharge under Reference Conditions.

P_{ABS} = Proportion of discharge that is removed due to abstraction

P_{RF} = Proportion of P_{ABS} that re-enters the system as return flow.

Px = additional loads of C

"N" = a function relating the reference condition concentration of C to discharge

"M" = a function relating the present state concentration of C ($[C]_p$) to discharge (Q) And: a, b, c and d are empirically derived constants.

Estimates of P_{ABS} and P_{RF} were derived from a comparison between natural runoff (Pitman runoff estimates as given in Midgley *et al.* 1994) and current discharge, as well as in consultation with hydrologists involved with the instream flow assessment. Estimates of point source loads were made in consultation with water quality specialists.

2.9.1 Points arising from this application

The Q-C plots for the Palmiet River are shown in Appendix B. Table 2.4 summarises the modelling results.

- As in the case of the Pienaars River, because of a lack of present-day water quality data, not all sites could be modelled. Because of this, only 2 out of a total of 4 sites (IFR 2, in the foothill region of the river, and IFR 4 in the transition region) were modelled.
- The most important consideration arising from this application was the suitability and usefulness of the refinement of the "generic model" for which good simulations were obtained for most water quality variables.

*Table 2.4 Summary of the results obtained using Q-C modelling for all water quality variables modelled for the **Palmiet River**. The source and time-period used for RC and PES discharge and water quality data is shown. For each variable, the trend in concentration with increasing discharge is shown as well as the regression coefficient (r^2), for the RC and PES. The accuracy of the simulation is given by the correlation coefficient between simulated PES and measured PES values. All concentrations expressed as mg/litre.*

IFR site	Water quality data		Discharge data		WQ Var.	RC Q-C relationship		PES Q-C relationship			Comments
	RC	PES	RC	PES		Q-C trend	r^2	Q-C trend	r^2	corr.coeff.	
1		No data									Not modelled
2	G4H029 -Q01 '87-'89	G4H029 -Q01 '93-'98	Simulat-ed	Simulat-ed	TDS	↓(log)	0.498	slight ↓(log)	0.344	0.455	PES just above RC
					PO_4^{-3}	slight ↑ (linear)	0.408	↑(linear)	0.807	0.895	Both PES and RC PO_4^{-3} levels increase with Q.
					Tot P	no data	-	no data	-	-	Not modelled
					NH_3	scatter	-	scatter	-	-	Data points scattered, not modelled
3		No data									Not modelled
4	G4H007 -Q01 '78-'88	G4H007 -Q01 '93-'98	Simulat-ed	Simulat-ed	TDS	↑(log)	0.839	↑(log)	0.514	0.678	Both RC and PES increase with Q.
					PO_4^{-3}	slight ↑(lin)	0.676	↑(linear)	0.883	0.940	
					Tot P	No data		No data			
					NH_3	slight ↓(log)	0.546	↓(exp)	0.494	0.098	Conc. rises sharply at low Q, difficult to simulate.

Q = discharge, C = concentration, RC= Reference condition, PES = Present ecological state, ERC = Ecological Reserve Class (previously termed Ecological Management Class).
Lin = linear, log = natural logarithm, exp = exponential, pow = power.

It was not always possible to obtain estimates of many of the required parameters (Pabs, Prf, Px etc.) however, and educated guesses had to be made of these values. By adjusting the values of the parameters empirically, as well as the values of the constants a, b, c, and d, good simulations could be obtained. Because accurate values of the above parameters were not available, the ability of the model to actually describe the system could not be verified, however. Furthermore, since the parameters were adjusted empirically to fit the regression line, exactly the same results were obtained with Q-C modelling as with mass balance modelling. In addition, setting up the generic equation is a complex, time-consuming process and it was easy to make mistakes. Thus, although the generic model was able to describe the Q-C relationships for the Palmiet River (as well as for the Olifants and the Breede systems), it was not particularly useful to use mass balance modelling, because accurate values of the various parameters (Pabs etc.) could not be obtained. Situations may well occur, however, when such information is available and the extra time and effort involved in setting up the equation will be justified.

- TDS at IFR 4, as well as phosphate at both sites, increased with discharge. This topic is discussed further under consideration of application of Q-C modelling to the Breede River.

2.10 Application to the Olifants River (Mpumalanga)

The Olifants River Ecological Water Requirement Assessment (OREWRA) entailed determination of the IFR for 16 sites, and was the first Reserve determination (as far as these authors are aware) in which steps were taken to integrate water quality and quantity numerically. Modelling of the downstream temperature of water released from an impoundment had been undertaken, however, as part of the Environmental flow assessment for the Lesotho Highlands Project (Skoroszewski 1997). The water quality modelling results for the OREWRA are documented in Malan (2001), and are therefore not discussed in detail here. The Q-C plots for all the sites modelled as part of the Olifants River study are shown in Appendix B. A table summarising the results for all the sites that were modelled is shown in Table 2.5. Due to lack of water quality data representative of the current state, modelling was not undertaken for IFR sites 2, 9 and 14B; due to unreliable discharge data, modelling was not carried out for IFR sites 4 and

6. There is extensive mining in the catchment, and because instream sulphate concentrations were a concern this variable was also modelled, as was fluoride at IFR sites 14B and 15.

2.10.1 Points arising from this application

The following points form this application are noteworthy.

- The recommended flow regime (as proposed by the specialists) is for *baseflow* - in other words, this represents the minimum flow required in the system to maintain the prerequisite state. Frequently, higher discharges would be present as a result of floods, freshes or additional flow (i.e. water that is not required for abstraction or impoundment). The predicted concentrations of water quality constituents in the case of those that showed a dilution effect with increased discharge, therefore represent the worst-case scenario. For a consideration of those constituents that increase with discharge, see section 2.11.
- Although ortho-phosphate increased slightly with discharge at two sites, the change was very slight. The recommended flow regime should therefore not lead to a change in assessment class for ortho-phosphate. Nitrates were found to give poor simulations and therefore the results are not recorded in Table 2.5.
- In contrast to the situation in the Breede and Palmiet Rivers, TDS in the Olifants River system, as in the Pienaars River, did not increase with discharge at any of the sites. The relationship between discharge and TDS is discussed in section 2.11.

***Table 2.5** Summary of the results obtained using Q-C modelling for all water quality variables modelled for the **Olifants River system**. The source and time-period used for RC and PES discharge and water quality data is shown. For each variable, the trend in concentration with increasing discharge is shown as well as the regression coefficient (r^2), for the RC and PES. The accuracy of the simulation is given by the correlation coefficient between simulated PES and measured PES values. All concentrations expressed as mg/litre.*

IFR site	Water quality data		Discharge data		WQ Var.	RC Q-C relationship		PES Q-C relationship			Comments
	RC	PES	RC	PES		Q-C trend	r^2	Q-C trend	r^2	corr.coeff.	
1	B1H002 Q01 & B1H006 Q01	B1H002 Q01 & B1H010 Q01	Simul-ated	B1H002 & B1H010	TDS	no change	-	slight ↓ (log)	0.341	0.597	Mass balance calculation used to derive WQ.
					SO_4^{-2}	↓ (log)	0.427	↓ (log)	0.312	0.479	
					PO_4^{-3}	↓ (linear)	0.016	↓ (log)	0.122	0.350	
3	B1H026 Q01	B1H015 Q01	Simul-ated	B1H015	TDS	slight ↓ (log)	0.044	slight ↓ (log)	0.066	0.24	
					SO_4^{-2}	no change	0.006	slight ↓ (log)	0.323	0.569	
					PO_4^{-3}	scattered	-	no change	-	-	'RC higher than PES
5	*B3H007Q01 ('92-'98)	B3H001 Q01	Simul-ated	Simul-ated	TDS	↓ (log)	0.233	↓ (power)	0.147	0.499	
					SO_4^{-2}	no change	-	↓ (log)	0.509	0.713	
					PO_4^{-3}	scattered	-	scattered	-	-	Not modelled
7	*B3H007Q01 ('92-'98)	B5H002 Q01	Simul-ated	Simul-ated	TDS	no change	-	marked ↓ (log)	0.513	0.716	No PES water quality data after 1988
					SO_4^{-2}	no change	-	↓ (log)	0.415	0.778	
					PO_4^{-3}	no change	-	slight ↓ (linear)	0.201	0.130	RC slightly higher than PES
8	B7H013 Q01 (entire data series)	B7H009 Q01 (entire data series)	Simul-ated	Simul-ated	TDS	↓ (linear)	0.473	marked ↓ (log)	0.779	0.889	
					SO_4^{-2}	no change	-	↓ (log)	0.778	0.883	
					PO_4^{-3}	slight ↓ (log)	0.026	slight ↑ (power)	0.537	0.708	

*Table 2.5 Summary of the results obtained using Q-C modelling for the **Olifants River system** cont.*

IFR site	Water quality data		Discharge data		WQ. Var.	RC Q-C relationship		PES Q-C relationship			Comments
	RC	PES	RC	PES		Q-C trend	r^2	Q-C trend	r^2	corr.coeff.	
10	B4H007 Q01	B4H011 Q01	Simul-ated	Simul-ated	TDS	↓(power)	0.859	↓(power)	0.797	0.879	
					SO_4^{-2}	no change	-	↓(power)	0.608	0.529	
					PO_4^{-3}	no change	-	no change	-	-	
11	Mean of B7H009 ('79-'84) & B7H013 Q01	B7H009 Q01 (entire data series)	Simul-ated	Simul-ated	TDS	↓(log)	0.791	marked ↓ (log)	0.798	0.894	RC & PES discharge derived by adding IFR 8 + IFR 10
					SO_4^{-2}	↓(power)	0.576	↓(log)	0.756	0.869	
					PO_4^{-3}	slight ↑ (linear)	0.065	slight ↑ (power)	0.521	0.687	
12	B6H001 Q01	B6H004 Q01	Simul-ated	Simul-ated	TDS	↓(log)	0.896	↓(log)	0.685	0.828	RC & PES similar values
					PO_4^{-3}	no change	-	no change	-	0.13	
13	*Mean of B6H004 & B7H007	*B7H007Q01 ('92-'97)	Simul-ated	B7H007	TDS	↓(linear)	0.328	↓(log)	0.256	0.894	RC WQ data derived from mean of Blyde R. and Olifants at Oxford ('92-'97).
					SO_4^{-2}	-	-	↓(linear)	0.143	0.838	
					PO_4^{-3}	slight ↑ (linear)	0.661	slight ↑ (linear)	0.754	0.887	
14B	B7H002 Q01	B7H019 Q01	Simul-ated	B7H019	TDS	no change	-	slight ↓ (log)	0.115	0.338	
					SO_4^{-2}	no change	-	slight ↓ (log)	0.102	0.319	
					F	no change	-	slight ↓ (log)	0.392	0.624	
15	*As above	*B7H015Q01	Simul-ated	Simul-ated	TDS	slight↓ (exp)	0.359	marked ↓ (log)	0.764	0.926	
					SO_4^{-2}	-	-	marked ↓ (log)	0.751	0.938	Note Mamba used as PS water quality
					PO_4^{-3}	slight ↑ (linear)	0.671	↓(log)	0.267	0.611	
					F	no change	-	↓(log)	0.772	0.928	
16	*As above	*B7H018Q01	Simul-ated	Simul-ated	TDS	no change	-	↓(log)	0.764	0.965	Same flow and RC water quality data as IFR 15

Q = discharge, C = concentration, RC= Reference condition, PES = Present ecological state, ERC = Ecological Reserve Class (previously termed Ecological Management Class). Lin = linear, log = natural logarithm, exp = exponential.

2.11 Application to the Breede River

The Breede River Basin study entailed (amongst other aspects) determination of the environmental flow requirement for six sites. Three sites were situated on the Breede River itself, one on the Molenaars River and two sites on the Riversonderend system (tributaries of the Breede River). The Q-C plots for all the sites that were modelled are shown in Appendix B, and the results are summarised in Table 2.6.

2.11.1 Points arising from this application

The most important point arising from the application of Q-C modelling to the Breede River Basin study was the Q-C trend of TDS in salinised catchments in the winter rainfall region. This is discussed in some detail below and is followed by a consideration of nutrients, which were also found, on occasion, to be positively correlated with discharge.

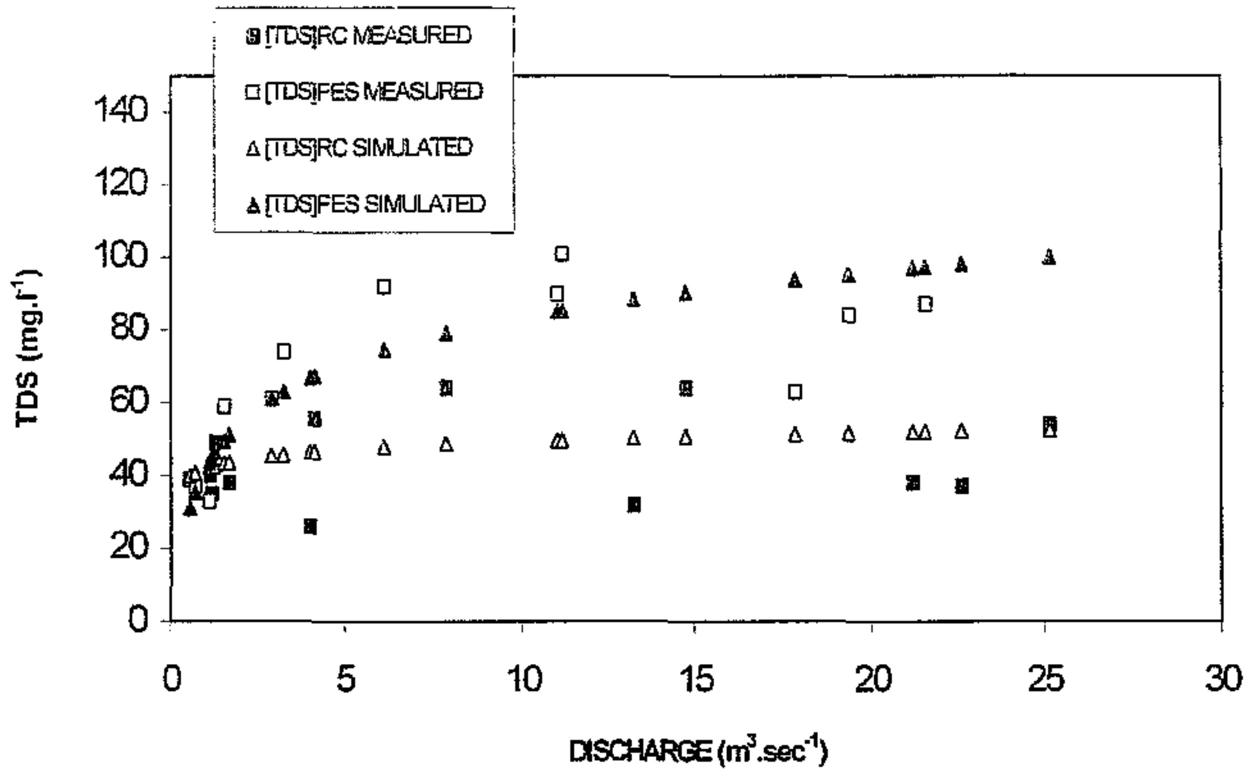
2.11.1.1 TDS Q-C trends in salinised catchments

Two sites on the Breede River, as well as one on the Palmiet River, showed marked increases in TDS with increasing discharge. An example of such a Q-C plot for IFR 1 (Mooiplaas) on the Breede River is shown in Figure 2.4. Examination of the seasonal TDS distribution for that site reveals that the highest instream levels of this constituent occur during winter and the lowest during summer. It can also be seen that the increase in TDS is coincident with the onset of the wet season in autumn (April). It was explained in section 1.6.1 that irrigation of surrounding agricultural lands during the hot dry summer results in a build-up of salts in the soil. The marked elevation of instream concentrations of this constituent during autumn and winter is the result of wash-off of salts from the surrounding catchment by rain. Another point of interest at this site is that by September, although rainfall (and hence streamflow) has started to decrease again as the year progresses towards summer, instream TDS levels are still very high. This illustrates the fact that TDS is not limiting in the surrounding soils, or put more simply, that the winter rainfall was not enough to wash out all the accumulated salts. No cases of positive Q-C trends for TDS were found on the Pienaars or Olifants River systems, probably because these catchments receive their rainfall primarily in the summer and hence a build-up of salts due to evaporation does not occur (or at least is much slower). Contributions to high instream TDS values from salinised groundwater has also been postulated to be a contributing factor to this problem in the Breede River catchment (Kirchner 1995).

***Table 2.6** Summary of the results obtained using Q-C modelling for all water quality variables modelled for the **Breede River system**. The source and time-period used for RC and PES discharge and water quality data is shown. For each variable, the trend in concentration with increasing discharge is shown as well as the regression coefficient (r^2), for the RC and PES. The accuracy of the simulation is given by the correlation coefficient between simulated PES and measured PES values. All concentrations expressed as mg/litre.*

IFR site	Water quality data RC	Water quality data PES	Discharge data RC	Discharge data PES	WQ Var.	RC Q-C relationship: Q-C trend	RC Q-C relationship: r^2	PES Q-C relationship: Q-C trend	PES Q-C relationship: r^2	PES Q-C relationship: corr.coeff.	Comments
1	H1H006 '79-'82 Breede R.	H1H006 '95-'99 Breede R.	Simulated		TDS	slight ↑(log)	0.043	↑(power)	0.704	0.826	TDS increases withQ
					$NO_2^- + NO_3^-$	no change	-	↑(power)	0.764	0.806	
					PO_4^{-3}	no change	-	no change	-	-	
					Na^+	scattered	-	↑(log)	0.665	0.816	
2	H1H018 Mol. R.	H1H018 Mol. R.	Simulated		TDS	no change	-	no change	-	-	RC & PES similar values
					$NO_2^- + NO_3^-$	no change	-	↓(log)	0.564	0.751	Molenaars River = tributary of Breede River
					PO_4^{-3}	no change	-	slightly ↑(lin)	0.055	-	
3	H4H017 '80-'82 Breede R.	H4H017 '94-'98 Breede R.	Simulated		TDS	-	-	↑then ↓ (polynomial)	0.546	0.620	Highly regulated system. ↑at low Q, ↓at high Q.
					$NO_2^- + NO_3^-$	–	-	↑(log)	0.830	0.911	
					PO_4^{-3}	-	-	↑then ↓ (polynomial)	0.295	-	
4	H7H006 Breede R.	H7H006 '94-'98 Breede R.	Simulated		TDS	↓(exp)	0.633	↓(exp)	0.584	0.728	data points scattered
					$NO_2^- + NO_3^-$	no change	-	↑(log)	0.810	0.896	Marked wash-off effect in PES
					PO_4^{-2}	no change	-	↑(log)	0.425	0.6517	
					Na^+	↓(exp)	0.588	↓(exp)	0.582	0.728	
6	H6H005 '79-'81 Baviaan R.	H6H005 '94-'98 Baviaan R.	Simulated		TDS	no change	-	no change	-	-	Baviaans River = tributary of Riversonderend.
					SO_4^{-2}	no change	-	no change	-	0.883	
					PO_4^{-3}	v. slight ↓ (lin)	0.08	v. slight ↑(log)	0.239	0.517	

Q = discharge, C = concentration, RC= Reference condition, PES = Present ecological state, ERC = Ecological Reserve Class (previously termed Ecological Management Class). Lin = linear, log = natural logarithm, exp = exponential, pow = power.



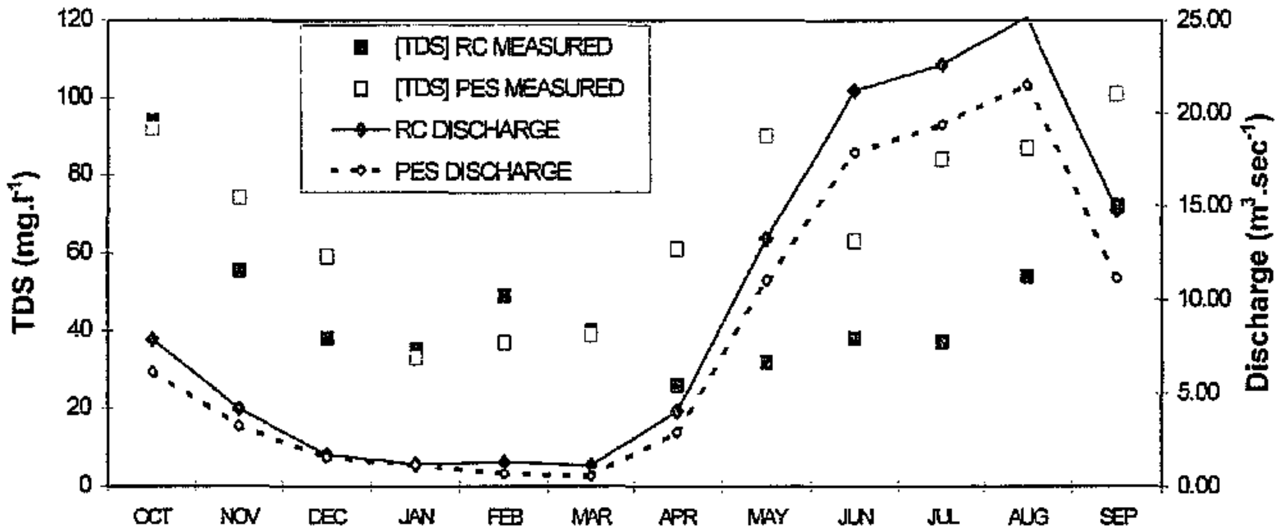


Figure 2.4 (Above) *Q-C plot of TDS (mg/litre) with flow (in $m^3.sec^{-1}$) for site IFR1 on the Breede River (Mooiplaas).* ***(Below)*** *Seasonal distribution of flow and TDS at IFR 1 for both the Reference Condition (RC) and the Present Ecological State (PES). Note the increased wash-off of salts during the onset of the wet season in the present state compared to natural.*

In section 2.2, the steps to be followed in making predictions of the concentrations of chemical constituents for the purpose of IFR assessments were explained. For a given month (and hence discharge), the concentration that would be expected under the Reference Condition and Present Ecological State is noted. The concentration that could be expected to occur under the new discharge for that month is obtained from the graph (Figure 2.5a). If discharge under the new regime is reduced compared to the present condition, due to a concentration effect, the predicted concentration will be higher. The above is valid for the typical situation in which the concentration of a constituent *decreases* with increased discharge. In the case of salinised catchments in the winter rainfall region described above, however, the above procedure would result lower predicted concentrations under reduced discharge. This is shown diagrammatically in Figure 2.5b. It is the contention of the project team that such predictions are likely to be erroneous. A consideration of the loading and sources of instream salinity can explain this. The loading of salts in the soils of the surrounding lands will remain constant, as will the rainfall. Thus the load of salts entering the river will be the same. If discharge is reduced, due to the concentration effect the instream levels of TDS must increase, or at least remain the same. Conversely if the predicted discharge is greater than under present day, it cannot be assumed that the instream concentration of TDS will be increased.

A factor that is critical to this analysis is the question of how discharge is to be changed (by means of upstream abstraction, construction of impoundments etc.) in other words the nature of the flow modifier. It was mentioned previously that predictions using Q-C modelling are only valid if the system is operated in the same way as it was when the data for setting up the rating curve were extracted. Predictions of concentration at IFR 1 would be very dependent on how the change in discharge was to be brought about. Because of this, it is recommended that Q-C modelling should not be used in such cases. If, for a given point on a river, the TDS loading due to diffuse pollution is known, it may be possible to obtain an estimate of predicted concentration using mass balance modelling, although it was not done in this project. In order to obtain reliable simulations of TDS a catchment-runoff model that can simulate loading of non-point source contaminants would need to be set up.

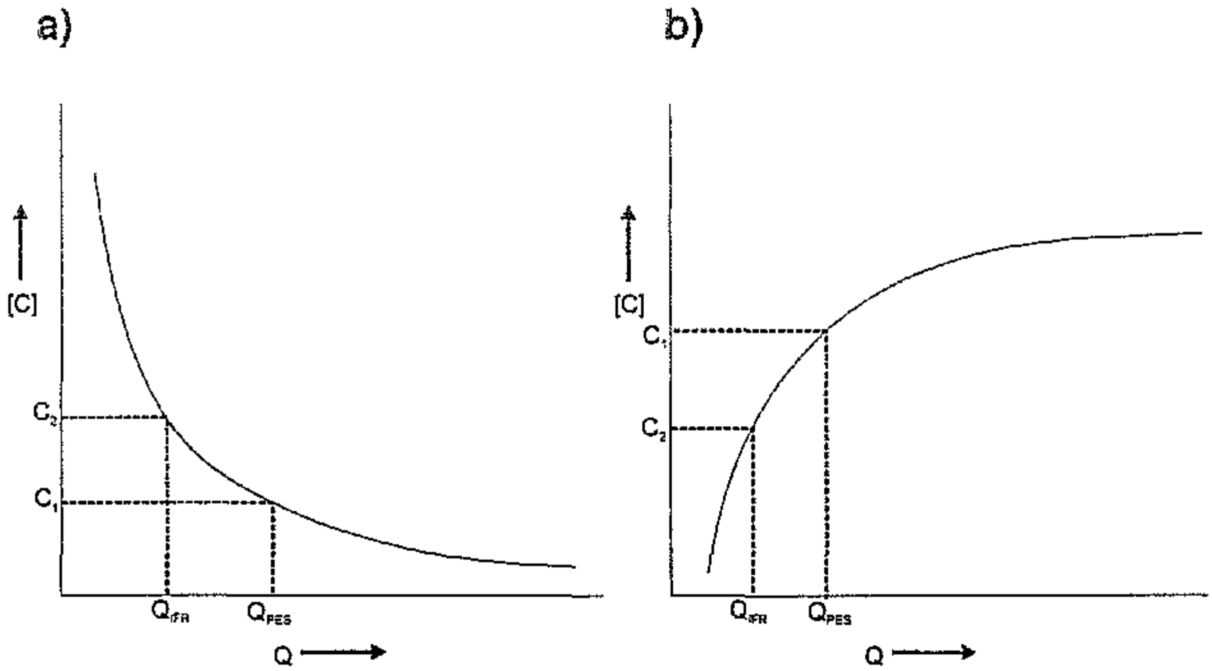


Figure 2.5 *Schematic representation of predicted concentrations for a given discharge.* ***a)*** *Classical dilution of a point source of pollution with increasing discharge.* ***b)*** *Increased concentration with increasing discharge.*

2.11.1.2 Other water quality variables that exhibit positive Q-C trends

Other chemical constituents, namely, ortho-phosphate, total phosphate and various forms of nitrogen (combined NO_2^- and NO_3^-, and total inorganic nitrogen), were also found, on occasion to increase in concentration with discharge. Such effects were not limited to catchments in the winter rainfall region of the country. For example ortho-phosphate increased slightly in WQ2 on the Pienaars River, as well as at IFR 8 and 13 on the Olifants River (Tables 2.3 and 2.5 respectively). From the discussion of generalised Q-C trends in section 1.6.1, it would appear that positive correlation between discharge and instream concentration of nutrients might well occur in areas where extensive agriculture is taking place, and thus in situations where pollutant loading is due mainly to non-point source rather than point sources of pollution. In addition, although TSS was not modelled in this study (because this variable is not monitored by DWAF on a regular basis), it is well documented that total phosphates are bound to sediments. It is also well known that sediments increase with discharge.

Increases in total phosphate with discharge were reported at some sites and this is likely to be due to the wash-off of soil and associated phosphate from the land into the river. The question of generalised Q-C trends in unimpacted rivers for different ecoregions of the country is considered further in Chapter 3.

From the above discussion it would follow that Q-C modelling should not generally be used to make predictions of concentrations in cases where a positive correlation with discharge is obtained. One exception may be found in the case of phosphates and other constituents, where elevated instream levels result from churning of benthic sediments or release from riverbanks, rather than wash-off from the surrounding catchment (Casey and Farr 1982). In such cases this would represent an instream pollutant source rather than an allochthonous, or off-land, input. Thus reduced discharge may well lead to lower concentrations of some constituents in the water column. Considering the usual lack of information that is normally the situation in many of South Africa's rivers, it will usually not be possible to ascertain the source of phosphates at a site. Thus to stay on the side of caution, unless additional data are available, predictions of concentrations for variables that exhibit a positive Q-C relationship at a given site should not be made.

2.12 Further generalisations

From an examination of the results from all four applications of the Q-C modelling process the following general conclusions can be made:

- The Present Ecological State Q-C relationship, can frequently be described using a logarithmic function.
- Compared to the Present Ecological State, the Reference Condition frequently shows little change with discharge and can often be described using a linear function. In some cases the Q-C plot is a horizontal line and thus concentration is independent of discharge (e.g. The Baviaans River (IFR 6, Breede River Basin study)).
- The Q-C trends of the conservative constituents Na^{+} and SO_4^{-2} usually mirrored those of TDS in the Breede and Olifants River systems respectively. This is to be expected since they each represent a major portion of TDS in the two systems.

- Point-sources of pollution show a characteristic negative Q-C trend. This effect is clearly shown by combined NO_2^- and NO_3^- in the Molenaars River (IFR 2, Breede River). The Molenaars is a largely unimpacted tributary of the Breede River. Western Cape Rivers are largely oligotrophic and the concentrations of most chemical constituents are low in this river (Appendix B). Due to nutrient-rich effluent from a trout farm on the bank of the river, however, NO_2^- and NO_3^- levels are relatively elevated at low discharges but become diluted at higher flows.
- Downstream of impoundments, TDS often shows little change with increasing discharge. This is due to mixing of water inflows of differing salinities within the impoundment. The resultant effect is that the salinity of the water flowing out of the impoundment tends to be attenuated. Thus extremes of highly saline or fresh water do not occur, and the resulting Q-C plot for a site downstream of the impoundment is only slightly negative. This effect was shown at IFR 1 and IFR 3 on the Olifants River, which are both downstream of major reservoirs.

2.13 Validation

The Q-C modelling method was validated by using an independent set of water quality and discharge data to that used to set up the model. The Q-C model for a given variable at a site was considered to be acceptable if more than 75% (9 out of 12) of the validation data points fell within the 95% confidence interval of the original data set. Only the model for the Present Ecological State but not the Reference Condition was validated.
The validation procedure was carried out in two different ways:

2.13.1 Validation method 1 (intercalating data)

In the first validation method that was tested, monthly median concentration values for the test data were calculated from the years 1993, 1995, 1997 and 1999. Monthly mean discharge values were also calculated using data from the same years. Median concentrations and mean discharge values for validation (both on a monthly basis) were calculated from the data for 1994, 1996, 1998 and 2000. The Q-C plot was drawn for the test data and the 95% confidence interval calculated as described in section 2.2. The validation Q-C plot was then constructed and the correspondence between the two simulated curves was assessed by noting the percentage of validation data points that fell within the 95% confidence interval.

2.13.2 Validation method 2

Q-C modelling for the various sites investigated in the project was carried out using water quality data up to 1998. In the second validation method therefore, water quality data from the appropriate monitoring station for the time period October1998-mid 2001 were used. A problem was encountered with the discharge data required for the validation procedure, however. For the majority of the sites previously modelled during the course of the project, simulated discharge data had been used. Such data were generated from rainfall-runoff models calibrated for the catchment and obtained from the relevant specialist responsible for hydrological modelling for the Reserve determination. Thus discharge data from 1999 onwards were not available. As an alternative, observed discharge data from the nearest gauging site could be used. It was considered that this should yield similar results to that using simulated data, but slight inaccuracies might occur because of the different sets of discharge data. Method 2 was therefore only employed for the few sites where observed discharge data rather than simulated data had been used.

2.13.3 Results of validation

2.13.3.1 Method 1

Good results were generally obtained using this validation method for all the sites that were modelled. Exceptions were cases where there was considerable scatter of measured data, resulting in low correlation coefficient (less than 0.6) between measured and simulated values. In such situations, usually fewer than 75% of the validation data points fell within the 95% confidence interval. As mentioned previously in this chapter, nutrients frequently yielded poor correlations between measured and simulated values. Consequently it was often nutrients for which the Q-C model could not be validated. Figure 2.6 shows the results of validation (using method 1) for IFR site 8 on the Olifants River. The validation curves obtained for EC, fluoride and pH are satisfactory, indicating that the Q-C model is valid for those water quality variables. Although there is greater scattering of data points, the validation curve for dissolved phosphate is also acceptable, since 75% of the points are within the confidence interval. In the case of combined

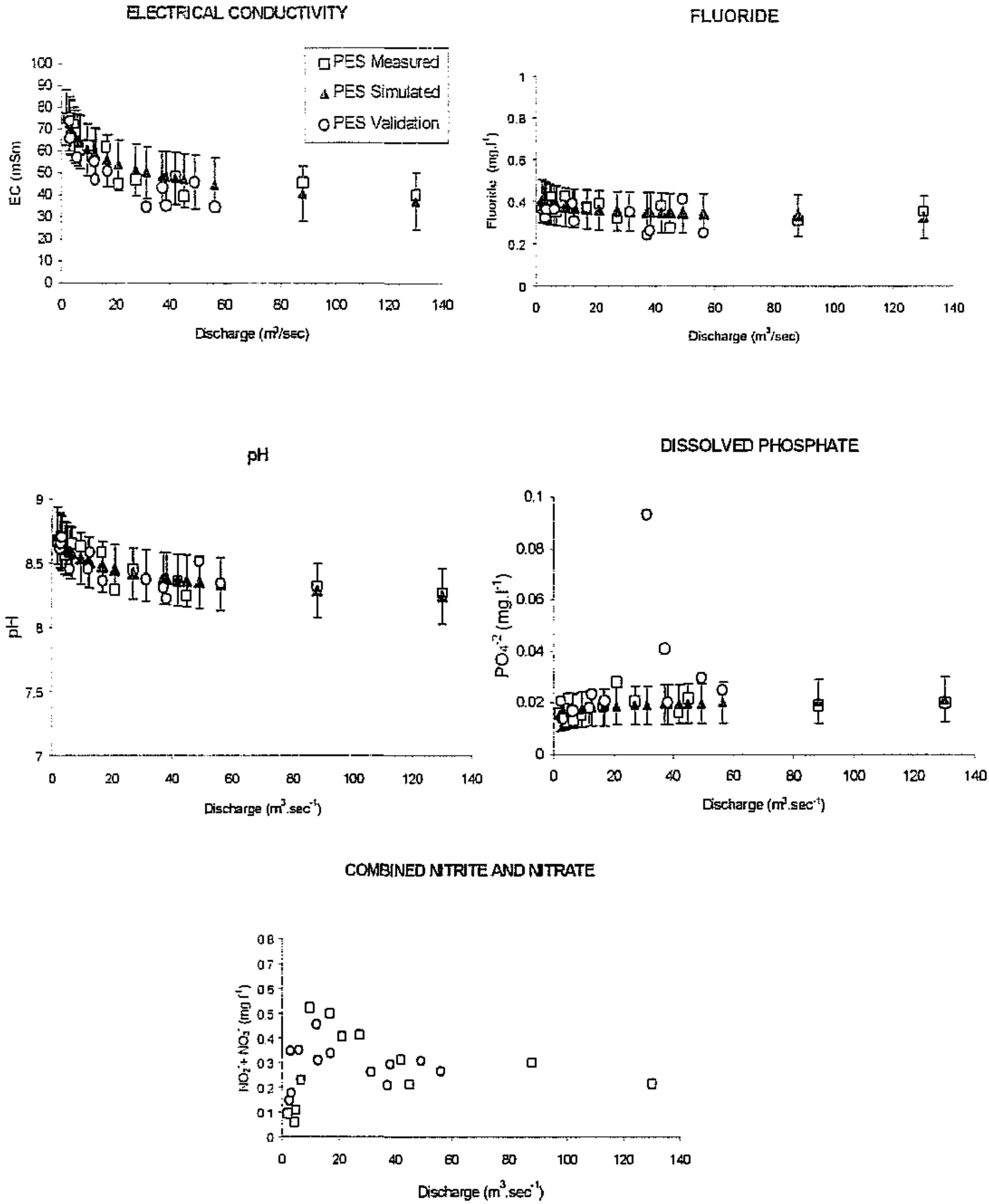


Figure 2.6 *Q-C plots for IFR 8 (Olifants River, Mpumalanga). The 95% confidence interval and the validation curve produced using Validation method 1 are shown. No significant relationship between stream flow ($m^3.sec^{-1}$) and concentration ($mg.l^{-1}$) could be derived for combined NO_2^- and NO_3^-.*

nitrate and nitrite however, no modelling could be under taken at all since the measured data points for the Present Ecological State were too scattered.

2.13.3.2 Method 2

In the second validation method, data subsequent to 1998 were used to construct the validation curve. This method was used only for sites where observed discharge data, rather than simulated data, had been used to set up the Q-C model (7 IFR sites in total). The results that were obtained using this validation method were mixed. For some sites, good validation curves were produced for all water quality variables that were modelled and at other sites, acceptable validation curves were obtained for conservative variables but not for nutrients. Interesting results were obtained for IFR 3 on the Olifants River and these are illustrated in Figure 2.7. A poor correspondence between the Q-C model and the validation model was obtained for TDS. The latter curve maintained a similar trend to that of the Q-C model curve (i.e. slight negative trend) but was consistently lower in concentration and below the lowest 95% confidence interval for all data points. A similar pattern was obtained for sulphate, except that these concentrations were not lowered to such an extent and remained just within the confidence interval. In the case of pH, all the validation data points were higher than the median values of the Q-C curve, but followed the same general trend. Nutrients (dissolved phosphate and combined nitrate and nitrite) were the exceptions in that the validation curve for both variables fell within the 95% confidence interval.

2.13.4 Discussion of the validation methods

It was explained in section 2.6 that the Q-C relationship derived for a particular site, is only valid as long as the system is operated in the same way. Thus when using Validation method 2, if there was a change in the Q-C relationship after 1998, the validation curve would be dissimilar to that of the test curve. For example in the case of IFR 3 on the Klein Olifants River, this site is a few kilometers downstream of the Middleberg dam. Because of mixing within the impoundment, the Q-C plot for TDS at that site shows only a slight negative trend with increasing discharge. The Q-C plot was constructed using data up to, and including, 1998. Exceptionally high rainfall was measured in parts of the Olifants River catchment during the summer of 1999 and 2000, however. The inflows of low salinity water into the Middleberg dam are likely to have resulted in lowered concentrations of many chemical constituents, including TDS,

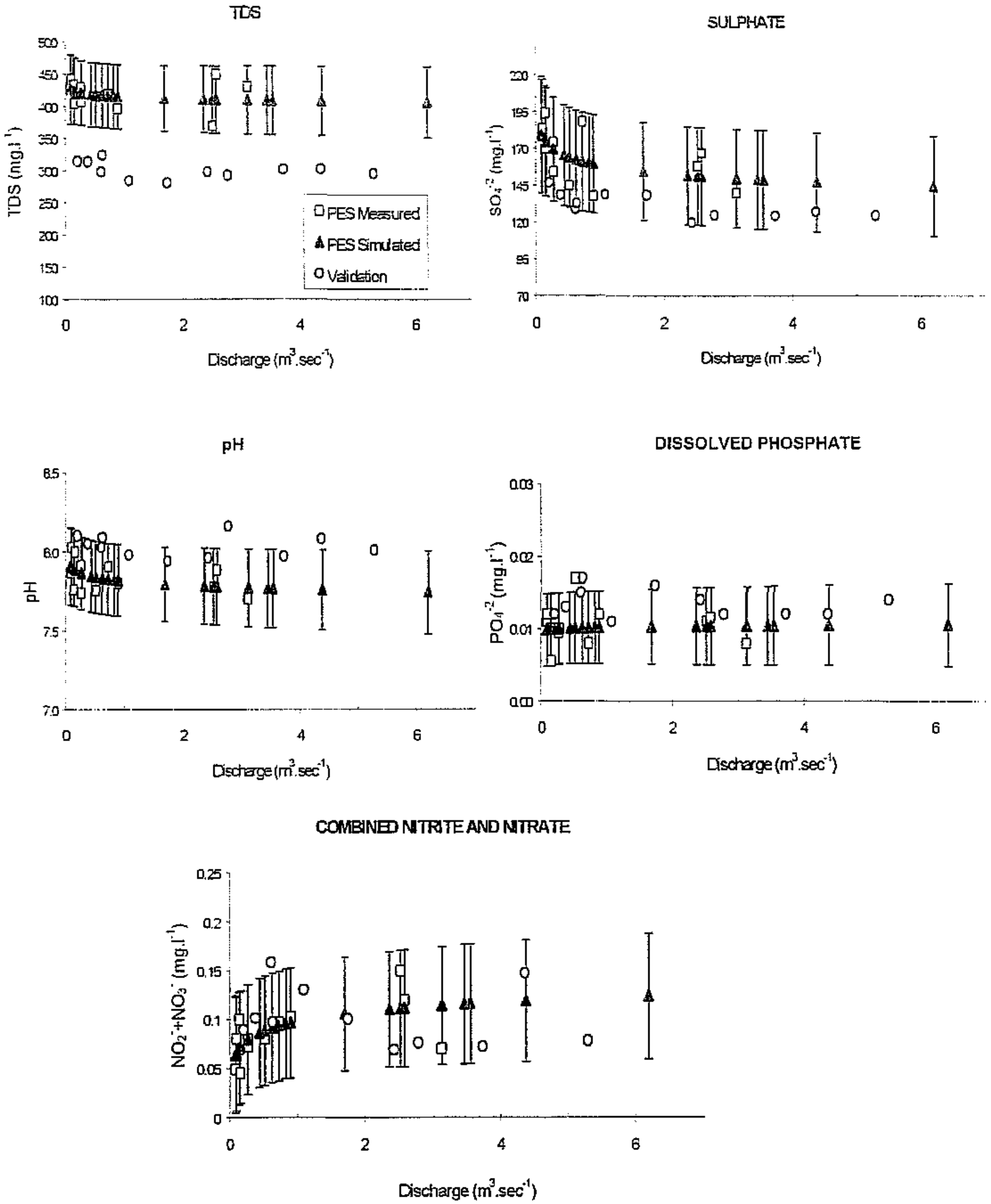


***Figure 2.7** Q-C plots for IFR 3 on the Klein Olifants River. The 95% confidence interval and the validation curve produced using Validation method 2 are shown.*

sulphate and organic acids within the impoundment. Consequently the Q-C plot for these constituents follows the same general trend as the curve generated using pre-1999 data, but is lower in the case of TDS and sulphate and higher in the case of pH due to the reduced constituent concentrations in the releases from the dam. Nutrients on the other hand do not seem to exhibit a similar dilution effect since the concentrations of the validation curve are within the confidence interval of the Q-C model. This is possibly due to a shift in the adsorbtion/desorbtion equilibria between nutrient ions and sediments within the impoundment. As a result, the concentration of phosphate and combined nitrate and nitrite in water released from the impoundment is more or less the same as pre-1999.

From the above discussion it can be concluded that Validation method 1, in which the record span of the test discharge and water quality data is intercalated with that of the validation data is the most appropriate method. The second validation method can be useful for examing the dynamics of discharge-concentration relationships at a site. It can also be used to test if the system is operated in the same way as when the Q-C model was derived. IFR sites at which the Q-C model could not be validated can still be used for making predictions of water quality. The Q-C model at such sites is not invalid, but it will however, reduce the degree of confidence that can be placed in the results. At sites where the Q-C model cannot be validated, if possible a second tier of water quality modelling (e.g. mass balance modelling, or QUAL2E) should be implemented.

2.14 Conclusion

Q-C modelling is a useful tool for providing estimates of predicted instream concentrations of chemical constituents for a given recommended discharge. There are limitations and assumptions inherent in the method, however, and these should be recognised when applying the technique in Reserve determinations. Thus Q-C modelling can only be considered to be a preliminary modelling technique and may sometimes need to be followed by application of more sophisticated modelling methods in some river reaches. The problem of non-point sources of pollutants, for which Q-C modelling is not suitable, is likely to increase in future. Because of this, attention should be directed towards setting up catchment run-off models for at least for key sites on rivers where extensive and conflicting demands are made on them. This is discussed further in Chapter 7, which considers integration of the modelling methods into the Reserve determination process.

CHAPTER 3

A PRELIMINARY INVESTIGATION OF REGIONAL PATTERNS OF DISCHARGE-CONCENTRATION TRENDS

by

J. Boulle and H. Malan

3.1 Introduction

In Chapter three of the literature review that accompanies this project (Malan and Day 2002) as well as in section 1.6.1 of this report, a synopsis is presented of the most common discharge-concentration (Q-C) trends reported in the literature. Despite substantial variation between sampling sites, and depending on the water quality constituent under consideration, these trends often exhibit common traits. In the previous chapter, the Q-C trends at sites on four different river systems were examined. Following from this work, a preliminary investigation was carried out to ascertain if different areas of the country exhibited different and characteristic Q-C trends. An additional aim was to compare the results obtained with the trends identified from the literature review. To avoid complications due to excessive anthropogenic effects, relatively unimpacted catchments were examined.

The aims of this part of the project are thus to:

- Ascertain how the concentrations of a number of chemical constituents vary with discharge at different sites in South Africa.
- Determine whether these responses are predictable and consistent within the same primary drainage region, ecoregion, or river type.
- Ascertain if these results are consistent with those identified from the literature.

3.2 Method used to assess regional Q-C trends

In order to examine regional patterns of change in water quality with discharge, within each drainage basin, sites were first identified that exhibited natural flow patterns. At each of these sites, an appropriate time segment was selected from the available data to reflect the least impacted water quality conditions. These water quality and discharge data for that time-period were then used to plot regional Q-C patterns. Only sites on perennial streams were examined. The water quality variables investigated were electrical conductivity (EC), pH and total alkalinity (TAL) since these are important system variables and define the type of water quality. Nutrients, namely combined NO_3^- and NO_2^-, as well as dissolved PO_4^{3-} were also examined. In addition, silicate (SiO_3^-) was studied as this element represents an example of a naturally occurring geolithic ion and is thus likely to be present in stream water due to weathering of rocks rather than as a result of pollution.

A detailed description of the method is given below.

1. Joubert and Hurly (1994) identified a number of DWAF gauging sites (352 in total) within South Africa that exhibited relatively natural discharge patterns, were situated upstream of all major impoundments or abstractions, had a minimum record span of 20 years, and comprised reliable data. These sites were used for further investigation in this project.
2. The chosen sites were screened to exclude those that were heavily polluted. Water quality data were downloaded from Water Quality on Disc, which contains regional water quality monitoring data collected by Department of Water Affairs and Forestry (DWAF) personnel. The criteria of Day, Dallas and Wackernagel (1998) were used to screen the data. Thus for each site, the percentage of samples for which electrical conductivity (EC) $\geq$500 mS/m, combined nitrate and nitrite (NO_3^- and NO_2^-) $\geq$0.5mg/l, and soluble phosphate (PO_4^{3-}) levels $\geq$0.1mg/l was calculated. The percentages obtained for each of the three water quality variables were averaged to provide an indication of the overall level of impact at each site. These averages were used to rank sites according to their water quality and data for the three least impacted sites from each primary drainage region were examined further.
3. In order to select a time period for data analysis, time-series were plotted for EC, combined nitrate and nitrite, and soluble phosphate, at each of the three sites within

each drainage basin. The time segment that was selected excluded periods when the system appeared to be impacted, and extended over at least 5 years. The minimum sample size was n = 25 for each variable (although for the majority of sites sample size was considerably larger – see Table 3.2). At sites where these criteria were not met, the next site on the ranked list was used. Thus at each site, an appropriate time segment was selected and used for all further investigation of trends.

4. For each of the selected sites and time periods, water quality data (where available) were examined for electrical conductivity (EC), pH, soluble phosphate (PO_4^{-3}), combined nitrate and nitrite (NO_3^-+NO_2^-), silicate (SiO_3^-) and total alkalinity (TAL). Average monthly discharge data for the appropriate gauging station and time period were obtained from DWAF.
5. In order to examine trends in water quality with discharge, the basic method as described for Q-C modelling (Figure 2.1) was followed. In brief, the discharge for each month of each year for the entire time period was used to calculate the average monthly discharge (i.e.12 discharge values). The corresponding median water quality values for each month, and for each variable, were also calculated.
6. These values were then used to plot water quality against discharge for each of the water quality variables and to examine trends at each of the selected sites within each drainage basin. Trendlines were fitted to these relationships, depending on the value of the correlation coefficient r^2 as described in section 2.2.
7. At sites where the correlation between water quality and discharge was poor, summer and winter data were examined separately to see if there was any improvement in the correlation. Summer was taken to be from October to March, and winter from April to September.

3.3 Results

3.3.1 Screening of sites for least impacted discharge and water quality

The complete list of gauging/water quality monitoring sites examined is given in Appendix C. The list was obtained from Joubert and Hurley (1994) and represents sites that were reasonably close to natural with respect to discharge. Also shown in the appendix, is the percentage of observations for the entire time-series that exceeded the specified water quality criteria of EC = 500 mS/m, combined nitrate and nitrite (NO_3^- and

NO_2^-) = 0.5mg/l, and soluble phosphate (PO_4^{3-}) levels = 0.1mg/l. In the final column of the table, the sample size at each site, for each of the three water quality constituents is given. The sample size for all water quality variables is not the same despite the fact that the same time period was used, because on sampling, not all variables were measured on every occasion. The above information is included as an appendix in the report because it represents a useful record of data sets that exhibit natural hydrological patterns in combination with water quality impacts ranging from slight to impacted.

The list of gauging/water quality monitoring sites that were used in the study after screening for impacted flow regime and water quality is given in Table 3.1, together with the name of the river on which the site is situated, and the Level 1 ecoregion in which it occurs (Kleynhans and Hill, 1999). Note that no sites in drainage regions D, F, or M could be considered to exhibit natural discharge patterns and so these regions are not considered further. Table 3.2 shows for each of the least-impacted sites, the median value of each of the water quality variables that were examined for Q-C trends with the standard deviation given in parenthesis. The time period of records that was used to calculate the median values is also recorded, as well as the sample size, which is given as a range since *n* is not the same for each variable. It can be seen from Table 3.2 that in the case of drainage regions N and P, only one station in the region exhibited un-impacted discharge patterns. It can also be seen from the table which sites were un-impacted with regard to water quality (e.g. E1H006) compared to those that were relatively salinised (e.g. P4H001). None of the sites chosen appeared to be particularly polluted by excessive nutrient levels.

3.3.2 Assessment of the Q-C trends

Initially, the use of daily flow data to represent the average discharge for a given date was investigated. This meant that water quality values recorded for a specific day were plotted against the average discharge measured for that same day rather than using monthly median water quality plotted against monthly mean discharge. This method proved to be far too time-consuming however, considering the amount of available data and the number of sites for which trends were examined (66 sites). In addition, as

Table 3.1 The gauging/water quality monitoring sites selected for investigation of Q-C trends and the corresponding primary drainage region and ecoregion.(CFM = Cape Folded Mountains).

Drainage region	Gauging station	Place name	Ecoregion
A	A6H018	Rasloop River, Sussensvale, Modderpoort	Central Highlands
	A6H019	Hessie se Water, Rietspruit	Central Highlands
	A6H021	De Wet Spruit, Groenvaley	Central Highlands
B	B4H005	Waterval River, Modderspruit	Central Highlands
	B6H003	Treur River, Willemsoord	Lebombo Uplands
	B7H004	Klaserie River, Fleur de Lys	Lebombo Uplands
C	C1H007	Vaal River, Goedgeluk, Bloukop	Highveld
	C2H026	Middelvlei Spruit, Middelvlei	Highveld
	C2H028	Rietfontein Spruit, Rietfontein	Highveld
E	E1H006	Jan Dissels River, Clanwilliam Commonage	Cape Fold Mountains
	E2H002	Doring River, Elands Drift, Aspoort	Cape Fold Mountains
	E2H003	Doring River, Melkboom	Western Coastal Belt
G	G1H012	Watervals River, Watervalsberge, Lower Watervals	Cape Fold Mountains
	G1H016	Kasteelkloof Spruit, Zachariashoek, Kasteelkloof Lower	Cape Fold Mountains
	G4H008	Klein-Jakkals River, Lebanon Forest Res.	Cape Fold Mountains
H	H2H005	Rooi-Elskloof River, Roode Els Berg	Cape Fold Mountains
	H3H004	Keisie River, Harmonie	Cape Fold Mountains
	H6H010	Waterkloof River, Waggensbooms Kloof	Cape Fold Mountains
J	J1H015	Bok River, Lot B	Cape Fold Mountains
	J3H013	Perdepoort River, Groenefontein, De Hoek	Cape Fold Mountains
	J3H018	Wynands River, Koetzers Kraal	Cape Fold Mountains
K	K3H001	Kaaimans River, Upper Barbierskraal	Southern Coastal Belt /CFM
	K4H003	Diep River, Woodville Forest Res.	Southern Coastal Belt/ CFM
	K5H002	Knysna River, Milwood Forest Res, Lower Streepbos	Southern Coastal Belt/CFM
L	L7H006	Groot River, Grootrivierspoort	Cape Fold Mountains
	L8H001	Wabooms River, Diepkloof	Cape Fold Mountains
	L8H002	Haarlem Spruit, Welgelegen	Cape Fold Mountains
N	N2H009	Volkers River	Eastern Uplands
P	P4H001	Kowie River, Bathurst, Wolfscrag	Southern Coastal Belt
Q	Q9H002	Koonap River, Adelaide	Eastern Uplands
	Q9H014	Koonap River, Frisch Gewaagd, Groenkop	Eastern Uplands
	Q9H016	Koonap River, Schurftekop	Eastern Uplands
R	R1H014	Tyume River, Kwa Khayaletu, Yantolas Loc.	Eastern Uplands
	R2H001	Buffalo River, Pirie Main Forest Res.	Eastern Uplands
	R2H008	Quencwe River, Braunschweig, Edendale	Eastern Uplands
S	S3H002	Klaas Smits River, Doornhoek, Wilgebosch	Eastern Uplands
	S6H001	Kubusi River, Stutterheim	Eastern Uplands
	S6H003	Toise River, Forkroad	Eastern Uplands
T	T2H002	Mtata River, Norwood, Umtata	Eastern Uplands
	T3H004	Mzintlava River, Slang Fontein, Kokstad	Eastern Uplands
	T3H009	Mooi River, Maclear	Eastern Uplands
U	U2H013	Mgeni River, Petrus Stroom	Eastern Uplands
	U4H002	Mvoti River, Mistley	Eastern Uplands
	U6H002	Mlazi River, Nooitgedacht, Bainesfield	Eastern Coastal Belt
V	V2H007	Hlatikulu River, Broadmoor	Eastern Uplands
	V1H041	Mlambonja River, Kleinerivier	Eastern Uplands
	V7H017	Boesmans River, Drakensberg Loc. 1	Great Escarpment
W	W5H004	Ngwempisi River, Bushmans Spruit	Highveld
	W5H006	Swartwater River, Zwartwater	Highveld
	W5H008	Bonnie Brook, Broadholms	Highveld
X	X2H010	North Kaap River, Bellevue	Lebombo Uplands
	X2H014	Houtbosloop, Sudwalaaskraal	Lebombo Uplands
	X3H006	Sabie River, Perry's Farm	Lebombo Uplands

Table 3.2 *List of the gauging sites used to examine Q-C trends. Also shown is the time-period, as well as the sample size used to derive median values of the water quality variables. The median value and standard deviation (given in parenthesis) for each variable during the chosen time-period is recorded. Note that sample size is given as a range since n is not the same for each variable. EC = electrical conductivity; NO = NO_3^- + NO_2^-; P = PO_4^{-3}, TAL = total alkalinity, Si = silicate.*

Gauging station	Time period used	Sample size	Median EC mSm^{-1}	Median TAL mg/l	Median Si mg/l	Median pH	Median NO mg/l	Median P mg/l
A6H018	77/12/05 – 98/01/21	123 - 159	3.4 (1.3)	13.1 (2.5)	4.9 (0.3)	6.5 (0.6)	0.04 (0.04)	0.007 (0.01)
A6H019	77/12/05 – 98/01/21	111 - 154	3.8 (1.2)	15.3 (4.9)	6.3 (0.6)	6.6 (0.6)	0.02 (0.02)	0.01 (0.01)
A6H021	77/12/26 – 82/01/27	26 - 46	3.5 (0.99)	9.9 (5.0)	5.5 (0.6)	5.5 (0.6)	0.02 (0.03)	0.007 (0.004)
B7H004	77/05/06 – 86/01/27	122 - 292	7.0 (1.3)	22.8 (6.8)	10.5 (1.5)	6.8 (0.5)	0.015 (0.02)	0.014 (0.02)
B4H005	78/10/12 – 86/12/01	117 - 216	14.0 (4.4)	61.7 (12.4)	11.5 (1.4)	7.2 (0.39)	0.075 (0.17)	0.009 (0.01)
B6H003	76/10/05 – 88/12/26	130 - 328	6.2 (2.3)	23.4 (13.13)	3.3 (1.1)	6.8 (0.65)	0.04 (0.09)	0.006 (0.01)
C2H026	79/05/02 – 97/12/03	211 – 289	6.0 (10.2)	15.6 (22.7)	3.9 (1.0)	7.0 (0.8)	0.04 (0.12)	0.01 (0.03)
C2H028	79/05/03 – 95/03/21	368 - 434	7.3 (2.1)	17.5 (5.0)	3.3 (1.2)	6.6 (0.6)	0.04 (0.16)	0.014 (0.015)
C1H007	76/10/04 – 98/02/24	846 - 1028	24.7 (12.0)	77.4 (42.6)	5.4 (1.8)	7.8 (0.5)	0.09 (0.92)	0.022 (0.03)
E1H006	78/01/04 – 83/06/30	77 - 216	4.5 (1.4)	5.1 (4.6)	2.7 (0.79)	5.5 (0.66)	0.02 (0.07)	0.007 (0.01)
E2H002	73/03/02 – 91/11/20	60 - 115	7.9 (7.7)	9.3 (10.1)	1.8 (0.59)	6.5 (0.88)	0.027 (0.08)	0.008 (0.01)
E2H003	72/05/13 – 91/12/23	90 - 304	28.3 (35.6)	20.0 (32.2)	2.0 (1.4)	7.0 (0.78)	0.03 (0.27)	0.01 (0.02)
G1H016	75/11/27 – 86/11/05	20 - 39	4.0 (2.6)	6.5 (4.2)	2.6 (1.9)	4.6 (0.8)	0.021 (0.01)	0.004 (0.002)
G1H012	77/10/18 – 97/04/24	247 - 383	7.3 (3.2)	5.4 (4.4)	2.3 (0.6)	5.1 (1.1)	0.022 (0.04)	0.009 (0.16)
G4H008	82/11/02 – 87/05/05	45 - 53	9.3 (2.1)	5.1 (4.1)	1.0 (0.6)	4.0 (0.7)	0.02 (0.03)	0.02 (0.01)
H2H005	73/11/29 – 86/12/31	67 - 160	2.1 (1.4)	5.8 (5.1)	2.4 (0.6)	5.0 (0.7)	0.026 (0.08)	0.012 (0.01)
H3H004	65/10/20 – 91/11/05	34 - 82	18.4 (13.1)	28.1 (20.3)	5.4 (1.1)	6.8 (0.6)	0.067(0.13)	0.012 (0.02)
H6H010	73/04/13 – 82/12/01	102 - 174	8.1 (2.29)	5.9 (9.2)	3.5 (0.7)	5.1 (0.9)	0.02 (0.05)	0.007 (0.01)
J1H015	75/02/12 – 82/12/02	80 - 137	1.6 (1.6)	4.8 (5.6)	1.6 (0.6)	5.2 (0.7)	0.028 (0.03)	0.005 (0.007)
J3H013	70/07/09 – 90/12/06	154 - 294	5.4 (3.9)	15.5 (10.5)	2.8 (0.8)	6.5 (0.7)	0.027 (0.08)	0.011 (0.02)
J3H018	69/04/11 – 97/03/12	169 - 217	26.2 (10.0)	76.9 (30.1)	3.5 (1.1)	7.6 (0.5)	0.05 (0.2)	0.01 (0.02)
K3H001	71/06/28 – 86/12/29	91 - 258	16.2 (12.0)	5.3 (13.9)	2.5 (0.8)	4.7 (0.7)	0.023 (0.05)	0.02 (0.03)
K4H003	71/08/10 – 97/01/13	227 - 254	22.8 (5.8)	7.1 (5.8)	2.8 (1.0)	5.7 (0.9)	0.054 (0.1)	0.011 (0.06)
K5H002	71/06/29 – 97/02/10	234 - 395	13.5 (6.6)	5.1 (11.0)	2.4 (0.6)	5.0 (1.0)	0.04 (0.08)	0.02 (0.03)

Table 3.2 List of the gauging sites used to examine Q-C trends continued.

Gauging station	Time period used	Sample size	Median EC mSm^{-1}	Median TAL mg/l	Median Si mg/l	Median pH	Median NO mg/l	Median P mg/l
L8H001	72/08/07 – 82/12/07	101 - 219	5.7 (1.3)	4.6 (9.1)	1.8 (0.4)	4.9 (0.9)	0.045 (0.05)	0.011 (0.01)
L8H002	72/08/07 – 82/12/07	86 - 202	7.1 (2.3)	5.8 (7.1)	1.9 (0.5)	5.5 (0.8)	0.026 (0.04)	0.009 (0.01)
L7H006	68/04/03 – 97/03/06	528 - 634	97.8 (159)	79.0 (35.9)	1.1 (1.3)	7.8 (0.5)	0.028 (0.63)	0.014 (1.13)
N2H009	78/10/05 – 97/04/17	694 - 771	108 (47.1)	227.3 (59.1)	9.1 (2.0)	8.2 (0.52)	0.461 (1.22)	0.089 (0.57)
P4H001	71/11/06 – 97/04/29	202 - 339	236 (130)	180.4 (53.0)	1.01 (2.5)	8.0 (0.48)	0.04 (1.5)	0.02 (0.27)
Q9H014	71/11/09 – 79/06/01	78 - 122	34 (17.2)	139.4 (66.5)	11.5 (2.6)	7.4 (0.4)	0.098 (0.6)	0.01 (0.05)
Q9H002	71/08/29 – 97/04/28	261 - 316	61.7 (34.2)	183.2 (98.1)	8.9 (2.1)	7.9 (0.5)	0.071 (0.75)	0.02 (0.14)
Q9H016	71/11/09 – 93/03/02	215 - 343	47.4 (17.9)	179.6 (76.0)	9.9 (2.7)	7.7 (0.6)	0.062 (0.59)	0.018 (0.14)
R1H014	71/11/08 – 83/02/15	101 - 277	7.0 (3.8)	23.9 (16.1)	7.7 (1.4)	6.7 (0.5)	0.059 (0.76)	0.013 (0.02)
R2H001	71/09/21 – 97/04/30	117 - 187	8.9 (3.5)	19.3 (10.7)	7.1 (1.5)	7.2 (0.6)	0.091 (0.14)	0.014 (0.02)
R2H008	72/01/26 – 97/04/30	241 - 295	41.6 (17.0)	83.7 (42.6)	9.4 (1.8)	7.6 (0.5)	0.161 (0.31)	0.017 (0.28)
S6H001	72/04/16 – 89/12/05	173 - 318	9.9 (2.8)	23.2 (8.4)	6.5 (2.2)	6.8 (0.6)	0.161 (0.15)	0.016 (0.04)
S6H003	72/01/27 – 97/04/16	285 - 430	26.8 (11.1)	73.2 (38.3)	7.5 (2.4)	7.5 (0.6)	0.09 (2.0)	0.022 (0.4)
S3H002	74/05/29 – 97/04/17	154 - 249	62.9 (17.9)	253 (76.2)	7.9 (2.6)	8.0 (0.4)	0.099 (0.33)	0.022 (0.06)
T3H009	71/09/18 – 97/02/25	428 - 574	7.1 (2.2)	28.4 (10.7)	6.5 (1.1)	7.5 (0.6)	0.039 (0.07)	0.013 (0.02)
T3H004	71/09/17 – 85/12/31	123 - 232	13.7 (3.2)	54.4 (13.6)	7.9 (1.3)	7.0 (0.5)	0.032 (0.09)	0.012 (0.08)
T5H003	76/11/24 – 87/12/09	114 - 306	5.4 (1.7)	21.5 (8.2)	4.7 (1.0)	6.8 (0.5)	0.031 (0.12)	0.008 (0.02)
U6H002	80/03/18 – 90/12/19	143 - 253	7.8 (2.0)	24.8 (6.1)	7.5 (1.0)	6.6 (0.54)	0.14 (0.16)	0.007 (0.013)
U4H002	77/06/15 – 92/05/28	208 - 384	10 4 (3.8)	30.4 (13.7)	6.2 (1.2)	7.0 (0.64)	0.03 (0.14)	0.01 (0.03)
U2H013	77/07/18 – 95/08/29	374 - 760	6.4 (2.0)	23.8 (10.0)	5.3 (0.8)	7.2 (0.66)	0.16 (0.3)	0.006 (0.03)
V7H017	77/07/18 – 97/04/23	215 - 407	7.4 (2.1)	32.7 (9.2)	8.7 (1.1)	7.4 (0.5)	0.025 (0.06)	0.013 (0.02)
V1H041	77/04/06 – 97/04/21	253 - 412	7.2 (2.0)	30.0 (7.9)	8.3 (1.0)	7.1 (0.6)	0.033 (0.08)	0.011 (0.02)
V2H007	76/09/21 – 97/04/30	215 - 363	4.3 (1.6)	19.3 (8.2)	4.5 (1.0)	6.9 (0.7)	0.031 (0.05)	0.011 (0.09)
W5H008	70/02/20 – 84/11/01	87 - 244	5.0 (1.9)	20.1 (10.8)	7.1 (1.2)	6.7 (0.6)	0.02 (0.09)	0.011 (0.01)
W5H006	77/08/16 – 96/08/16	135 - 240	5.5 (2.4)	20 (9.3)	5.3 (0.9)	6.9 (0.7)	0.076 (0.1)	0.012 (0.02)
W5H004	77/04/29 – 88/05/24	101 - 258	10.1 (2.6)	38.5 (17.5)	8.2 (1.4)	6.9 (0.4)	0.04 (0.13)	0.01 (0.01)
X2H014	66/08/02 – 87/12/29	223 - 423	9.1 (2.7)	38.7 (15.0)	6.8 (1.5)	7.0 (0.5)	0.03 (0.03)	0.007 (0.01)
X2H010	72/03/27 – 87/12/30	107 - 227	9.2 (1.5)	38.2 (7.7)	13.5 (2.0)	7.0 (0.5)	0.02 (0.04)	0.01 (0.02)
X3H006	69/11/26 – 87/07/29	151 - 345	9.8 (2.2)	37.5 (12.9)	6.0 (0.8)	7.0 (0.5)	0.14 (0.1)	0.01 (0.01)

discussed in section 2.5, the Q-C trends tended to be obscured by the variability in response. It was decided therefore to use monthly data as specified in the Q-C method for further analysis.

Problems were encountered in assessing the trends in Q-C relationships. In general, there was a large range in r^2 values, making it difficult to decide what should be accepted as a "good" or a "bad" correlation. Three categories were arbitrarily assigned as follows:

- $r^2 > 0.5$ = a **good** correlation between discharge and concentration;
- **$0.2 < r^2 < 0.5$** = an **average** correlation and
- **$r^2 < 0.2$** = a **poor** correlation.

It is recognised that these criteria are not particularly stringent, and much stricter criteria were used for Q-C modelling used in Reserve assessments. In addition, at different sites even within the same drainage region, the various chemical constituents exhibited disparate trends, which made the classification of "typical" responses with regard to drainage region or ecoregion very difficult. On examination of the Q-C trends at each site, a number of classes of response emerged (see Figures 3.1 to 3.4). These could be broadly divided into four categories on the basis of the way in which the selected water quality variables responded to changes in discharge. The four categories are as follows:

- <u>Category 1:</u> All four non-nutrient variables (pH, EC, TAL and SiO_3^-) at a site decrease either logarithmically or linearly with an increase in discharge. No consistent correlation between water quality and discharge is evident for NO_3^-+NO_2^- or for PO_4^{-3}.
- <u>Category 2:</u> EC, TAL and SiO_3^- decrease logarithmically or linearly with discharge at all sites. No consistent trend is evident for the relationship between pH and discharge. No consistent correlation between water quality and discharge is evident for NO_3^-+NO_2^-, or for PO_4^{-3}.
- <u>Category 3:</u> pH, EC and TAL decrease logarithmically or linearly with discharge. A correlation between SiO_3^- and discharge is evident, but the exact relationship is inconsistent. Silicate either increases, decreases, or remains constant with

increasing discharge. No correlation between water quality and discharge is evident for NO_3^-+NO_2^-, or for PO_4^{-3}.

- **Category 4:** This category contained sites where the correlation between all or most of the water quality variables and discharge is very low. There is either considerable scatter of data points, or else the data points are in a horizontal line, indicating that concentration is independent of discharge.

Figures 3.1 to 3.4 provide representative examples of the types of responses that were found at different sites. Each figure represents one of the four categories of site Q-C relationships that were observed. Figure 3.1 shows the Q-C trends for B6H003 (Treur River). This site was classified as category 1, because all four non-nutrient constituents (EC, pH, TAL and SiO_3^-) exhibited strong negative correlations with discharge. Correlations for the nutrients with discharge were weak, however. Figure 3.2 represents a category 2 site (U6H002, Mlazi River). Electrical conductivity, TAL and SiO_3^- are negatively correlated with discharge. In contrast to category 1 however, there was no correlation between pH and discharge. In addition, nutrients showed considerable scatter. An example of a site exhibiting category 3, Q-C relationships (S3H002, Klaas Smits River) is shown in Figure 3.3. Three out of four of the conservative variables exhibited good ($r^2 \geq 0.5$) negative, relationships with discharge. Silicate, on the other hand was found to be inconsistent, as were combined nitrite and nitrate, as well as phosphate. In the case of category 4 sites (as exemplified by K4H003 Diep River, Southern Coastal belt) the relationship between concentration and discharge was weak for all, or most of, the water quality variables examined.

It has been reported in the literature that some water quality variables, e.g. pH can vary seasonally (Dallas *et al.* 1998). When summer and winter data were examined separately at those sites where the correlation between water quality and discharge was poor, some improvement in the correlation was found for certain variables, in that the value of the regression coefficient r^2 was increased. This improvement was not consistent, however, and in some cases splitting data into winter and summer values made the relationship between water quality and discharge even less clear. This avenue of investigation was therefore not pursued further.

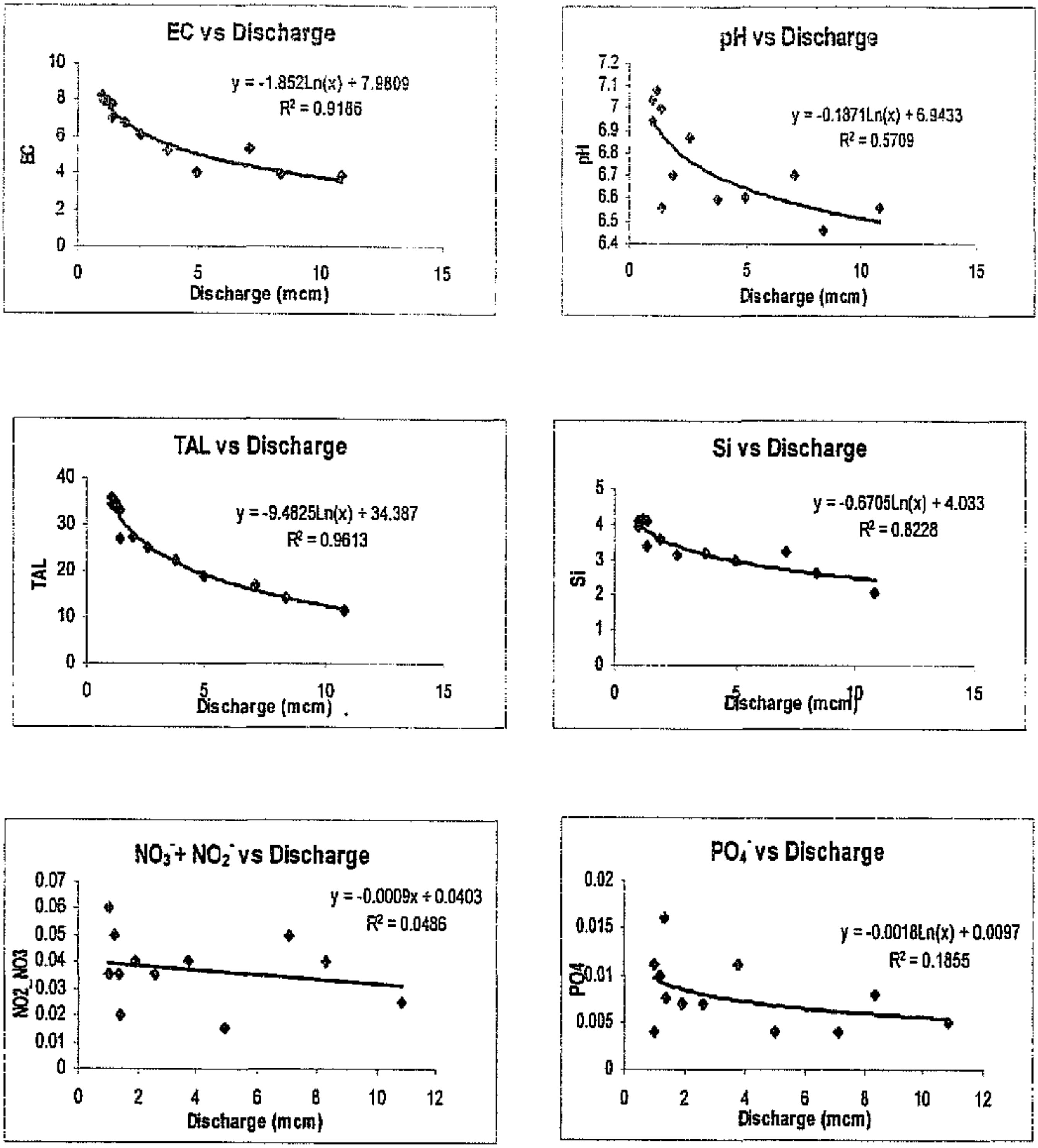


Figure 3.1 *The relationship between a number of water quality variables and discharge at site B6H003 (Treur River, Mpumalanga). These relationships are representative of those described for "Category 1": The correlation of pH, EC, TAL and SiO_3 with discharge is high, and these variables decrease either logarithmically or linearly with discharge. There is only a poor correlation between NO_3^-+NO_2^-, PO_4^{-3} and discharge. SiO_3, TAL, NO_3^-+NO_2, and PO_4^{-3} given as mg/litre, EC as mSm^{-1}, and discharge as million cubic metres/month (mcm).*

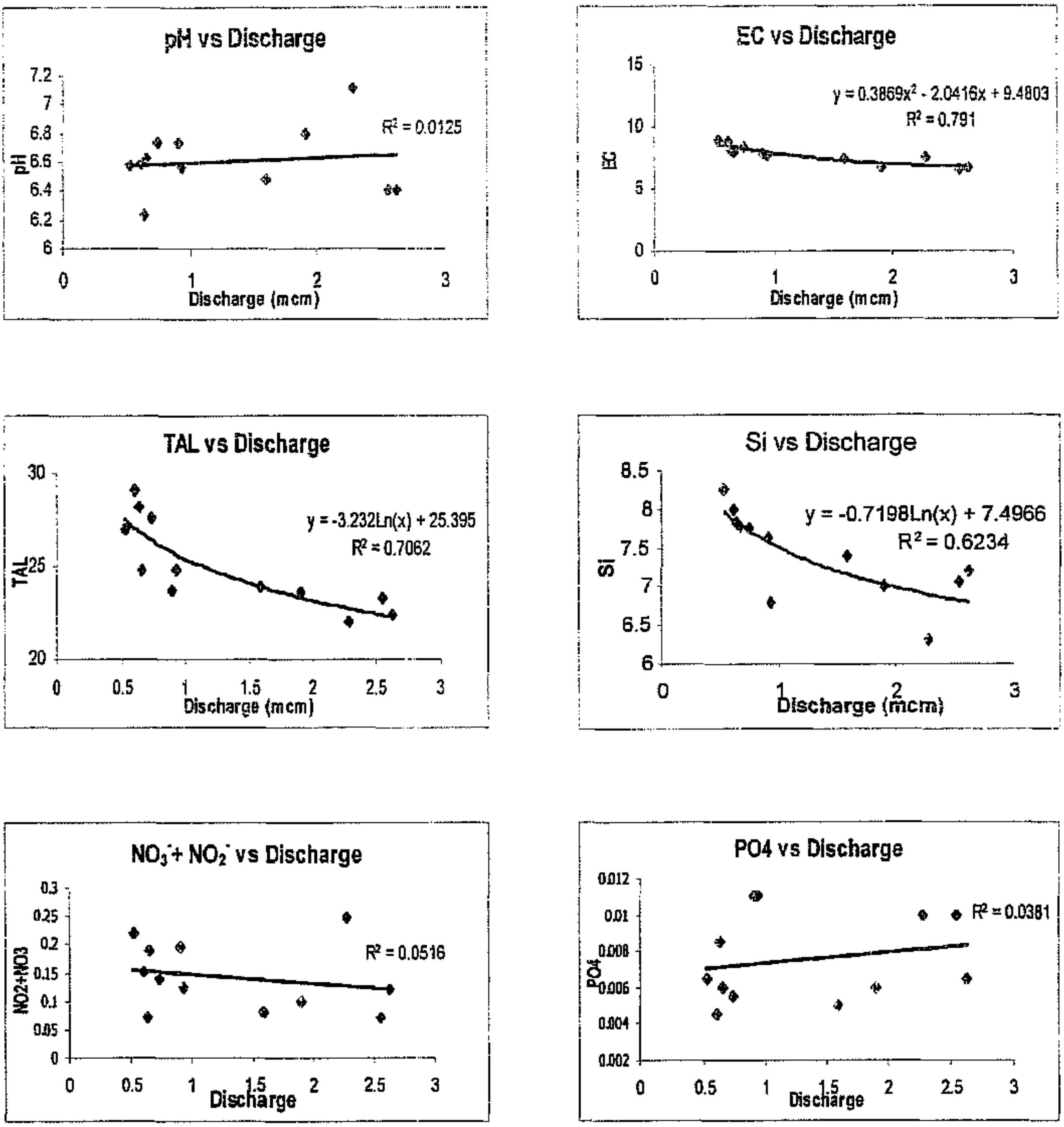


Figure 3.2 *The relationship between a number of water quality variables and discharge at site U6H002 (Mlazi River, Kwazulu-Natal). These relationships are representative of those described for "Category 2": No clear correlation is evident between pH and discharge, whilst EC, TAL and Si all decrease either logarithmically or linearly with discharge. The correlation of both* NO_3^-+NO_2^- *and* PO_4^{-3} *with discharge, is also poor.* SiO_3, *TAL,* NO_3^-+NO_2, *and* PO_4^{-3} *given as mg/litre, EC as* mSm^{-1}, *and discharge as million cubic metres/month (mcm).*

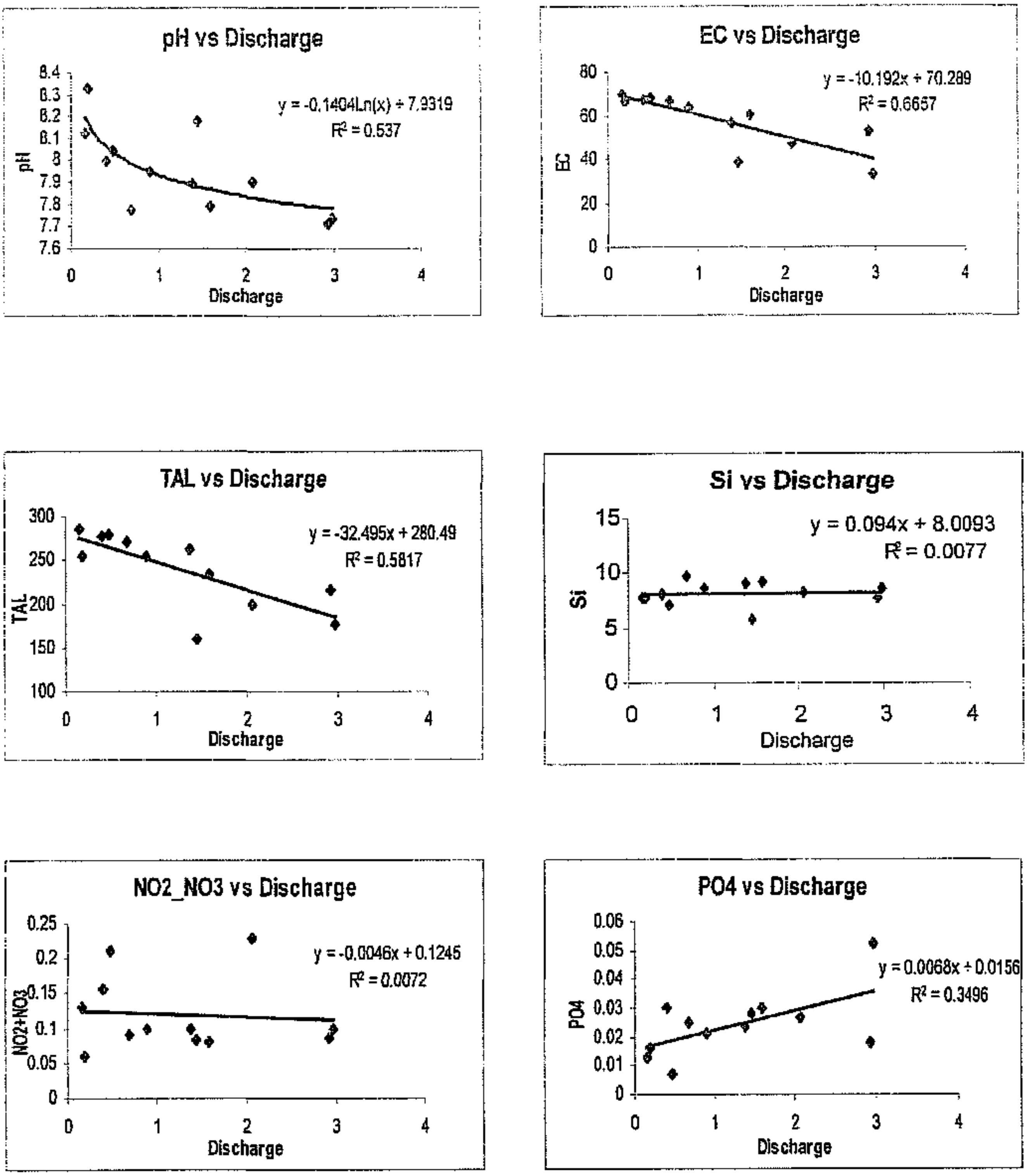


Figure 3.3 *The relationship between a number of water quality variables and discharge at site S3H002 (Klaas Smits River, Eastern Cape. These relationships are representative of those described for "Category 3": pH, EC and TAL all decrease either logarithmically or linearly with discharge. The correlation between Si and discharge is inconsistent across sites within a catchment. The same is true for* NO_3^-*+*NO_2^- *and* PO_4^{-3}*. Although* PO_4^{-3} *seems to be better correlated with discharge at this site, this trend was not found at any other sites within the catchment.* SiO_3*, TAL,* NO_3^-*+*NO_2*, and* PO_4^{-3} *given as mg/litre, EC as* mSm^{-1}*, and discharge as million cubic metres/month.*

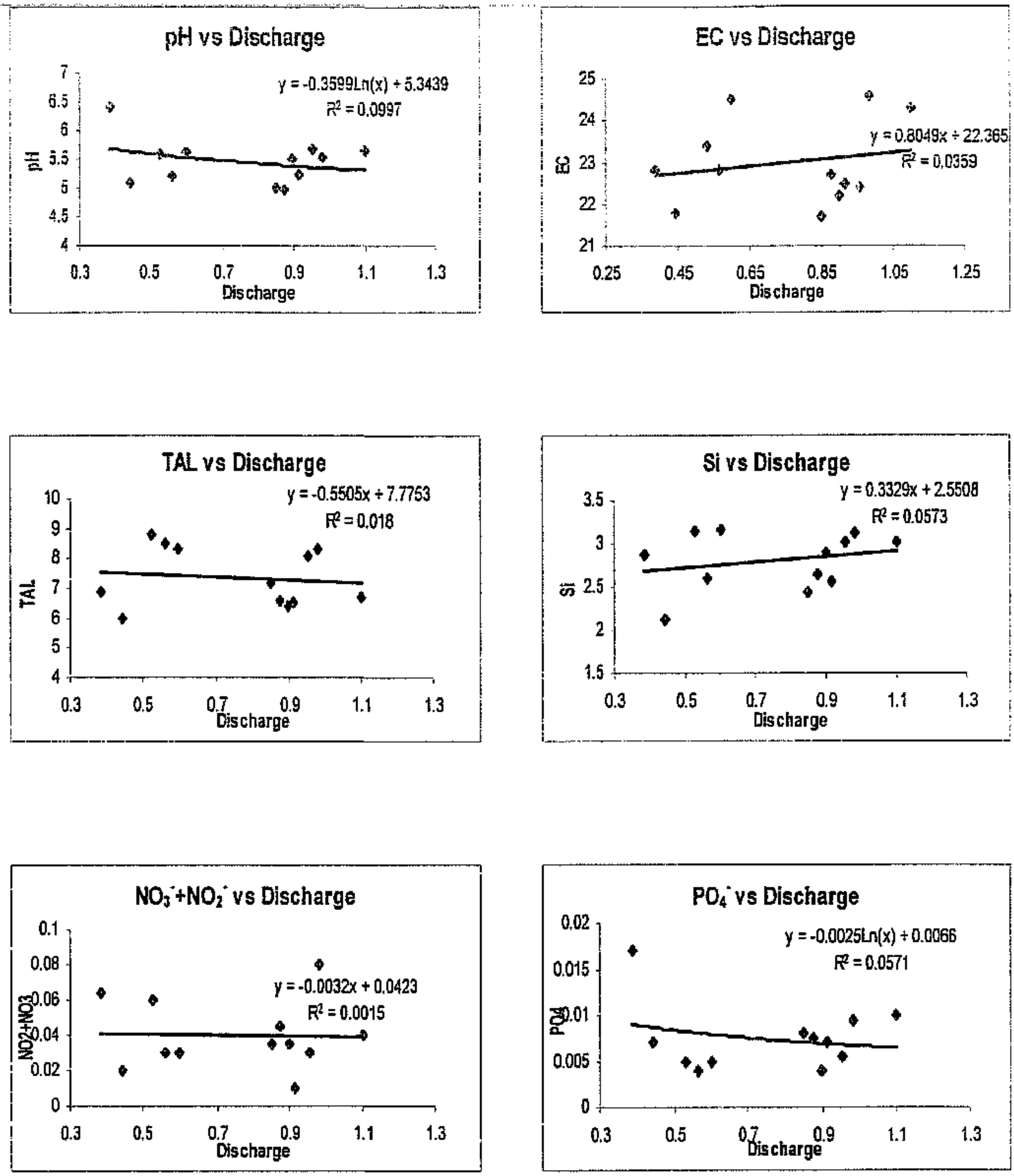


***Figure 3.4** The relationship between a number of water quality variables and discharge at site K4H003 (Diep River, Southern Coastal Belt). These relationships are representative of those described for "Category 4": In this category, either the correlation of water quality with discharge is very weak, or the nature of the correlation (i.e. log, linear etc) between water quality and discharge is highly inconsistent across the stations investigated within the catchment. SiO_3 ,TAL, NO_3^-+NO_2, and PO_4^{-3} given as mg/litre, EC as mSm^{-1}, and discharge as million cubic metres/month.*

3.3.3 Primary drainage regions and Q-C trends

An attempt was made to examine the Q-C trends for all selected sites within a given drainage region, in order to assess if these were consistent. These results are presented in Table 3.3, which summarises the category of Q-C trends that are evident in each primary drainage region. For each site, the trend category was determined. An overall trend category was then assigned to the drainage region by taking the mode category (i.e. the most common) from the three monitoring stations. The consistency of patterns of individual water constituents (i.e. the number of monitoring stations within a drainage region at which the described patterns was evident) was recorded for each drainage region. Consistency was recorded as high when a trend was evident at all sites, average when trends were evident at only two sites and poor if there is a lack of any particular trend. In cases where the trend category was different for all three sites in the same drainage region (i.e. catchments A and H) an overall category number was not assigned. Table 3.3 also shows the strength of the correlation between the concentration of chemical constituents and discharge at all the sites in the drainage region. A strong (good) correlation implies that the r^2 value were greater than 0.5 for most of the variables within that region. An average correlation implies that most of the r^2 values were between 0.3 and 0.5 and a poor correlation that the r^2 values were generally less than 0.3.

In almost all catchments, EC and TAL consistently decreased either logarithmically or linearly with an increase in discharge. Although not tested statistically, it appeared that in catchments in which EC and TAL were low, the dilution effect with increased discharge was not so marked. This is to be expected as the influx of "fresh" surface water at higher flows would not be so different from water largely derived from baseflow and thus a marked change in concentration would not be expected. In the case of pH and SiO_3^-, the correlation of these variables with discharge was not always as clear as in the case of EC and TAL. These variables, either decreased, increased or were independent of discharge.

Table 3.3 A summary of the trends in water quality that are evident within each primary drainage region, including the strength of the correlation between water quality and discharge, as well as the consistency of these patterns for individual constituents between sites in the same catchment.

Drainage Region	Site	Correl-ation*	Consistency within drainage region**	Site Category	Overall Category	Trends within drainage region
A	A6H018 A6H019 A6H021	Average	Low	3 4 1	?	Although a correlation between some water quality varaibles and discharge was evident, the trends observed were inconsistent between the sites.
B	B4H005 B6H003 B7H004	Good	High	2 1 1	1	Log decrease in pH, EC, TAL and Si with discharge. No trends evident for nutrients.
C	C1H007 C2H026 C2H028	Poor	Low	3 4 4	4	Unclear and inconsistent correlations between water quality variables and discharge.
E	E1H006 E2H002 E2H003	Average	Average	1 1 1	1	Trends not very consistent.
G	G1H012 G1H016 G4H008	Average	Average	4 4 4	4	Unclear and inconsistent correlations between water quality variables and discharge.
H	H2H005 H3H004 H6H010	Poor	Average	1 4 2	?	pH, EC and TAL all decrease with discharge. The relationship between Si and discharge is inconsistent. Station H3H004 is inconsistent with this category.
J	J1H015 J3H013 J3H018	Average -poor	Low	4 1 4	4	One site (J3H013) shows clear decrease in pH, EC, TAL and Si with discharge. For remaining sites, no patterns are evident.
K	K3H001 K4H003 K5H002	Poor	High	4 4 4	4	Poor correlation of water quality with discharge. No trends evident.
L	L7H006 L8H001 L8H002	Poor	High	4 4 2	4	No consistent correlation for any water quality variables.
N	N2H009	Average	Only one site	4	-	Only one site, therefore difficult to establish trends.
P	P4H001	Poor	Only one site	4	-	Only one site within the drainage region. No clear correlation of water quality variables with discharge.
Q	Q9H002 Q9H014 Q9H016	Good	Average	3 4 3	3	Log decrease in pH, EC and TAL with discharge. Trends for Si and nutrients are unclear. One site (Q9H014) is inconsistent with trends observed at other two sites.

* A strong (good) correlation implies that r^2 values are greater than 0.5 for most of the variables within that drainage region. Average correlation implies that most r^2 values are between 0.3 and 0.5, and a poor correlation implies that R^2 values are generally less than 0.3.
** Refers to the number of stations within a drainage region at which the described patterns are evident. Consistency is high when a trend (or the lack of any particular trend) is evident at all sites, average when trends are evident at 2 sites and poor when only one site displays the described trends.

Table 3.3 A summary of the trends in water quality that are evident within each primary drainage region continued.

Drainage Region	Site	Correl-ation*	Consistency within drainage region**	Site Category	Overall Category	Trends within drainage region
R	R1H014 R2H001 R2H008	Average	Average	3 1 3	3	Log or linear decrease in pH, EC, and TAL with discharge. Trends for nutrients and Si are less clear.
S	S3H002 S6H001 S6H003	Good	High	3 1 3	3	Log decrease in pH, EC and TAL with discharge. Si and nutrients are unclear
T	T3H004 T3H009 T5H003	Average	Average	3 1 3	3	pH, EC and TAL both decrease with discharge. The relationship between Si and discharge is inconsistent. There is no correlation between NO_3+NO_2 or PO_4 and discharge.
U	U2H013 U4H002 U6H002	Good	Average	2 4 2	2	Logarithmic decrease in EC, TAL and Si with discharge. Trends for pH and nutrients are unclear.
V	V1H041 V2H007 V7H017	Good	High	2 2 2	2	EC, TAL and Si decrease (either logarithmically or linearly) with discharge. Trends for pH and nutrients are unclear.
W	W5H004 W5H006 W5H008	Average	Low	2 2 2	2	Trend for pH is unclear. Log decrease in EC, TAL and Si with discharge. Nutrient trends are unclear.
X	X2H010 X2H014 X3H006	Good	High	1 1 1	1	Mostly log decrease in pH, EC, TAL and Si with discharge. No trends evident for nutrients.

* A strong (good) correlation implies that r^2 values are greater than 0.5 for most of the variables within that drainage region. Average correlation implies that most r^2 values are between 0.3 and 0.5, and a poor correlation implies that R^2 values are generally less than 0.3.
** Refers to the number of gauging stations within a drainage region at which the described patterns are evident. Consistency is high when a trend (or the lack of any particular trend) is evident at all sites, average when trends are evident at 2 sites and poor when only one site displays the described trends.

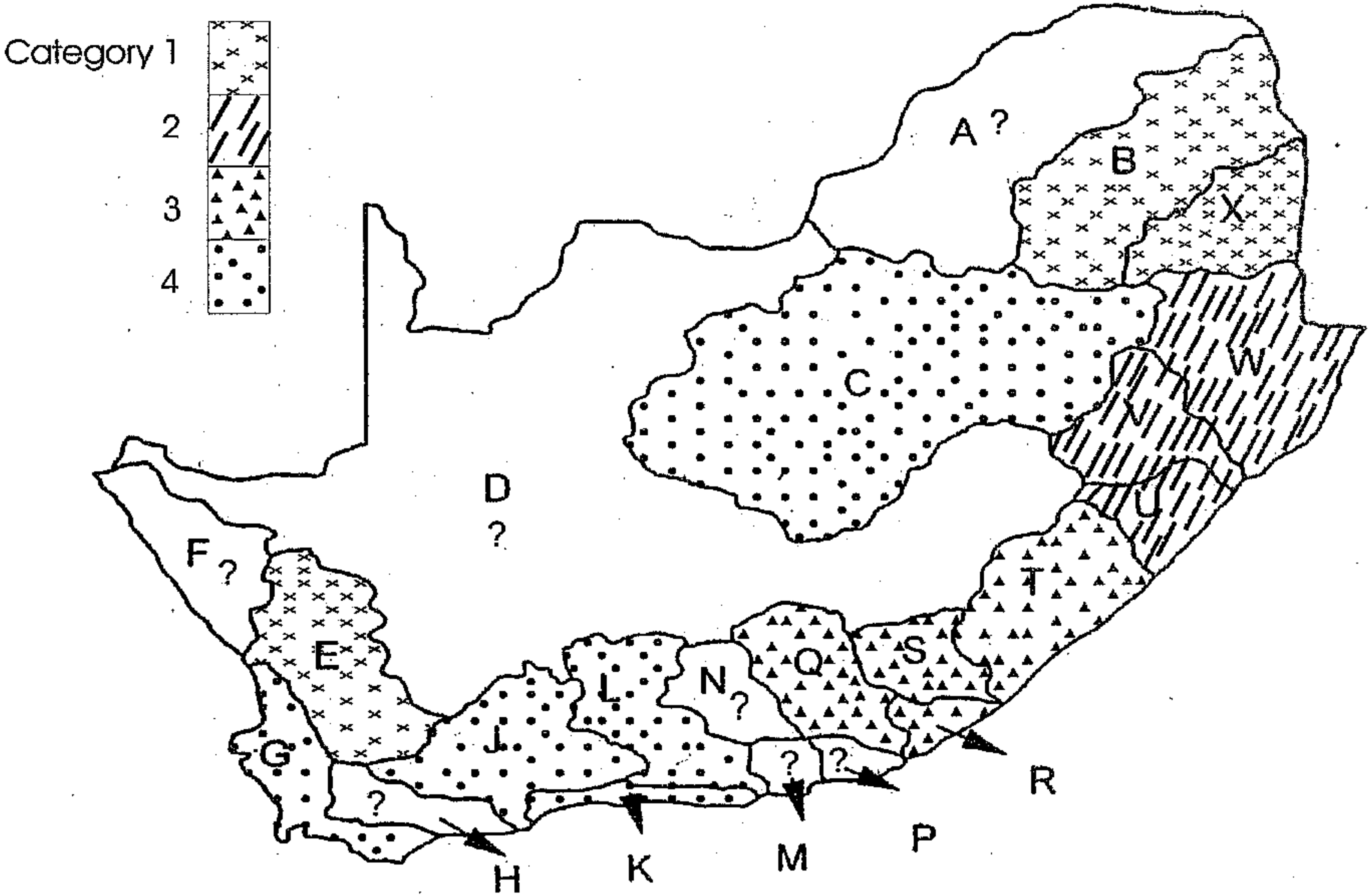


Figure 3.5 *Map showing the primary drainage regions of South Africa (as delineated by DWAF), and the overall Q-C trend category exhibited by each region. (Adapted from Day et al. 1998). Some regions are indicated with a question mark either because there were insufficient natural discharge data (D, F, M), too few unpolluted stations (N, P) or the consistency of trend amongst stations was low (A, H) for the overall trend category to be determined.*

As expected, the non-conservative constituents, $NO_3^-+NO_2^-$ and PO_4^{-3} did not usually show a very strong correlation with discharge in any of the catchments compared to the conservative variables mentioned above. Both of these constituents, either, increased slightly with discharge, decreased slightly or showed no relationship within the same drainage region. Only two of the sites (R2H008 and S6H001) were slightly impacted with regard to nutrients (relatively high combined nitrate and nitrite level). Because of this a comparison of nutrient Q-C trends for non-impacted with polluted sites could not be carried out.

The consistency of patterns within each primary drainage region was generally above average. In other words, the described trends were evident at two or more of the gauging stations sampled within that catchment. Only two catchments, namely catchments A and H, showed a low level of consistency. Thus, at the level of primary drainage region, within the limitations of the small data set, rivers appeared to display fairly consistent patterns with regard to the response of water quality to changing discharge. An examination of Figure 3.5 shows that to some extent, drainage regions that showed the same Q-C trend category tended to be located in the same area of the country. For example, B and X were both classified as Category 1, and represent adjacent catchments. Similarly, U,V,W (Category 2) and Q,R,S,T (Category 3) and G, J, K, L, (Category 4) are situated close to each other. From the above it may be postulated that if more sites from these regions were investigated, primary drainage region A may well exhibit Category 1 Q-C trends, and region H Category 4 trends.

3.3.4 <u>Ecoregions and Q-C trends</u>

In order to investigate if there was any correlation between ecoregion and the type of Q-C response, classification at individual monitoring sites was extended to Level 1 ecoregions. Table 3.4 summarises the list of drainage regions, the type(s) of ecoregion found in each catchment, and the overall Q-C trend category. It can be seen from the table that some catchments are comprised of more than one ecoregion. In addition, some ecoregions exhibited more than one category of Q-C trend. As an alternative approach, the location of the individual monitoring sites was plotted on a map of the ecoregions for South Africa. Plotting the location of individual sites did yield some correlation between ecoregion and Q-C category (data not shown). In some ecoregions, such as the Southern Coastal Belt, responses are consistent both at the catchment level and at an ecoregion level. This is to be expected as adjacent catchments are often in the same ecoregion. In most cases however, the response of water quality to changes in discharge varied within an ecoregion. In the Eastern Uplands for example, a number of responses are evident. Thus it would appear that consistency exists at a catchment level, but that this consistency does not extend to entire ecoregions.

Table 3.4 *A list of the catchments (primary drainage regions) used, the ecoregion within which they are situated and the Q-C trend category. ? indicates that a clear trend category could not be distinguished and – that too few sites were available for analysis.*

Catchment	Ecoregion	Trend category
A	Central Highlands	?
B	Central Highlands/Great Escarpment Mountains/Lowveld	1
C	Highveld/Central Highlands	4
E	Cape Fold Mountains	1
G	Cape Fold Mountains	4
H	Cape Fold Mountains	?
J	Cape Fold Mountains	4
K	Southern Coastal Belt/CFM	4
L	Cape Fold Mountains	4
N	Eastern Uplands	-
P	Southern Coastal Belt	-
Q	Eastern Uplands	3
R	Eastern Uplands	3
S	Eastern Uplands	3
T	Eastern Uplands	3
U	Eastern Uplands/Eastern Coastal Belt	2
V	Eastern Uplands	2
W	Highveld	2
X	Lebombo Uplands/Great Escarpment Mountains	1

3.4 Discussion

The following conclusions can be drawn from this preliminary investigation of regional patterns of discharge-concentration trends.

- Nutrients (NO_3^-+NO_2^- and PO_4^{-3}) showed largely unpredictable responses to changes in discharge at any site. The other variables (pH, SiO_3^-, EC and TAL) usually decreased either logarithmically or linearly with discharge. Different catchments showed slight variations on this theme (these variations are represented in the different categories). In general, it was difficult to categorise trends from all the graphs produced, often because of the range of correlation values and the absence of any particular pattern at a site.

- Discharge-concentration trends seem to be fairly consistent at the level of primary drainage region. In catchments where the consistency of trends was low, this normally meant that the value of the correlation coefficient (r^2) was also low, in other words no particular trend was evident. It must be kept in mind however, that only three sites per drainage region were examined. Thus, although tentative predictions could be made as to the likely Q-C patterns at a site, this requires confirmation by examination of more sites.
- Q-C trends were not as consistent within an ecoregion as within primary drainage regions. In some ecoregions, for example (e.g. the Southern Coastal Belt), responses were the same at both a primary drainage and ecoregion scale, but most ecoregions encompassed a variety of responses. Division of the country into Level 1 ecoregions is based on considerations of physiography, climate, geology and soils as well as potential natural vegetation (Kleynhans and Hill 1999). There was also some correspondence between the water quality management regions as proposed by Day *et al.* (1998) and the categories of Q-C patterns exhibited by the different drainage regions (data not shown). This second system of delineation of South African rivers is based on consideration of many inorganic chemical variables including conductivity, pH, fluoride, silicate, ratios of inorganic ions. In section 1.6.1, a brief discussion was given of the factors that influence geohydrological responses in rivers (this is discussed in more detail in the literature review to this project: Malan and Day 2002). It was explained that amongst other factors, the climatic or geological region in which a site is situated is very important in determining the Q-C trend, in addition to catchment land-use. Thus it is to be expected that the categories of Q-C trends that were described in this report should to a greater or lesser extent correspond with ecoregions or water management regions as described above. Time was too limited however to pursue these avenues of research further and to examine exactly which factors show the most significant correlation.
- River type (i.e. mountain stream, foothill, lowland river, etc.) was also not taken into account in this study. Because sites were chosen that were relatively unimpacted with regard to discharge regime and were thus situated above impoundments, the sites all tended to be either mountain streams or foothill rivers.

- The sites examined in this study were relatively unimpacted with regard to water quality, and in particular were not polluted by nutrients. Results from the literature survey of Q-C trends suggest that at non-impacted sites, inorganic nitrogen and phosphates may decrease, or show little change with increased discharge. In catchments where there is extensive agriculture or urban development however, nutrients may well increase with discharge, due to wash-off from the surrounding land. These observations were not supported by the results from this survey, since even in unimpacted catchments, nutrients showed a wide variety of responses. It would be useful to extend the investigation to nutrient-enriched sites however in order to ascertain if an increase in concentration of phosphorus or nitrogen with increasing discharge, at low flows, always occurred.
- The separation of data into summer and winter values showed inconsistent results. Some variables (e.g. electrical conductivity) generally showed a slight improvement (i.e. increase) in correlation with discharge when the data set was split, whilst others showed no change. It is possible that some clearer patterns would emerge if more sites were examined, but based on the few sites that were examined, no conclusions could be drawn.
- One of the regional trends that was noticeable during the water quality modelling carried out for Reserve determinations was the marked increase in TDS (and usually combined nitrates and nitrites) with discharge at some sites. This was found to occur in salinised catchments of the winter rainfall region (section 2.11.1). Such a Q-C trend was not noted in this part of the project however (drainage regions H and G) because salinised sites (high EC) were not considered.

In conclusion, some, but not all of the original aims of this part of the project have been met. A review has been undertaken of the Q-C trends for sites throughout the country and it has been found that, to a limited extent, these responses are predictable and consistent within the same primary drainage region. More sites in each region need to be investigated with regards to the Q-C trends exhibited by individual water quality constituents, however. The reasons for this are firstly, to establish whether the four categories chosen are suitable, and secondly, to ascertain if the overall category assigned to the primary drainage regions is appropriate. Thus, although interesting results have already been obtained, the work carried out so far can only be considered to be preliminary.

CHAPTER 4

DEVELOPMENT OF THE TIME-SERIES MODELLING METHOD

4.1 Introduction

The work for this chapter arises from collaboration with Prof. C. Palmer, Centre for Aquatic Toxicology, Institute for Water Research (CAT-IWR), Rhodes University, during the Reserve determinations for the Olifants (Mpumalanga) and Breede (Western Cape) Rivers. Using discharge-concentration relationships derived from Q-C modelling, hydrology time-series were converted to time-series of concentration, for selected sites on both rivers. In the case of TDS, time-series of salinity were then further transformed to those of perceived ecological stress using ecotoxicological data. The ecotoxicological data were obtained from experiments conducted at CAT-IWR as part of the WRC-funded project (Project number K5/1108) titled "The integration of water quality tools for the ecological Reserve into a risk-based DSS". This chapter therefore presents the results of a synergy between the two projects and is documented in the final report of each. The results of Q-C modelling, as well as time-series modelling for the OREWRA project (Olifants River) are documented as Appendix VII (Malan 2001) in the assessment of water quality for the study (Palmer and Rossouw 2001).

Chapter 7 of this report gives an outline of an IFA (Instream Flow Assessment) and how water quality is currently integrated into the process, as well as potential future developments in linking water quality and quantity. It is explained that the IFR process no longer ends with the specialist workshop and the production of a recommended discharge regime in the form of a table showing months of the year and discharge for maintenance and drought years. In order to comply with the needs of the DWAF yield model (Water Resources Yield Model; WRYM), an IFR model has been developed. The output from the IFR model is a time-series of discharge that, for each calendar month, specifies the percentage of time that the modified discharge regime is at or above maintenance levels, between maintenance and drought, and at drought levels. Using WRYM and the output from the IFR model, different flow scenarios can be generated. A Scenario workshop is then held in which the ecological specialists assess the environmental impacts of the various flow scenarios suggested by the yield modeller, and rank them. Although the types of information that could be derived using Q-C modelling were found to be suitable for the needs of the IFR workshop (Chapter 7), they were not suitable for comparing different flow scenarios as presented at the scenario workshop. Converting discharge time-series to concentration or stress time-series is an attempt to assess and rank the various alternative flow scenarios with regard to their water quality consequences.

In addition, there has been a movement away from expressing the implications of altered water quality in terms of *hazard* for the biota (how toxic a given concentration of chemical constituent is) to expressing it in terms of *risk* (the likelihood of a given concentration occurring). This is in line with developments in other countries. The philosophy and reasoning behind this movement is discussed fully in Jooste, Mackay, Scherman *et al.* (2000). Thus, water quality modelling as carried out in Reserve determinations for the Olifants and Breede Rivers entailed the following. Firstly, the stress (hazard) that a predicted concentration of chemical constituent would be likely to exert on the aquatic biota was determined using the results of ecotoxicological testing. This was carried out for salinity only. Secondly, the risk to the biota was defined as the combination of the concentration of salinity predicted, with the frequency and duration of occurrence. This was determined by linking concentration (and hence stress) to the modified discharge time-series and deriving duration curves.

In summary then, concentration time-series were developed:

- So that the water quality consequences of different discharge scenarios generated using the WRYM model could be assessed and ranked.
- In order to make water quality assessments risk-based rather than hazard-based.

4.2 Methods used

Discharge time-series were converted to time-series of concentration (concentration profiles) using the computer package "TSOFT". This software was developed at IWR, Rhodes University, by Prof. Hughes and is described in detail in Hughes, Forsyth and Watkins (2000). The steps taken to obtain the concentration profiles are explained below and are summarised in Fig. 4.1. The process is carried out separately for each water quality variable (but see section 4.2.1) at each IFR site.

1. *Prepare a transformation matrix for both the Reference condition and Present ecological state for the given chemical constituent C.*
In addition to a discharge time-series, TSOFT requires a transformation matrix as input. The matrix must consist of exactly 20 discharge values and the corresponding concentration values (or concentration values and corresponding stress levels, see later). In order that the profiles obtained be as accurate as possible, the matrix should cover the entire discharge range.
i) The discharge time-series to be transformed is examined in TSOFT and the range over which flows occur at that site is determined. The approximate 1:10 year flood flow is also noted.
ii) The range of discharges for which the Q-C model has been set up is examined. The Q-C model uses monthly mean discharge values and thus will not extend to the full range shown by the time-series. The function linking discharge and concentration derived by Q-C modelling can be used to calculate the corresponding concentration for the middle portion of the matrix.
iii) The relationship for the extremes of discharge (i.e. very low flows – those less than the lowest monthly mean value, and floods, or flows higher than the maximum monthly mean flow) is then derived. The relationship linking discharge and concentration derived using Q-C modelling can also be used to calculate the corresponding concentration for each discharge, although expert judgement is needed. A full Q-C graph, which utilises all the Present ecological state water quality data points (rather than monthly medians) with corresponding discharge can be examined to help in this regard. In particular the discharge can be determined above which the concentration of C remains constant (and is usually the same as in the Reference condition). The matrix is set up to cover

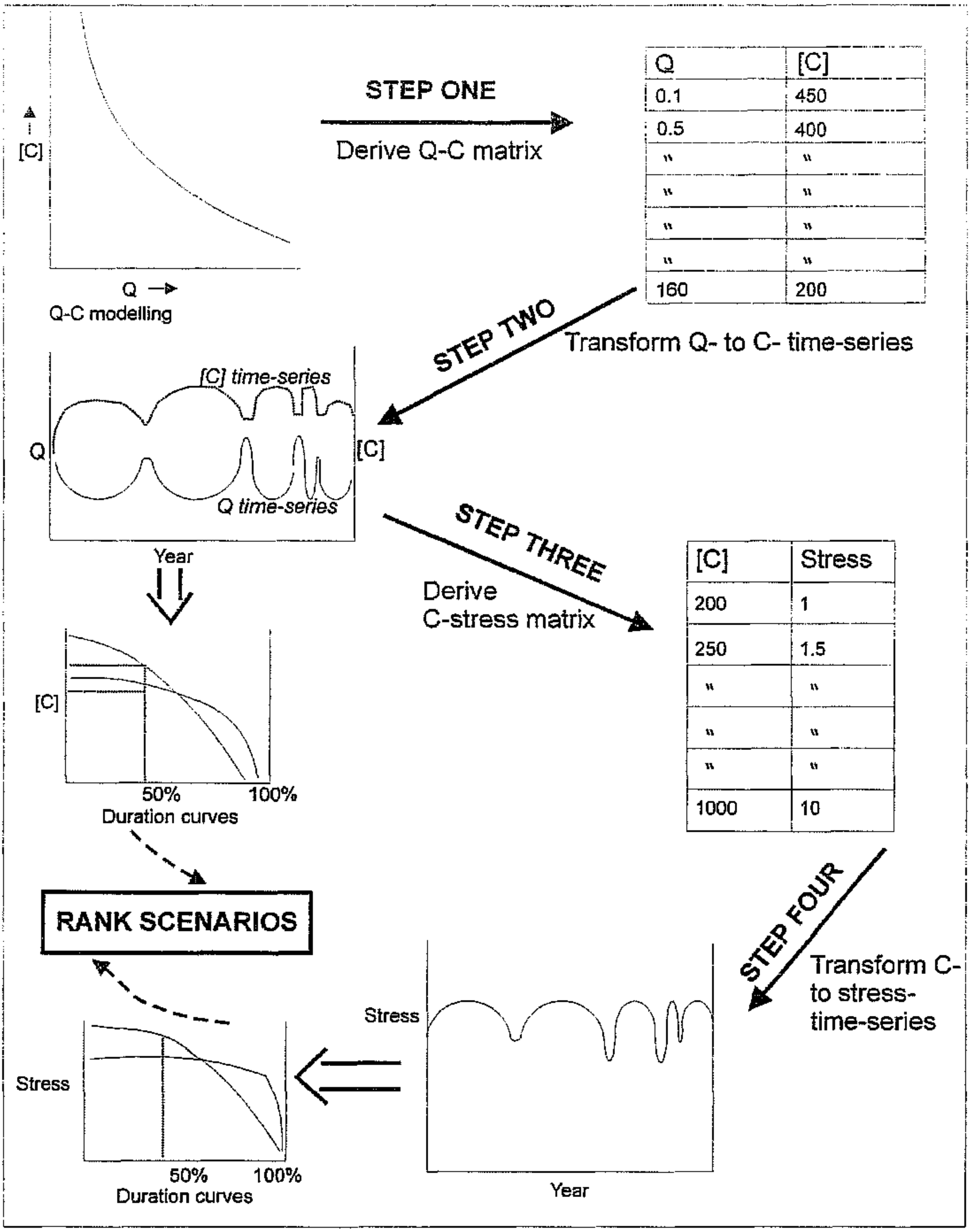


Fig. 4.1 Summary of the steps for preparation of concentration and stress time-series and comparison of different flow scenarios.

flows ranging approximately from the lowest discharges that occur at the site to approximately the 1:10 year flood. An example of a transformation matrix for converting discharge to TDS is given in Figure 4.2, for IFR site 13 (Tulani) on the Olifants River. The monthly Q-C plot from which the matrix was derived is also shown.

2. *Use the transformation matrix to transform the discharge time-series to a time-series of corresponding concentration of the chemical constituent C.*
The output from TSOFT is a time-series plot of date or time (usually year) versus concentration. An example from site 13 on the Olifants River is shown in Figure 4.3. The concentration time-series can then be manipulated using facilities built into the software. Thus duration curves can be generated for different flow scenarios and using these, the water quality consequences of different discharge regimes can be compared (Figure 4.4).

If data are available, the following additional steps can be taken.
3. *Prepare a transformation matrix to convert concentration to corresponding stress level.*
In a similar manner to concentration time-series, stress time-series can be generated using concentration-stress relationships derived from ecotoxicological parameters. The stress index was expressed on a scale of 1-10, where 1 represents very low stress (low concentration) and 10 extreme stress. A matrix linking stress and concentration was set up and used to convert concentration time-series to stress time-series. Different stress matrices were drawn up for different rivers. This is discussed further in section 4.2.2.

4. *Use the* stress *transformation matrix to transform the concentration time-series to a time-series of corresponding stress levels.*
This is carried out using TSOFT, in an analogous manner to the preparation of concentration time-series. Duration curves can also be derived.

5. *The concentration and stress profiles arising from several different flow scenarios are ranked according to their perceived impact on the aquatic biota.*
This is discussed further in section 4.8.

4.2.1 Points to note

The following aspects were deemed to be of importance.

- Because of the inherent inaccuracies in extrapolating to flows for which no observed TDS values are available, the transformation matrix is set up to cover only up to the 1:10 year flood flows. Floods equal to, or larger than this, are all set at the minimum concentration value. The fact that the concentration of most variables usually decreases as discharge is increased (section 2.11.1.1) is considered to be justification for this approximation. Thus, it is usually during periods of low flow, rather than during floods that water quality problems are most likely to occur. Exceptions do occur however (for example, TDS in salinised catchments of the winter rainfall region, as well as nitrates and phosphates in some rivers). Predictions using Q-C modelling are not recommended in such situations and only limited modelling (e.g. for the Reference condition), can usually be carried out. This is discussed further in section 4.5.
- The Q-C plot for the Present ecological state is used to prepare the transformation matrix and hence the concentration profiles of different flow scenarios. If a concentration profile that is representative of the natural state is required however, the Q-C relationship derived from Reference condition water quality and discharge data is used to set up the transformation matrix.
- As in the case of Q-C modelling, the predicted concentration time-series is based on the assumption that the current pollution load will remain the same.
- Concentration time-series were prepared for selected IFR sites on the Olifants and Breede Rivers. The sites were chosen prior to the scenario workshop and represented key IFR sites, including those where insufficient water was available to meet the demands of the Reserve as well as the existing water users (in other words, where the demand for water was likely to be greater than the supply).
- TDS time-series were produced for all the key sites mentioned above. The reasons for focusing on this chemical constituent were as follows. Firstly, TDS was considered by the project team to be one of the most important water quality constituents in the Olifants system and to be important in determining the overall present water quality status of a water resource. This is supported by the fact that TDS is classed as a system variable in South African Water Quality Guidelines (DWAF 1996). Secondly, the CAT-IWR team had already carried out preliminary salinity toxicity testing in other catchments and an experimental protocol for this

variable had been developed. Other water quality variables can also be modelled in the same manner and can provide useful information. For example, fluoride profiles were prepared for the IFR site 15 on the Olifants River (section 4.4.1.2), and combined nitrates and nitrites for a site on the Molenaars River section (section 4.5). Taking into account the limitations and approximations in the modelling methods (sections 2.5 and 4.3), any water quality variable that is suitable for Q-C modelling and gives good simulations can potentially be converted to concentration time-series.

- Time-series of environmental variables have been used in other countries to compare the impacts of different discharge regimes. For instance, Waddle (1998) reported the use of time-series of suitable habitat area (expressed as Weighted Usable Area) derived from PHABSIM (Physical Habitat Simulation System), a methodology developed in USA to assist in the determination of the EFR for rivers (Milhous 1998). The term "suitable habitat area" refers to the area within a river reach that is suitable for a target fish species. Discharge time-series were converted to habitat time-series using a habitat-flow relationship and habitat duration curves from different discharge regimes were compared. By comparing the duration curves, the different flow regimes could then be ranked according to the likely impact on the habitat availability for the target fish species.

4.2.2 Use of ecotoxicological data

Using ecotoxicological testing, the toxicity of either Na_2SO_4 or NaCl (whichever was the most important chemical constituent, in terms of mass, of TDS in the system under consideration) was determined. The effect of different concentrations of either salt on the percentage survival of mayfly nymphs exposed in an artificial stream situation was examined. These results were used to predict the effect that a given level of TDS would have on the aquatic biota in the river. From this information the stress that might be exerted on the system at a given level of salinity was inferred. Toxicity testing was undertaken by the CAT-IWR team at Rhodes University. The results from these experiments are documented in Palmer and Rossouw (2001) and Palmer, Rossouw, Malan *et al.* (*in prep.*). Due to financial, time and logistical constraints, only the toxicity of Na_2SO_4 was investigated in the case of the Olifants River and only NaCl in the Breede River. These salts were found to be the major constituents of TDS in their respective systems. This finding is consistent with the fact that the major source of salinity pollution in the Olifants River is from mining, whereas in the Breede River it is largely due to agricultural return flows. Also due to financial, time and logistical reasons, the concentration-response relationship was investigated for only one genus of aquatic invertebrate, namely mayfly nymphs, *Tricorythus spp.* Both acute (96 hour) and chronic (10-day) experiments were carried out in an artificial stream situation to which differing concentrations of Na_2SO_4 or NaCl had been added. The criterion of toxicity used was total immobility of the organism, rather than death. Ecotoxicological parameters such as [1]LC_1 (the concentration required to cause death of 1% of the test population) LC_5, and the chronic effects value (CEV) etc. were calculated from the experimental results. These were then used to relate concentration, to biotic response.

It is acknowledged that the stress experienced by an organism in a given reach is likely to be the cumulative effect of salinity, and of effects caused by other factors such as temperature, toxic substances, hydraulic habitat etc. In this work, the term "stress" refers to that effect brought about by salinity (or another chemical constituent under consideration) alone. So far no effort has been made to integrate the different factors to obtain an overall stress value. This is a major inadequacy in the method that requires further investigation.

Ecotoxicological data were used in two different ways in the Olifants River Ecological Water Requirement Assessment (OREWRA) and the Breede River Basin study.

- The results from salinity testing were used to delineate the TDS assessment categories. This was necessary because the original method of percentage deviation from reference condition as specified in the RDM manual (DWAF 1999) did not correspond to the categories defined using SASS scores (Palmer and Rossouw 2001). In other words, the results from biomonitoring indicated that the resource was in a much better present state than would be indicated from a consideration of TDS concentration. This is fully discussed in the final report for the WRC project K5/1108 "The integration of water quality tools for the ecological Reserve into a risk-based DSS".

[1] The ecotoxicological parameters LC_1 and LC_5 should more correctly be designated as EC_1 and EC_5, but to avoid confusion with electrical conductivity Palmer *et al.* (*in prep.*) use the former.

- The results from salinity testing formed the basis for the stress-concentration matrix. This was then used to transform concentration time-series to time-series of salinity-induced stress.

4.3 Assumptions and limitations in the method

It was explained in Chapter 2 that due to the many factors that can influence Q-C relationships, the predictions made using Q-C modelling are not particularly accurate. Using these relationships to obtain time-series of concentrations therefore represents a further approximation. Consequently, the specific results of this type of modelling (e.g. the percentage time that a chemical constituent will be within a given assessment category) will therefore not be particularly accurate. Nonetheless, the method is useful for comparing and ranking different flow scenarios with regard to their water quality consequences and (within limits), the resultant stress on the biota. Uncertainty in modelling (both Q-C and time-series) may sometimes make it difficult to distinguish between different scenarios and it is important that further work be done to incorporate confidence intervals into concentration time-series and duration curves.

It was also explained in Chapter 2 that the predictions of concentration are valid only if the system is operated in the same manner as used to derive the Q-C relationships. In the yield modelling that was undertaken for the OREWRA project, no information was available as to how the relative sources (from the main river, tributaries etc.) of water would change between flow scenarios. Nor was attention given to the effect amelioration of point sources of pollutants would have on the loading of constituents in the system. Thus the actual effects on water quality could not be determined, and all predictions are made on the premise that the source of water for all flow scenarios was the same. Obviously, this is likely to be a simplistic assumption. Qualitative statements such as "the discharge from tributary X, which carries good quality water should be maintained in order to ensure that salinity at site Z downstream of the confluence is not compromised" were included in the scenario report to encompass likely effects from changes in the source of water. Although it is likely to increase the complexity of the scenario phase, there is an urgent need to include more sophisticated water quality modelling techniques in order to ensure that the water quality Reserve is attained. This aspect is considered in more detail in Chapter 7.

In addition to those mentioned above, various other simplifications of the real situation are made when applying this method and are discussed below. A more complete description of the assumptions made and the limitations in using ecotoxicological parameters to infer water quality-induced stress for the aquatic biota is given in Palmer *et al.* (*in prep*).

- Since the results from Q-C modelling are used to prepare concentration profiles, any limitations that apply to Q-C modelling are also likely to apply to time-series modelling.
- Unless there are measured water quality data for very low and very high discharges, extrapolation to these regions is likely to be inaccurate.
- In converting time-series of discharge to time-series of salinity (and consequently stress) it is assumed that TDS concentration is dependent only on discharge. This might not always be a valid assumption. In the case of Mamba for instance (IFR site 15, Olifants River) it was noted that extremely high TDS values were sometimes experienced in February, when flow was relatively high. This was presumably due to increased output from a point-source during that period. Other activities may also lead to sporadic increases in other water quality variables that are independent of discharge. Scouring of the Phalaborwa barrage for example, has been cited as the cause of downstream peaks in sediment and total phosphate concentrations.
- In the case of the Breede River Basin study, daily discharge data were transformed to daily values of concentration using a Q-C relationship derived using monthly water quality data. This process obviously involves inaccuracies.

The following assumptions and limitations in the method pertain to the extrapolation from concentration time-series to time-series of the stress on aquatic biota. A more detailed discussion of these factors is presented in Palmer *et al.* (*in prep.*).

- Because of limited time and financial resources, ecotoxicological testing was carried out using only Na_2SO_4 (in the case of the Olifants River) and NaCl (in the case of the Breede River) and these results were extrapolated to make predictions of the effect of TDS. Total dissolved solids (TDS) is a combination of many chemical compounds and not just Na_2SO_4 or NaCl. Because of extensive mining, sulphate is the most important contributor to TDS (salinity) in the Olifants catchment. In a similar manner NaCl is assumed to be the most appropriate surrogate for TDS in the Breede River. Because of limited time and financial resources, ecotoxicological testing was carried out using only one genus (three species) of invertebrates, and these results were used to derive the implications of changed salinity conditions for the entire invertebrate assemblage. It is very likely that some species may well be more sensitive to the surrogate TDS salts that were selected and similarly that others are likely to be less sensitive.

- Ecotoxicological testing was carried out in artificial streams in which environmental parameters were controlled. It is difficult to extrapolate the levels of stress experienced under artificial conditions to those in the field. In addition, In the case of the Olifants River, the experiments were carried out using filtered Grahamstown tap water to which Na_2SO_4 had been added. Due to differences in chemical speciation of Na_2SO_4 in filtered tap water compared to river water, the toxicity of these chemical compounds in the field may not be the same as in the laboratory.
- Lethal concentration parameters (i.e. LC_1, LC_5), were used to infer chronic effects on invertebrate populations.
- As noted above, the stress experienced by the aquatic biota is the cumulative effect of salinity as well as possible effects caused by other chemical constituents and physical variables (temperature, dissolved oxygen), hydraulic habitat, biotic effects such as predation etc. Ignoring the other stressors and considering just salinity is a gross simplification, despite this we propose that this is a useful approach for comparative purposes.

4.4 <u>Application to the Olifants River</u>

TDS profiles were prepared using different discharge time-series in order to compare the various flow scenarios with regard to the likely resultant water quality consequences. In the case of the Reserve determination for the Olifants River, the discharge time-series used were generated using the Water Resources Yield Model (WRYM). This was part of a system analysis to assess whether the ecological Reserve and demands of existing users could be supplied from the water available in the system (Louw and Maré, 2001).

A range of flow scenarios was examined and the consequent salinity implications assessed. This was carried out by preparing salinity duration curves, and by examining the percentage time that salinity would be in each assessment category. These results were presented at a Scenario workshop attended by specialists from different ecological disciplines that were involved in the Reserve determination for the system. The specialists were able to make use of these data in assessing the effects of and ranking the acceptability of each flow scenario with regard to their particular discipline.

4.4.1 Results from the Olifants River system

The results from Q-C modelling for all the modelled sites on the Olifants River are presented in Chapter 2 and in Appendix B. The results for salinity and stress time-series modelling for IFR site 13 (Tulani) are presented as an illustration of the type of results that were obtained. Also discussed are the results for fluoride at IFR site 15 (section 4.4.1.2).

4.4.1.1 *TDS at Tulani (IFR site 13)*

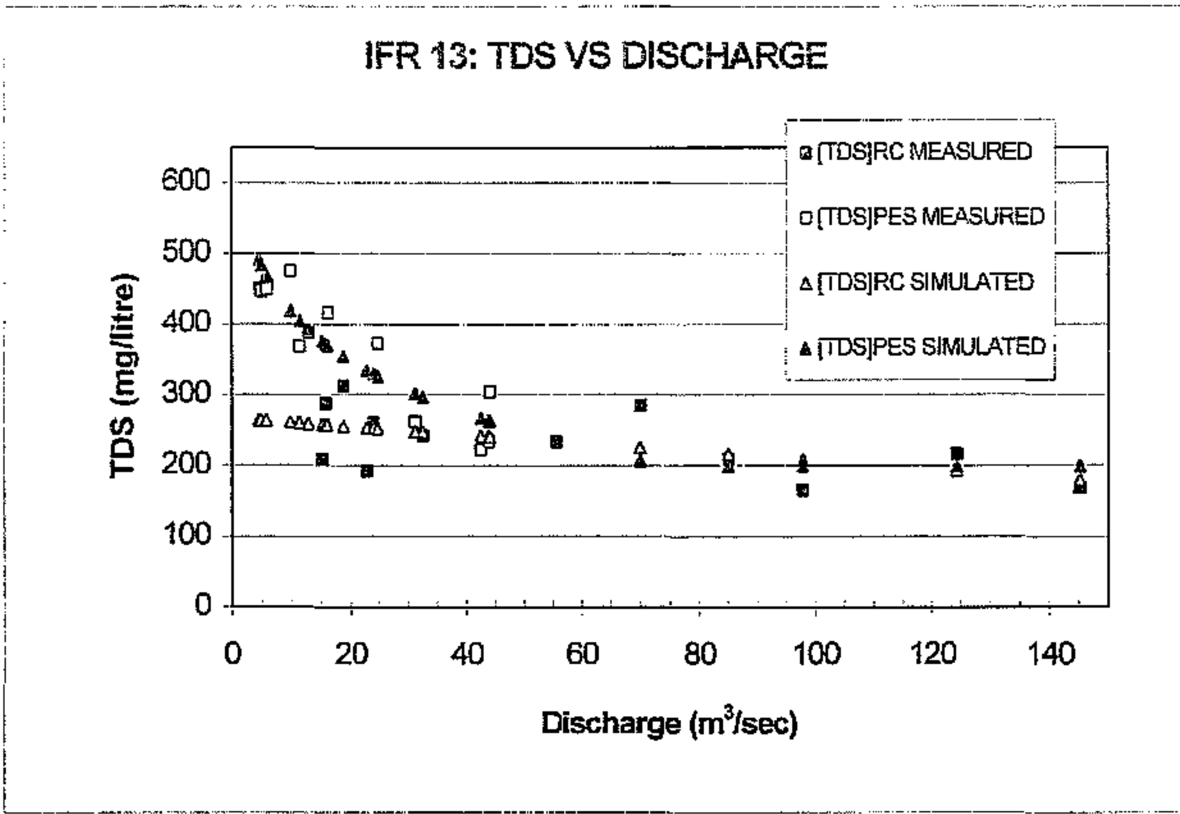
This IFR site is on the Olifants River, downstream of the confluence with the Blyde River and upstream of the Phalaborwa barrage and the confluence with the Selati River (Figure 5.1). The discharge data for the Reference condition were simulated data provided by the hydrologists involved in the OREWRA project. Discharge data that represented the current situation at the site were derived from data collected by DWAF at the nearest gauging station B7H007 (Olifants River at Oxford). The water quality data for the Reference condition for this site were derived from the mean of the Blyde River and the Olifants River at Oxford (B6H004Q01 data set 1992-1997 and B7H007Q01 data set 1992-1997). This was necessary because there were no pre-impact data available from Oxford. Present day water quality was taken from Oxford (B7H007Q01 data set 1992-1997).

i) Concentration profiles

Figure 4.2 shows the Q-C plot for TDS at this site as well as the matrix that was used to convert discharge to concentration using TSOFT. A fairly typical Q-C relationship is exhibited, in that the Present ecological state concentration of TDS is considerably higher than the Reference condition levels at low flows, and that TDS shows a negative trend with increasing discharge. At flows of approximately 70 m^3sec^{-1} and higher, there was little difference between the Reference condition and Present ecological state concentrations of TDS. As a result, in the transformation matrix, flows greater than 70 $m^3.sec^{-1}$ were assigned a concentration value of 200 mgl^{-1} TDS.

The discharge time-series and derived concentration time-series for one of the flow scenarios considered (equivalent to the present day discharge) is shown in Fig. 4.3. Concentrations range between 630 and 200 mgl/litre as prescribed by the transformation matrix. Figure 4.4 shows the mean monthly discharge (a) and mean monthly salinity (b) for IFR 13 as predicted under the different flow scenarios using TSOFT. The values are calculated using the entire flow time-series (1920-1987). The various flow scenarios that were compared are given in the legend at the foot of the figure. In brief the scenarios ranged from present day discharge ("No-IFR") to scenarios in which drought was allowed to occur in the system 10% or 20% of the time (named "10% drought" and "20% drought" respectively).

The flow scenarios considered in the scenario modelling phase of the OREWRA are outlined in Malan (2001), and a comprehensive description is given in Louw and Maré (2001).



Discharge (m^3/sec)	Salinity (mg/litre)	Discharge (m^3/sec)	Salinity (mg/litre)
1	631	25	326
3	530	31	302
5	491	35	289
6	469	43	267
10	419	48	253
11	404	56	236
13	392	70	207
16	372	98	200
19	354	125	200
23	334	145	200

Figure 4.2 *TDS Q -C plot for IFR site 13 on the Olifants River. Also shown is the transformation matrix used to convert discharge to salinity in preparation of the concentration time-series for the same site.*

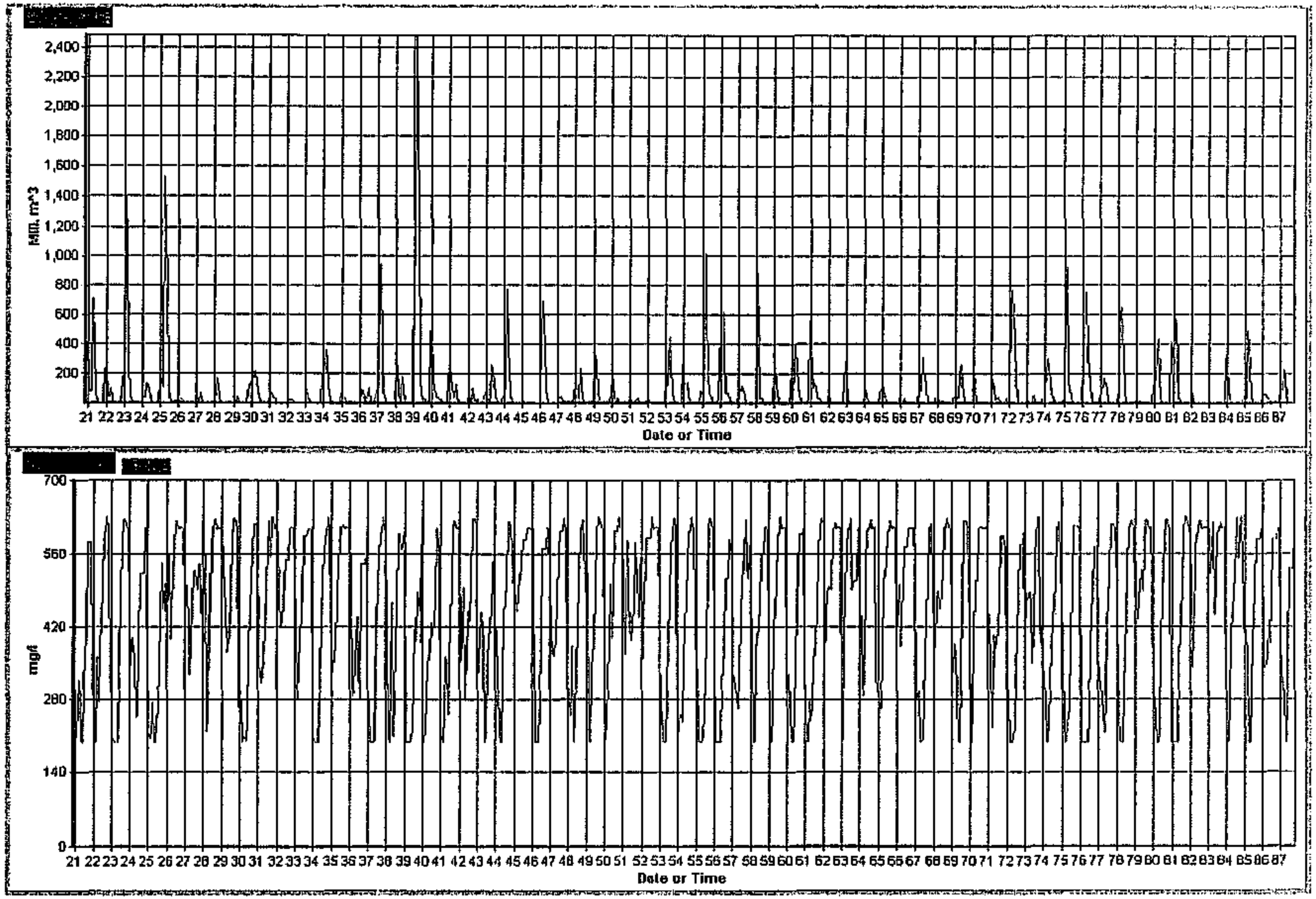


Figure 4.3 *Time-series of present day stream flow in million m³/month (top) and TDS in mg/l (bottom) for IFR 13 (Tulani) on the Olifants River. The x-axis is given in years (1921-1987).*

Figure 4.4 Mean monthly discharge in million cubic meters (a) and mean monthly salinity in mg/litre (b) for IFR 13 as predicted under the different flow scenarios using TSOFT. Values calculated using the entire flow timeseries (1920-1987). Flow scenarios given in the box at the foot of the page.

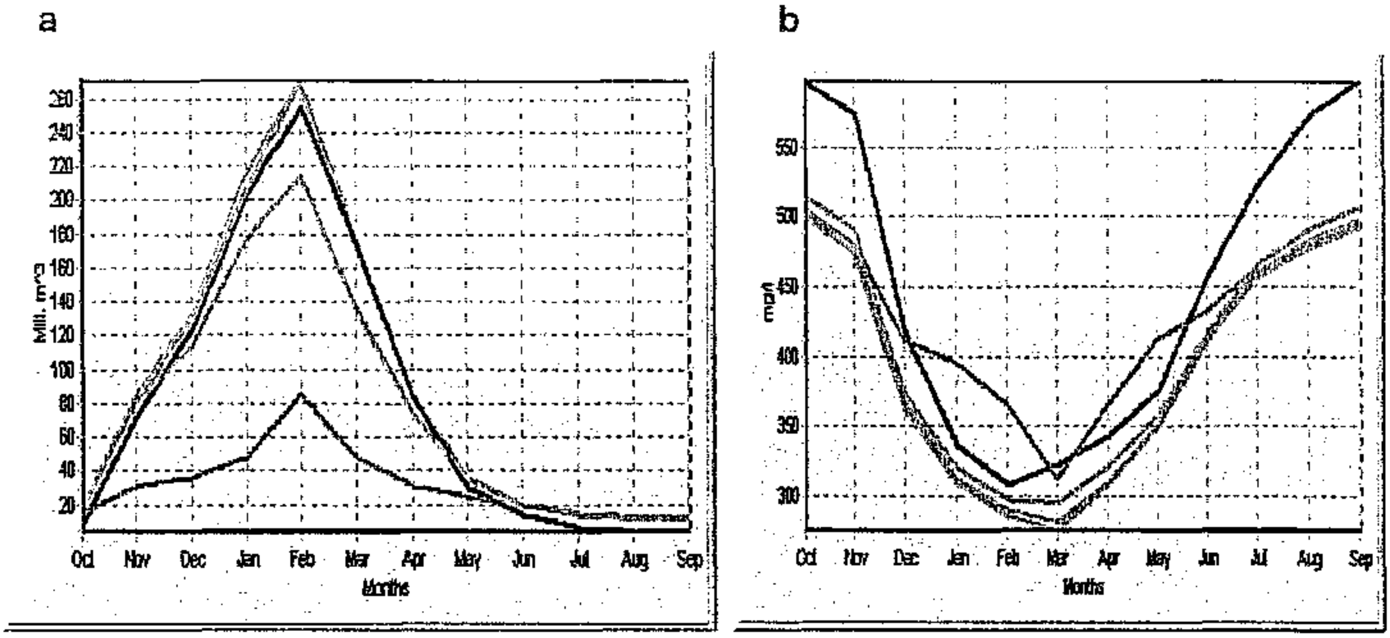
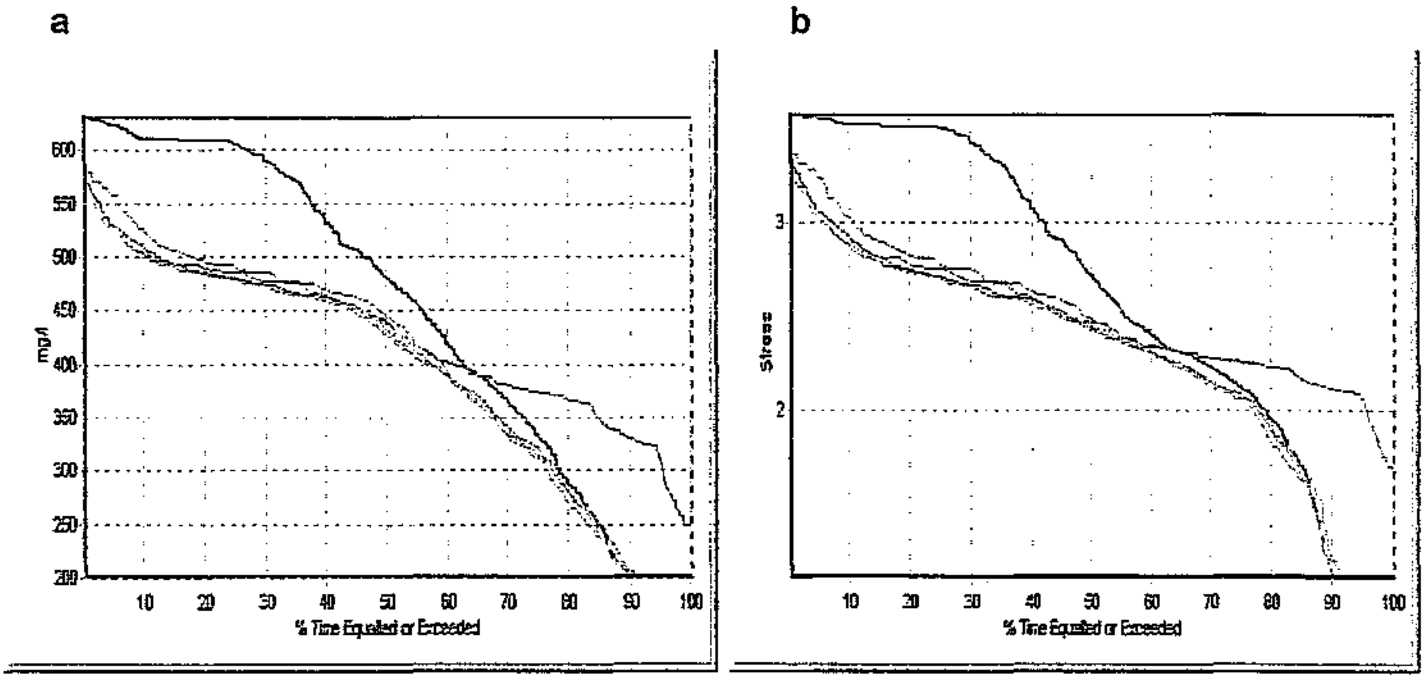


Figure 4.5 *Salinity in mg/litre (a) and stress (b) duration curves for IFR 13 as predicted under the different flow scenarios. Stress derived using a scale of 0-10 as described in Table 4.1.*



No-IFR	10% No-flood
IFR-rule	20% No-flood
Required IFR	PES Alternate

Figure 4.5 shows the concentration duration curves (or more correctly exceedence curves, Hughes, D. IWR, Rhodes University *pers. comm.*), for all the flow scenarios examined at site 13. The "No-IFR" scenario (present day scenario) appears to represent the worst water quality consequences since TDS reaches the highest concentrations under this scenario. Indeed under this scenario TDS would be in a D category (i.e. higher than 520 mg/litre) 42% of the time, whereas for all the other scenarios, TDS will be above 520mg/litre for only 5% of the time. Figure 4.5 also shows that approximately 22% of the time, all scenarios except IFR-rule (this represents just the IFR as determined by the specialists, with no additional discharge) would be in a B category, or better, for TDS (less than 300 mg/litre). Under the IFR-rule flow scenario, TDS would be less than this threshold only 4% of the time. Thus with regard to the water quality scenarios, for site IFR 13, the "No-IFR" flow scenario was ranked worst, followed by the "IFR-rule", and the other four flow scenarios equally in third place.

ii) Stress profiles

Stress time-series were prepared using salinity-stress relationships derived from ecotoxicological results. The stress index was expressed on a scale of 1-10, where 1 represents very low stress (low salinity) and 10 extreme salinity-induced stress. Table 4.1 shows the ERC (Ecological Reserve Class, formerly known as Ecological Management Class, Louw, D. IWR Environmental, *pers. comm.*), TDS and sulphate concentration, and the corresponding value of the stress index. Also indicated are the toxicity descriptor (given in terms of confidence level for the various LC values) as well as the likely biotic response. The derivation of this table is described fully in Palmer *et al.* (*in prep.*). The stress matrix that was used for all sites on the Olifants River is shown in Table 4.2. Stress profiles were analysed for frequency and duration of the highest stress levels as in the case of salinity profiles. Finally, the different flow scenarios were ranked according to the likely increasing severity for aquatic biota.

Table 4.1 *A comparison of the Ecological Reserve Class (ERC), concentration of TDS* and sulphate, stress index value, toxicity descriptor and biotic response descriptor for the Olifants River (from Palmer et al., in press).*

ERC	TDS (mg/litre)	Sulphate (mg SO_4/litre)	Stress level	Toxicity descriptor	Biotic response descriptor
A			0	<measurable response	
	195	130	1	measurable threshold (<low 95% CL)	
B	295	200	2	Threshold below which there is a less than 5% risk of mortality if exposure period > 10 days	Slight reduction in species abundance and health
C	520	350	3	Threshold below which <1% risk of mortality if exposure >4days	
D	780	530	4	Chronic LC_1, ie. 1% risk for exposure longer than 10 days	All species present in short term – infer risk to critical life-history stages.
E	890	602	5		
	1000	675	6		
F	1400	945	7	Chronic LC_{99}, ie. 99% risk for exposure longer than 10days	Most species disappear
	1800	1215	8		
	4400	2972	9		
	7000	4730	10	Upper limit of 99% risk (upper 95% CL for LC_{99})	No survivors

* Note that TDS was calculated by the authors by multiplying the electrical conductivity of the Na_2SO_4 solution by 6.5.
CL = confidence level, LC = lethal concentration

Table 4.2 Transformation matrix used to convert TDS concentration to associated stress levels for aquatic invertebrates in the Olifants River (see Table 4.1).

TDS (mg/litre)	Stress level	TDS (mg/litre)	Stress level
120	0.10	890	5.0
150	0.50	900	5.5
195	1.0	1000	6.0
220	1.5	1200	6.5
295	2.0	1400	7.0
450	2.5	1600	7.5
520	3.0	1800	8.0
600	3.5	3000	8.5
780	4.0	4400	9.0
800	4.5	7000	10.0

The salinity-induced stress duration curves for IFR 13 are shown in Figure 4.5 b. Similar results were obtained to the salinity profiles. Consequently, it can be seen that ranking of the stress indices yields similar results to those obtained from considering salinity.

4.4.1.2 *Fluoride at Mamba (IFR site 15)*

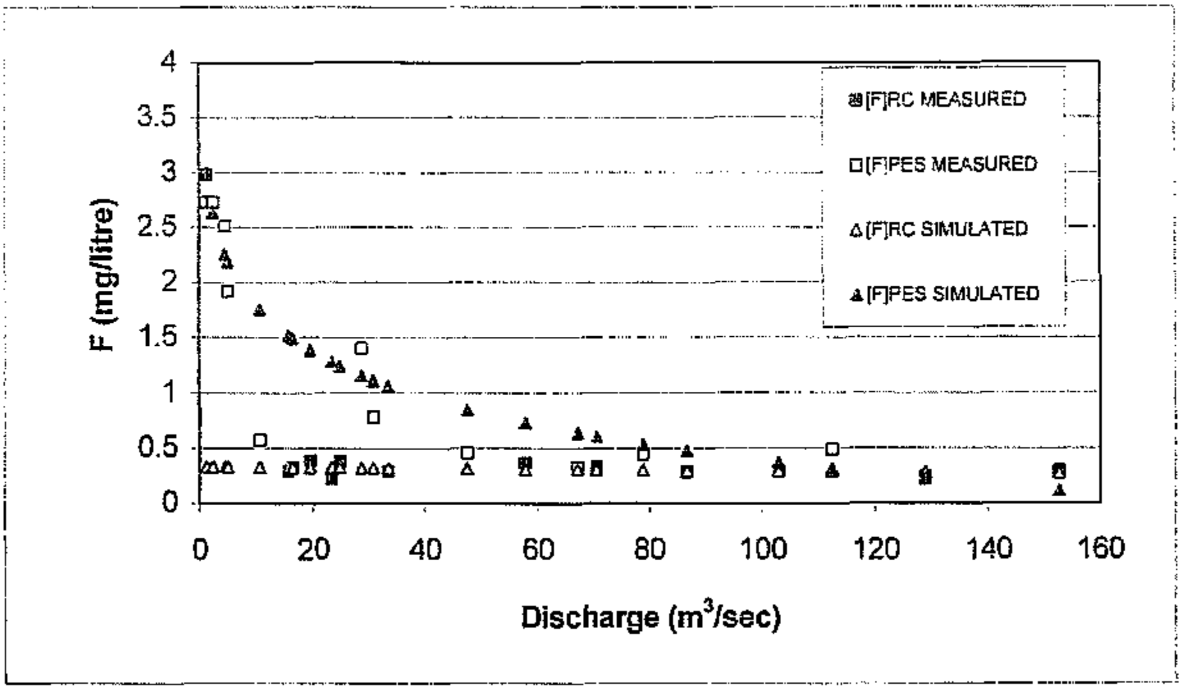
Fluoride pollution, originating from the mining complex at Phalaborwa, was found to be a key water quality issue at IFR site 15, as well as downstream in the Kruger National Park (Palmer and Rossouw 2000. This constituent was therefore modelled in order to compare fluoride profiles under natural, present day and under the recommended IFR regime, which was designed to achieve a B category river with regard to the water quantity Reserve. These results were not included in the Scenario workshop, which only considered salinity and flow scenarios generated from the yield model. In addition, stress profiles were not generated, as insufficient data were available to link fluoride concentration and biotic response.

Figure 4.6 shows the Q-C plot for fluoride at IFR 15. The fluoride duration curves generated for natural, present-day, as well as for the flow regime recommended by the specialists to attain a B category river (with regard to water quantity) are shown. According to the South African water quality guidelines for aquatic ecosystems (DWAF 1996) the upper limit of the target water quality range (TWQR) for fluoride is 0.75 mg/litre. The same work also gives the CEV (chronic effect value) as 1.5 mg/litre fluoride and the AEV (acute effect value) as 2.54 mg/litre fluoride. The South African water quality guidelines define the CEV as the concentration at which there is expected to be a significant probability of measurable chronic effects in up to 5% of the species in the aquatic community. The AEV on the other hand is defined as that concentration of a constituent above which statistically significant acute toxic effects are expected to occur.

From Figure 4.6 it can be seen that under natural conditions, the instream fluoride concentration would always have been well within the TWQR. Under the present day levels of pollution and accompanying reduced levels of discharge, however, fluoride concentration is equal to, or above, the CEV approximately 68% of the time, and above the AEC 34% of the time. Figure 4.6 also shows that under the recommended flow regime, although the peak values of fluoride will no longer occur, neither will the very low values. Thus without mitigation of pollution the water quality Reserve for toxic substances is not likely to be attained.

4.5 Application to the Breede River Basin study

Time-series modelling of water quality was not part of the Terms of Reference for the Breede River Basin study. In addition, of the six designated IFR sites, only two were considered to be suitable for this type of modelling. Therefore only very limited time-series modelling was undertaken in this catchment. Nevertheless, useful insight was gained in considering the suitability of various sites as well as the data that were required. These aspects are discussed below.



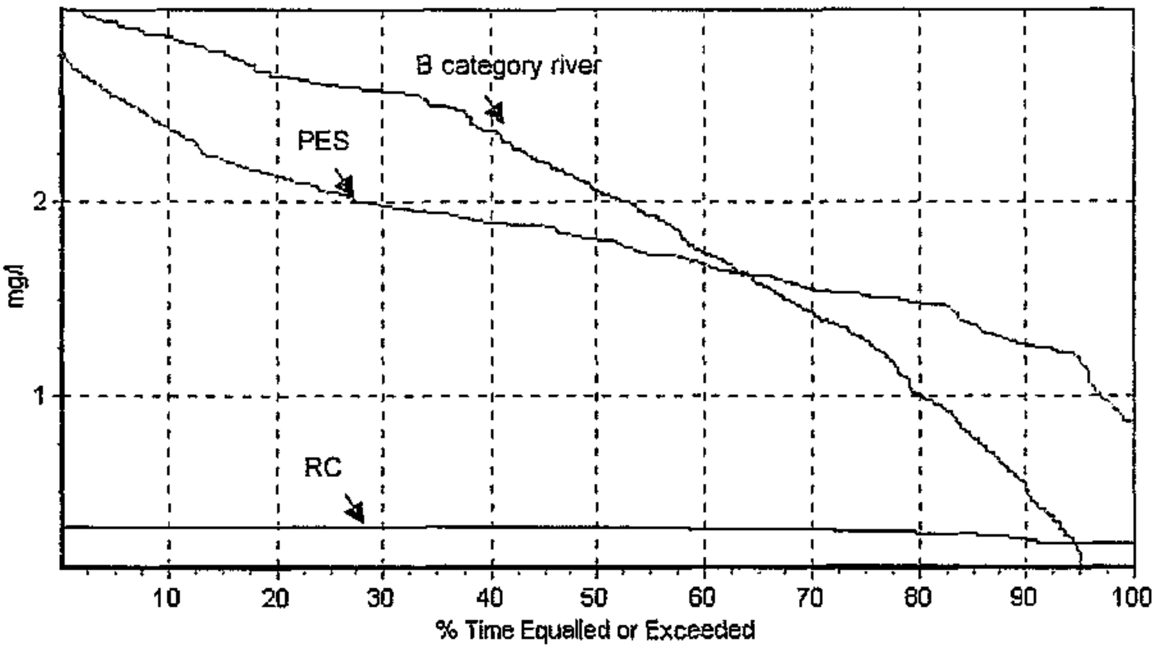
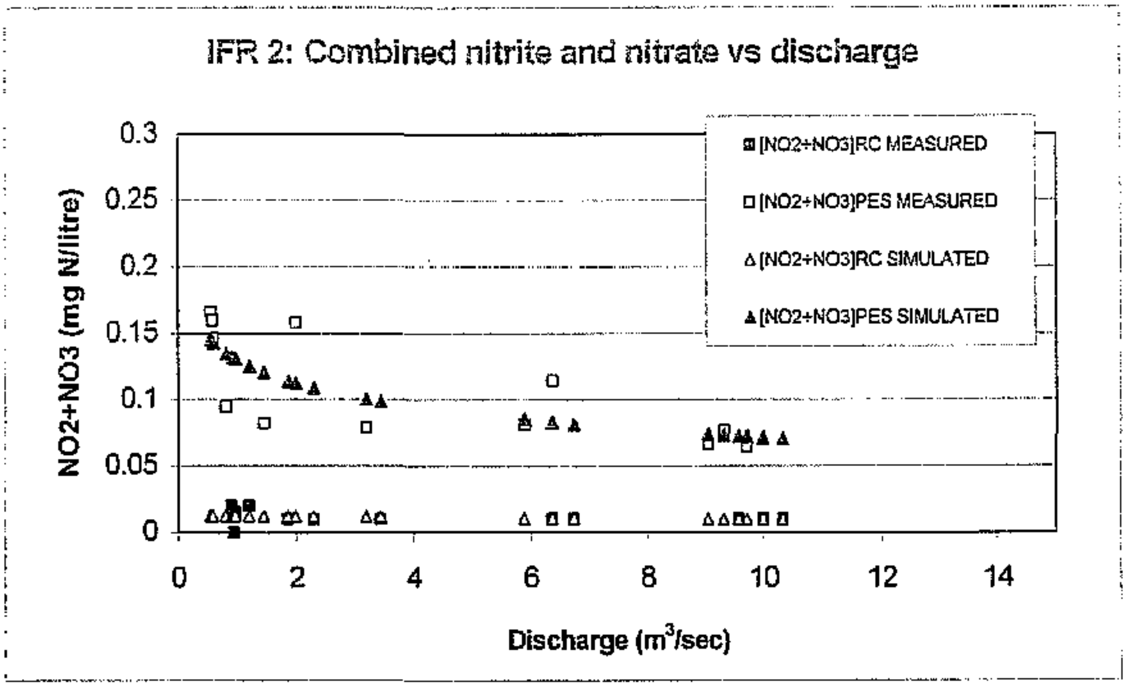


***Figure 4.6 (above)** Q-C plot for fluoride at IFR 15 on the Olifants River. Note the elevated concentration at low discharges in the present state (PES) compared to the Reference Condition (RC). **(below)** Fluoride duration curves for RC, present state and under the recommended flow regime for a "B" discharge category river.*

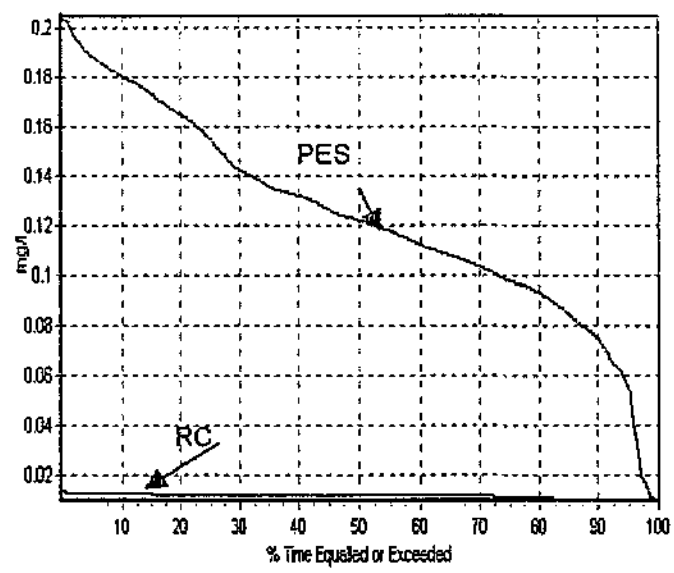
The results from Q-C modelling for the Breede River Basin study are given in Appendix B. The following points were noted when assessing the feasibility of time-series modelling for this system:

- Two of the sites, namely IFR 1 and IFR 3 showed wash-off of TDS during the onset of winter rains. In section 2.11.1.1 the validity of using such Q-C plots to make predictions of water quality was considered, and it was concluded that this method should not be used in such cases. Thus, although time-series modelling is probably valid for Reference condition and present day conditions it should not be used to predict the water quality implications for a future discharge regime.
- Another two sites, namely IFR 2 on the Molenaars River and IFR 6 on the Baviaans River (part of the Riversonderend system), were relatively unimpacted with regard to TDS (although some nutrient enrichment was apparent at IFR 2, see this section, further on). Current TDS levels were largely the same as the Reference condition levels. In addition, the Q-C plots show very little variation in TDS concentration with discharge. As a result, all flow scenarios result in very similar salinity duration curves, and are all within the limits required to attain the water quality Reserve. Time-series modelling in such cases does not yield very useful results.
- Q-C plots for IFR site 5 (Riversonderend) could not be modelled accurately because no water quality monitoring station was close by. Attempts to model the water quality at this site using data from the either the closest upstream and or the downstream station, or the mean value of the two, were not successful.
- IFR site 4 was suitable for Q-C and time-series modelling of TDS, since this variable showed a marked decrease in concentration with increased discharge. TDS duration curves were generated for the Reference condition and Present ecological state, however the discharge time-series for the flow regime recommended at the specialist workshop was not available at the time of writing this report (Louw, D. IWR, Rhodes University *pers. comm.*).
- There was slightly increased nutrient loading at IFR 2 resulting from the discharge of effluent from a trout farm into the Molenaars River (Rossouw and Kamish 2001). A reasonably good correlation between measured combined nitrate and nitrite concentration and simulated values was obtained (correlation coefficient = 0.75). Concentration time-series modelling was carried out, and the results are presented in Figure 4.7. The Q-C plot shows that levels of combined nitrate and nitrite are very



a)

b)



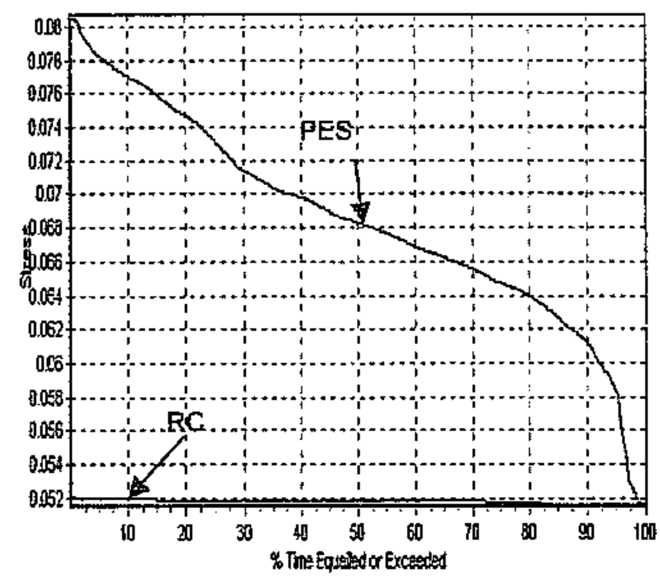


*Figure 4.7 **(Above)** Q-C plot for combined NO_3 and NO_2 at IFR 2 (Molenaars River). Note the very low concentrations of these constituents in the natural state. **(Below)** Duration curves for concentration (a) and stress (b) for combined NO_3 and NO_2 in the Reference Condition (RC) and the present state (PES). The curve for the RC (both concentration and stress) is just above the X-axis.*

low at this site in the natural state, and are largely independent of discharge. As a result, the Reference condition line on both the concentration and stress duration curves is just above the X-axis. The Q-C plot also shows that the Present ecological state concentration of combined nitrate and nitrite is elevated compared to Reference condition and decreases with increasing discharge, and is thus typical of a point source of pollution. Although a tentative nitrogen-induced stress duration curve is shown in Fig. 4.7, attempts to convert time-series of concentration to stress were hampered by a lack of quantitative data linking concentrations of combined nitrates and nitrites to biotic response. The most significant effect of nutrient enrichment on water resources is the change in ecological state (e.g. from oligotrophic to mesotrophic) and community-level processes rather than a toxic effect on the biota *per se* (DWAF 1996). Considerably more research is required into the effects of eutrophication on macroinvertebrate communities in different ecoregions before a confident prediction could be made as to the stress exerted by different levels of nitrates and nitrites (and other nutrients) on an aquatic organism.

4.6 Ecological interpretation of the results

In the OREWRA study the situation was frequently encountered such as that depicted in Figure 4.5 a) and b). Namely, that under the recommended IFR flow scenario whilst occasions of very high concentrations of salinity (or another constituent) would be avoided, periods of very good quality water would also be removed. This is a consequence of the fact that it is normally during the period of high discharge that water would be harvested for other users. The overall result is an attenuation of the current salinity profile so that the extremes of very high and very low concentration would no longer occur. Whilst the removal of episodes of poor water quality is obviously advantageous, the effect of removing periods of "fresh" water is not clear. It is thought that such pulses of low salinity water may act as environmental cues, perhaps initiating spawning in fish, or other ecologically important activities. This matter requires further research and should form part of a detailed study of the effects of implementation of the IFR for key rivers.

More attention should also be paid to the relationship between water quality and biotic response. Although stress profiles were developed they were not particularly useful in these studies because the response between concentration and stress was roughly linear. Therefore Q-TDS and TDS-Stress duration curves were very similar and the same ranking of flow scenarios was obtained with both. It is conceivable however that other water variables would have very different concentration-stress relationships. For example combined nitrate-nitrite at very low concentrations may well be more stressful (due to lack of nutrients) than at slightly higher concentrations (increased nutrient availability). As concentration of nitrogen is increased further bringing about a change in trophic status of the water body, biotic stress may well increase again. The nature of this response would be very dependent on the biotic community under consideration as well as the natural water quality conditions of that site. Transformation of concentration profiles to stress profiles using such matrices may yield interesting results. Until more basic research is carried out into the effect of water quality on the abundance, composition and functioning of key elements of the biota however, reliable concentration-stress matrices for nutrients cannot be derived.

4.7 Conclusion

Time-series modelling can only be used at sites where there is a strong relationship between discharge and concentration. It is a useful tool for comparing the water quality that will result from different discharge time-series and for ranking such scenarios. Similarly, stress time-series can also be potentially useful in ranking the implications of different scenarios for the aquatic biota. It should be remembered however that only one water quality variable at a time can be modelled using this technique. In real rivers, aquatic biota are exposed to multiple co-stressors. Attention still needs to be given to the integration of different water quality variables to obtain the actual stress that would be experienced by the biota. Due to the additive, synergistic and antagonistic effects between individual chemical constituents and with the physical attributes of water bodies, this is not an easy task. This topic is considered further in Chapter 6.

CHAPTER 5

APPLICATION OF QUAL2E TO ENVIRONMENTAL FLOW ASSESSMENTS

5.1 Introduction

In Chapter 1 it was suggested that rather than a single model, a hierarchy of water quality models should be used as part of the assessment of environmental flows. The choice of modelling method would depend on the particular circumstances, or factors pertinent to a given Reserve determination, the most important factors being, availability and completeness of discharge and water quality data; the time within which the modelling exercise should be completed; and financial constraints. This chapter investigates an application of a second-tier model, namely QUAL2E, to the environmental flow determination process. It examines where and how the modelling method might fit into the current Reserve methodology and whether it would be useful in this regard.

The method was examined by means of an application of the model to the lower reaches of the Olifants River, Mpumalanga. The QUAL2E modelling exercise was carried out after the environmental flow (IFR) workshops for the Olifants River Environmental Flow Requirement Assessment (OREWRA) and so the results (unlike those from Q-C modelling) were not used to investigate the consequences of the recommended water quantity Reserve. The specific objectives of this application of QUAL2E were as follows:

- to examine the type of output (results) that could be obtained using this modelling method and to explore their usefulness in the context of Environmental Flow Assessments
- to obtain an idea of the time, level of expertise and data required in setting up QUAL2E

- to investigate problems or difficulties likely to be encountered in applying QUAL2E to South African rivers
- to compare Q-C modelling and QUAL2E in respect to the objectives given above
- to predict the concentrations of TDS (and phosphate) that would occur under the flow regimes recommended by the specialists at the IFR Workshop and compare those with the results obtained from discharge-concentration modelling
- to investigate, using QUAL2E, the measures needed (for example control of point sources) to bring the concentrations of the water quality variables considered within the proposed Resource Quality Objectives for the river in the Kruger National Park

5.2 Application of QUAL2E

5.2.1 Background to the model

The following description of QUAL2E is taken largely from the United States Environmental Protection Agency (US EPA) web site. The address is given at the end of this document in the section "Useful web site addresses". It is a public-domain model (supported by the US EPA) and can thus be downloaded from the Internet at no cost.

QUAL2E (Enhanced stream water quality model) is an instream, riverine, water quality model that simulates the major reactions of nutrient cycles, algal production, benthic and carbonaceous demand, atmospheric re-aeration and their effects on the dissolved oxygen balance. It can predict up to 15 water quality constituent concentrations (including up to three conservative constituents) and has been developed as a planning tool for estimating TMDLs (total maximum daily loads) in the USA. It can be semi-dynamic if required, in that some water quality variables, such as DO and algal growth, can be allowed to vary diurnally. Discharge (such as from headwaters or point source loads) is presumed to remain steady with time, however. The model can be used to study the impact of waste loads on instream water quality, by predicting the instream concentration for each river reach for a given instant in time. It can also be used to identify the magnitude and quality characteristics of non-point waste loads as part of a field-sampling program. The main attributes and characteristics of QUAL2E are summarised in Table 5.1. QUAL2E-UNCAS is a refinement of the original model that has

Table 5.1 *Major characteristics and attributes of the instream water quality model QUAL2E (taken mainly from Wurbs 1995).*

NAME	AUTHOR/ DEVELOPER	CHARACTERISTICS AND APPLICATIONS	ADVANTAGES	DISADVANTAGES	AVAILABILITY	COMMENTS
QUAL2E and QUAL2E-UNCAS	US EPA, CEAM	Well mixed streams. Up to 15 constituents can be modelled. 1D, deterministic, steady-state, with some semi-dynamic features. QUAL2E-UNCAS takes into account the stochasticity of natural systems. **Variables** = TDS, N, P, DO, *E.Coli*, BOD, Chl, conservative constituents, and temperature.	Well known and used widely in the world. Reasonably user friendly. Good for identifying non-point loads.	Relatively simple but needs expert input. Requires values for many parameters if modelling non-conservative constituents. No catchment run-off module included. Steady state, therefore needs to be re-run for low-, medium- and high-flows.	Public domain. Expertise available locally	Can be used to calculate flow required to yield a pre-determined DO level (not available for other constituents). Has been used to model eutrophication on Vaal River Rossouw and Quibell (1993) and water quality in the Kuils River (Ninham Shand 1999).

US EPA = United States Environmental Protection Agency. CEAM = Centre for Exposure Assessment Modelling.

been developed to take into account the stochasticity of natural systems. It was not used in this project because it was felt that the simpler version of the model should first be examined.

5.2.2 <u>Description of the study area</u>

The portion of the river that was modelled, significant hydrological features, and the position of IFR sites and DWAF gauging and water quality monitoring stations are shown schematically in Fig. 5.1. The Olifants River (Mpumalanga), and this section of river in particular, was chosen for application of QUAL2E (version 3.12, February 1995), for the following reasons.

- A Comprehensive Reserve Determination had been carried out for the Olifants system (Palmer and Rossouw 2000) in which one of the authors (HLM) was involved. Thus, not only was the system familiar, but specialists in other fields (for example hydraulics) were available for confirming the values of parameters required in the model.
- It offered an opportunity for a comparison of the results derived from the Discharge-Concentration (Q-C) modelling method and QUAL2E, using the same stretch of river.
- Mamba is a key IFR site since the Selati River flows into the Olifants River in this region and carries water of extremely poor water quality. This has a significant effect on the quality of water flowing into the Kruger National Park, an ecologically sensitive area.
- Because this section of river includes the confluence with the Selati River, the effects on water quality of differing water quality management scenarios (i.e. varying the pollutant loading between the two headwaters) could be examined.
- The availability of data (both discharge and water quality) for this stretch of river is considered to be reasonably good compared to some other parts of the country.

5.2.3 <u>Implementation of the model</u>

5.2.3.1 Conceptualization of the system

The system was divided into headwaters, reaches and junctions. Headwaters represent an upstream "starting point" to the system and a discharge value must be specified for each one. From Figure 5.1 it can be seen that the two headwaters for the section of the Olifants system that was modelled, were the Selati River and the mainstem river.

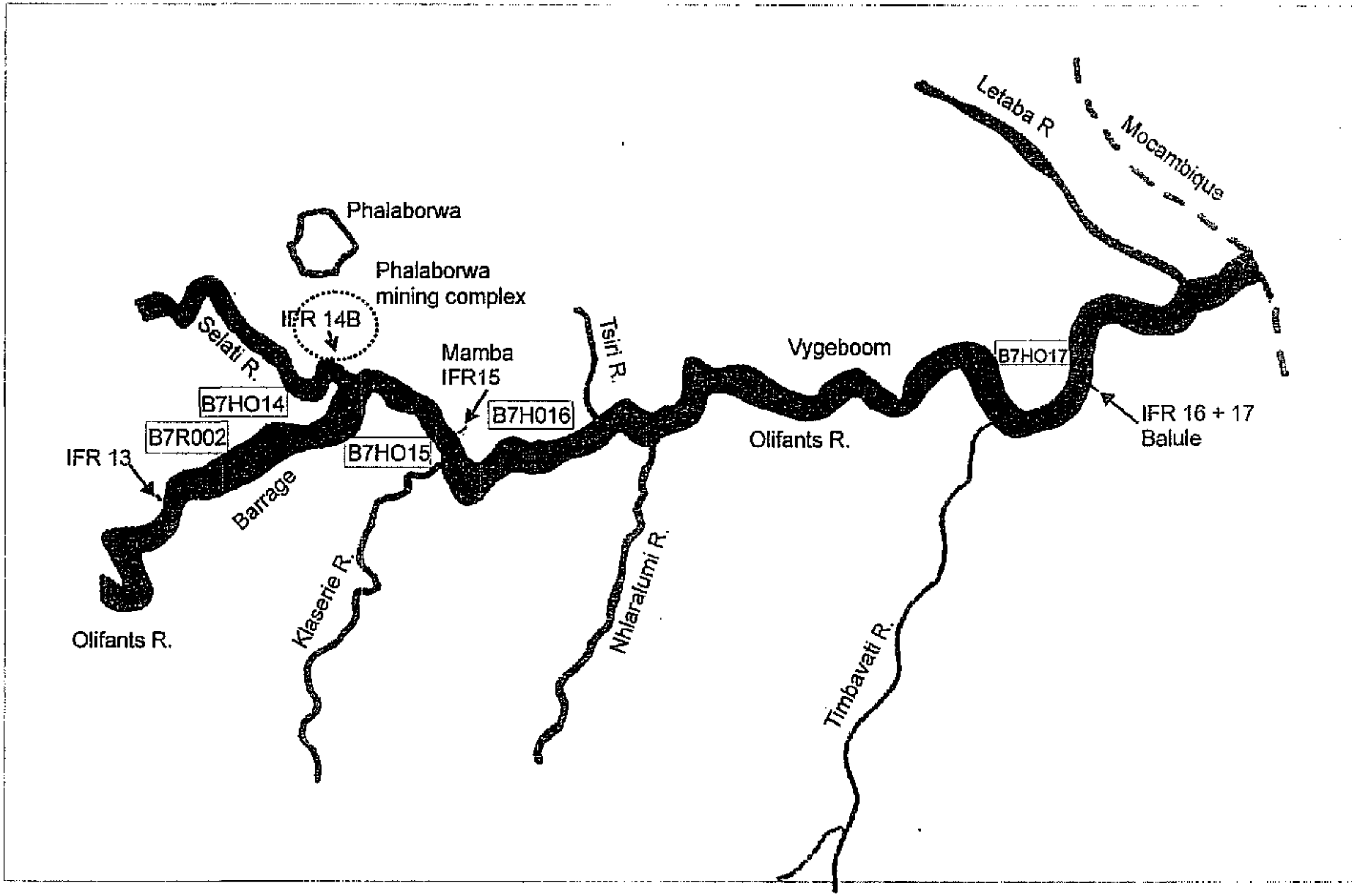


Figure 5.1 Schematic representation of the reaches of the Olifants and Selati Rivers that were modelled in this application of QUAL2E. Significant hydrological features (tributaries and impoundments), as well as the position of IFR sites, gauging and water quality monitoring stations are shown. The length of the Olifants River shown in the figure is approximately 100 km.

The outflow from the Phalaborwa barrage was used as the headwaters for the Olifants River. In the case of the Selati River, the other headwater, it has been reported that most of the pollutant loads that enter the system are discharged just above both the confluence with the Olifants River, or above the monitoring site B7H019 (N. Rossouw, Ninham Shand, Cape Town, *pers. comm.*). Thus it was considered unnecessary to consider reaches further upstream.

Reaches are lengths of river along which it is assumed that hydraulic characteristics are constant. Reaches were delineated by examining 1:50 000 topographical maps of the area. Hydrological features such as a confluence (junction) with a major tributary (e.g. Selati River), or the presence of a monitoring site were used to delineate the boundaries of the reaches. In addition, changes in gradient (identified from contour lines on the maps) were also used. QUAL2E requires that each reach be sub-divided into computational elements within which water quality is assumed to be homogenous (Brown and Barnwell 1987). Although the length of a reach can vary, the length of the computational elements is the same throughout the system and an arbitrary value of 2 km was chosen. The presence of any point sources and abstraction points must also be entered into the model (although the latter was not relevant in this stretch of the river). All minor tributaries (namely the Klaserie, Tsiri, Timbavati and Nhralalumi rivers) were treated as point sources. Figure 5.2 shows the arrangement of the computational elements for the portion of the system considered. Each reach is numbered, and it can be seen from the figure in which reach and in which computational element each tributary joins the system.

5.2.3.2 The data required

In order to set up QUAL2E, climatological, hydraulic, discharge and water quality data are required. These were obtained as described below.

Some climatological data (wet and dry bulb temperature, wind speed and barometric pressure) were obtained from a statistical summary of climatic data for South Africa (Weather Bureau 1986), and other data were requested directly from the South African Weather Bureau (see 'Useful web sites').

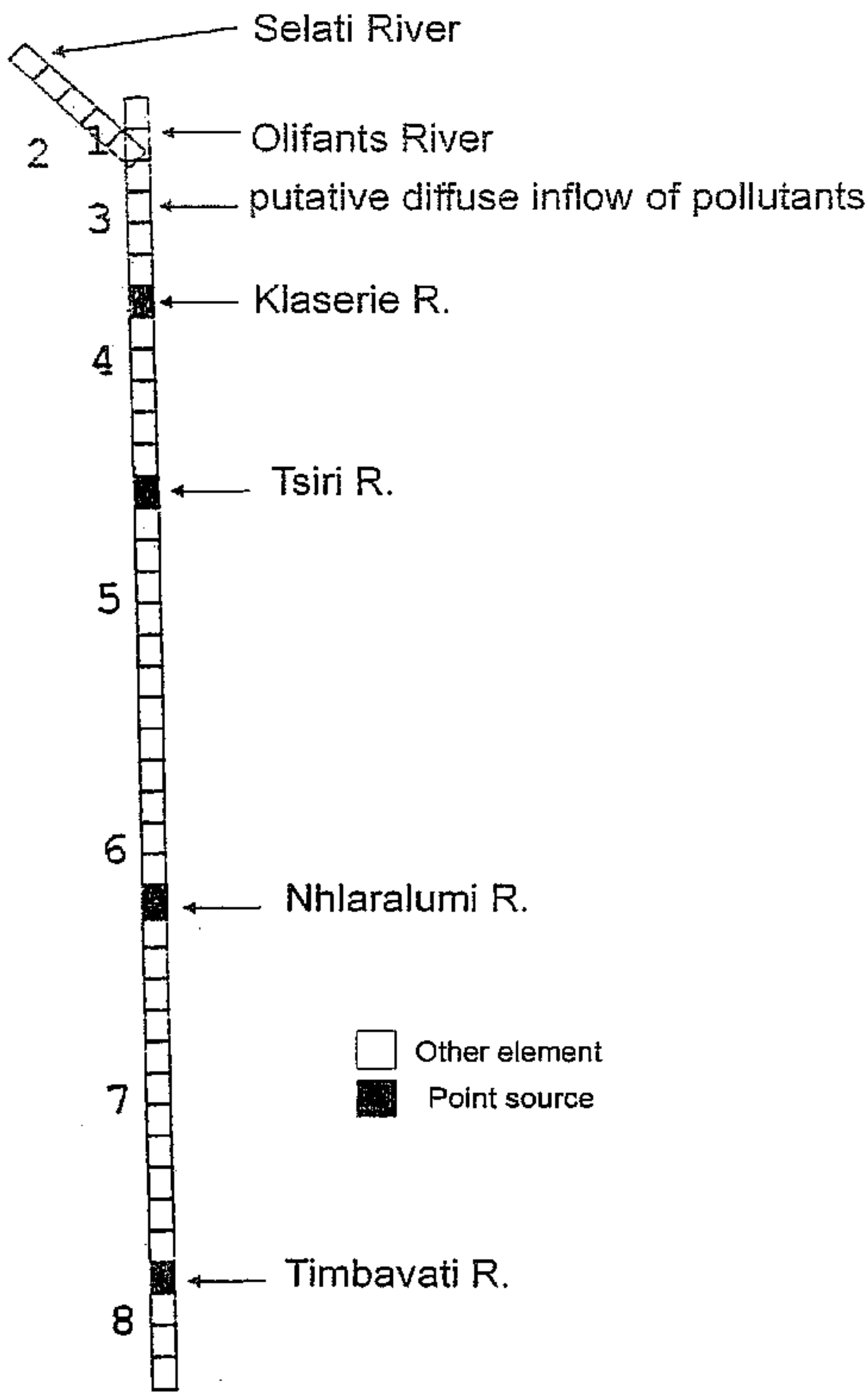


Figure 5.2 *Division of the modelled reaches of the Olifants and Selati Rivers into computational elements (each 2 km long). Reaches were comprised of differing numbers of computational elements as described in the text. Reach numbers and names of tributaries (indicated as point sources) are shown.*

Hydraulic data, in the form of the gradient of river reaches, were obtained from 1:50 000 topographical maps. An estimation of the slope of the channel sides, as well as the channel width, was made from an examination of the channel cross-sections provided for the IFR workshop. Values of Manning's roughness coefficient (*n*) for this section of the river were obtained from the hydraulic engineer involved in the IFR workshop (A. Birkhead, Streamflow Solutions, Johannesburg, *pers.com.*).

Water quality data (i.e. the concentrations of the chemical constituents under consideration) were obtained either by contacting DWAF directly, or from the "Water Quality on Disk" CD (web sites listed at the end of this report). Discharge data were also obtained from DWAF. In addition, data concerned with flow from Phalaborwa barrage were obtained from Lepelle Northern Water Board (E. Coetzee, Lepelle Northern Water Board, Phalaborwa, *pers. com.* 2001).

5.2.3.3 Parameter values

The model requires the values of several parameters (e.g. dust attenuation factor, dispersion coefficient) as input. To a large extent these parameters are not required when simulating conservative constituents, but were needed for simulation of phosphate (and, although not applicable to this modelling exercise, for simulation of temperature, nutrients, chlorophyll concentration or DO). When it was necessary to specify a value, these were chosen from one of the following sources:

- from an examination of the default values in the model,
- from values used for other applications of QUAL2E to South African rivers (e.g. the Kuils River, Ninham Shand 1993; and the Vaal River, Rossouw and Quibell 1993),
- from Bowie, Mills, Porcella *et al.* (1985).

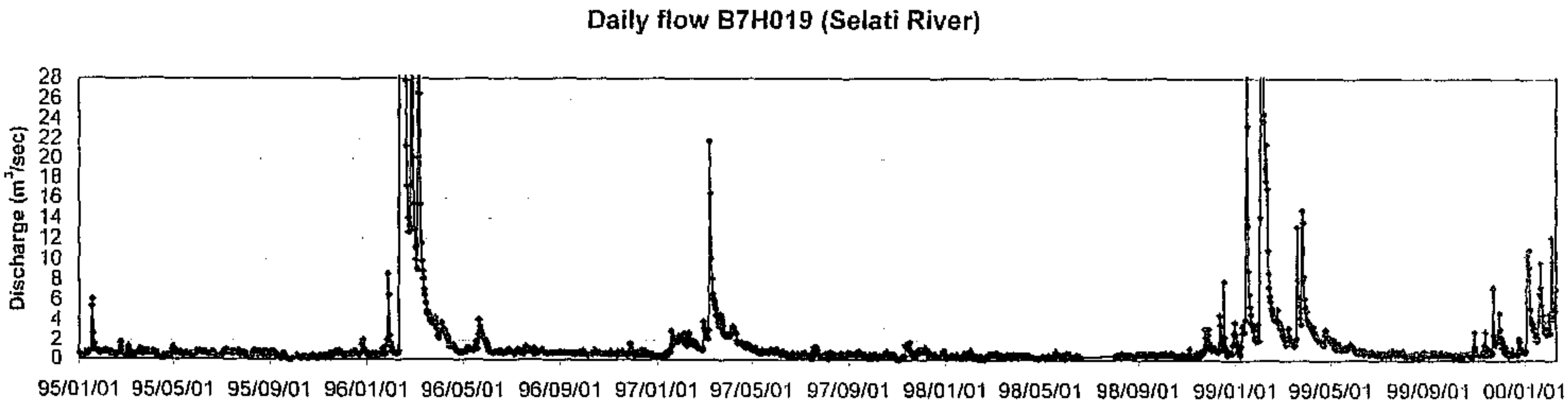
5.2.3.4 The water quality variables simulated

The system was first calibrated and then validated for TDS, followed by the other conservative constituents, chloride and sulphate. A non-conservative variable, namely phosphate was then modelled. TDS was simulated because it is an important system variable and also because it had been used for time-series analysis (Chapter 4). Sulphate was simulated because it is a major contaminant in this section of the river. Chloride was modelled as an additional conservative variable in order to compare results with those obtained for TDS. Simulations of phosphate were carried out with a view to examining the application of QUAL2E to a non-conservative constituent.

5.2.3.5 The management scenarios considered

QUAL2E is, at least with regard to the variables investigated in this application, a steady-state model and thus during each simulation, discharge was assumed to remain constant. The model was therefore calibrated using winter discharge values, because an examination of the time-series for two key gauging stations in the system showed that discharge was relatively constant during the winter months (Fig. 5.3). The second reason for using this portion of the hydrological year was that since discharge is at a minimum, this is when water quality is likely to be the worst (see the discharge-concentration plots for IFR site 15, Mamba, Figure 5.9). The model was calibrated using discharge and water quality data from winter 1999 and was validated using an independent set of data from 1998. Input discharge values were taken as the monthly mean flow for June, July and August of the appropriate year. Input concentrations of the water quality variables were taken as the median of all the samples for June, July and August 1999 (i.e. one value each for TDS, sulphate, chloride and phosphate). The model was validated using water quality data (median values) for the winter months of 1998.

The major aim of this application was to investigate the usefulness of QUAL2E within an assessment of the ecological Reserve. One of the most important products that arises from the IFR determination, from the point of view of water quality modelling, is the recommended flow regime. For Mamba, the recommended maintenance baseflow as specified by the specialists in the BBM workshop, for the winter months of June, July and August was 9.4, 8.2 and 7.4 m^3/sec respectively. This represents an increase in discharge compared to the mean present day discharge for the same months. The junction with the Selati River is just upstream of this site, and thus theoretically the recommended increased discharge at Mamba could be achieved by increasing outflow from either source. The effects of variation in discharge on water quality were initially simulated by assuming the same input (both volume and concentration) via the tributaries and point sources as during calibration. The amount of water (but not the quality) that was released from the Phalaborwa barrage was then increased in the model (to bring about the required increased discharge at Mamba). The gauging station at Mamba is on the Olifants River near to the western boundary of the Kruger National



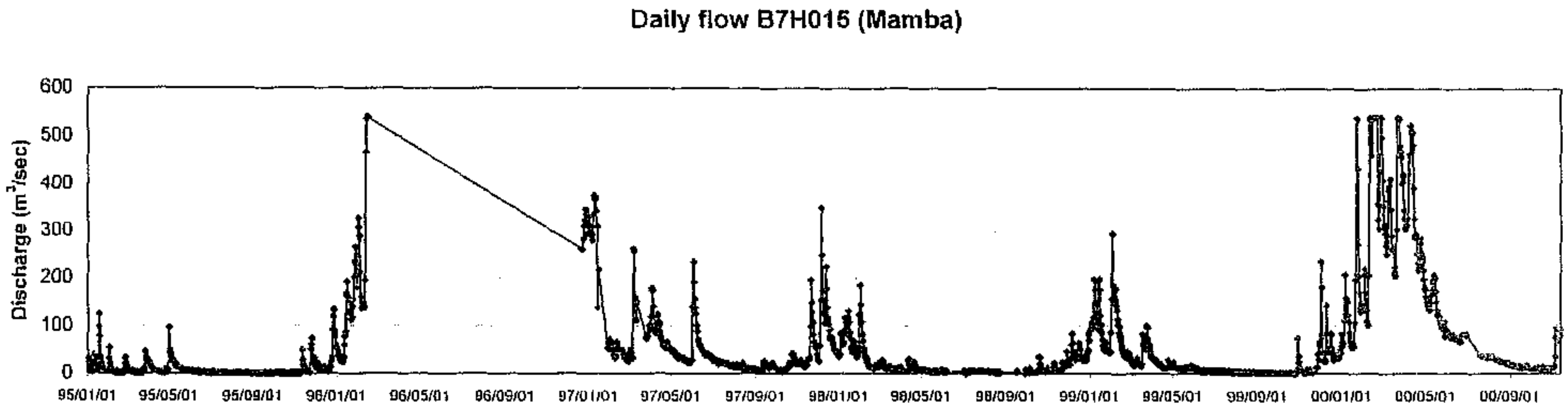


Figure 5.3 *Hydrological time-series for two gauging weirs on the lower Olifants River. Note the low, steady-state discharge during the winter months.*

Park. Tributaries entering the river downstream of this point are considered to be largely natural in water quality (N. Rossouw, Ninham Shand. Cape Town, *pers. comm.*). This is reflected in the fact that water quality for Balule, which is the most downstream point that was modelled (Fig 5.1), is considerably improved compared to Mamba. Thus it was considered that if discharge and pollutant loads were manipulated so that water quality was acceptable at Mamba, then it would follow that at Balule it would be at least the same, if not better.

The following scenarios were considered:

- **Scenario 1:** Total discharge at Mamba equal to the minimum winter *maintenance* baseflow as recommended at the IFR workshop (assuming the same proportional input, in terms of volume from the tributaries, and point-sources as during calibration). In addition, it was assumed that the input concentration of the chemical constituents under consideration (i.e. the concentration at the headwaters) was the same as during calibration.
- **Scenario 2:** Total discharge at Mamba equal to the minimum winter *drought* flow as recommended at the IFR workshop (assuming the same input, both in volume and concentration from the tributaries and point-sources as during calibration).
- **Scenario 3:** Amelioration of pollutant loading in order to investigate the effect of varying the proportion of discharge (and the concentration of chemical constituents) originating from different sources.

5.3 Results

5.3.1 Simulation of discharge

The observed values (for 1999) used to calibrate discharge are given in Table 5.2. This table also serves to illustrate the availability of relevant data. There was a discrepancy between the discharge recorded at Mamba and the sum of the recorded discharges from the Phalaborwa barrage and the Selati River, the flows recorded at Mamba being slightly higher (by approximately 0.2 m^3/sec) than the calculated discharge. It was assumed that this was due to diffuse flow into the Olifants river, just upstream of Mamba, since in addition the concentration of TDS was also higher at this site than would otherwise be expected (section 5.3.2).

For the lower reaches of the river, including the gauging weir at Balule, no discharge data were available. The inflow from the minor tributaries (namely the Klaserie, Tsiri, Timbavati and Nhralalumi Rivers) was also largely unknown. There was considerable dilution of TDS and other conservative constituents between Mamba and Balule, however. The median TDS concentration for winter 1999 (June, July and August) was 455 mg/litre, whilst for the same period at Balule it was 420 mg/litre. Downstream dilution in concentration was also recorded for chloride and sulphate. This reduction in the concentration of conservative constituents downstream of Mamba indicated that the contribution to total discharge in the lower Olifants River from these tributaries, even during the dry season, could not be ignored. An estimate of this volume was made by using QUAL2E to calculate the discharge required to obtain the observed concentrations of TDS, Cl^- and SO_4^{-2} at Balule (water quality data available for 1999 only). This was found to be approximately 1.2 m^3/sec. Observed and simulated discharge for 1999 (as well as 1998, see below) are presented in Fig. 5.4. The figure shows a graph of distance upstream from the lowest point of the system (i.e. Balule) versus discharge at that site. The two available measured data points (the outflow from Phalaborwa barrage and Mamba) are shown. The point at which inflow from the tributaries enters the mainstream river (e.g. Selati River at 80 km upstream from the lower boundary) is also indicated.

Table 5.2 *Observed discharge values used for* ***calibration*** *of the model. Mean taken of June, July and August values for 1999. (KNP = Kruger National Park).*

Reach number	Site name	Gauging station	Mean monthly discharge (m^3/sec)	Comments
1	Barrage outflow		4.4	Data from Lepelle Water
2	Selati River	B7H019	0.69	
3	Mamba	B7H015	5.2	
6	Tsiri	B7H016	No data	
8	Balule/KNP Hiking trail	B7H017/ B7H018	No data	
Tributary	Timbavati	B7H020		No data for '99. Some records for '94, 96, 2000

Attempts were then made to validate the model using discharge data from winter 1998, and using the same values for diffuse flow at Mamba as well as for the input from the lower tributaries (Fig. 5.4). The agreement between observed and predicted discharge values was not as close as that obtained during calibration. This is shown by the fact that the predicted discharge at Mamba was slightly lower than the mean measured value of 4.19 m^3/sec.

Table 5.3 *Observed discharge values used for validation of the model. Mean taken of June, July and August values for 1998. (KNP = Kruger National Park).*

Reach number	Site name	Gauging station	Mean monthly discharge (m^3/sec)	Comments
1	Barrage outflow		3.1	Data from Lepelle Water
2	Selati River	B7H019	0.46	
3	Mamba	B7H015	4.19	
6	Tsiri	B7H016	No data	
8	Balule/KNP Hiking trail	B7H017/ B7H018	No data	
Tributary	Timbavati	B7H020		No data for '99. Some records for '94, 96, 2000

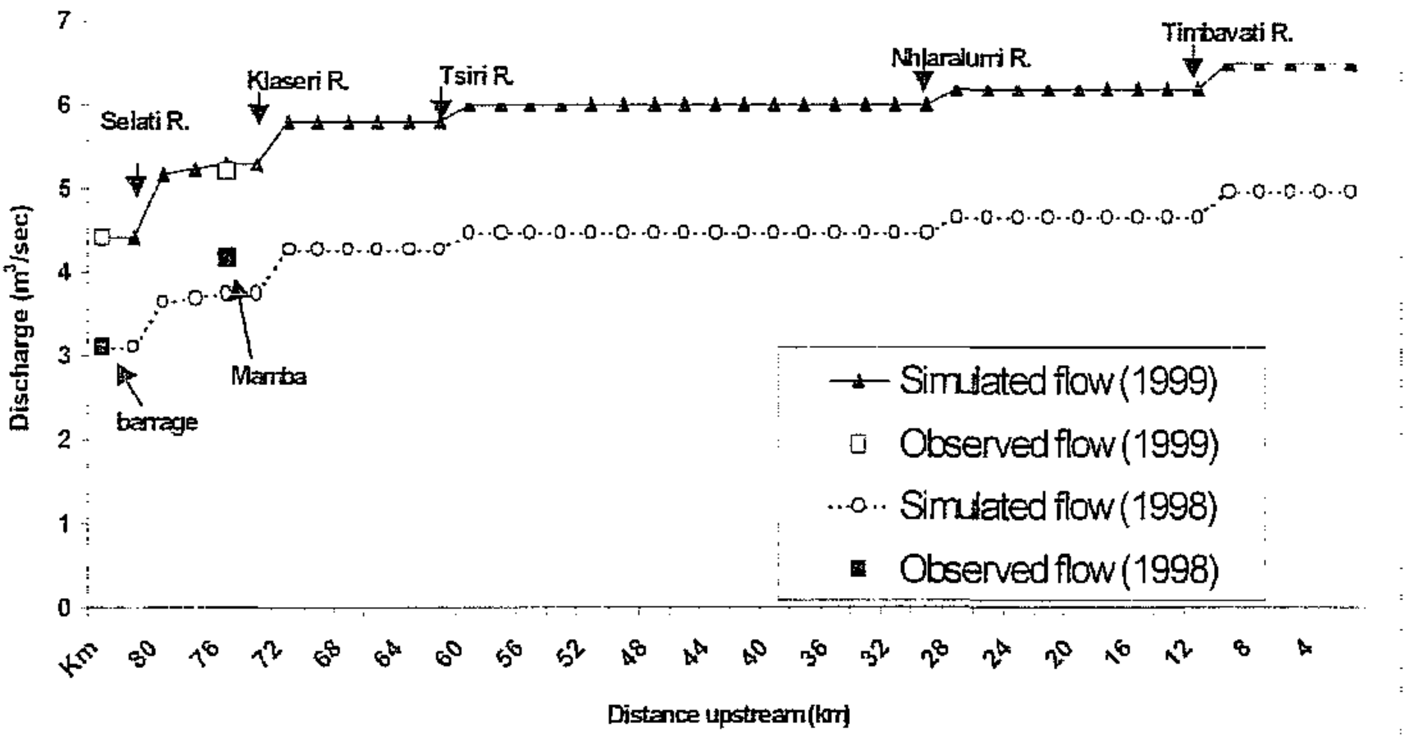


***Figure 5.4** Observed and simulated discharge along the length of the lower Olifants River as predicted by QUAL2E. Discharge values for the period of winter 1999 and 1998. The point at which each tributary enters the system is indicated.*

5.3.2 Simulation of conservative constituents

The availability of water quality data, as well as the median measured concentrations for winter 1999 is indicated in Table 5.4. No data for the barrage outflow were available and therefore input concentrations were estimated from those of Oxford on the Olifants River (B7H007Q01) the nearest upstream monitoring site. It was mentioned in the previous section that, in order to account for the discrepancy between measured discharge at Mamba and the sum of the two headwaters, diffuse flow was considered to enter the system just below the junction with the Selati River and above Mamba weir. Thus the model was set up so that a volume of water (termed "incremental flow" in QUAL2E) was allowed to enter the system at reach 3 (Figure 5.2). The concentrations of TDS, Cl^- and SO_4^{-2} in this inflow were adjusted empirically, since no water chemistry data were available.

Figure 5.5 shows the change in concentration of TDS, Cl^- and SO_4^{-2} along the length of the river for the winter low-flow period 1999. The accuracy of the simulations was evaluated by comparing simulated values with measured values (at Mamba and Balule). The figure shows that a fairly good correspondence was obtained between the observed and simulated values for all three conservative constituents.

The model was then re-run using discharge and concentration input values appropriate to 1998 (Tables 5.3 and 5.5 respectively). The same discharge values for diffuse flow at just above Mamba and inflow from the lower tributaries were used as for 1999. In addition, the same concentrations were also used for these two additional inflows. The results are shown in Fig. 5.6. Unfortunately no water quality data were available for Balule for that year. This made it very difficult to assess the accuracy of the model since predicted values could only be compared to the observed data at Mamba. The simulations underestimated the concentration of all three conservative constituents, especially TDS (predicted value 653 mg/litre, median observed value 898 mg/litre) and SO_4^{-2} (predicted value 176 mg/litre, median observed value 285 mg/litre).

Table 5.4 *Observed values of water quality variables used for calibration of the model. Median taken of June, July and August values for 1999.*

WQ variable	Reach number	Site name	Monitoring station	Monthly conc (mg/litre)	Range	Sample size
TDS Chloride Sulphate ortho-phosphate Total phosphate	1	Barrage outflow	B7R002	No data		
TDS Chloride Sulphate ortho-phosphate Total phosphate	2	Selati River	B7H019	1565 166 644 0.395 No data	903-1757 149-190 240-752 0.267-0.491	n = 4 n = 5 " "
TDS Chloride Sulphate ortho-phosphate Total phosphate	3	Mamba	B7H015	455 49 99 0.033 0.06	422-617 45-66 90-170 0.031-0.048 0.05-0.08	n = 3 " " " "
All variables	6	Tsiri	B7H016	No '99 data		
TDS Chloride Sulphate ortho-phosphate Total phosphate	8	Balule/KNP Hiking trail	B7H017/ B7H018	420 47 93 0.022 0.056	348-600 40-70 62-176 0.01-0.06 0.033-0.109	n = 7 " " " "

Table 5.5 *Observed values of water quality variables used for validation of the model. Median taken of June, July and August values for 1998.*

WQ variable	Reach number	Site name	Monitoring station	Monthly conc (mg/litre)	Range	Sample size
TDS	1	Barrage outflow	B7R002	No data		
Chloride						
Sulphate						
ortho-phosphate						
Total phosphate						
TDS	2	Selati River	B7H019	1936	1896-2021	n = 5
Chloride				219	214-224	"
Sulphate				849	838-943	"
ortho-phosphate				0.308	0.305-0.362	"
Total phosphate				No data		
TDS	3	Mamba	B7H015	898	608-1066	n=3
Chloride				49	72-131	"
Sulphate				99	155-386	"
ortho-phosphate				0.033	0.027-0.05	"
Total phosphate				0.06	0.07-0.11	"
All variables	6	Tsiri	B7H016	No '98 data		
TDS	8	Balule/KNP Hiking trail	B7H017/ B7H018	No '98 data		
Chloride						
Sulphate						
ortho-phosphate						
Total phosphate						

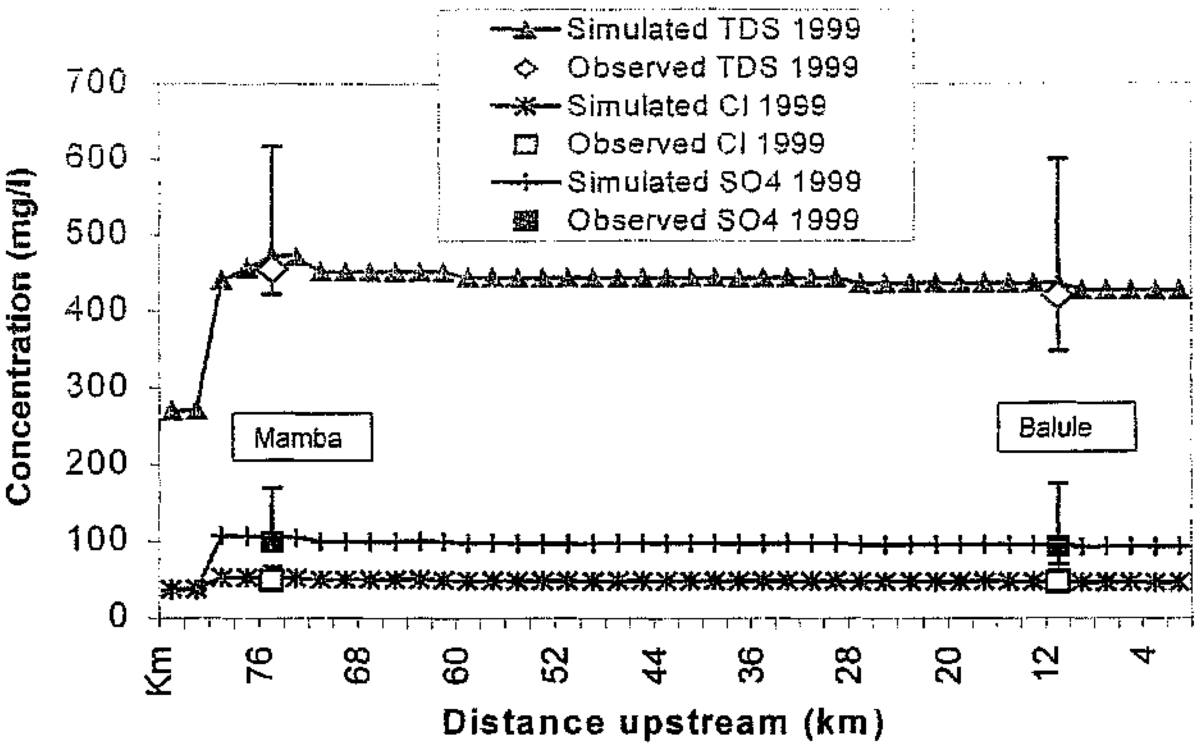


Fig. 5.5 *Observed and simulated concentrations of TDS, Cl^- and SO_4^{-2} along the length of the river for 1999.*

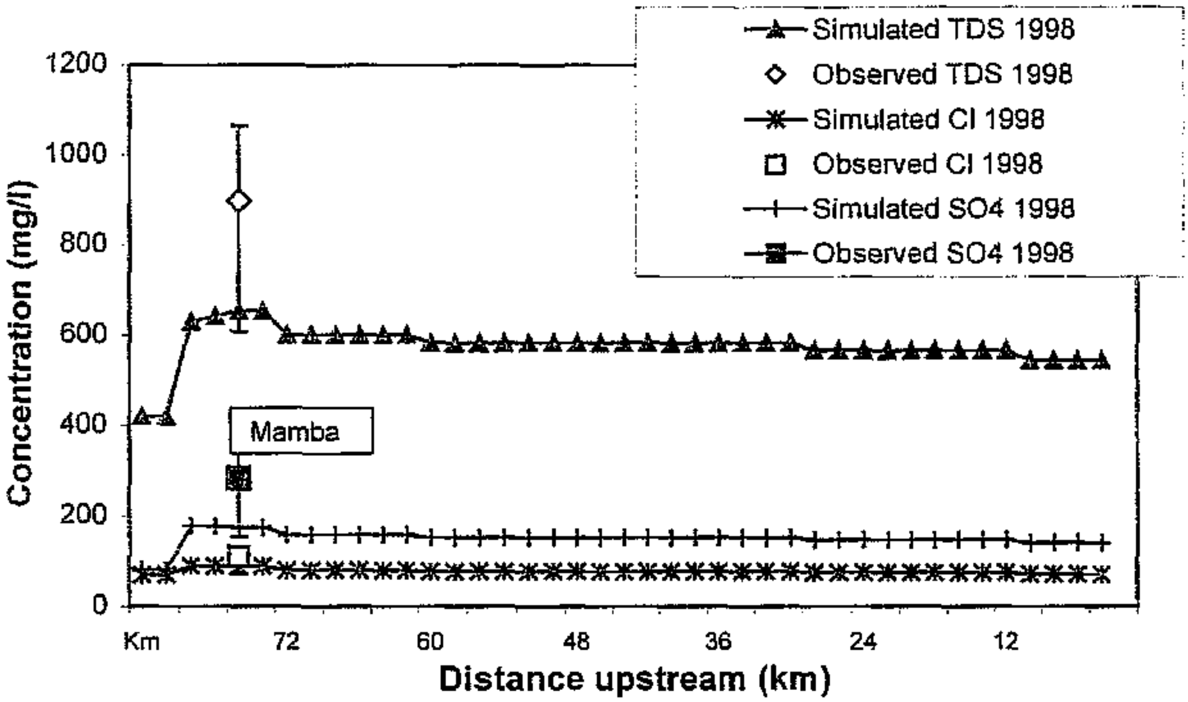


Fig. 5.6 *Observed and simulated concentrations of TDS, Cl^- and SO_4^{-2} along the length of the river for 1998.*

5.3.3 Simulation of phosphorus concentrations

The various transformations that phosphorus can undergo in QUAL2E are shown in Fig. 5.7. The values of parameters determine the extent to which interconversion between the various forms of phosphorus occur, such as, for example, the proportion of organic phosphorus that is allowed in the model to be converted to dissolved phosphate. It is necessary to estimate the values of these parameters in order to set up the model. DWAF records the concentration of ortho-phosphate and (less frequently) total phosphorus at many monitoring sites, including some of those on the lower reaches of the Olifants River. QUAL2E, on the other hand is able to simulate dissolved and organic phosphorus (Brown and Barnwell 1987), but not total phosphorus. Thus inorganic phosphorus adsorbed to suspended sediments is not included in the model. A rough estimate of organic phosphorus concentration might be made by subtracting ortho-phosphate from total phosphorus.

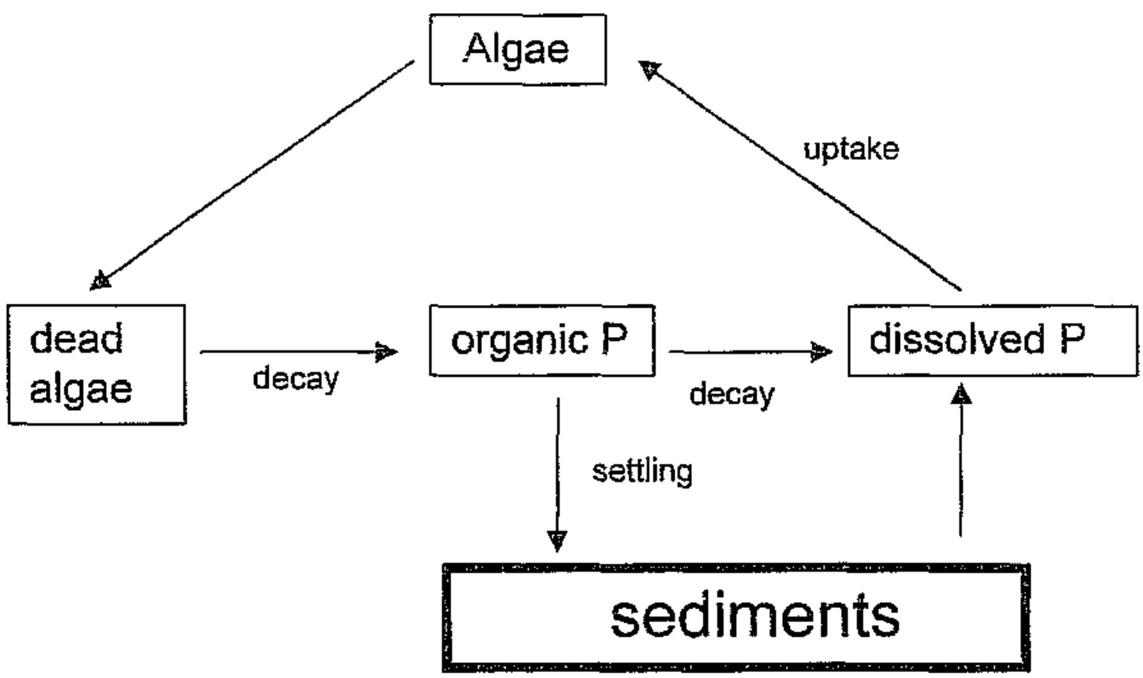


***Figure 5.7** Some of the interconversions of phosphorus considered in QUAL2E.*

This approach is probably not valid for the Olifants River, however, since it carries a reasonably high load of suspended sediments, even during the low flow months (N. Rossouw, Ninham Shand, Cape Town, *pers. comm.*). In addition, as shown in Table 5.4, the only site at which total phosphorus data were available was at Mamba (B7H015Q01). Attempts were made to model ortho-phosphate alone but these were not particularly accurate and, considering the uncertainties in the discharge data and the lack information on water quality, this was not pursued further.

5.3.4 Application of QUAL2E to predict the effect of differing management scenarios

In section 5.2.3.5, three different management scenarios were presented. The scenarios varied from each other either in terms of the total discharge that was allocated to Mamba, or with regard to the contribution of different sources to the overall pollutant loading of the system. The results of the different discharge and management scenarios with regard to the concentration of the three conservative variables (TDS, Cl^- and SO_4^{-2}) are presented below.

5.3.4.1 Scenario 1; Maintenance baseflow

In this scenario, the total discharge at Mamba was set equal to the minimum winter baseflow recommended at the IFR workshop. The minimum baseflows recommended for June, July and August at this site, were 9.4, 8.2 and 7.4 m^3/sec respectively. The model was therefore run using a total flow at Mamba equal to 7.4 m^3/sec. The inflow from the Selati River was set at 0.64 m^3/sec, which is the mean monthly discharge (for the winter months) for the period 1995-1999 (the winter monthly discharge for 1999 being 0.69 m^3/sec). The diffuse inflow considered to enter the system just above Mamba, as well as inflows from the lower tributaries were set the same as during calibration. The discharge from the barrage was then set at 6.6 m^3/sec, the value that was calculated to produce a resultant discharge at Mamba equal to 7.4 m^3/sec. In addition, it was assumed that the concentration of the water quality constituents originating from each headwater or each tributary was the same as during calibration. A summary of the input values for discharge and water quality used in the simulation of this scenario is given in Table 5.6.

Under the conditions prescribed by the above scenario, the resulting discharge at Balule, the most downstream point in the system, was predicted to be 8.6 m^3/sec. The predicted concentrations of the conservative constituents at Mamba and Balule under minimum maintenance baseflow are recorded in the top two data lines of Table 5.7. QUAL2E predicts that the instream concentration of TDS at Mamba that will occur from the scenario detailed, is 435.5 mg/litre and that of chloride and sulphate is 49 and 89 mg/litre respectively. The water quality at Balule is predicted to be considerably better than at Mamba. Note that these same values could also be derived from a simple mass balance calculation (section 2.7.1) as shown below for TDS at Mamba.

$$\text{Conc. TDS at Mamba} = \frac{\text{Load from barrage + Load from Selati + Diffuse load}}{\text{Total discharge at Mamba}}$$

$$= \frac{(6.6*280) + (0.64*1564) + (0.2*1900)}{(6.6+0.64+0.2)} = \mathbf{434.0\ mg/litre\ TDS}$$

Table 5.6 *Input values of discharge and concentration of conservative constituents for Scenario 1 (i.e. discharge at Mamba equal to the lowest recommended maintenance baseflow for a winter month).*

Site	**Discharge (m^3/sec)**	**Concentration (mg/litre)**		
		TDS	**chloride**	**sulphate**
Barrage outflow	6.6	280	38	36
Selati River	0.64	1564	166	643
Diffuse inflow @ Mamba	0.2	1900	30	30
Lower tributaries: Klaserie R. Tsiri Nhlaralumi Timbavati	 0.5 0.2 0.2 0.3	 200 200 200 200	 15 15 15 15	 30 30 30 30

Table 5.7 Predicted concentrations of conservative constituents (mg/litre) for Scenario 1 (i.e. for the recommended maintenance and drought baseflows at Mamba of 7.4 and 2.1 m^3/sec respectively).

Concentrations predicted by QUAL2E				
Site	**Discharge (m^3/sec)**	**Predicted concentration**		
		TDS	**Chloride**	**Sulphate**
	Recommended maintenance Q			
Mamba	7.4	435.5	49	89
Balule	8.6	395	44	81
	Recommended drought Q			
Mamba	2.1	821	76	221
Balule	3.3	595	54	152

5.3.4.2 Scenario 2; Drought baseflow

In this scenario, the total discharge at Mamba was set equal to the minimum winter baseflow during drought periods as recommended at the IFR workshop. This discharge was 2.1 m^3/sec. The model was therefore run setting the total flow at Mamba = to 2.1 m^3/sec. The inflow from the Selati River remained set at 0.64 m^3/sec (the mean monthly discharge, for the winter months, for the period 1995-1999, and the value used in Scenario 1). Because no other data were available, the diffuse inflow into the river immediately above Mamba, as well as inflows from the tributaries, were left at the same values as used during both calibration, as well as for Scenario 1. Discharge from the barrage was then set at 1.26 m^3/sec, the value that was calculated to produce a resultant discharge at Mamba equal to 2.1 m^3/sec. Discharge at Balule under this scenario was predicted to be 3.3 m^3/sec. In the case of pollutant loading from the different sources, it was assumed that the concentration of the water quality constituents originating from each headwater or each tributary was the same as during calibration.

The predicted concentrations of the conservative constituents at Mamba and Balule under minimum maintenance baseflow are given in the third and fourth data lines of Table 5.7. Not surprisingly, the instream concentration of all three variables is predicted to be higher under the drought discharge regime than under maintenance flows. This is to be expected since the total discharge (and hence the dilution capacity) is reduced. Another expected result is that the overall water quality at Balule is better than for Mamba. This is a result of the comparatively good quality water that enters below Mamba from the four lower tributaries.

5.3.4.3 Scenario 3; Reduction of pollution sources and variation in pollutant loading from the headwaters

i) Reduction of non-point source pollution

In order to investigate the effect of reduction of pollution, the model was adjusted so that the diffuse pollution arising from the mining complex at Phalaborwa was removed. Consequently, the loading of TDS, chloride and sulphate in the diffuse pollution that enters the system just above Mamba was reduced and these concentrations set to the same values as for the outflow from the Phalaborwa barrage (namely: TDS =280, Cl^- = 38, SO_4^{-2} = 36 mg/litre).

The concentrations of the three conservative variables under Scenario 3 that are predicted by QUAL2E can be seen in Figure 5.8. QUAL2E predicts that along the entire length of the modelled portion of Olifants River, TDS would be below 400 mg/litre. The proposed Ecological Reserve Category (Ecological management class) for Mamba is a "C". Therefore the instream TDS concentration at this site should be maintained below 520 mg/litre, which is the upper boundary for a "C" class in this system (Palmer and Rossouw 2000). The concentrations of the constituents that are predicted for Mamba (at 72 km in Fig. 5.8) are TDS = 392, Cl^- = 49 and SO_4^{-2} = 89 mg/litre. The predicted concentrations for Balule (at 0 km in Fig. 5.8) are TDS = 371, Cl^- = 45 and SO_4^{-2} = 83 mg/litre. Note that these values can be checked using simple mass balance calculations as for Scenario 1. The calculations were slightly more laborious for Balule since more sources (the four downstream tributaries, as well as the input from the Olifants River at Mamba) needed to be taken into account, but the same values were obtained as predicted by QUAL2E.

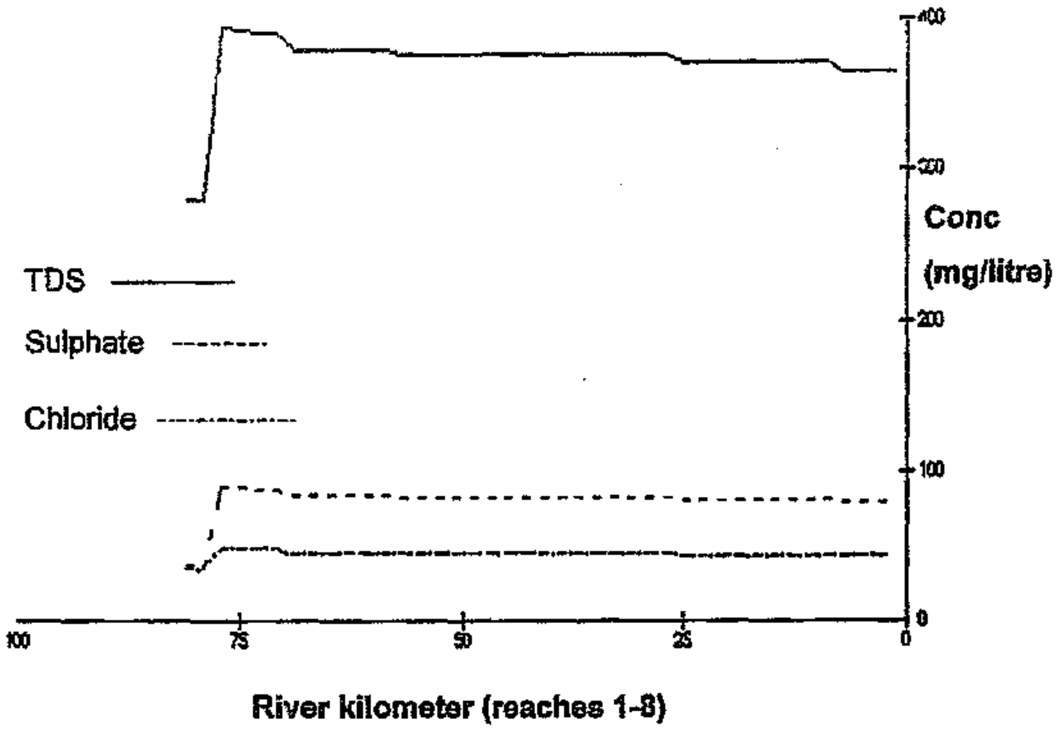


Figure 5.8 *Example of the type of graphical output obtained from QUAL2E. This simulation shows the predicted TDS, Cl^- and SO_4^{-2} concentrations along the length of the Olifants River if the diffuse pollution source above Mamba is controlled (TDS reduced to 280mg/litre). The lower broken line is Cl^- and the upper broken line is SO_4^{-2}.*

ii) Reduction of point-sources of pollution

In order to investigate the relative contributions of pollutants from the two headwaters on the total pollution load at Mamba, a further simulation was carried out. In this case the input data were the same as those shown in Table 5.6 (including the TDS concentration and discharge in the diffuse pollution entering the system above Mamba being set at 1900 mg/litre and 0.2 m^3/sec). The concentrations of chemical constituents entering the system via the Selati River (i.e. as a point source) were reduced considerably, however. The values used were TDS = 520, Cl^- = 50 and SO_4^{-2} = 50 mg/litre. These concentrations were chosen arbitrarily.

QUAL2E predicted that under this water quality management scenario the resulting concentration of TDS, Cl^- and SO_4^{-2} at Mamba would be 345, 39 and 37 mg/litre respectively, and at Balule, 328, 36 and 36 mg/litre. Thus under this scenario TDS for the river reaches that were modelled would be in a "C" class for TDS. Receiving water quality objectives for chloride and sulphate were not set as part of the Reserve determination for the Olifants River and thus no comment can be made as to the achievement of any specified limits for those variables.

5.3.5 Comparison of the results obtained from QUAL2E and Q-C modelling

The results obtained from the two types of water quality modelling, QUAL2E and Q-C modelling, were compared for Scenarios 1 and 2. Discharge-concentration modelling had not been undertaken for chloride, so the results for this constituent could not be compared. The Q-C plots for TDS and SO_4^{-2} at Mamba and for TDS at Balule are shown in Fig. 5.9 and a comparison of the results obtained using both methods is shown in Table 5.8.

Considering the predicted concentrations at Mamba under Scenario 1 (section 5.3.4) the total discharge at Mamba was set as 7.4 m^3/sec. The predicted concentrations of TDS and SO_4^{-2} at this site are shown in Table 5.6 (and to facilitate comparison, again in Table 5.8). From the Q-C plots for Mamba as well as from Table 5.8, it can be seen that the predicted TDS concentration for a discharge of 7.4 m^3/sec at this site is 436 mg/litre using QUAL2E, and 1058 mg/litre using Q-C modelling. In a similar manner the TDS and sulphate concentrations for both Mamba and Balule can be compared for both scenarios using the two types of water quality modelling. In all cases there is a very large difference between the predicted concentrations. The concentrations predicted by Q-C modelling are always higher than those predicted by QUAL2E. In addition, the predicted concentrations for TDS concentration at Balule are lower than for Mamba, whichever model is employed. Possible reasons for the discrepancy in the predictions obtained with the two different modelling methods are given in section 5.4.2.

Table 5.8 Comparison of concentrations of conservative constituents (mg/litre) predicted at Mamba and Balule using QUAL2E and Q-C modelling under the recommended maintenance and drought flow regimes (Scenarios 1 and 2 respectively).

Concentrations predicted by QUAL2E				
Site	Discharge (m^3/sec)	Predicted concentration		
		TDS	Chloride	Sulphate
	Recommended maintenance Q			
Mamba	7.4	435.5	49	89
Balule	8.6	395	44	81
	Recommended drought Q			
Mamba	2.1	821	76	221
Balule	3.3	595	54	152
Concentrations predicted by Q-C modelling				
Site	Discharge (m^3/sec)	Predicted concentration		
		TDS	Chloride	Sulphate
	Recommended maintenance Q			
Mamba	7.4	1058	Not modelled	343
Balule	8.6	500	Not modelled	Not modelled
	Recommended drought Q			
Mamba	2.1	1407	Not modelled	573
Balule	3.3	643	Not modelled	Not modelled

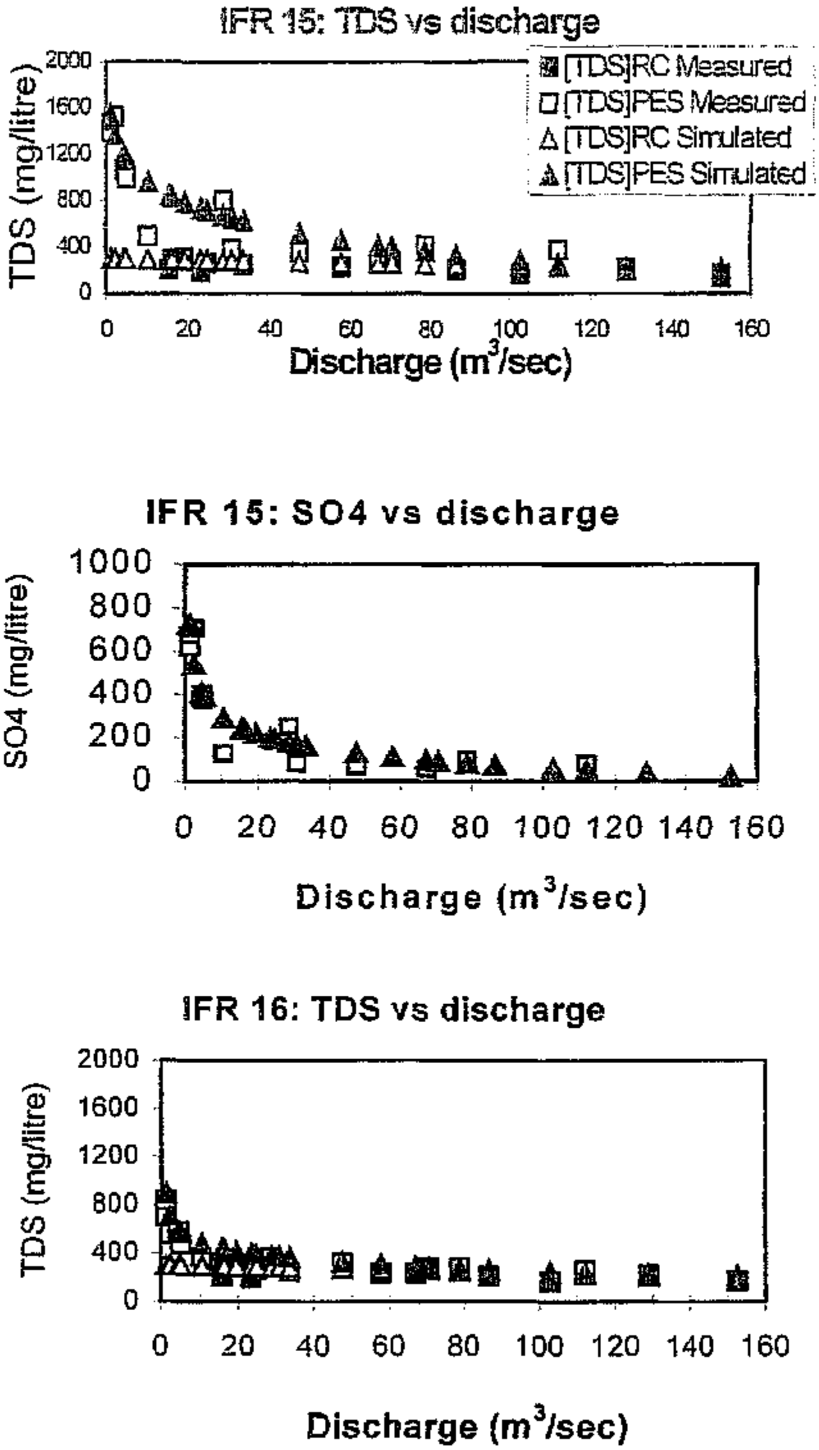


Figure 5.9 *Q-C plots for TDS and sulphate (SO_4^{-2})for Mamba (Olifants River: IFR 15) and TDS for Balule (Olifants River: IFR 16).*

5.4 Discussion

5.4.1 General considerations on using QUAL2E

5.4.1.1 *Accuracy of the predictions*

The accuracy of the predictions (for both discharge and concentration) that were obtained using QUAL2E was evaluated by comparing simulated values with measured values. From Figure 5.4 it can be seen that, with regard to discharge, the correspondence between measured and simulated values for both 1999, and to a lesser extent for 1998, appeared to be reasonably good. There were only two sites with measured discharge data, however, and none further downstream than Mamba. Consequently it is difficult to assess how realistic the discharge simulations are. In addition, there was a lack of discharge data for calibration at some sites, necessitating the use of estimates. It was not possible to obtain an accurate water balance for the system, since the observed discharge at Mamba was more than the sum of the contributions from the Phalaborwa barrage and the Selati River. In order to account for this additional volume of water, it was assumed that there was diffuse flow entering just above Mamba, and this was incorporated into the model. The volume of this diffuse source was slightly different for 1999 and 1998, although in the various simulations, this difference was ignored and the discharge attributed to diffuse inflow was set at 0.2 m^3/sec.

The main causes of uncertainty with regard to discharge were:

- the lack of any discharge data for the Olifants River downstream of Mamba Weir. Thus the discharge from the downstream tributaries had to be estimated. This was carried out by manipulating flow from these sources until an acceptable (i.e. in agreement with measured data), dilution of conservative substances at Balule was obtained;
- very limited discharge data for the downstream tributaries;
- no data to indicate the volume (if any) of diffuse flow entering the system immediately above Mamba, and no information to indicate how this might change from year to year.

From Figures 5.5 and it can be seen that, with regard to water quality, the correspondence between measured and simulated values for all three conservative constituents for 1999 is relatively good. This result is likely to be misleading, however, since it was necessary to adjust the pollutant concentrations in the diffuse source until a good simulation for TDS, chloride and sulphate was obtained at Mamba. Similarly, in the absence of any measured water quality data, the concentration in the lower tributaries was adjusted until a reasonable correspondence was obtained between measured and simulated values at Balule. For validation (1998) the same pollutant loadings for the diffuse source and the lower tributaries were used as for calibration. Poor correspondence was obtained between simulated and observed values for TDS and sulphate (Figure 5.6), although the results for chloride are acceptable. There is no reason to assume that the pollutant loading for the diffuse source and lower tributaries should remain the same from year to year. Furthermore, no data are available to indicate what the values actually are. The main causes of uncertainty with regard to predictions of water quality are as follows:

- because of a lack of water quality data for the outflow from the Phalaborwa barrage, input concentrations for this headwater were estimated from the nearest upstream site on the Olifants River, namely from Oxford (B7H007). Due to releases of freshwater from the Blydepoort dam, water quality at Oxford is variable and may not reflect the outflow from the barrage;
- no data were available to a) confirm a non-point source of pollutants entering the system above Mamba and b) indicate the concentration of pollutants in this source;
- no water quality data were available for Balule for 1998 for testing the model.

Because estimates needed to be made for several key areas, the confidence in any predictions that were made of either discharge or water quality were considerably reduced. Furthermore, better simulations may have been obtained by using instantaneous values for both water quality and discharge (matched data sets) for all sites, rather than taking the median or monthly value over three months. Monthly values were used, however, because this is the temporal resolution used in the Building Block Methodology.

5.4.1.2 The management scenarios considered

The junction with the Selati River is just upstream of Mamba, and thus theoretically the increased discharge (the recommended IFR), could be supplied to this site by increasing outflow from either source. In practice, however, there is an impoundment on the Olifants River (the Phalaborwa Barrage) just upstream of Mamba but not in the lower reaches of the Selati River. It was therefore considered that the most realistic management scenario was to increase discharge at Mamba by supplying more from the barrage. As a result, in the case of Scenario 1, which simulated the recommended maintenance discharge at Mamba, the volume of discharge and concentration of constituents entering the system from the Selati River, from the diffuse pollutant source and from the lower tributaries, remained the same as during calibration. Only the volume of water (but not the water quality) released from the barrage was altered. In the case of Scenario 2, the recommended drought flow at Mamba was simulated, and once again, only the volume of water released from the barrage was altered. This second scenario is likely to be somewhat unrealistic, however, since under conditions of drought, the discharge arising from natural runoff would decrease. Discharge from the mining complex in the form of effluent entering the Selati River and diffuse inflow above Mamba, on the other hand, is not likely to decrease significantly under drought conditions. In such situations, the concentration of chemical constituents will increase in the Selati and Olifants Rivers. More accurate simulations could have been made by taking these factors into account, but because of the generally poor level of simulation in other areas, it was considered to be unprofitable to pursue this further.

One of the major advantages of implementing QUAL2E is that it enables a more profound understanding of the system than is obtained using Q-C modelling. For instance, one of the assumptions made at the IFR workshop was that there was no significant increase in discharge between Mamba and Balule during the winter months. The detailed examination of the system which was necessary when implementing QUAL2E showed that there is considerable dilution of conservative substances in this stretch of river and that the assumption is possibly erroneous. It should be noted, however, that this is not a serious error in that it would have led to an under estimation of discharge at Balule and thus would have led to predictions of worse water quality conditions than are actually likely to occur.

5.4.1.3 Suitability of QUAL2E to South African conditions

Because of the inaccurate simulations that were obtained from this application of QUAL2E it is difficult to assess how suitable the modelling method is for general use in South Africa. From the point of view of data availability, the system that was modelled is considered by the authors to be reasonably well monitored with regard to discharge and water quality, compared to many other parts of the country. Despite this, because of a lack of data for both discharge and constituent concentrations for key points on the system, a realistic model could not be set up. In addition, it should be noted that this was the situation for conservative substances, the simplest water quality variables to model (Dortch and Martin 1989). In the case of the non-conservative constituent, phosphorus, no simulations at all could be carried out. Thus it can be concluded that without the collection of field data gathered specially in order to calibrate the model, it is not likely that realistic simulations could be carried out in South Africa using QUAL2E. This conclusion has important implications for the Reserve determination process and is discussed further in section 5.4.3.

In setting up QUAL2E for a specific river system, special attention may need to be given to allow for evaporation from the river surface during summer, as in the simulations for the Vaal River carried out by Rossouw and Quibell (1993). This is especially necessary if low-flow situations need to be evaluated in wide, shallow rivers from which evaporation may lead to significant loss of water. The fact that only organic and dissolved phosphorus but not total phosphorus is considered by QUAL2E is a further disadvantage. In highly turbid systems a large proportion of the phosphate load may be bound to sediments (N. Rossouw, Ninham Shand, Cape Town, *pers. comm.*). Omitting this component may result in inaccurate simulations of this water quality constituent. On the other hand DWAF do not carry out determination of particulate or organic phosphorus on a routine basis. Thus using QUAL2E for simulations of phosphorus in South African rivers may pose difficulties. Despite limitations with regard to discharge and water quality data, QUAL2E has been used in this country with reasonable levels of success for predicting instream concentrations of several water quality variables including non-conservative constituents and *E. coli* (Rossouw and Quibell 1993; Ninham Shand 1999; Steynberg, Venter, de Wet, *et al.* 1995). In addition, it is also widely used in Australia as the water quality module of IQQM (Integrated quality-quantity model) which has been used in environmental flow assessments (Podger, Sharma and Black, 1994;

Simons, Podger and Cooke 1996; Podger and Hameed 2000). Furthermore, QUAL2E is one of the most commonly used instream water quality models used world-wide.

5.4.2 Comparison of QUAL2E and Q-C modelling

Table 5.8 shows a comparison of the water quality predicted for two different discharge scenarios using QUAL2E and Q-C modelling. These results are very dissimilar and serve to emphasise the major difference between the two approaches: namely that Q-C modelling is a purely empirical method, whereas QUAL2E is more deterministic. Mamba is immediately downstream of the confluence with a major tributary in which the water quality is usually considerably different (worse) than the mainstem river. The Q-C plot for Mamba encompasses all the pollutant loading scenarios that have been recorded at that site, but cannot differentiate between different scenarios. When using QUAL2E however, different management scenarios (i.e. pollutant loading from various sources) can be specified and indeed specific scenarios were chosen and were used to make the predictions of resultant water quality. We have little idea of how realistic these scenarios are. This is the result firstly of a lack of general relevant discharge and water quality data, in that it is not clear how accurately the model describes the system. Secondly there is a lack of *information* arising from the IFR workshop in that only the required discharge (maintenance or drought) at a site is specified but not *how it should be achieved*. In other words the relative proportion of discharge that should come from each source is not prescribed. This lack of consideration of water quality management scenarios is a major omission in the IFR process and is discussed further in Chapter 7. With regard to the results shown in Table 5.8 it is considered that the predictions of neither model is likely to be particularly accurate for Mamba. The results of Q-C modelling are probably more accurate in this instance than those of QUAL2E in that they are based on observed data for Mamba. As long as the recommended discharge regime is supplied in the same manner (i.e. the same proportion of pollutants from the two headwaters) as during the time period used to derive the Q-C relationship, the predictions using this model should be reasonably good approximations. The results obtained from QUAL2E, on the other hand, have the potential to be very accurate for a specified water quality management scenario. In order to achieve realistic predictions with QUAL2E, it is necessary first to obtain accurate data to calibrate the model successfully and secondly to have detailed data describing the specific management scenario that is required.

With regard to the time, effort and amount of data that is required by the two approaches, the modelling methods differ considerably. The data requirements of QUAL2E have been discussed elsewhere in this chapter and are reasonably extensive. Discharge-concentration modelling on the other hand usually makes use of the median Reference and Present Ecological State water quality concentrations that are generated during the assessment of water quality (Chapter 2). As a result, Q-C modelling normally takes approximately a day per site (depending on the number of water quality variables modelled – typically three). Depending on the complexity of the system that is simulated, setting up QUAL2E can easily take several weeks. Furthermore, a certain level of expertise is necessary when using QUAL2E, whereas in the case of Q-C modelling, as long as the modeller is aware of the limitations of the technique, it can be used by unspecialised technicians.

As can be seen from the previous discussion, Q-C modelling gives only estimations of instream water quality and is a purely empirical method. QUAL2E is more mechanistic and enables an understanding of the sources and sinks of a given chemical constituent within the system. If only conservative constituents are of importance at a site, and if the system is relatively simple (i.e. not too many sources and sinks) then it should only be necessary to use Q-C modelling. Simple mass-balance calculations can be used in addition to differentiate between different management scenarios (i.e. differences in source pollutant loads). If on the other hand, the site under consideration is complex, with several sources of pollutants, QUAL2E should be used as it aids an understanding of some of the factors contributing to instream water quality. Furthermore, if temperature or non-conservative constituents are of importance at a particular site (including BOD, DO), it would also be necessary to set up QUAL2E. In all cases where QUAL2E is employed it would usually be necessary to initiate a sampling programme.

5.4.3 Application of QUAL2E to Environmental Flow Assessments

It is the opinion of the authors that there is a place for the instream water quality model QUAL2E in the Reserve determination process. The following points need to be considered however when incorporating QUAL2E into such assessments.

- Discharge and water quality data usually need to be collected in the field in order to calibrate QUAL2E. This has implications with regard to the time, expertise required and costs involved in integrating water quality and quantity. Therefore this model would probably only be used as part of a Comprehensive Reserve Determination. The necessity for the collection of data in the field and the accompanying increase in costs would make the exercise too expensive for a lower level of Reserve determination.
- Because of the increased time and financial requirements mentioned above, QUAL2E should be used only at key sites. These would be sites that are complex in terms of sources and sinks of water quality constituents. In other words QUAL2E is useful for performing mass-balance calculations in complicated systems where there are many loads originating from tributaries and point sources, or several points of abstraction. In addition, if detailed predictions of non-conservative water quality variables are required, this modelling method should be employed.
- The extra effort required in setting up QUAL2E would only be justified if, in addition to the situations mentioned above, different water quality management scenarios are to be considered.

5.5 Conclusion

Because of a lack of water quality and discharge data it was not possible to obtain reliable predictions of the instream concentrations of water quality variables for the Lower Olifants River. The exercise was beneficial, however, for assessing the potential usefulness of QUAL2E within the Reserve process in situations where data are not limiting. It is concluded that QUAL2E can be effective in addressing some of the limitations of the Q-C modelling method – in that it can predict the outcome of different pollutant loading scenarios and can be used to simulate non-conservative water quality constituents as well as physical variables such as temperature and DO. These advantages however come at a price – that of additional data requirement. This in turn means that QUAL2E takes much longer to set up, and requires expert knowledge, both of which contribute to a greater cost of implementation. Thus QUAL2E should be used at key sites for Comprehensive Reserve determinations only where Q-C modelling and mass balance calculations are not suitable.

CHAPTER 6

PREDICTING THE IMPLICATIONS OF CHANGED WATER QUALITY FOR THE BIOTA

6.1 Introduction

As reported in the introductory chapter to this report, not only can discharge have a marked effect on water quality - but water quality can also exert a profound effect on the functioning, structure and abundance of aquatic biota. All organisms exhibit tolerance ranges for environmental variables, including those pertaining to water quality, within which they can survive and reproduce. Outside of these tolerance ranges, processes such as growth and fecundity can be compromised. Because the species forming an assemblage in a stream exhibit different tolerance ranges, changes in water quality will affect some species more than others. As the alteration in water quality becomes more pronounced, some species will not be able to survive and will disappear from the community whilst other, more tolerant species, will start to establish themselves. The philosophy behind the macroinvertebrate biomonitoring system "SASS" makes use of the differing tolerance ranges of individual families. A more comprehensive discussion of biomonitoring, as well as methods that have been used in the literature to predict the effects of water quality on the aquatic biota, is given in the literature review that originated from this project (Malan and Day 2002).

Because natural ecosystems are complex and highly variable, it is extremely difficult to predict the effect that changed water quality might have on them. At best, only general predictions can be made. This problem is exacerbated by the fact that there are few data linking water quality and biota, especially with regard to indigenous organisms, whilst the variability of ecological systems necessitates the compilation of extensive field data sets. In addition, water quality variables can interact with one another, leading to additive, antagonistic and synergistic effects.

Whilst taking the above caveats into account, attention was given to developing a method that could be used to predict the implications of altered water quality for the aquatic biota. The result is a set of steps termed the "Biotic Protocol" which represents a structured way for assessing the likely effects of changed water quality on macroinvertebrates. The method takes into account existing sources of relevant information, such as documented tolerance ranges from ecotoxicological studies, as well as the South African water quality guidelines. It is, however, based primarily on the examination of historical records, where the presence of macroinvertebrate families (and hence derived SASS scores) is linked with measurements of water quality variables. These data are recorded in the Biobase, a database of linked historical chemical and biological data, as well as in the Rivers database, which was developed to contain the sampling results obtained from the national biomonitoring initiative, the River Health Programme (section 1.9.4). This protocol can be used either within the Reserve determination process, or in other situations where the effect of altered water quality (for instance resulting from increased effluent discharge) on the biota needs to be assessed. The individual steps of the protocol are discussed below. This is followed by an application of the Biotic protocol to the Lower Olifants River (section 6.5) and the Middle Olifants River (section 6.6).

6.2 The Biotic Protocol

6.2.1 General outline

The Biotic protocol is carried out separately, for each critical water quality reach or a particular site (e.g. an IFR site), for the proposed discharge regime. The steps of the protocol are summarised in Figure 6.1.

i) Using water quality modelling, tabulate and identify the critical values for each significant water quality variable. This is carried out separately for the:

a) The Reference Condition,

b) The Present Ecological State

c) The Future Predicted State.

Figure 6.1 Summary of the Biotic Protocol

i) Using water quality modelling, tabulate and identify the critical values for each water quality variable. This is carried out separately for the;

a) Reference Condition,
b) Present Ecological State,
c) Future Predicted State.

ii) Compare the critical values (Ccrit) with the Target water quality range and identify the water quality variables (typically two or three) likely to pose the most serious risk.

iii) Compare ecotoxicological parameters, if available, with the critical values in order to estimate the toxicity of the variable in question.

iv) Consult the Biobase and Rivers database for sampling data characteristic of the ***Reference Condition*** *for;*

a) The specific river in question, and/or
b) similar systems (i.e. in the same ecoregion and type of river).

v) Consult the Biobase and Rivers database for sampling data characteristic of the ***Future predicted state*** *for;*

a) The specific river in question, and/or
b) similar systems (i.e. the same ecoregion, and type of river).

vi) Compare taxa lists for the Reference Condition and Future predicted state. Derive a theoretical SASS score and a tentative Assessment class for the Future predicted state.

vii) Include input from any other biotic tolerance indices and databases that may be relevant.

viii) Synthesize a scenario for the aquatic biota that is likely to be the consequence of the proposed change in discharge. Assign the future Assessment class (A-F).

Any type of water quality modelling can be used (e.g. discharge-concentration, QUAL2E) providing that monthly values for a given water quality variable can be predicted. From these simulations the critical values can be identified. In the case of TDS, nutrients or other chemical constituents, the critical concentration (Ccrit) is defined as the maximum monthly value to be recorded during the entire year (for example, see Table 2.2). In the case of pH or DO, the critical value would be the monthly value that shows the greatest deviation from the Reference Condition. It is important to note that three sets of discharge, water quality and biological data are involved in the protocol, namely those characteristic of the Reference Condition, those characteristic of the Present Ecological State and those representing the Future Predicted State (under the recommended discharge regime). It is useful to tabulate the minimum, median and maximum monthly values for each of the important water quality variables at the site under consideration, and for each of the three scenarios. An example of such a set of information is shown in Table 6.3.

ii) Compare the critical values (Ccrit) with the Target water quality range and identify the two or three water quality variables likely to pose the most serious risk at that particular site.

Target water quality ranges (TWQR) are the recommended ranges of concentrations of chemical constituents and values of physical variables that should not be exceeded in aquatic ecosystems. These values are documented in the South African Water Quality Guidelines (DWAF 1996). Two or three critical water quality variables are selected, since this is considered to be a practical number to work with. A larger number of water quality variables can be examined, although this tends to make the process cumbersome. If only one variable is chosen, on the other hand, interactions between water quality variables (and possible antagonistic, synergistic or additive effects) are not taken into account. The choice as to which water quality variables are likely to pose the most serious threat to the biota requires expert judgement. Toxins are rated as the most important followed by system variables.

iii) Compare ecotoxicological parameters, if available, with the critical concentrations in order to estimate the toxicity of the variable in question.

Check ecotoxicological databases for data on the water quality variables under consideration, both for indigenous organisms, as well as for laboratory organisms in international databases (see section 1.9.4 for possible sources of ecotoxicological data). Ecotoxicological parameters such as the LOEC (the lowest concentration that brings about an observed effect) and LC_{50} (the concentration that corresponds to a 50% cumulative probability of death of the test population) can be compared with the critical values predicted for the recommended discharge regime. Thus an idea of the toxicity of individual water quality constituents can be obtained.

iv) Consult the Biobase and Rivers database for sampling data characteristic of the ***Reference Condition*** *for:*

a) the specific river in question;

b) similar systems (i.e. other rivers of the same type and in the same ecoregion).

The aim of this step is to identify macroinvertebrate taxa that are likely to be present at the IFR site in question under unimpacted conditions. This can be done by setting the theoretical SASS and ASPT score equal to values representative of the Reference Condition for that given ecoregion and type of river (see note 1, section 6.2.2), and then filtering the data in order to identify samples from least-impacted sites. An alternative way is to search the databases for sampling occasions in which the median monthly concentrations or values of physical variables are within 15% of the values expected in the Reference Condition.

v) Consult the Biobase and Rivers database for sampling data characteristic of the ***Future Predicted State*** *for:*

a) the specific river in question, and/or

b) similar systems (i.e. other rivers of the same type and in the same ecoregion).

Extract sampling data where the values of the water quality variable are similar to the critical values. Data are screened by setting the variable values equal to, or higher than, the critical value ± 10%. Lists are compiled of the macroinvertebrate taxa that can be expected to survive under the predicted water quality conditions. If no data are available in which all water quality variables (i.e. two or three) are similar to the critical value, the databases are again interrogated setting only fewer variables equal to 90% of the critical value. A list of invertebrate taxa likely to be found under such a water quality scenario is compiled. A lower confidence can be placed in the predictions in this case however, since the situation is less representative of the future water quality scenario.

vi) Compare taxa lists for the Reference Condition, Present Ecological State and Future Predicted State. Derive a theoretical SASS score and a tentative likely Assessment Category for the Future Predicted State.

Tables are drawn up recording the taxa expected under the Reference Condition and Present Ecological State (if required) and under water conditions pertaining to the Future Predicted State. SASS scores from different sites and sampling occasions that are characteristic of a given water quality scenario (i.e. either the Reference Condition, Present State or Future Predicted State), are combined by taking the median. A theoretical SASS score for the predicted impacted state can be calculated and from this a tentative, future Assessment Category (see note 2, section 6.2.2). Possible shifts in species composition including occurrence of nuisance species and loss of rare/key species, should also be noted at this point.

vii) Include input from any other tolerance indices (e.g. the Fish Index of Kleynhans, 1999) or databases that may be relevant.

This step can be included if suitable data and expertise, for example on fish or riparian vegetation, are available.

ix) Synthesize a likely scenario for the aquatic biota as a result of the proposed change in discharge. Assign the future Assessment Category (A-F).

This step incorporates expert knowledge of the particular river of concern. Consideration of factors such as the occurrence of vulnerable sites (e.g. spawning sites) in a given water quality reach, sensitivity of juvenile life stages, etc., should be taken into account. In addition, the potential effects on the biota of likely alterations to hydraulic habitat should also be assessed. It will be necessary to liaise with other biological specialists at the IFR workshop to derive this information. It is also necessary at this step to consider how representative the sampling data were that were used to make predictions of the impact. It was explained in step *v)* that the databases are first interrogated to obtain sampling occasions in which three water quality variables are equal to, or greater than 90% of the critical value. If no suitable sampling occasions are available, data are extracted in which two variables fit the required critera. The fewer variables that represent the Future Predicted State, the lower the confidence in the prediction. The final step is to reassess the tentative class assigned in step *vi)* in the light of the information obtained above and to produce a final future Assessment Category.

6.2.2 Points to note

1) SASS and ASPT (Average score per taxon) values used for identifying Reference conditions will vary according to the ecoregion (or Water quality management region; WQMR), as well as the type of river. The values recorded in Table 6.1 were taken from Dallas, Day, Musibono and Day (1998) for the Western Cape and from DWAF (1999) for the rest of the country. Those for the non-acidic regions should be considered as tentative until further confirmation. It is important that the water quality and biological specialists involved in a Reserve determination liaise with one another in order to decided on what is characteristic of the Reference Condition for the river in question. Version 4 of SASS was used in the development of this protocol, although version 5 (SASS5) is now available. Both forms are compatible with the protocol although in order to compare scores from different sampling occasions accurately it is necessary to first ensure that they are all converted to the same version of the index.

Table 6.1 *SASS4 score and ASPT values used to identify least impacted sampling sites in the Western Cape as well as for other Water Quality Management Regions (WQMR).*

WQMR	River type	SASS4 score ≥	ASPT ≥
Southern and western coast	Mountain stream	140	7.5
	Foothill	120	7.5
	Transitional	85	6.5
	Lowland	50	5.0
"A" class river		>140	>7.0

2) A tentative Assessment Category for the Future Predicted State can be derived from the theoretical SASS4 score by using Table 6.2. As in the case of SASS scores that are characteristic of Reference Condition (Table 6.1), the values of SASS4 scores that delineate the different Assessment Categories also require further research in order to confirm the exact values.

Table 6.2 A comparison of Assessment category, river condition, and associated SASS4 scores and ASPT values.

Class	Condition	SASS4 score	ASPT value
A	Excellent	> 140	> 7
B	Very good	121 – 140	6 – 7
C	Good	101 – 120	5 – 6
D	Fair	61 – 100	4 – 5
E	Poor	31 – 60	3 – 4
F	Very poor	< 30	< 3

(Taken from DWAF (1999) Resource directed measures for protection of water resources, Integrated report, January 1999) SASS4 = South African scoring system version 4, ASPT = average score per taxon.

6.3 General considerations

- The Biotic protocol has been devised as a means of checking all potential sources of useful information in a systematic manner. Because of the complexity of ecological interactions, no single source of information (whether ecotoxicological, or historical data from the Biobase or other databases) is enough to give a full understanding of the likely consequences of a proposed change in water quality for the biota. By including all sources of relevant information, a better prediction of the likely effects can be made. New databases that link, for example macroinvertebrate taxa and water quality, should they become available, could also be accessed.
- If the Biotic protocol is used in the context of a Reserve determination, some of the steps would be carried out as part of other activities. For example, lists of macroinvertebrate taxa are compiled by the invertebrate specialist for the Present Ecological State and also possibly for the Reference Condition. Present and Reference Condition water quality is also derived. It is probably necessary to apply the Biotic protocol only to critical reaches in which either water quality is very poor, or sensitive species are present, or if the reach has a high score on the Importance and Sensitivity index (Kleynhans and O'Keeffe 2000). In addition, it should be necessary to examine only the most extreme discharge scenarios (i.e. the ones that are likely to affect water quality most severely).

- As a consequence of the two points mentioned above, it is envisaged that water quality modelling would be carried out before and at the IFR workshop (and the results presented as the "water quality consequences"). On the other hand, assessing the implications of the predicted water quality scenario for the biota would be carried out after the workshop, once the prescribed flow regime and associated water quality is known.
- Interactions between water quality variables are taken into account by the manner in which the databases are interrogated. The databases are searched for sampling data in which as many variables as possible are simultaneously (approximately) equal to 90% of the predicted critical values. The records obtained are also filtered to ensure that they are obtained from the same ecoregion and type of river. The biological data (i.e. lists of macroinvertebrate taxa) characteristic of the given water quality scenario are then used to extrapolate to what taxa might be expected under the recommended discharge regime. If no sampling occasions are recorded in the database where all three variables were similar to the critical values, the process is repeated by interrogating the database for two variables, or even one. The smaller the number of variables that are in the same concentration range as the critical values, the lower the confidence that can be placed in those predictions. Ideally, more than three water quality variables should be used but, as mentioned previously, this is not practicable. In addition, due to the limited data that are available, the chances of finding sampling occasions recorded in the database in which more than three water quality variables are simultaneously in the range of the critical values becomes small.
- At many steps in the protocol, specialised knowledge of water quality and the macroinvertebrates characteristic of the river under consideration are required. Expert knowledge is needed for at least the following aspects:

  a) choice as to which water quality variables are likely to exert the most profound effects on the biota;

  b) identification of potential nuisance species or key species that are likely to be lost.

  c) synthesising information on the integrated effects of the proposed water quality scenario on the biota.
- In step v) of the protocol, the databases are searched for sampling data in which the concentration of the water quality variable(s) of concern is set at 90% of the critical

concentration. This value is arbitrary, and may need to be increased or decreased depending on availability of data, avoiding the situation where, for example, the critical value of TDS is 400 mg/litre, but sampling results in which TDS = 399 mg/litre is excluded. In data-rich situations, an upper limit to the critical value (Ccrit) should also be placed. For example:

$$90\%\text{Ccrit} \leq C \geq 110\%\text{Ccrit}$$

It is also necessary to limit the data in this way in cases where the water quality for Future predicted state is an improvement on the Present state (see section 6.5).

- The usefulness of ecotoxicological data can sometimes be limited. For example, TDS, being a composite rather than pure chemical constituent, is not listed in most databases. Similarly, nutrients usually exert an effect on aquatic ecosystems by altering the trophic state of an ecosystem, rather than having a toxic effect *per se.* Ecotoxicological data are most useful when toxic substances (i.e. those that exert a deleterious effect at very low concentrations) are present in a system and the likely concentrations under the recommended discharge regime can be predicted (see section 6.4). Sources of ecotoxicological data are given in section 1.9.4.
- Copies of the Biobase (on compact disk), in conjunction with a manual titled "The biological and chemical database: User manual" (Dallas, H.F. and Janssens, M.P. 1998; WRC Report No. TT 100/98) are available through the Water Research Commission. Information on the Rivers database can be obtained through the Rivers Health Programme. The website addresses are given at the end of this document.

6.4 Limitations and approximations in the method

The following limitations and approximations apply.

- The major stumbling block in the use of this protocol is the frequent lack of data linking water chemistry with the presence of macroinvertebrate taxa. As the River Health Programme becomes implemented in all provinces of the country, this situation should eventually be rectified to some extent. It is essential however, that in addition to the collection of biomonitoring data in the above programme, water quality data should be collected at the same site and at the same time.
- The protocol does not directly take into account any change or loss of hydraulic habitat that may occur under the new discharge scenario. For example, from a

consideration of the water quality under a given future scenario, it may be predicted that a given taxon will be present. If that taxon was riffle-dependent, however, and riffles would no longer be present all the time under the new recommended discharge scenario, that taxon is unlikely to become established. Nevertheless, loss of hydraulic habitat is accounted for indirectly in the protocol, in that it can be considered during the final step. This involves integration of all information, and empirical adjustment of the final future Ecological Reserve category.

- Aspects other than water quality and habitat availability may influence the presence of a given species in a given reach. This involves factors such as the presence of refugia and recolonisation sites. Although the water quality in a reach may be improved so that it is again suitable for certain species for instance, if there are no refugial sources of individuals to recolonise the site, the future presence of that species at that site is unlikely.
- Another important consideration follows from the ones above. Because a given taxon is not found in the database under particular conditions of water quality, this does not necessarily mean that the species cannot tolerate those conditions. The databases are patchy, and it may well be that the organism can survive such water quality but either this fact has not been recorded, or the taxon in question has not previously encountered such conditions. This point emphasizes the need for the compilation of extensive databases linking water quality and biomonitoring data.
- The Biotic protocol makes use of the "Critical values" to assess the potential effect of future water quality on the biota, where this refers to the maximum monthly concentration that is likely to occur over an entire year, in the case of chemical constituents (and the maximum deviation from natural in the case of pH, DO and temperature). It was noted in Chapter 4 that one of the likely consequences with regard to recommended environmental flow regimes is that, whilst periods of very poor quality water might be avoided, periods of extremely good quality might also no longer occur in a system. The overall effect on water quality of such a discharge regime would be to raise the mean annual concentration of chemical constituents. To infer the potential effects of a water quality scenario in which the annual mean concentration is raised, sites (or sampling occasions for the same site) would need to be compared which were identical in ecoregion, type of river and availability of habitat. In addition, the critical concentrations of the most significant water quality would also need to be similar. The only difference would be that the minimum and

possibly the median concentrations would need to differ between the two sites. It is not clear at this stage if the protocol would be suitable to distinguish these more subtle changes in a water quality scenario such as that described above. The biotic protocol as it now stands, is a relatively coarse method for inferring potential effects on the biota.

- In addition to the concentration of chemical constituents that are predicted for the system, the length of time that biota would be exposed to that concentration should also be considered. That is, not only should the hazard be considered but also the risk. In the protocol at the moment, there is no provision to compare water quality scenarios where, for example, a critical concentration would now occur for three months of the year, where previously it occurred one month of the year. As stated above, whether the protocol is able to predict the consequences of subtle changes in water quality is not clear and requires further research.
- The Biotic protocol (and the biomonitoring method, SASS) rely heavily on the presence or absence of taxa at a site. There has been criticism that the loss of taxa is possibly too extreme an effect, and that cognizance should be taken of shifts in abundance, rather than the presence, or absence of taxa (Dallas 2000). For the moment, SASS in its present form is the method that is recommended for assessing river health in South Africa. With regard to the Biotic Protocol, it needs to be as pragmatic as possible and since it is dependent on the data that are collected, it makes use of the most commonly used biomonitoring tool. Should that biomonitoring tool change however, adjustments might need to be made to the protocol.
- Some of the assumptions that apply to Q-C modelling are also of relevance in application of the Biotic protocol, namely, that predictions are made on the assumption that the pollution load will remain the same as at present. Furthermore, since the recommended maintenance and drought baseflows represent the minimum discharge for each month, predicted water quality, and therefore the potential impact on the biota, represents the "worst case scenario".
- According to Dallas and Janssens (1998), the application of SASS scores to historical data in the Biobase provides a crude means of ascertaining the degree of pollution. There are discrepancies in the way data were collected and recorded in the original papers on which the database is based. Some samples were collected from a single biotope, whereas others covered more than one. Some of the records represent one site visit and others a number of visits over several years. For this

reason the SASS scores in Biobase as well as those arising from the Biotic protocol, are approximate and should be referred to as "derived" or "theoretical" SASS scores (Dallas, FRU, UCT, *pers. comm.*).

- The situation might often occur where, because of lack of data, a significant water quality variable at a site is not modelled. Elevated loads of TSS, for example, are considered to have a considerable impact on riverine biota (Dallas and Day 1993), and yet measurement of this water quality constituent is limited in most rivers in the country. If no quantitative data are available for a site, but it is suspected that that variable is exerting a major impact, the effect can be incorporated qualitatively by adjusting the final Assessment class.

6.5 Application to the Lower Olifants River

The Biotic protocol was applied to IFR site 15 (Mamba) on the lower Olifants River, Mpumalanga. This site is on the section of river that was modelled using QUAL2E in Chapter 5 (Figure 5.1). The reasons for choosing this site were:

- Water quality is persistently poor in this reach of the river.
- The site is on the western (upstream) boundary of the Kruger National Park and is therefore ecologically important.
- Considerable progress in the River Health Programme has been made in Mpumalanga, and as a result, reasonably extensive biomonitoring data are available for this region in the Rivers database (Dallas, H., FRU, UCT. *pers. comm.*).

6.5.1 Results

The likely water quality consequences for the aquatic macroinvertebrates at IFR site 15 under the recommended flow regime are discussed below. In order to elucidate the method and to highlight some of the associated issues, the results are presented in the form of the sequential steps of the protocol.

i) Using water quality modelling, tabulate and identify the critical values for each significant water quality variable. This is carried out separately for;

a) the Reference Condition;

b) the Present Ecological State;

c) the Future Predicted State;

The results from water quality modelling (using the Q-C method) are summarised in Table 6.3. The Q-C plots for the water quality variables that were modelled at Mamba and that were used to derive the data in the above table, are given in Appendix B. Table 6.3 shows for each water quality variable, the minimum, median and maximum monthly concentration that could be expected during the hydrological year. This is reported as the simulated concentrations of each variable for the Reference Condition and Present Ecological State, and the predicted concentrations for the Future Predicted State. Two different flow regimes are included in the latter, namely the recommended maintenance baseflow and the drought baseflow. It is important to note that both the recommended maintenance and drought flow regimes represent an *increase* in discharge compared to the current discharge regime at that site and that water quality will be improved with either. The most improved water quality scenario would result from implementation of maintenance baseflow. Implementation of drought baseflow would result in considerably less improved water quality. It was decided that the implications to the biota of the drought baseflow should be investigated as this represented the "worst case scenario". Thus the effect of the following critical concentrations of chemical constituents (expressed in mg/litre) on the biota were investigated:

TDS = 1421

Sulphate = 586

Fluoride = 2.7

Total inorganic nitrogen (TIN) = 0.29

ortho-Phosphate = 0.07

Table 6.3 Minimum, median and maximum concentrations (mg/litre) of the water quality variables modelled at IFR 15 on the Olifants River, under the Reference Condition (natural), present day, and future (maintenance and drought) discharge regimes. Concentrations predicted using the Q-C method. The TWQR is indicated as well as whether the predicted maximum concentration will fall within this range (√) or will exceed it (X).

Variable	Flow regime	Predicted/measured concentration (mg/litre)			TWQR	Max within TWQR?
		Min.	**Median**	**Max.**		
TDS	Natural	212	266	300	≤ 15% deviation from Reference Condition	√
	Present	252	816	1541		X
	Maintenance	778	956	1074		X
	Drought	1167	1323	1421		X
Sulphate	Natural	-	-	-	Not in SA Guidelines	-
	Present	52	235	727		-
	Maintenance	217	288	343		-
	Drought	394	499	586		-
Fluoride	Natural	0.274	0.317	0.34	TWQR = 0.75 mg/litre CEV = 1.5 mg/litre AEV = 2.54 mg/litre	√
	Present	0.314	1.5	3.00		X
	Maintenance	1.37	1.45	2.00		X
	Drought	2.2	2.5	2.70		X
TIN	Natural	0.13	0.14	0.17	< 0.5 mg/litre = oligotrophic	√
	Present	0.275	0.28	0.32		√
	Maintenance	0.28	0.28	0.29		√
	Drought	0.28	0.28	0.29		√
ortho-P	Natural	0.016	0.017	0.02	≤ 15% deviation from Reference Condition 0.025-0.25 mg/litre = eutrophic	√
	Present	0.023	0.036	0.09		X
	Maintenance	0.034	0.041	0.05		X
	Drought	0.052	0.063	0.07		X

TDS =total dissolved solids, TIN =total inorganic nitrogen, ortho-P =ortho-phosphate. CEV =chronic effect value, AEV =acute effect value. TWQR =target water quality range as specified in the South African Water Quality Guidelines (DWAF 1996).

ii) Compare the critical values (Ccrit) with the Target water quality range and identify the two or three water quality variables likely to pose the most serious risk.

The Target water quality range (TWQR) for each of the chemical constituents, as recommended in the South African water quality guidelines (DWAF 1996) is also shown in Table 6.3. The final column of the table indicates whether or not the maximum value (the critical concentration) falls within the TWQR. It can be seen that of the five water quality variables modelled, only total inorganic nitrogen (TIN) would be within the target water quality range (TWQR) under the recommended discharge regime. Sulphate is not listed in the South African water quality guidelines and in addition, there were no suitable Reference Condition data available for Q-C modelling. It is suspected however, by comparing the values with other sites on the same system that SO_4^{-2} is considerably elevated at this site. The median Reference Condition values for sulphate in the middle and upper parts of the river ranged between 3 and 20 mg/litre. The maximum monthly concentration of sulphate that is currently recorded in the river at Mamba is over 700 mg/litre. Thus it is likely that this water quality constituent may be exerting a major impact on the biota at IFR site 15.

All of the other water quality variables that were modelled for Mamba were outside of the TWQR for the Present Ecological state. In addition, under both the recommended discharge regime for maintenance flow and drought flow, the TWQR would also not be attained. As a result, four of the water quality variables, namely; TDS, fluoride, ortho-phosphate and sulphate were chosen for further investigation. It was decided that only after viewing data available for relevant sites would the choice be made as to which were the most significant water quality variables.

iii) Compare ecotoxicological parameters, if available, with the critical values in order to estimate the toxicity of the variable in question.

No data could be found in the EPA Aquire database regarding ecotoxicological parameters for TDS, ortho-phosphate or sulphate, although limited data for fluoride were found. The LC_{50} for fluoride (i.e. the concentration resulting in a likelihood of death for 50% of a test population within 24 – 48 hours) was found to range from 26 to 680 mg/litre. This was derived from the data for *Daphnia* and three species of caddisfly. These concentrations are much higher than those predicted to occur at Mamba. A probability of death for 50% of the test population is, on the other hand, an extreme impact. No other useful ecotoxicological information could be found in this database.

The CEV and AEV for fluoride are reported in DWAF (1996) as 1.5 and 2.54 mg/litre respectively. The predicted critical concentration for fluoride (2.7 mg/litre) is thus beyond these values. The CEV (the chronic effect value) is defined by DWAF (1996), as the (upper) concentration limit that is safe for most populations even during continuous exposure. The AEV (acute effect value) is the concentration at, and above which, statistically significant acute adverse effects are expected to occur. These data show that fluoride at the concentrations currently present in the system, as well as under the recommended drought baseflow is likely to be exerting acutely toxic effects on the biota. It is therefore likely to be a major factor in determining the presence or absence of sensitive taxa in that reach of the river.

iv) Consult the Biobase and Rivers database for sampling data characteristic of the ***Reference Condition*** *for:*

a) The specific river in question.

b) Similar systems (i.e. in the same ecoregion and type of river).

A review of the macroinvertebrate taxa found under present conditions as well as historically at Mamba is given in Palmer (2000). The author cites eight SASS4 taxa that were historically found at Mamba (Moore and Chutter 1988), but that have not been found subsequently during the biomonitoring programme. These were Perlidae, *Hydra*, Heptageniidae, Pleidae, Nymphulidae, Haliplidae, Hydraenidae and Lymnaeidae. A comprehensive list of the taxa to be expected under Reference conditions is not given in Palmer (2000), however.

The Biobase and Rivers databases were then interrogated for sampling data characteristic of the Reference Condition by setting the theoretical SASS score equal to 140. The value of this SASS4 score used to identify Reference conditions was chosen after consideration of Table 6.1. Only sites in the lowveld were examined that were in the foothill, gravel-bed sub-region (Dallas, H., FRU, UCT *pers. comm.*). Two records were found (one in each database) that fitted this criterion, both from Mamba. These sampling occasions are indicated as records 6 and 13 respectively in Table 6.4. Mamba is designated as KNP06 in the Biobase and B7OLIF-Mamba in the Rivers database. The taxa that were found during these sampling occasions and that are considered to be representative of the Reference Condition are listed in Table 6.5. Note that despite a very high SASS score of 165 for record 13, the concentration of fluoride was relatively high (2.9 mg/litre). An examination of the taxa present on this sampling occasion (see further on) indicated that intolerant taxa were present. It is suspected that the chemical analysis for fluoride was inaccurate. Alternatively, sampling may have occurred at the beginning of an episode of high fluoride, before the biota had responded to the poor water quality.

v) Consult the Biobase and Rivers database for sampling data characteristic of the ***Future predicted state*** *for:*

a) the specific river in question;

b) similar systems (i.e. the same ecoregion, and type of river).

Three sites in the databases were found to fit the necessary criteria (i.e. in the same ecoregion, and the same type of river with similar water quality conditions) and were deemed to be suitable to use for infering biological information. The three sites were Mamba itself, Vygeboom and Balule. The last two sites are downstream of Mamba on the Olifants River and within the Kruger National Park (Figure 5.1). Like Mamba, they are both in the lowveld, and are representative of foothill gravel-bed rivers. Table 6.4 shows a summary of all the sampling data for the three sites, including the name of the site, the sampling date, as well as the concentrations of the four water quality variables at the time of sampling (or in the case of the Biobase, at approximately the time of sampling). The derived SASS score and ASPT are indicated, as are the biotopes that were sampled, and the database from which the information was extracted. Graphs were drawn of concentration versus derived SASS score for each of the chemical constituents in order to ascertain the correlation between the two factors. These graphs are shown in Figure 6.2. There is a negative trend between concentration and SASS score for TDS, fluoride and sulphate. The derived SASS value seemed to be largely independent of the concentration of ortho-phosphate however. This is not surprising since ortho-phosphate is not a toxin. In common with other nutrients, the major effect of elevated concentrations of these constituents on ecosystems, is to alter the trophic state rather than to exert a toxic effect. Thus, since the values of ortho-phosphate were not particularly high (see for example the values of phosphate in the Pienaars River, Appendix B), this variable was omitted from further considerations.

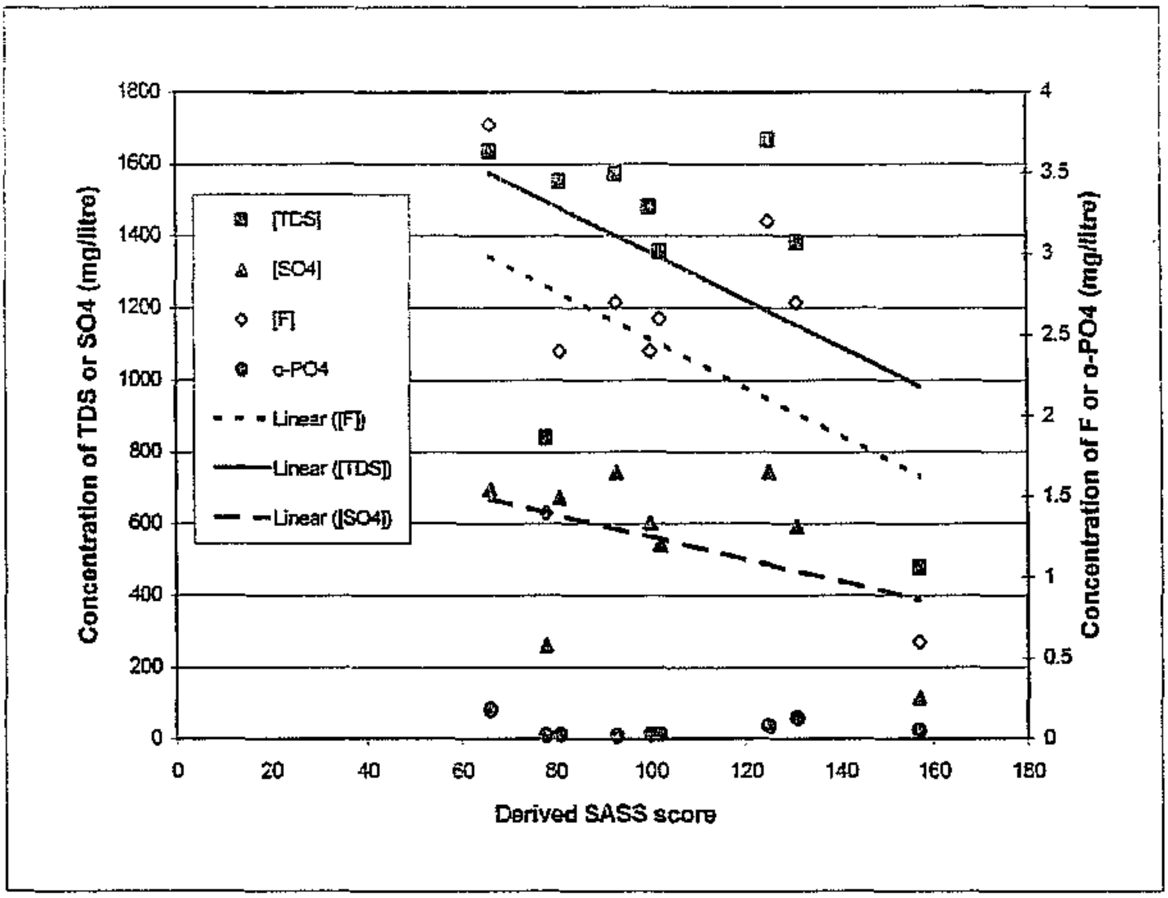


***Figure 6.2** Graph of concentration plotted against derived SASS score for the four water quality variables (TDS and SO_4^{-2} on the left Y-axis and fluoride and ortho-phosphate on the right Y-axis) considered to be the most significant at Mamba on the Olifants River.*

Table 6.4 *Summary of the sampling data from Mamba, Vygeboom and Balule stored in the Rivers database and the Biobase. Included are the site name, sampling date, and the concentrations of the significant water quality variables at the time of sampling. The derived SASS score and ASPT, the number of biotopes sampled, as well as the database from which the information was extracted, are given. Sampling occasions used to infer the taxa likely to be present in the Future Predicted State are indicated in bold.*

Record number	Site	Site visit	[F]	[TDS]	[SO4]	[ortho-P]	SASS4 Score	ASPT	No. of Families	No. biotopes sampled	Database
1	**B7OLIF -Mamba**	**7/6/93**	**2.7**	**1572**	**744**	**0.02**	**93**	**4.2**	**22**	**All biotopes**	RIVERS DATABASE
2	**B7OLIF -Mamba**	**29/7/93**	**2.7**	**1383**	**592**	**0.13**	**131**	**4.7**	**28**	**All biotopes**	
3	B7OLIF-Mamba	28/6/94	2.6	1518	660	0.06	-	-	-	All biotopes	
4	B7OLIF-Mamba	19/7/94	3.2	1667	745	0.08	125	5.2	24	All biotopes	
5	B7OLIF-Mamba	1/8/95	3.8	1637	696	0.18	66	5.1	13	All biotopes	
6	B7OLIF-Mamba	7/10/98	0.6	478	115	0.05	157	5.6	28	All biotopes	
7	B7OLIF-Vyge	8/10/98	0.6	448	-	0.02	73	4.9	15	All biotopes	
8	B7OLIF-Balule	6/6/93	1.4	841	262	0.022	78	7.8	15	All biotopes	
9	**B7OLIF -Balule**	**29/7/93**	**2.4**	**1551**	**672**	**0.025**	**81**	**4.8**	**17**	**All biotopes**	
10	**B7OLIF -Balule**	**20/7/94**	**2.6**	**1356**	**542**	**0.028**	**102**	**5.1**	**20**	**All biotopes**	
11	**B7OLIF -Balule**	**1/8/95**	**2.4**	**1481**	**601**	**0.025**	**100**	**5.6**	**18**	**All biotopes**	
12	B7OLIF-Balule	8/10/98	0.8	558	155	0.019	62	4.4	14	All biotopes	
13	KNP06 Mamba	1983-1986/10/03	2.9	225	11	0.05	165	6.1	27	3	BIOBASE
14	KNP06 Mamba	1983-1986/04/09	0.3	288	18	0.04	125	6.0	21	2	

ASPT=average score per taxon, Vyge = Vygeboom, The data in Table 6.4 were inspected for occasions on which the concentration of the water quality variables were equal to or higher than approximately 90%, but less than 110% of the critical value. Namely:

Ccrit TDS = 1420 (≈ 1560 –1280) mg/litre

Ccrit sulphate = 580 (≈ 525 –640) mg/litre

Ccrit fluoride = 2.7 (≈ 2.4 –3.0) mg/litre

The sampling occasions that were used to infer the taxa likely to be present in the Future Predicted State were records number 1,2, 9, 10 and 11. These are indicated in bold in Table 6.4. Records 4 and 5 were considered to representative of the present state. The data recorded in the Biobase were not found to be useful for deriving information with regard to the recommended discharge regime as the TDS and sulphate levels were too low. In addition, record 3 (from the Rivers database) was omitted because no biological data were collected on that sampling occasion.

vi) *Compare taxa lists for the Reference Condition and Future predicted state. Derive a theoretical SASS score and Assessment category for the Future Predicted State.*

Lists of the taxa present on the sampling occasions indicated in Table 6.4 were compiled. These lists are presented in Table 6.5, which shows the water quality (the concentration of fluoride, TDS and sulphate), the macroinvertebrate families that were present at the time of sampling, as well as the derived SASS4 score. The data are arranged from left to right in terms of increasing fluoride concentration. The two columns on the left-hand side, show the taxa expected under Reference conditions (Biotic protocol, step *iv*). Columns 3, 4 and 5 are similar to the critical value for TDS (1420 mg/litre), fluoride (2.7 mg/litre) and to that of sulphate (580 mg/litre) and were collected at Balule. Columns 6 and 7 are also representative of the Future Predicted State and were obtained from Mamba. Columns 8 and 9 represent the present state water quality at Mamba.

By comparing the taxa lists representative of the Reference Condition (columns 1 and 2) with those that represent the Present Ecological State (columns 8 and 9) and Future Predicted State, the taxa that have been lost from the system compared to the Reference Condition could be deduced (underlined in Table 6.5).

TABLE 6.5 Lists of invertebrate taxa and SASS scores for individual sampling occasions. Taxa that have been lost compared to the Reference condition are underlined. Taxa that may be regained in the system under the Future Predicted State shown in bold.

REFERENCE CONDITION		FUTURE PREDICTED STATE (UNDER DROUGHT BASEFLOW)					PRESENT STATE	
RECORD 13 (KNP06) 1983-1986/10/03	RECORD 6 (Mamba) 7/10/98	RECORD 11 (Balule) 1/8/95	RECORD 9 (Balule) 29/7/93	RECORD 10 (Balule) 20/7/94	RECORD 1 (Mamba) 7/6/93	RECORD 2 (Mamba) 29/7/93	RECORD 4 (Mamba) 19/7/94	RECORD 5 (Mamba) 1/8/95
F=2.9, TDS=225, S=11	F=0.6, TDS=470, S=115	F=2.4,TDS=1481, S=601	F=2.4, TDS=1551, S=672	F=2.6, TDS=1356, S=542	F=2.7, TDS=1570, S=744	F=2.7, TDS=1383, S=592	F=3.2, TDS=1667, S=74	F=3.8, TDS=1637, S=69
<u>AESHNIDAE</u>								
	ANCYLIDAE					ANCYLIDAE		
						ATHERICIDAE		
BAETIDAE 3 TYPES	BAETIDAE 1 TYPE	BAETIDAE 2 TYPES	BAETIDAE 1 TYPE	BAETIDAE 2 TYPES	BAETIDAE 1 TYPE	BAETIDAE 1 TYPE	BAETIDAE 2 TYPES	BAETIDAE 2 TYPES
	BELASTOMATIDAE		**BELASTOMATIDAE**		**BELASTOMATIDAE**	**BELASTOMATIDAE**		
CAENIDAE	CAENIDAE	CAENIDAE	CAENIDAE	CAENIDAE	CAENIDAE	CAENIDAE	CAENIDAE	CAENIDAE
CERATOPOGONIDAE	CERATOPOGONIDAE	**CERATOPOGONIDAE**		**CERATOPOGONIDAE**				
CHIRONOMIDAE	CHIRONOMIDAE	CHIRONOMIDAE	CHIRONOMIDAE	CHIRONOMIDAE	CHIRONOMIDAE	CHIRONOMIDAE	CHIRONOMIDAE	CHIRONOMIDAE
COENAGRIONIDAE	COENAGRIONIDAE	COENAGRIONIDAE	COENAGRIONIDAE	COENAGRIONIDAE	COENAGRIONIDAE	COENAGRIONIDAE	COENAGRIONIDAE	
CORDULIIDAE			CORDULIIDAE	CORDULIIDAE	CORDULIIDAE	CORDULIIDAE	CORDULIIDAE	
CORIXIDAE	CORIXIDAE	CORIXIDAE			CORIXIDAE	CORIXIDAE	CORIXIDAE	
			CULICIDAE	CULICIDAE	CULICIDAE	CULICIDAE	CULICIDAE	
							ELMIDAE/DRYOPIDAE	
DYTISCIDAE			DYTISCIDAE		DYTISCIDAE	DYTISCIDAE		
ECNOMIDAE			ECNOMIDAE					
ELMIDAE/DRYOPIDAE	ELMIDAE/DRYOPIDAE			ELMIDAE/DRYOPIDAE				
				GERRIDAE	**GERRIDAE**	**GERRIDAE**		
GOMPHIDAE	GOMPHIDAE	GOMPHIDAE	GOMPHIDAE	GOMPHIDAE	GOMPHIDAE	GOMPHIDAE	GOMPHIDAE	GOMPHIDAE
GYRINIDAE				**GYRINIDAE**	**GYRINIDAE**			
HIRUDINEA	HIRUDINEA	HIRUDINEA		HIRUDINEA	HIRUDINEA	HIRUDINEA	HIRUDINEA	HIRUDINEA
HYDROPHILIDAE		HYDRAENIDAE	HYDROPHILIDAE			HYDRACHNELLAE	HYDRACHNELLAE	
						HYDROPHILIDAE		
<u>HYDROPSYCHIDAE 3 TYPES</u>	<u>HYDROPSYCHIDAE 3 TYPES</u>	HYDROPSYCHIDAE 2 TYPES	HYDROPSYCHIDAE 1 TYPE		HYDROPSYCHIDAE 1 TYPE	HYDROPSYCHIDAE 1 TYPE	HYDROPSYCHIDAE 1 TYPE	HYDROPSYCHIDAE 2 TYPES
		LEPTOPHLEBIIDAE (pH>6.6)						
HYDROPTILIDAE	HYDROPTILIDAE							HYDROPTILIDAE
<u>HYDROZOA</u>								
	LEPTOPHLEBIIDAE (pH>6.6)						LEPTOPHLEBIIDAE (pH>6.6)	LEPTOPHLEBIIDAE (pH>6.6)
	LIBELLULIDAE	LIBELLULIDAE	LIBELLULIDAE	LIBELLULIDAE	LIBELLULIDAE	LIBELLULIDAE	LIBELLULIDAE	LIBELLULIDAE
	MELANIIDAE				MELANIIDAE	MELANIIDAE	MELANIIDAE	
	MUSCIDAE			NAUCORIDAE		MUSCIDAE		
	<u>NANTANTIA (SHRIMPS)</u>							
NAUCORIDAE	NAUCORIDAE					NAUCORIDAE	NAUCORIDAE	
	NEPIDAE		NEPIDAE		NEPIDAE	NEPIDAE	NEPIDAE	
	NOTONECTIDAE			**NOTONECTIDAE**	**NOTONECTIDAE**	**NOTONECTIDAE**		
	<u>OLIGONEURIDAE</u>							
OLIGOCHAETA		OLIGOCHAETA		OLIGOCHAETA			OLIGOCHAETA	OLIGOCHAETA
	PHYSIDAE				**PHYSIDAE**	**PHYSIDAE**		
<u>PLANARIIDAE</u>	<u>PLANARIIDAE</u>							
<u>PLANORBIDAE</u>								
SIMULIIDAE	SIMULIIDAE	SIMULIIDAE		SIMULIIDAE	SIMULIIDAE		SIMULIIDAE	SIMULIIDAE
	SPHAERIIDAE	SPHAERIIDAE		SPHAERIIDAE		SPHAERIIDAE	SPHAERIIDAE	SPHAERIIDAE
TABANIDAE	TABANIDAE	TABANIDAE	TABANIDAE	TABANIDAE	TABANIDAE	TABANIDAE	TABANIDAE	TABANIDAE
<u>THIORIDAE</u>								
TRICHOPTERA (CASE CADDIS 2 TYPES)		TRICHOPTERA (CASE CADDIS 2 TYPES)	TRICHOPTERA (CASE CADDIS 1 TYPE)	TRICHOPTERA (CASE CADDIS 2 TYPES)	TRICHOPTERA (CASE CADDIS 1 TYPE)	TRICHOPTERA (CASE CADDIS 1 TYPE)	TRICHOPTERA (CASE CADDIS 1 TYPE)	
TRICHORYTHIDAE	TRICHORYTHIDAE						TRICORYTHIDAE	
	VELIIDAE	VELIIDAE	VELIIDAE			VELIIDAE	VELIIDAE	
SASS 185	157	100	81	102	93	131	125	66

F=fluoride, TDS = total dissolved solids, S= sulphate

These include:
Aeshnidae, one type of Hydropsychidae, Hydrozoa, Nantantia, Oligoneuridae, Planariidae, Planorbidae, and Thioridae.

A comparison of taxa present during conditions representing the Future Predicted State (columns 3-7) with those present during the poorest water quality (columns 8 and 9) was used to infer the taxa that would likely be regained in the system under the improved water quality scenario that should result from the recommended flow regime. These are shown in bold in Table 6.5. Taxa that may be *regained* in the system under the improved water quality scenario include:
Belastomatidae, Ceratopogonidae, Dytiscidae, Gerridae, Gyrinidae, Notonectidae and Physidae.

A tentative derived SASS score was for the Future predicted state was estimated to be about 100. This value was obtained by taking the median of the SASS scores for columns 3 -7. From table 6.2, this would indicate a future Assessment class of "C/D".

vii) Include input from any other biotic tolerance indices that may be relevant e.g. the Fish Index (Kleynhans, 1999).
This step was not attempted in this application of the protocol.

viii) Synthesize a scenario for the aquatic biota that is likely to be the consequence of the proposed change in discharge. Assign the future Ecological Reserve class (A-F).
It was considered by the project team that the predicted improvement in Assessment Category from "D" (Palmer 2000) to "C/D" was too conservative. This arises from the fact that the proposed flow regime represents an increase in the present-day discharge and that the critical concentrations were taken from those expected under the drought baseflow. The latter is likely to be imposed for less than 20% of the time at the site, and baseflow represents the worst-case scenario (i.e. the minimum volume of water that would be allowed in the system). In addition, the position of the site and potential recolonisation was taken into account. The poor quality water originates to a large extent from the Selati River, which is a few kilometres upstream from Mamba. Water quality in the mainstem Olifants River is considerably better than at Mamba and thus faunal populations exist that would be likely to recolonise Mamba, should the conditions there

improve. Thus it is considered that under the recommended flow regime, the likely future assessment class will be "C".

6.5.2 Points arising from the application

- The choice as to which water quality variables can be considered to be most significant is not always easy. At some sites the lack of data will preclude Q-C modelling for some variables and in any case this type of modelling cannot be used for dissolved oxygen or temperature. These two water quality variables are likely to be very important with regard to impact on the biota. Application of the Biotic protocol to Mamba was aided by the fact that usually when fluoride levels were high, the concentrations of TDS and sulphate were also high. This might not always be the case, however, if for example there were two or more upstream point source of pollutants, discharging effluent at different times. In such situations, when the level of one pollutant was high, the other might be low. This would make using the protocol considerably more difficult.
- Ecotoxicological data that are available in databases are not always easy to interpret, and if possible, a toxicologist should be consulted.
- As far as possible, given the constraints of the limited data, sampling data from the same time of the year were compared. This is important since the composition of taxa (and thus the SASS score) will vary during the year, the highest scores being recorded during spring (Dallas 2000).
- There is a danger in using data that are representative of Future Predicted State and the Present state from the same site, unless these data sets are well separated in time. For example, due to previous poor water quality, record 7, used to help derive the taxa characteristic of the Reference Condition may not have contained all the taxa that would normally be present under non-impacted conditions.
- An interesting observation is that records 7 and 12 (at Vygeboom and Balule respectively) indicate low derived SASS scores and yet the concentrations of TDS, and fluoride are also relatively low. There are several possible reasons for this. Firstly sulphate concentration for record 12 is relatively high, and this may have resulted in the very low SASS figure of 62. The sulphate concentration was not determined in record 7. The low SASS score in record 12 may have been the result of a previous period of very poor water quality from which on 8/10/98 the biota had not yet recovered. The poor condition of macroinvertebrates at Vygeboom on

8/10/98 is not so easily explained since the SASS score for the previous day at Mamba (upstream of Vygeboom) was extremely high (157). This may well be a consequence of a tributary carrying poor quality water from the mining operations at Phalaborwa, which joins the Olifants River downstream of Mamba. Because of the anomalous derived SASS scores and water quality values these two sites were not used to infer biological information during this application. Thus it is possible that (currently unmonitored) toxins, other than fluoride and sulphate, may be exerting a deleterious effect in the lower reaches of the Olifants River.

6.6 Application to the Middle Olifants River

The Biotic protocol was also applied to a site on the middle Olifants River. This was carried out as part of an Honours degree at UCT by David Smith, and took the form of a critical appraisal of the protocol. It had been hoped that a river, other than the Olifants, could be used to apply the method, but unfortunately this was not possible due to a lack of biomonitoring data in the Palmiet and Breede Rivers. The study examined an application of the protocol to IFR 5, which is downstream of the Loskop dam. It was difficult to find clear differences in the assemblage of predicted macroinvertebrates compared to those presently in the system. This is likely to be due to the fact that there is only a small difference in predicted water quality, compared to the Present Ecological State. The project therefore focused on the development of statistical techniques to aid both in application of the protocol as well as interpretation of results obtained. These techniques included the use of a Water Quality Index, as well as multi-dimensional scaling plots of invertebrate assemblages. The work that was carried out is preliminary and consequently the results are not presented in this report.

6.7 Potential developments to the Biotic protocol

There are several aspects of the protocol that could be expanded or improved.

- Given the development of suitable tools (namely databases relating distribution and water quality) it should be possible to extend the protocol to make likely predictions of altered water quality on fish and possibly on riparian vegetation.

- One of the major problems that was encountered in applying the Biotic protocol was in obtaining the integrated effect of more than one water quality variable on the biota. In the application of the protocol to Mamba on the Olifants River, water quality scenarios were primarily ranked according to the measured fluoride concentration. It might be instructive to investigate the use of water quality indices. Using a water quality index the integrated impact from more than one water quality variable could be determined.
- In many situations, due to a lack of data, or of sophisticated water quality modelling, detailed assessments of some water quality variables may be lacking. In such cases it may be possible to draw up some "rules of thumb" to account for synergistic and antagonistic effects between water quality variables.

  For example;

  - a combination of high predicted NH_3 concentrations in combination with high pH should be regarded as serious.

  - a combination of high predicted concentrations of a metal pollutant in combination with low pH should be regarded as serious.

  - similarly, predicted conditions of high water temperature accompanied by low levels of dissolved oxygen.

  - a situation where high levels of nutrients are predicted might be mitigated against if this is likely to occur in combination with high TSS loads and high discharge (i.e. if the risk of eutrophication is reduced).

  This type of information could be used to modify the final Assessment Class, or could be included as comments in the compilation of the predicted water quality scenario.
- Average score per taxon (ASPT) is considered to be a more reliable assessment of water quality than SASS scores (Dallas 1997). The reason for this is because it excludes variability between SASS scores obtained from more than one site that originate from differences in the availability of distinct habitats. In the application to the Olifants River, however, ASPT was not found to be particularly useful. This was largely because the values of this parameter did not appear to change markedly with altered water quality. This aspect requires further investigation.
- More work needs to be done to check the SASS4 score and ASPT value that are used to identify the Reference Condition. Specifically scores for ecoregions/bioregions other than the Western Cape need to be derived or confirmed.

6.8 Conclusion

The Biotic protocol offers a framework within which, given the development of appropriate tools, it should be possible to make predictions of the effect of altered water quality on other biotic components such as fish or riparian vegetation. Aquatic ecosytems are complex, however, and many interactions between biotic and abiotic components occur, some of which are known and can possibly be quantified, and others of which are totally unknown. In addition, aquatic ecosystems are highly variable, and different responses to stressors are possible under different circumstances. Thus large databases are required in order to make reliable predictions of biotic responses to changes in water quality impacts. The initiation of the River Health Programme and development of the Rivers database is an invaluable opportunity to obtain more data linking water quality and biotic response. The use of this protocol has highlighted the importance of co-ordinating the collection of discharge data, but especially of water quality and biomonitoring data. To make the most effective use of limited resources and to optimize the benefits of the River Health Programme, water quality and biomonitoring data should be collected at the same time and in the same place so that the relationships between water quality and biotic response can be elucidated.

CHAPTER 7

INCORPORATION OF PREDICTIONS OF WATER QUALITY AND BIOTIC RESPONSE INTO RESERVE ASSESSMENTS

7.1 Introduction

The tools developed and used in this project to predict water quality or biotic response can be employed in many different situations. Mass balance modelling can be used, for instance to calculate the instream concentration of a given chemical constituent if the pollutant loading of the system were to be altered. The Biotic Protocol could be used to estimate the potential effect of such a management change on aquatic organisms. One of the major uses of these tools, is nevertheless, within Reserve determinations and for this reason integration of the tools into this process is discussed in detail.

Because instream concentrations of chemical constituents and values of physical variables frequently change in response to altered discharge, it is necessary to ensure that in setting the ecological Reserve with regard to the quantity and timing of discharge, that the Resource Quality Objectives (RQOs) with regard to water quality will also be met. In order to do this, quantitative predictions of the concentrations of chemical constituents and values of physical variables that can be expected for a given discharge need to be made. In other words, some form of water quality modelling is required. Prior to this project, only qualitative predictions of water quality were usually made during environmental flow determinations in this country (Palmer, Malan and Day 2000). The situation is frequently encountered in which the recommended discharge for a given reach of river is reduced compared to current flow regime. Although it can be predicted that the instream concentrations of chemicals arising from a point source discharging into that reach are likely to increase, the exact amount cannot be ascertained without some form of modelling.

In this chapter, the process currently followed in South Africa when determining the environmental flow requirement of a river reach is outlined. This is followed by a brief discussion of how each of the tools developed or used in the project should fit into the process, as well as what information and benefit each could provide. Also included is a consideration of the circumstances under which each tool should or should not be used. Finally a framework is presented for the integration of water quality modelling, as well as prediction of likely impacts on aquatic biota, into Reserve assessments for South African rivers.

7.2 Outline of an Environmental Flow Assessment

An Environmental flow assessment (frequently termed "Instream flow assessment" in South Africa) can be defined as "an assessment of how much of the original flow regime of a river should continue to flow down it in order to maintain specified valued features of the river ecosystem" (King *et al.* 1998). The riverine features of concern may be as specific as a species of fish or tree that should be retained in the system. Alternatively, the goal may be to rehabilitate a given river reach from, for example, an "E" Ecological Reserve category to a "C". An Environmental flow assessment (EFA) is then carried out in order to obtain an estimate of how much water can be abstracted for a specified level of impact on the riverine ecosystem.

To put this chapter into context a brief outline of the environmental flow determination process is given below. A more detailed discussion is given in Chapter 5 of the literature review to this project (Malan and Day 2002). It was noted in Chapter 1 that the South African National Water Act (No. 36 of 1998) requires that the ecological Reserve be determined for all significant water bodies. The currently accepted method in South Africa for a comprehensive determination of the water quantity component of the reserve is the Building Block Methodology (BBM), which represents the minimum procedural requirement (DWAF 1999). DRIFT (Downstream Response to Imposed Flow Transformations) is a new developing methodology for determination of environmental flow requirements in South Africa (Brown and King 2000; King, Brown and Sabet *in press*). It is not currently in official use as part of the Reserve Determination Process, but was applied in parallel with the BBM during the EFA for the Breede River Basin study. An outline of both methodologies is given below.

7.2.1 The Building Block Methodology

This outline is largely taken from King and Louw (1998), Tharme and King (1998) and from the BBM manual (King, Tharme and de Villiers 2000).

The BBM is based on the assumption that the natural flow regime of a river can be broken up into different kinds (i.e. differing in magnitude, timing, frequency and duration) of flow. The different kinds of flow are the building blocks of the new hydrological regime and each are considered to exert differing impacts on ecological functioning. A modified, reduced flow regime can be constructed, representing the EFR, which incorporates the flows that are perceived to be essential or important and that are characteristic of the natural hydrological regime. Water that is not perceived to play a specific role (such as that required for the shifting of any sediments, provision of enough depth for fish, or floods required to act as environmental cues for fish spawning etc.) is potentially available for abstraction. Decision as to which flows are important is made in a workshop attended by a group of experienced river scientists. The scientists involved usually include those knowledgeable in the fields of freshwater fish, aquatic invertebrates, riparian vegetation, riverine habitat integrity, fluvial geomorphology, hydraulics, hydrology and water quality. In addition, if the EFA is conducted in response to the proposed development of a water resource, specialists experienced in management of such projects are also frequently included. Prior to the workshop, a document is produced by the specialists which describes the present state (and usually the Reference condition) of various aspects of the river system.

Financial constraints frequently limit the collection of new data, one exception to this however is the surveying of cross-sectional profiles at specific BBM sites along the river. These sites are identified within the study area and each has an EFR described for it. The hydraulic engineer surveys each site throughout the hydrological year (i.e. during periods of both low and high discharge) and measures various parameters such as water height, velocity, river width etc. From analysis of the data and examination of the riparian vegetation, cross-sections of each BBM site are produced. Using such diagrams, river scientists are able to calculate the relationship between discharge and the water depth that is required for a specific ecological feature e.g. to keep water moving over riffle areas during the dry season. Motivations from each discipline are made and the flow requirements for maintenance years (i.e. when "normal" rainfall occurs) and those for drought years are recorded for each month. Floods during normal and drought years are specified in volume, frequency, timing and duration. The prescribed flow regime is used by DWAF to calculate the amount of water available for use in the catchment and in the planning of water development schemes (but see section 7.2.3, below).

7.2.2 <u>The DRIFT Methodology</u>

Downstream Response to Imposed Flow Transformation (DRIFT) is a holistic environmental flow methodology that has developed out of the BBM. It was largely refined during the Lesotho Highlands Water Project (Brown and King 2000). DRIFT also involves groups of riverine specialists in a workshop situation, but according to the authors, DRIFT differs from the BBM in the following ways:

- The BBM constructs a recommended flow regime from nothing, whereas DRIFT uses present-state hydrology as the starting point and then describes the consequences for the river of further reducing (or, if relevant, increasing) the flow at different times of the year.
- One of the principle products of a DRIFT assessment is a database that can be queried to produce the biophysical consequences of a wide range of prescribed flow regimes and management options (scenarios). On the other hand, the BBM requires identification of a single predetermined condition and a single flow regime is recommended to facilitate maintenance of that condition. In the most recent applications of the BBM (in Comprehensive Reserve determinations), however, flows and their associated consequences have been designed to comply with a specific future ecological Reserve category (A – D). Using a Decision Support System (Hughes and Münster 1999), these results can be extrapolated to give low confidence estimates of flows for any other class that is required.
- DRIFT was specifically developed in order to link the biophysical consequences of altered discharge regimes and the social impact for the population at risk along the river. Although social importance can be included in the BBM the links between

change in river flow and economic costs of mitigation and compensation for loss of river services are not well defined.

7.2.3 Post- workshop developments

Prior to 1997, the water quantity aspect of the BBM ended with the drawing up of a recommended flow regime table that was comprised of monthly discharge values for maintenance and drought years. In addition, the required floods during normal and drought years were specified in terms of volume, frequency, timing and duration. According to Louw, Hughes and Birkhead (2000) it became increasingly apparent that the IFR process could not end at this point. Only after determination of the Reserve could the "excess" water that was available for other users (termed the system yield) be calculated. The prescribed IFR flow regime as represented by the values in the table was found to be incompatible with the Water Resources Yield Model (WRYM) which is used by DWAF to determine yield. There were two main reasons for the incompatibility. Firstly, there was no indication in the output from the BBM of the percentage time the system should be at drought, how often flows should be intermediate between drought and maintenance, and in wet years, how often flows should be above maintenance requirement. This type of information, in addition to the recommended flow regime, is now determined by the specialists at the IFR workshop. Secondly, there was no indication in the recommended flow regime as to when droughts should occur and when floods should be released. Louw *et al*. (2000) go on to say that to overcome the two above problems the IFR model has been developed. This model uses the output of the IFR workshop, is linked, via a reference time-series to climatic cues in the catchment, and makes use of a set of rules defined by the specialists which specify when, and for how long, low flow and flood events should occur. The output from the IFR model is a flow time-series that, for each calendar month, specifies the percentage of time that the modified flow regime is at or above maintenance, between maintenance and drought, and at drought. This is summarised in the form of flow duration curves for each month. The next step is for the yield modeller to assess the impact of supplying the Reserve on the yield and on the existing requirements on the system. If both the Reserve and the existing users cannot be met, often by making slight adjustments to the flow regime (i.e. by generating different discharge scenarios), all demands on the system can be satisfied. A Scenario workshop is held in which the ecological specialists assess the environmental impacts of the various flow scenarios suggested by the yield modeller, and rank them according to the perceived severity for the aquatic biota.

7.3 Natural versus anthropogenic water quality impacts

Although it has long been recognised that water quality should be addressed within the IFR process, how this should take place has only recently become clear. Current thinking is that cognisance should be take of any present day impairment to water quality (since this is likely to affect what species of fish and invertebrates are currently present in the system). In addition, predictions are required as to the concentration of chemical components and values of physical variables under the new, proposed flow regime. When setting the environmental flows, however, the discharge specified for any one month should not be set higher than required by the biota and fluvial geomorphology in order to provide dilution flows (Tharme and King 1998). Poor water quality should rather be ameliorated by the control of pollutants at source. If "additional" volume is utilised to dilute problematic constituents this should not be considered to be part of the IFR or the water quantity Reserve of that river. It is a fact of life that many South African rivers are impacted due to high levels of pollutants. Thus, if point and diffuse sources of pollution are not addressed and implementation of IFR regimes leads to a reduction in base flow, deterioration of water quality is a likely result. The current approach therefore is to predict the likely *water quality consequences* of the recommended flow regime in the absence of pollution control (Palmer and Rossouw 2000).

In the context of IFR determinations it is important to distinguish between "natural" and "anthropogenic " water quality problems. Natural water quality problems would include instances where due to the geology of the surrounding catchment, water draining that region is naturally saline resulting in elevated concentrations of salts in the river. The aquatic biota in such rivers are adapted to high salinity levels. Implementation of an IFR flow regime for which the maintenance and drought flows represent a reduced discharge volume compared to natural, may well result in unacceptably high levels of TDS, however. In such cases, incorporation of dilution flows into the flow requirement (the water quantity Reserve) would be necessary and acceptable. As an example, the lower reaches of the Swartkops River (Eastern Cape Province) exhibit naturally high salinity values due to the underlying geology being derived from old marine sediments. Under natural hydrological conditions, salinity levels are not excessively high in these reaches.

If the flow regime were to be substantially reduced on the other hand, salinity might be expected to reach concentrations that would be stressful to some invertebrate species (N. Rossouw, Ninham Shand, Cape Town, *pers. comm.* 2000). In such a situation, the water quantity reserve as determined from consideration of the hydraulic requirements of the biota and geomorphology, might need adjustment in order to attain acceptable water quality.

7.4 Water quality modelling and the Reserve determination process

The proposed role in Reserve assessments of each the water quality models is discussed below.

7.4.1 Discharge-concentration (Q-C) modelling

It is proposed that Q-C modelling be undertaken for each IFR site (for which data are available), as part of an Intermediate or Comprehensive Reserve determination. This method is the lowest tier of recommended models (section 1.8.3). It should be used to screen sites in order to assess whether the water quality component of the Reserve would be attained under the proposed maintenance and drought baseflows. It should be kept in mind that the Q-C method gives only estimates of predicted concentrations and that predictions are for baseflows – the lowest recommended flows, and thus represent the "worst-case scenario" with regard to water quality.

Using concentration-flow modelling, depending on the availability and reliability of data at each IFR site, the following information can be obtained:

- flow-concentration relationships for some key water quality variables;
- estimates as to how many months of the year the water quality component of the Reserve would be attained, as well as the likely Assessment Category (A, B, C etc.), for each water quality constituent of concern;
- in what month the worst water quality would be likely to occur and what concentrations could be expected;
- the extent of deviation of predicted concentrations from those specified under the Reference Condition.

- what discharge, in the absence of pollution control, would be required to dilute pollutants in order to attain the Resource Quality Objectives;
- In the case of "natural water quality problems", what discharge would be needed to dilute the chemical constituents in order to attain the water quality component of the Reserve. Note though, that in the Reserve Assessments that were examined during this project, no cases of "natural water quality problems" were considered to occur.

The degree of confidence in the accuracy of the simulations for each IFR site can be assessed by taking the following factors into consideration:

- the completeness of the data-set used to assess Reference Condition (RC) water quality for each chemical constituent, as well as an assessment of how representative that data-set is of the natural state;
- the completeness of the data-set for each water quality variable that is used to assess water quality for the Present Ecological State (PES);
- how representative the water quality data (both RC and PES) are of the IFR site under consideration. This depends largely on how close the monitoring station used as data source is to the IFR site, and if a hydrological feature (e.g. minor tributary, weir etc.) is situated between the IFR site and data source site;
- the reliability of the discharge data. This can be assessed by consultation with the hydrologist for the project;
- the accuracy of the water quality simulations. This is indicated by the correlation coefficient between measured and predicted values.

Predictions of water quality should not be made using Q-C modelling:

- if the available PES water quality data do not satisfy the requirements as laid out in the Resource Directed Measures manual (DWAF 1999). In other words, a minimum of 60 water quality data points is required covering the entire hydrological year. In the absence of RC data, predictions of future water quality can still be made, but the extent of deviation from natural can not be assessed;
- if the nearest water quality monitoring station to the IFR site is in a different water quality reach from the site. This may be either because of the distance between the two is too great, or because a significant hydrological feature, e.g. large tributary, major point source of pollution or impoundment is situated between the monitoring station and the IFR site. In other words if no data are available that are considered

to be representative of the water quality at the IFR site, modelling should not be carried out;

- if accurate present day discharge data for the water quality reach under consideration are not available;
- if the correlation coefficient between measured and predicted values is less than 0.7, the simulations should be discarded. It can be concluded in such cases that factors other than discharge are influencing instream concentrations. Predictions of water quality in such cases should be made using QUAL2E (or possibly a catchment run-off model);
- if the concentration of the water quality variable exhibits a marked *increase* in concentration with increasing discharge (section 2.11.1) and if this is likely to be due to wash-off from the surrounding land. In a few cases, positive Q-C trends may be a result of the release of constituents (especially nutrients) from instream sources, such as sediments. If this can be verified, then predictions of concentrations can be made using the Q-C method.

7.4.2 Mass balance modelling and QUAL2E

Mass balance modelling or QUAL2E should be used for IFR sites where initial screening using Q-C plots has indicated that, under the recommended discharge regime, the water quality component of the Reserve may not be attained during all months. A Mass balance model or QUAL2E should be set up for such sites (or reaches of river) and used to assess the implications of different water quality management scenarios. Thus the effect of different sources and loading of pollutants on instream water quality should be examined. These results would be employed in the Flow scenario phase (Step F, Table 7.1). A consideration of different water quality management scenarios is not currently incorporated into implementation of Reserve determinations, but it is a strong recommendation by the project team that they should be.

Mass balance modelling can be used in the form of the generic equation if the necessary data on *inter alia* percentage return flows are available (section 2.7.1). If such data are not available, different water quality management scenarios (i.e. different pollution loadings) can be assessed using simple mass balance calculations (see example in section 5.3.4.1). Because Mass balance modelling is not suitable for non-conservative constituents, it should be used in situations in which:

- conservative chemical constituents are of importance;
- simple situations are to be considered (i.e. where only two to five well-defined sources of pollutants impact on the reach under consideration).

QUAL2E is able to model conservative and non-conservative constituents but requires extensive data (section 5.4.1.3). This model, therefore, should only be set up as part of a Comprehensive Reserve determination in which a field sampling programme is included. In addition it can be used:

- if non-conservative chemical constituents (nutrients, dissolved oxygen, temperature, chlorophyll) are of concern;
- if complex situations are to be considered (i.e. multiple sources and sinks of pollutants are present which impact on the reach under consideration);
- if sufficient expertise and finance are available to set up the model.

7.4.3 Concentration time-series modelling

Concentration time-series modelling should be employed when different flow scenarios need to be compared with regard to the effect on instream water quality, and the discharge data are in the form of time-series (i.e. generated by the IFR model or the Yield model, section 7.2.3). Because of the major approximations and assumptions that are made in the preparation of the concentration time-series they should not be used to make quantitative predictions. Thus, for example, the precise percentage of the time that a given water quality constituent will be in each Assessment Category under a particular flow scenario, should not be determined using this method. This technique should rather be used to compare and rank flow time-series in terms of the water quality that will result from each. Concentration time-series modelling can be used in the IFR workshop (Step C, Table 7.1) to compare the concentration duration curves arising from natural, from present-day and from the recommended flow regimes. It can also be used at the Flow Scenario workshop (Step F) to compare the water quality consequences of different flow regimes generated by the Yield model (WRYM). Concentration time-series modelling should be carried out:

- for key IFR sites (i.e. where there is a likelihood that the water quality component of the Reserve may not be attained);
- for the water quality variable that is likely to be the most influential in terms of potential impacts on the biota;
- during either, or both, the IFR workshop and the Scenario modelling phase.

Concentration time-series modelling should not be undertaken:

- for sites at which there is a low confidence in the Q-C model (e.g. a correlation coefficient <0.7, or a positive correlation between constituent concentration and discharge, or poor representivity of data);
- if different water quality management options are considered. The method has the potential to be useful in such situations but needs more research before it can be used to obtain accurate results.

7.4.4 Catchment run-off models

Catchment run-off models represent the third and most sophisticated tier of water quality modelling that was suggested (section 1.8.3). Such models were not examined in this project and thus details cannot be specified as to exactly which models should be used, and in which situations. The project team recommends, however, that a suitable catchment run-off model be set-up and used for key catchments, (i.e. those in which intense demands are made in terms of excessive use of water and release of pollutants) and where non-point source pollutants are an important issue. This would include catchments where a large proportion of land-use is given over to informal settlements or to agriculture. In section 2.11.1.1, a site on the Breede River, Western Cape was discussed. Due to salinisation of the surrounding soil, TDS increases with increasing discharge, at least in the low-flow portion of the discharge range. Discharge-concentration modelling cannot be used in such situations to make reliable predictions of the concentration of TDS that would result from a given discharge regime. In order to make such predictions, detailed knowledge would be required as to how that modified flow regime would be brought about, exactly what flow modifiers (e.g. impoundment, instream abstraction) would be employed and how they would be operated, and how the generation of run-off (and thus of diffuse pollution loading) would be affected. Because of the distributed nature of the pollution, mass balance modelling or QUAL2E would not be very successful in predicting realistic water quality and thus a catchment run-off model is required. Catchment run-off models require extensive data, expertise and can take a considerable time to set-up and thus should only be employed in specialised cases.

7.5 General considerations on water quality modelling and Reserve assessments

The following points should be noted when incorporating water quality modelling into Reserve assessments.

- During the BBM, baseflows are specified for each month of the year. This is for both maintenance years (when the normal rainfall and hence streamflow is expected) as well as for drought years. Using water quality modelling (Q-C, mass balance, QUAL2E), the concentration of each water quality variable can be predicted for each month under the prescribed regime. Baseflows represent the lowest flow allowed for a given month at that site. In the case of TDS, sulphate and other chemical constituents, which usually decrease in concentration with increased discharge, the predictions from concentration-flow modelling, therefore, represent the "worst case scenario".
- Due to stratification in deep impoundments, the drawoff level can exert a profound effect on the water quality of releases downstream. Thus it would sometimes be necessary when carrying out water quality modelling to first model releases from a dam. This would then be followed by instream modelling of the reaches below the dam wall.
- There is increasing recognition that manipulation of the timing of effluent discharge may represent a useful management option (Grayson and Doolan 1994). If effluents can be stored they can be released during periods of high flow so that although the total annual load is not affected, concentrations in the river during low flows are reduced with potential ecological benefits. In order to do this and at the same time to ensure that the water quality Reserve is not exceeded, some form of mass balance modelling would be necessary.
- The results of water quality modelling pertain to the entire reach in which a given IFR site is located. In the IFR workshop, descriptive predictions can also be made which are applicable to a specific IFR site. For example, a situation might occur where nutrient concentrations are relatively high and the cross-section for a given IFR site

indicates that there would be areas of non-flowing water during the dry season. The water quality specialist could then predict that there was a likelihood of algal blooms in back-waters during such periods. An examination of the water velocity at individual IFR sites can also give insight into movement of sediments and impact on water quality. Further examples of such qualitative predictions for the localized areas around individual IFR sites are described in Palmer, Malan and Day (2000).

7.6 Predictions of impacts on the aquatic biota and the Reserve determination process

7.6.1 The Biotic Protocol

The Biotic protocol represents a structured manner in which all relevant information is collated in order to make an assessment of the likely effects a proposed flow regime, and the resulting water quality scenario, will have on the aquatic biota. The products arising from an application of the protocol include lists of invertebrate taxa that are likely to be lost (or regained) in the system under the proposed flow regime, predicted theoretical SASS scores and the predicted Assessment Category. Thus the protocol can be employed to check that at key IFR sites the recommended flow regime will result in the required Resource Quality Objectives (RQO) with regard to macroinvertebrates. A flow regime must first be recommended by the specialists at the workshop, before the Biotic protocol can be applied. Because of the length of time it requires to apply the protocol and because of the different data sources that need to be accessed, in practice, when using the BBM to determine the flow requirement, the protocol can only be applied after the workshop. This is indicated in Table 7.1, where the protocol is shown as Step D. If the environmental flow requirement is determined using DRIFT on the other hand, the consequences of successive flow reductions are recorded. If the proposed flow reduction scenarios are known before the specialist workshop, both Q-C modelling as well as the Biotic protocol can be applied prior to the meeting. Thus, for each flow scenario, the water quality consequences, as well as the likely implications for the macroinvertebrates can be assessed. These results can then be presented at the environmental flow workshop, and if necessary, river reaches can be identified in which pollution must be ameliorated in order for the RQOs for macroinvertebrates to be met.

7.6.2 Use of stress time-series

Stress time-series are of potential use in assessing the implications of different discharge time-series for the biota. They would be used in conjunction with and following on from, concentration time-series. As is the case for concentration time-series, it is recommended that stress time-series be used to compare and rank different flow scenarios with regard to likely impacts on the biota. They should not be used to obtain quantitative values of the exact duration of different stress levels. Stress time-series can be employed most usefully when considering the effect of toxic substances and salts. They should not be used for modelling of nutrients since the principle mode of impact of these constituents is by altering the trophic state of a system rather than a direct toxic effect on organisms.

7.7 Scenario modelling

One of the major limitations of the IFR methodology as it now stands, is that how a given system will be operated (and hence the sources and proportions of different flows), is not usually known. Limited scenario modelling is undertaken, in which no consideration is usually given to the manner in which the system will be operated. In other words no information is available as to how the relative sources of water would change between flow scenarios. Thus, the actual effects on water quality can not be determined, and all predictions are made on the premise that the source of water for all flow scenarios will be the same as at present. Although this is adequate if the water quantity component of the Reserve is to be determined, for assessing the water quality of the Reserve, it is simplistic. Qualitative statements such as "the discharge from tributary X, which carries good quality water should be maintained in order to ensure that salinity at site Z, downstream of the confluence is not compromised," were included in the scenario report for the OREWRA. This was intended to encompass likely effects on pollutant loading resulting from changes in the source (tributaries, impoundments etc.) of water. Although it is likely to increase the complexity of the scenario phase, there is an urgent need to include more realistic scenarios that consider the water quality of the various water sources. It should be possible using a combination of mass-balance modelling and time-series modelling to obtain more accurate predictions of water quality and thus ranking of flow scenarios. The use of DRIFT and in particular the database that is generated during its application, should be investigated to assess the suitability of including different water quality management scenarios.

7.8 A framework for incorporation of water quality into the Reserve

A framework has been developed (Table 7.1) which indicates how, and at what stage, water quality should be incorporated into the Reserve determination process. The proposed framework also incorporates the predictions arising from water quality modelling, and the assessment of the likely implications of changed water quality for the aquatic biota. It has been compiled by liaison with IFR specialists and from involvement with actual Reserve determinations. The framework is compatible with both methods of determining the IFR namely, with both the BBM and the DRIFT method.

The framework is comprised of three major phases, depending on whether the activities take place, before, during, or after the Environmental Flow (IFR) workshop. Each phase is sub-divided into steps, which in turn are comprised of individual work components. The first step in the Pre-IFR workshop phase entails assessing the water quality of the resource. This includes an examination of ecoregions, point-sources of pollution and catchment land-use, as well as division of the resource into water quality reaches, in which the concentrations of chemical constituents and values of physical variables are assumed to be homogeneous. All pertinent water quality data are assembled and examined for completeness. Reference conditions (RC) and present ecological state (PES) water quality is defined for individual river reaches and the Reserve Categories (management classes) are determined. For each water quality variable of concern, a numerical value for the Reserve (the RQO) is assigned. All this information is recorded in the IFR starter document. The above activities are part of the general water quality assessment that is routinely carried out during Reserve determinations. One of the major products arising from these activities is the generation of monthly median concentration values (for RC and PES) which are used in Q-C modelling. The second half of this phase (Step B) is concerned with integration of discharge and water quality. Discharge-concentration plots for each chemical constituent of concern are drawn up for each IFR site for which there are data. In addition, the spreadsheet templates are prepared for use in the IFR workshop.

Table 7.1 Framework for the incorporation of predictions of water quality, as well as the implications of altered water quality for the biota, into Reserve assessments. (WQ = water quality, RC = Reference condition, PES = present ecological state, RQOs = resource quality objectives).

<table>
<tr><th>Phase</th><th>Step</th><th>Work component</th><th>Product</th><th>Use of product</th></tr>
<tr><td rowspan="11">PRE-INSTREAM FLOW REQUIREMENT WORKSHOP</td><td rowspan="7">Step A: Water quality assessment</td><td>1. Identify resource & delineate boundaries, chose IFR sites etc.</td><td>Maps of resource</td><td rowspan="2">Used for entire process</td></tr>
<tr><td>2. Identify ecoregions, significant hydrological features, point sources etc.</td><td>WQ reaches</td></tr>
<tr><td>3. Determine Reference condition</td><td>Monthly median values</td><td>Step B</td></tr>
<tr><td>4. Determine Present ecological state</td><td>Monthly median values</td><td>Step B</td></tr>
<tr><td>5. Assign provisional Ecological Reserve Class</td><td></td><td></td></tr>
<tr><td>6. Assign WQ Reserve</td><td>WQ categories and RQOs for each variable and site</td><td>Step C</td></tr>
<tr><td>7. Record findings</td><td>WQ Starter document</td><td>Used for entire process</td></tr>
<tr><td rowspan="4">Step B: Integration of WQ & quantity (Pre-workshop)</td><td>8. Check RC & PES monthly median values for suitability for modelling</td><td rowspan="2"></td><td rowspan="2"></td></tr>
<tr><td>9. Consult with hydrologist. Obtain appropriate discharge data</td></tr>
<tr><td>10. Prepare Q-C modelling spreadsheets.</td><td>Q-C relationships for each variable at each IFR site</td><td rowspan="2">Step C and Step F</td></tr>
<tr><td>11. Prepare concentration duration curves</td><td>Concentration duration curves</td></tr>
<tr><td rowspan="3">INSTREAM FLOW REQUIREMENT WORKSHOP</td><td rowspan="3">Step C: Integration of WQ & quantity (during workshop)</td><td>12. Use Q-C modelling to predict concentrations and WQ classes</td><td>"Water quality consequences"</td><td>Step E</td></tr>
<tr><td>13. If "natural WQ impacts", use Q-C modelling to motivate for Quantity Reserve</td><td>Recommended flow regime that will attain RQOs for water quality</td><td>Step E</td></tr>
<tr><td>14.If required, use concentration duration curves to assess recommended flow regime compared to natural and present-day</td><td></td><td></td></tr>
</table>

Table 7.1 Framework for incorporation of predictions of water quality and implications for the biota, continued.

Phase	Step	Work component	Product	Use of product
POST-INSTREAM FLOW REQUIREMENT WORKSHOP	**Step D:** Assess implications of predicted WQ for biota	15. Apply Biotic protocol to key IFR sites, for critical recommended discharges	Predicted derived SASS scores, Ecological Reserve Category, lists of taxa	**Step E**
	Step E: Document Predicitons of WQ and effects on aquatic biota	16. Record results of WQ modelling	Water quality modelling report	**Step F**
		17. Record results of Biotic protocol	Report on expected effects on the aquatic biota	
	Step F: Assessment of flow Scenarios generated by Yeild model	18. Use Q-C relationships to transform discharge to concentration time-series for key sites		
		19. Prepare concentration duration curves	Concentration duration curves	**Step G**
		20. Rank flow scenarios with regard to WQ impact	Flow scenarios ranked wrt water quality implications	**Step G**
		21. Use ecotoxicological data to transform concentration to stress time-series for key sites (if suitable data available).		
		22. Prepare stress duration curves	Stress duration curves	**Step G**
		23. Rank flow scenarios with regard to WQ impact	Flow scenarios ranked wrt implications for biota	
	Step G: Document results of time-series modelling	24. Record results of time-series modelling	Report on flow scenario modelling	

The second phase of the framework involves the tasks that arise during the IFR workshop itself (Step C). It is suggested that the qualitative, site-specific manner in which water quality is currently incorporated should be retained. In addition, broad-scale predictions using results from Q-C modelling should also be employed. At the workshop predictions of water quality in response to the exact flow regimes prescribed by the IFR practitioners are produced (the "water quality consequences"). Concentration duration curves can be used at this stage, if required, to compare the water quality consequences arising from the natural, the present-day and the recommended flow regime. Occasionally, cases of "natural water quality impacts" may be encountered. In such cases, Q-C modelling can be used to calculate the volume of water required to dilute naturally occuring chemical constituents, so that the water quality component of the Reserve would be obtained.

The final phase of the process is carried out after the IFR workshop. The implications of the predicted water quality for the biota are assessed, in other words, the likelihood of the Resource Quality Objectives for aquatic macroinvertebrates being attained under the proposed flow regime, and current pollution loading is ascertained. This phase may, or may not, include a comparison of flow scenarios. Presently, such comparisons are made at a Flow Scenario workshop which is held if there are such heavy demands on the system in question that both the ecological Reserve and the demands of existing users in the catchment cannot be satisfied (Louw *et al.*, 2000). In order to make realistic predictions of water quality it is necessary to take into account the various sources of water as well as the loads of pollutants that can contribute to the total discharge at a particular site on a river. Although incorporation of management scenarios might greatly increase the complexity of the process of quantifying the Reserve, it is an essential step if realistic predictions of water quality are to be made. Concentration duration curves can be used at this stage to compare and rank different flow scenarios. Stress duration curves may also be of use if suitable data are available. The final step (Step G) is to document the results of the Scenario modelling phase (both of flow scenarios as well as water quality management scenarios). If DRIFT is used as the methodology for assessing the environmental flow requirement, the results contained in the database would be reported at this stage.

CHAPTER 8

CONCLUSIONS AND RECOMMENDATIONS

8.1 Major conclusions arising from this project

The following major conclusions have been reached during the course of this project:

- Some form of water quality modelling is essential within Instream Flow Assessments to ensure that in setting the ecological Reserve with regard to water quantity, the water quality component of the Reserve will also be attained.
- In order to make realistic predictions of water quality it is necessary to take into account the various sources of water as well as their pollutant loads that can, or do, contribute to the total flow at a particular site on a river. Thus consideration of different water quality management (or system operation) scenarios, in addition to flow scenarios, is important in terms of Reserve assessments. At present, this aspect is not usually considered. Although, incorporation of management scenarios would increase the complexity of the entire process, it is an essential step if realistic predictions of water quality are to be obtained.
- No single, water quality modelling method possesses all the attributes that are required. Furthermore, the attributes exhibited by the models would not necessarily be required in all situations. Therefore a hierarchy of methods should be employed, with three tiers of modelling complexity:
    - A simple discharge-concentration (Q-C) regression method.
    - Mass-balance modelling or QUAL2E.
    - A catchment runoff model.
- Discharge-concentration (Q-C) modelling is a useful method for estimating the instream concentration of chemical constituents that will arise from implementation of a given flow regime. It can be used as a screening tool to identify IFR sites where the water quality component of the Reserve is not likely to be attained under the

recommended flow regime. There are limitations in the method, however, and these should be clearly recognised when applying the model.

- Q-C modelling is suitable for conservative substances, less successful in modelling nutrients and cannot be used for dissolved oxygen or temperature. It should also be used with caution when making predictions of constituents that exhibit a positive correlation with discharge.
- Mass balance models (either spreadsheet or QUAL2E) can be used to examine the results of different water quality management scenarios.
- QUAL2E can be useful for providing a more mechanistic description of the processes affecting water quality in a given reach compared to the empirical approach used in Q-C modelling. It has fairly extensive data requirements, however, and frequently additional data will need to be collected. Thus in the context of Reserve determinations it is suitable for use as part of a Comprehensive Reserve Assessment for key IFR sites. It is not suitable for Rapid or Intermediate levels of Reserve determination. This model can be most usefully employed in situations where: there is a complex situation of pollutant loading; when nutrients, dissolved oxygen or temperature need to be modelled; and when different management scenarios need to be assessed.
- The problem of non-point sources of pollutants, for which Q-C modelling is not suitable, is likely to increase in future. Because of this, attention should be directed towards setting up catchment run-off models, at least for key sites on rivers where extensive and conflicting demands are made on them and for critical reaches in which there is extensive loading from diffuse pollutants.
- Preliminary research indicates that patterns in Q-C trends at un-impacted sites do show similarity within the same primary drainage region. Further work is required, however, to confirm these findings.
- Despite the approximations in the derivation of concentration time-series, these can be used for comparing and ranking different flow scenarios, as generated by the IFR or yield model (WRYM), with regard to the likely effect on water quality.
- Attention needs to be paid to the further development of concentration time-series, primarily in the use of confidence intervals, and in the use of spell analysis (i.e. examining the number of times and the length of time that the concentration of a given variable will be above a threshold value). Attention also needs to be given to

the use of concentration time-series in combination with mass balance modelling. This would aid in interpretation of the implications of different water quality management scenarios.

- The use of stress time-series shows potential for ranking flow scenarios with regard to likely impacts on the aquatic biota. The method is probably most suited to toxic constituents or system variables rather than nutrients. More research is required, however, to link the concentration of chemical constituents and values of physical variables with biotic response. In particular, methods to integrate co-stressors need to be investigated and the method, in some manner, needs to be validated using field data.
- The Biotic protocol can be useful for assessing the likely implications of a proposed water quality scenario for benthic macroinvertebrates. It can be used to identify taxa that may be lost or regained to a system under a recommended flow regime as well as to make predictions of the theoretical SASS scores and Assessment class likely to be attained. The Biotic protocol is, however, entirely dependent on the availability of suitable data. It needs further refinement by applying the method to catchments that are situated in a variety of ecoregions.
- The potential of the Biotic protocol for predicting the likely implications of a proposed water quality scenario for other sectors of the aquatic biota (for example, fish and riparian vegetation) should be explored.
- From the point of view of water quality, DRIFT appears to be more flexible than the BBM. The use of a database means that the results from modelling of different water quality management scenarios can be recorded and the optimum scenario chosen. In addition, using DRIFT, the discharge regime can be made available before the IFR workshop so that the water quality consequences as well as the consequences for the biota (obtained from the Biotic protocol) can be derived. These results can then be used in the workshop to further refine the recommended flow regime (although it is permissible to increase the water quantity component of the Reserve to allow for dilution only in the case of "natural water quality impacts").
- The tools developed in the course of this project, although not specifically examined for this purpose, should be useful in a wider field than Environmental Flow and Reserve determinations. The current trend in water quality management both internationally as well as in South Africa away from the control of effluent discharge to resource directed measures. Thus rather than considering the composition and

volume of point sources of pollutants, the water quality of the receiving water body is of primary concern. Simple Mass balance modelling can be used to predict the instream concentrations of contaminants that would result from the loading of different point sources. The Biotic Protocol could be used to assess the likely impacts of a particular water quality scenario (for example the addition or removal of a point-source) on aquatic invertebrates. Thus, the water quality scenario to be examined need not necessarily be one that is predicted from implementation of the Reserve.

8.2 Recommendations and future research areas

The following recommendations arise from this project:

- In order to arrive at realistic predictions of water quality, attention must be directed towards incorporation of water quality management (or system operation) scenarios into the Reserve process. Mass balance modelling could then be used to predict the water quality consequences of each scenario. Attention should be given to the use of the DRIFT database to store and analyse the results from such a modelling exercise so that the optimum management scenario can be chosen.
- There should be extensive co-ordination between monitoring networks, in particular, between that of water quality and biomonitoring. Collecting measurements of water quality as well as biological data (presence/absence of macroinvertebrate taxa and SASS scores) at the same time and same place maximises the usefulness of both. This enhanced usefulness is primarily because links between water quality and the presence or absence of specific taxa can then be made.
- Research should be carried out to confirm the SASS scores and ASPT values that form the boundaries for macroinvertebrate Assessment categories (Table 6.2). It should also be confirmed if these are valid for all regions of the country. The values of SASS scores that are characteristic of the Reference Condition (Table 6.1) also need confirmation.
- In the OREWRA study the situation was frequently encountered where, under the recommended IFR flow scenario, whilst occasions of very high concentrations of salinity (or other constituent) were avoided, periods of very good quality water would also be removed. This is a consequence of the fact that it is normally during the

period of high flow that water would be harvested for other users. The overall result is an attenuation of the current salinity profile so that the extremes of very high and very low concentration would no longer occur. Whilst the removal of episodes of poor water quality is obviously advantageous, the effect of removing periods of "fresh" water is not clear. This matter requires further research and should form part of a detailed study of the long-term effects of implementation of the IFR for key rivers. It should be noted that this is an urgent research need, not only to test the predictions of water quality modelling, and of the Biotic protocol, but to assess the effectiveness of the entire Reserve determination process.

- At present, the form in which water quality data are available in South Africa is not optimal. Water quality simulations as developed in this project (and also for the Reserve) require examination and comparison of segments of the time series for given variables. The relevant data are available in several different forms. Despite this, there is no completely reliable, easily accessible manner that will enable a user to view the entire time series and that will also calculate summary statistics for different time segments. Manipulation of data to obtain the required information can be time consuming. In addition, linked water quality and discharge data should be directly available from DWAF. This would avoid errors due to the use of the incorrect rating curve etc.
- Some statistical aspects of the Q-C method may need refinement. This includes the use of r^2, and correlation coefficients as well as testing the significance of non-linear relationships. Some Q-C relationships may be better described by more sophisticated relationships than those available in the software package used (Microsoft Excel). This requires further investigation.

8.3 Conclusion

Several tools have been developed that can be used to integrate water quality and quantity within Reserve assessments. They are complementary and given the assumptions and limitations in the methods, can yield useful information.

Discharge-concentration modelling can be used to predict the concentration of chemical constituents that could be expected at a given discharge. It is a rather simplistic approach, as it does not take into account the origin of the water (and hence water quality) into account and cannot distinguish between instream and catchment effects. It is, however, a step in the right direction towards integrating water quality into the IFR process and is an improvement on the purely qualitative estimates given previously.

Because of the inherent extrapolations and simplifications in the time-series modelling method it is not particularly accurate for predicting the exact proportion of time that a water quality variable is likely to be in each assessment category. Nevertheless, it is a useful tool for comparing the water quality that will result from different flow time-series and for ranking such scenarios.

It is also acknowledged that the Biotic protocol is simply a structured manner for assessing the data available that links water quality and the presence or absence of macroinvertebrate taxa. Predictions of the potential implications of a predicted water quality scenario that are made using this protocol are likely to be a best, tentative. This is a consequence of the variation in natural ecosystems. Using this method, however, does give an indication of the taxa that may be gained or lost in a system, and thus potential impacts can to some extent be ascertained.

In conclusion, prescribing environmental flows for a river, especially systems that are large and complex, is not an easy task. Methods for determining the quantity and timing of flows that are required have evolved in South Africa over several years. The field of water quality lags considerably behind that of quantity and this is the first major attempt to integrate the two fields within Reserve assessments. Because of increasing demands on a limited resource, water quality nationally is likely to become a pressing problem. This project presents the first step in a process that will see water quality modelling playing a more important role in the management of aquatic resources in South Africa. Development of the predictive tools and integration of these into the Reserve determination process has been brought about by participation in the process and accompanied by liaison with practitioners in the field of environmental flow assessments. It is likely, however, that much development of the method is still needs to be carried out and feedback is required from water resource managers in order to make the tools as useful and relevant as possible. This is especially as recommended flow regimes are implemented in South African rivers and Resource Quality Objectives need to be met.

Despite the approximations and assumptions that are inherent in the tools that have been developed, these methods represent an attempt to predict the water quality that will be experienced under a proposed IFR regime as well as to assess the implications for the aquatic biota. It is the opinion and hope of the project team that the use of these tools will aid in the integration of water quality and quantity and should result in a more balanced approach to the use of water resources and enhanced protection of aquatic ecosystems.

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USEFUL WEB SITES

Institute for Water Quality Studies (DWAF):
http://iwqs.pwv.gov.za

DWAF water quality database:
http://iwqs.pwv.gov.za/wq/map

River Health Project:
http://www.csir.co.za/RHP

Water Quality on Disk:
http://envweb.csir.co.za/water/wqcd/index.html

Water Research Commission:
http://www.wrc.org.za

US Environmental Protection Agency:
http://www.epa.gov

US EPA ecotoxicology (ECOTOX) database:

http:/www.EPA.Gov/ecotox

US Geological Survey, surface water and water quality models information:
http://smig.usgs.gov/smic

APPENDIX A

Given below are the equations used to calculate the 95% confidence interval for predictions of concentrations using the Q-C method. All equations taken from Underhill (1985).

$C_{xx} = \sum x^2 - (\sum x)^2/n$

$C_{xy} = \sum xy - (\sum y)(\sum x)/n$

$C_{yy} = \sum y^2 - (\sum y)^2/n$

Linear equation:

$y = a + bx$

Slope:

$b = C_{xy}/C_{xx}$

Y intercept:

$a = \bar{y} - b\bar{x}$

Correlation coefficient r:

$r = C_{xy} / \sqrt{C_{xx}\,C_{yy}}$

Residual standard deviation:

$RSD = \sqrt{(C_{yy} - bC_{xy})/(n-2)}$

Confidence interval:

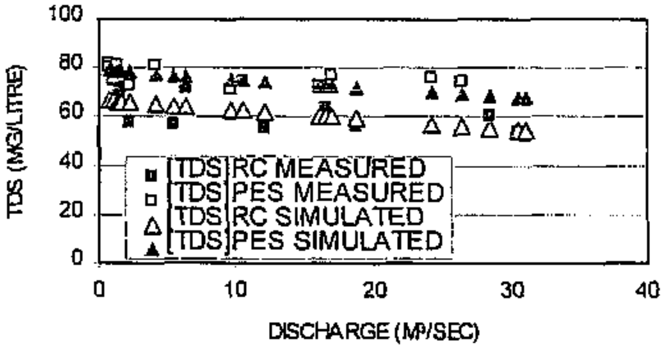
$= \text{predicted value} \pm t_{n-2} \times RSD \times \sqrt{1 + 1/n + \dfrac{(x-\bar{x})^2}{C_{xx}}}$

APPENDIX B

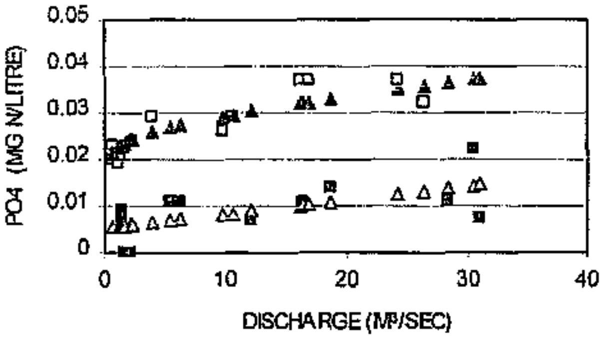
The discharge-concentration graphs for the various water quality variables obtained at each IFR site, are given below.

Palmiet River

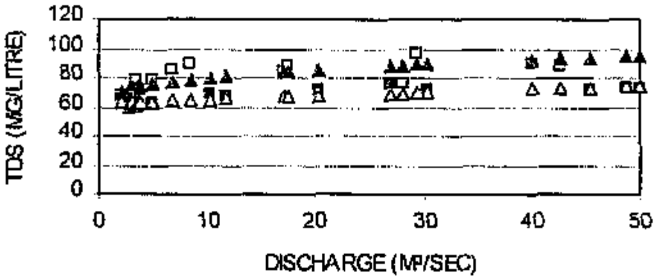
PALMIET IFR 2: TDS



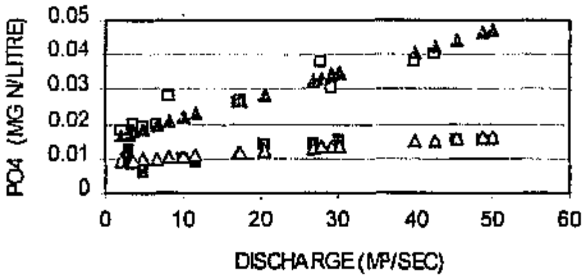
PALMIET IFR 2: PO4



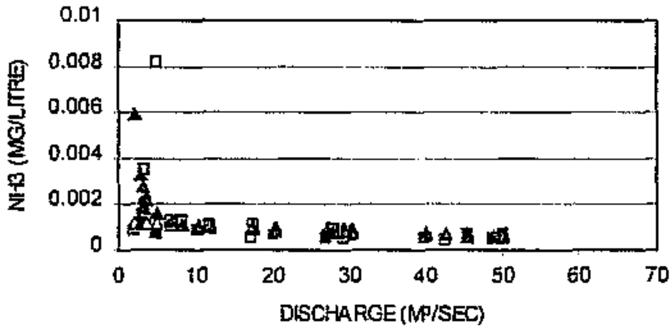
PALMIET IFR 4: TDS



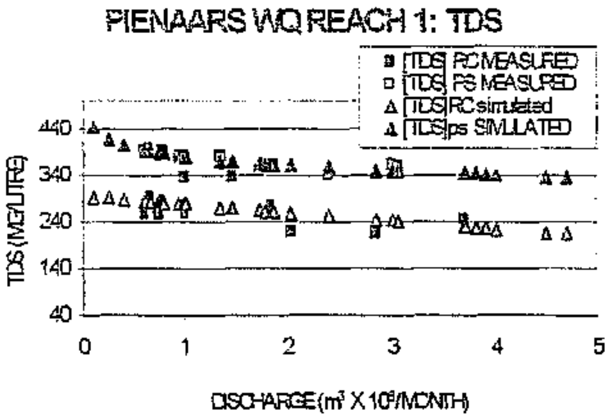
PALMIET IFR 4: PO4

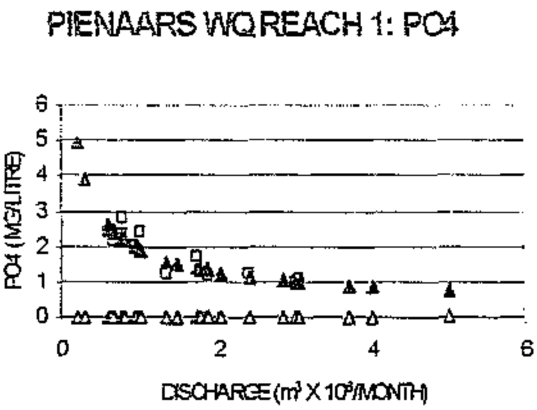


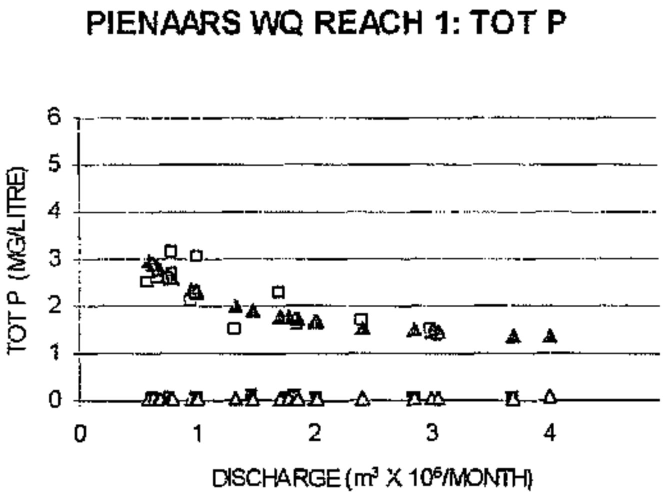
PALMIET IFR 4: NH3

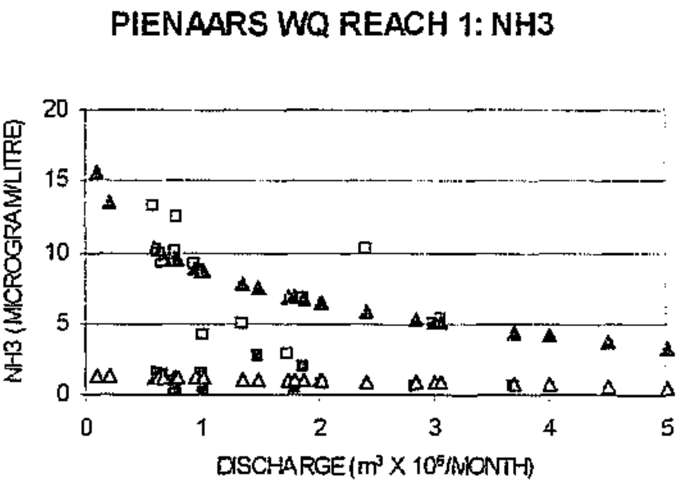


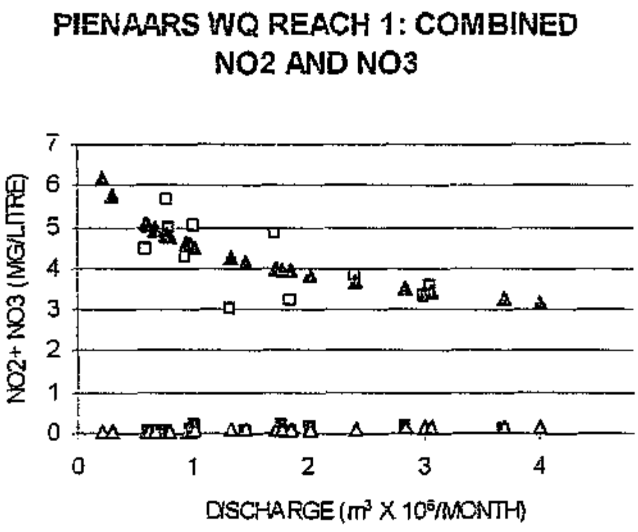
Pienaars River

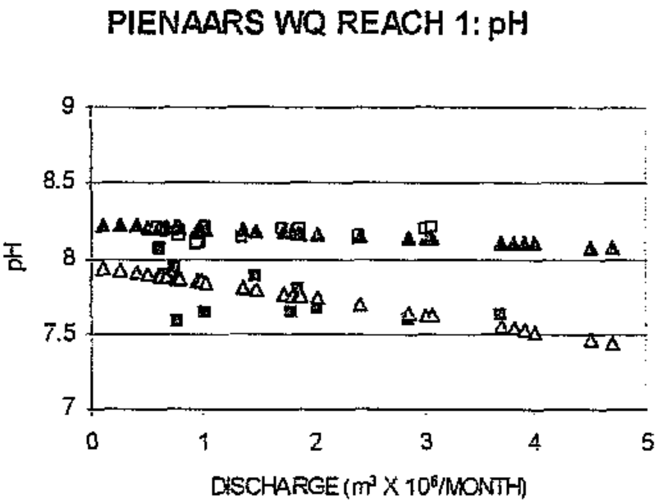






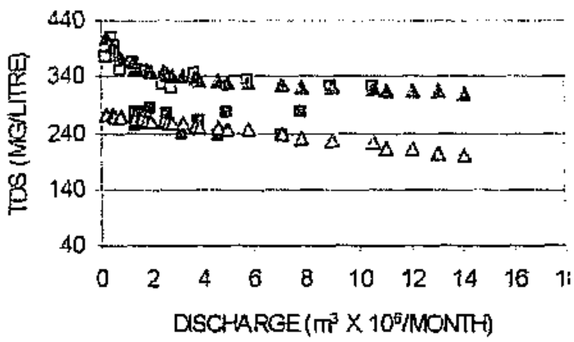




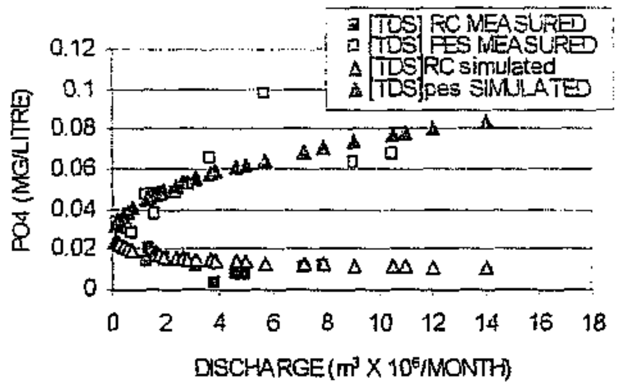


Pienaars River

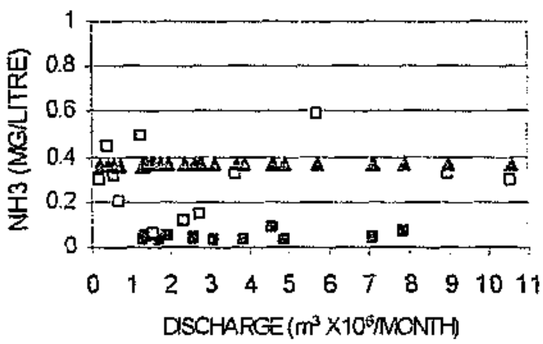
PIENAARS WQ REACH 2: TDS



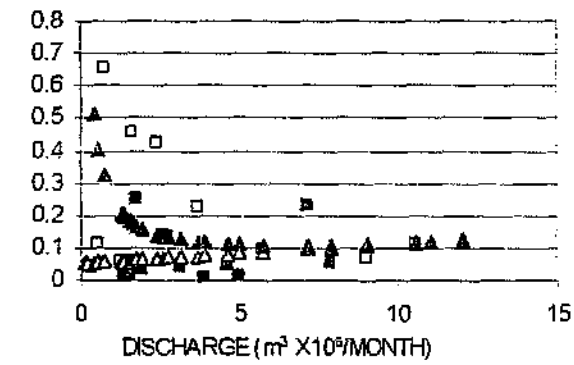
PIENAARS WQ REACH 2: PO4



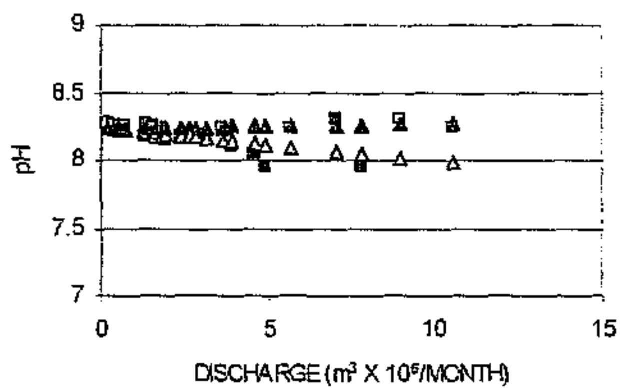
PIENAARS WQ REACH 2: NH3



WQ REACH 2: COMBINED NO2 AND NO3

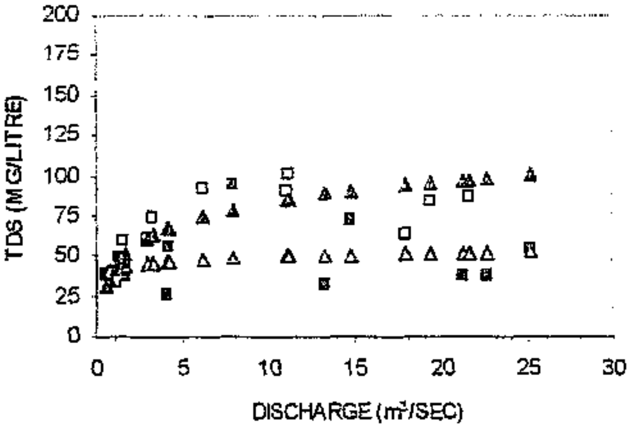


PIENAARS WQ REACH 2: pH

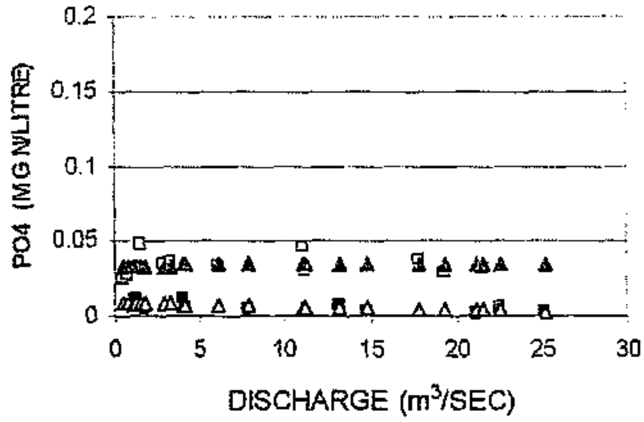


Breede River

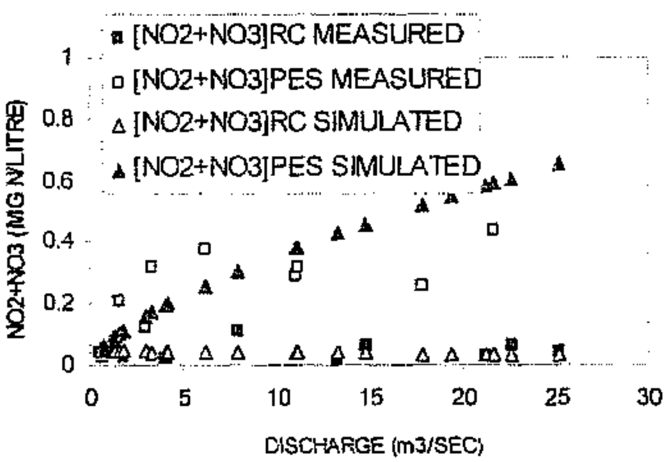
IFR 1: TDS VS DISCHARGE



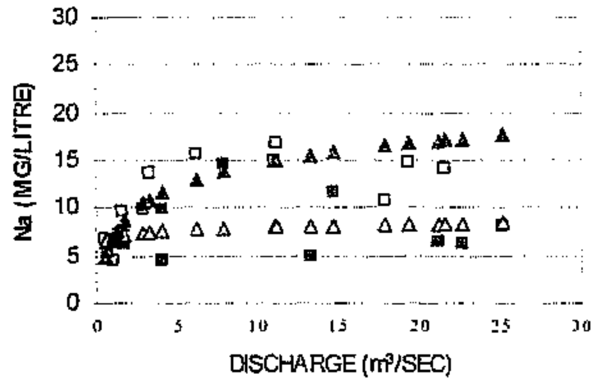
IFR 1: PO4 VS DISCHARGE



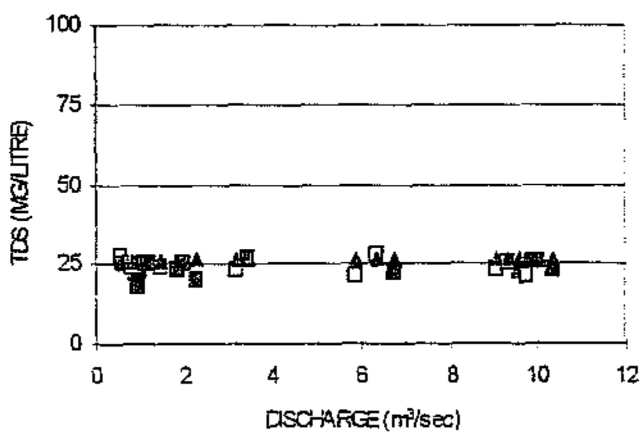
IFR 1: NO2+NO3 VS DISCHARGE



IFR 1: Na VS DISCHARGE



IFR 2: TDS VS DISCHARGE



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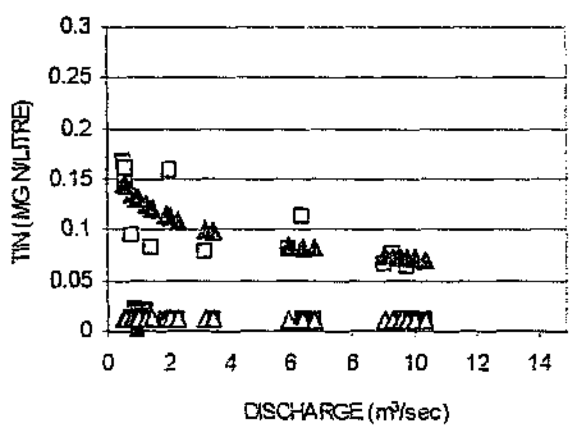
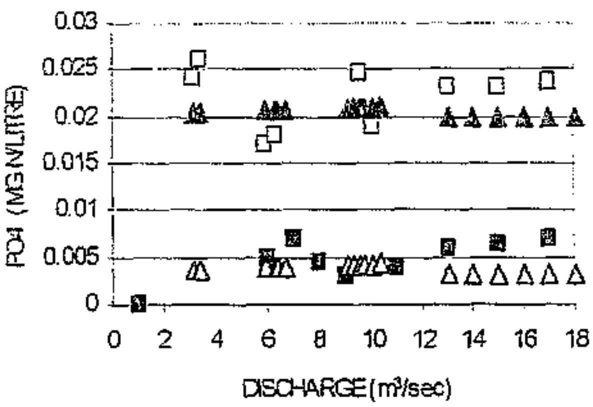
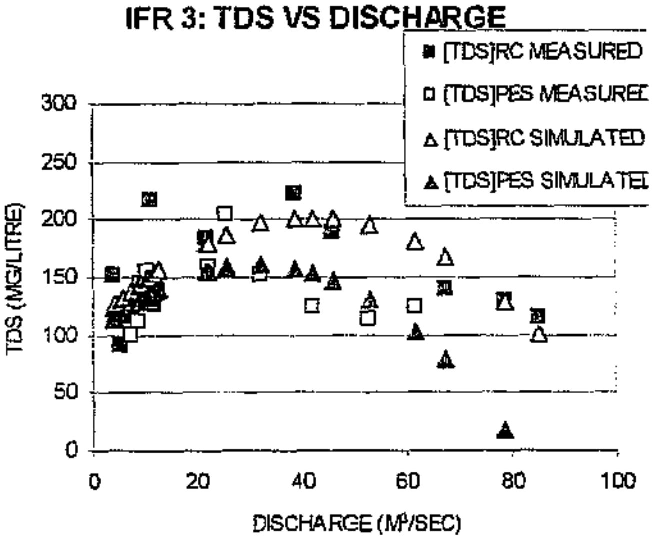


Breede River

IFR 2: PO4 VS DISCHARGE



IFR 3: TDS VS DISCHARGE



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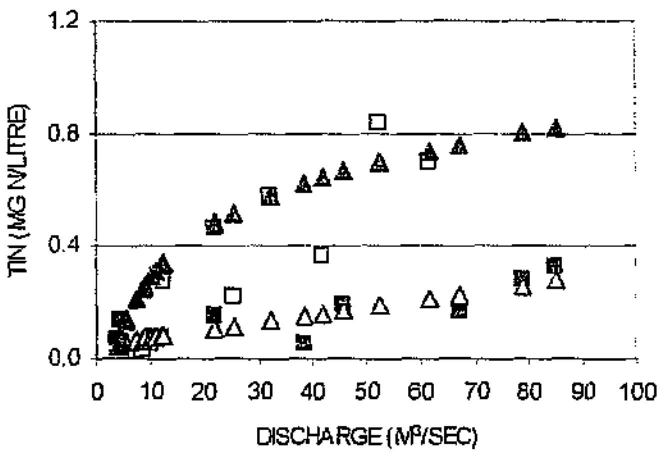

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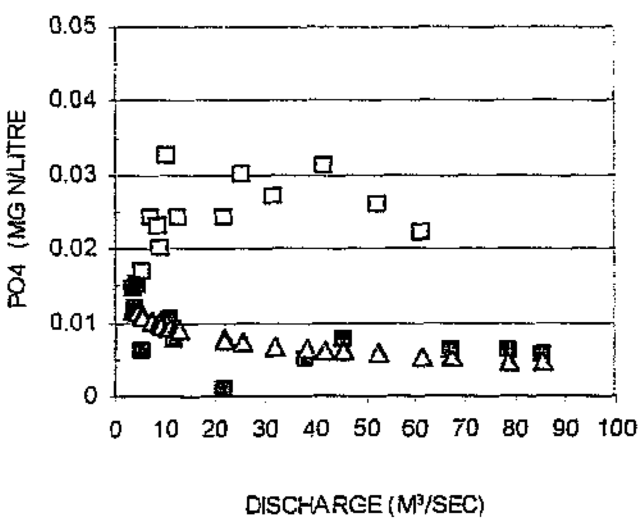
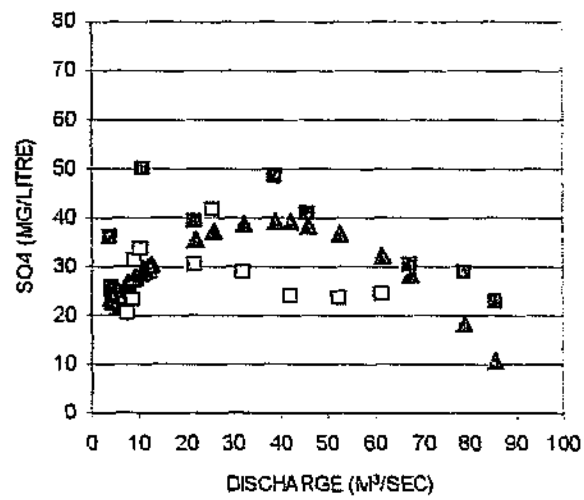
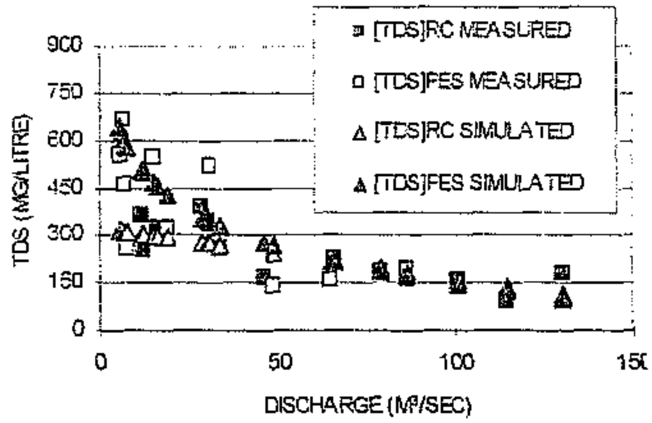


IFR 3: Na VS DISCHARGE

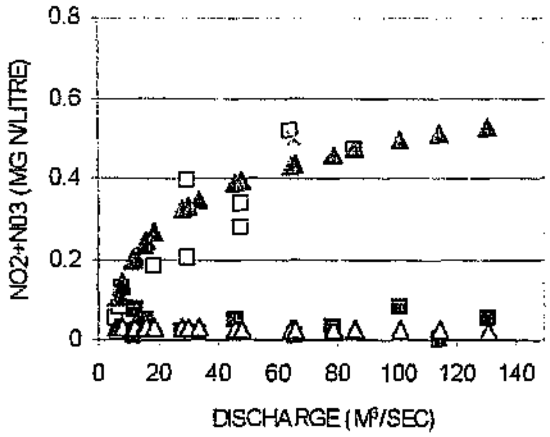


Breede River

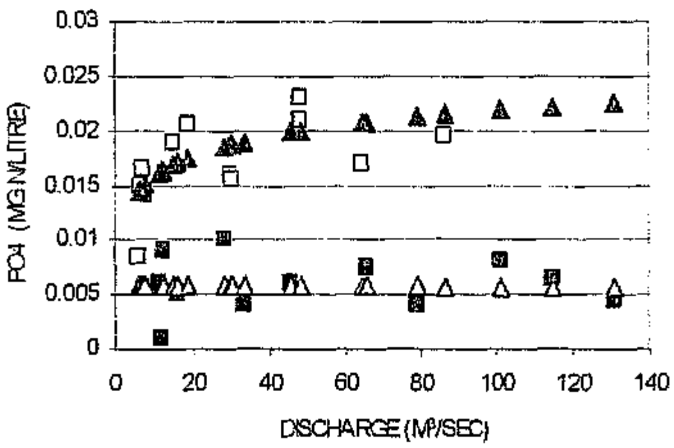
IFR 4: TDS VS DISCHARGE



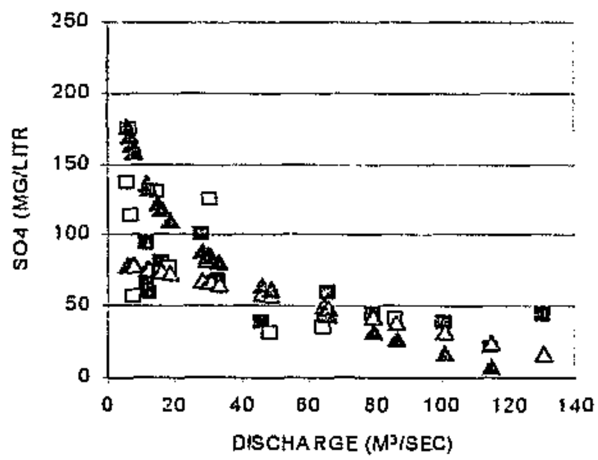
IFR 4: NO2+NO3 VS DISCHARGE



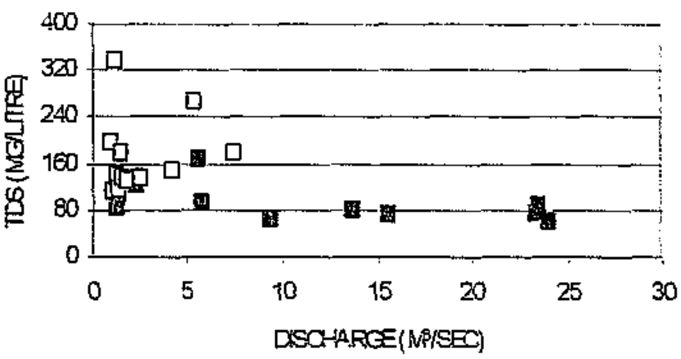
IFR 4: PO4 VS DISCHARGE



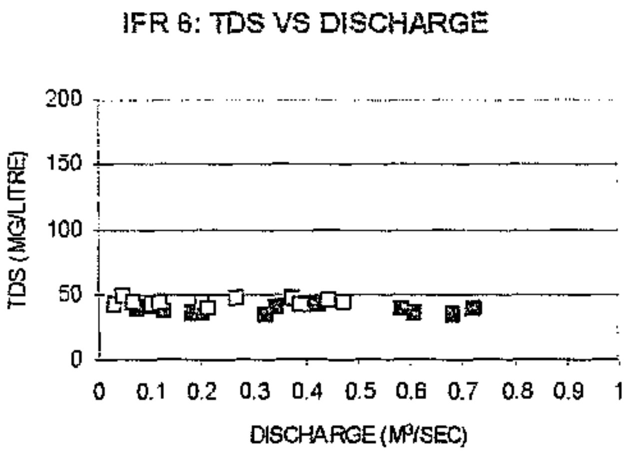
IFR 4: Na VS DISCHARGE

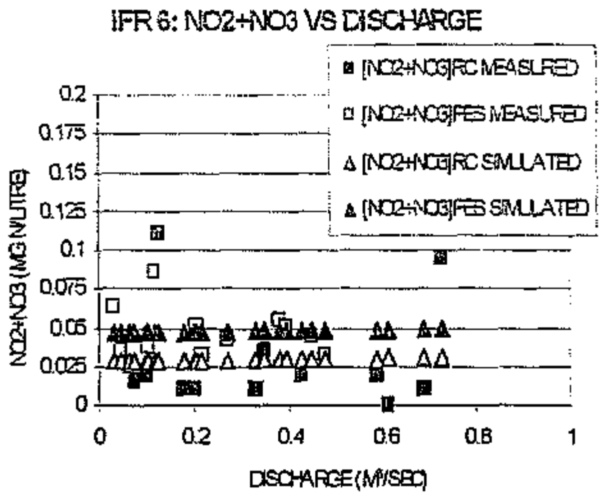


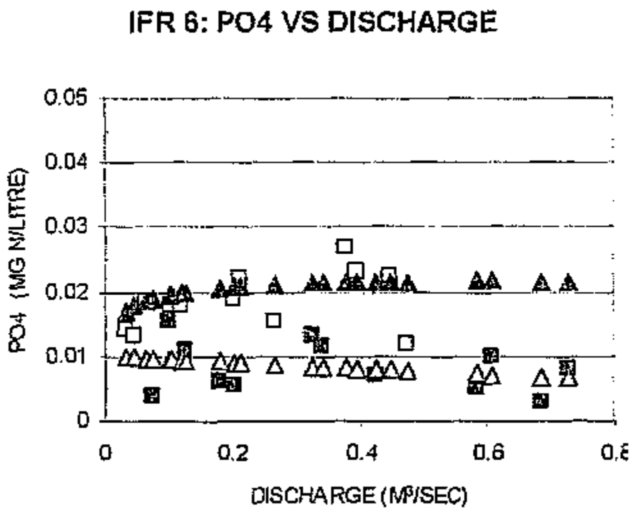
IFR 5: TDS VS DISCHARGE

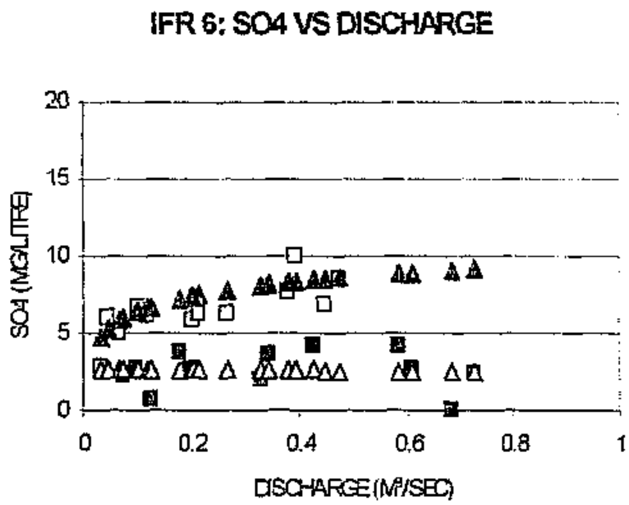


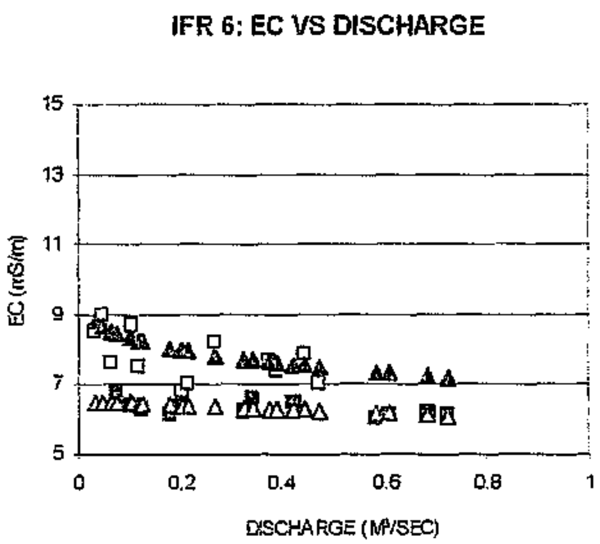
Breede River

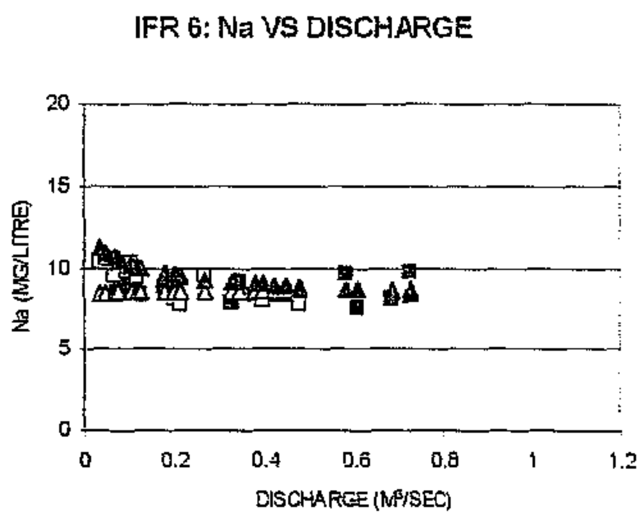






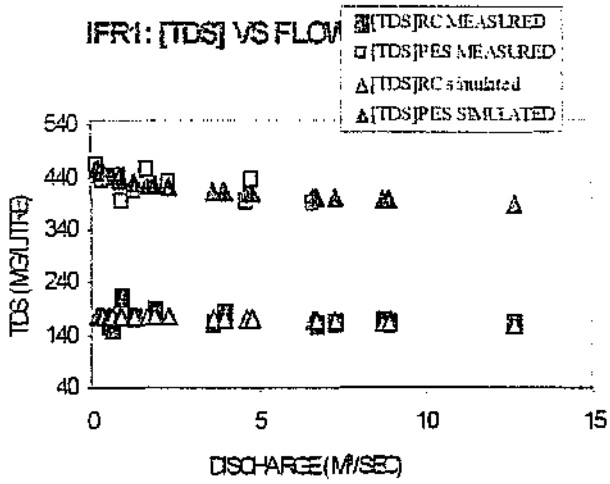




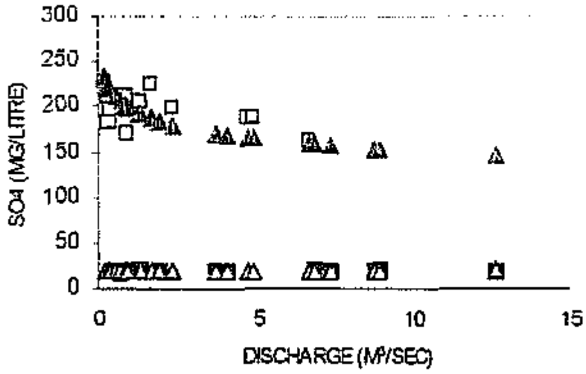


Olifants River

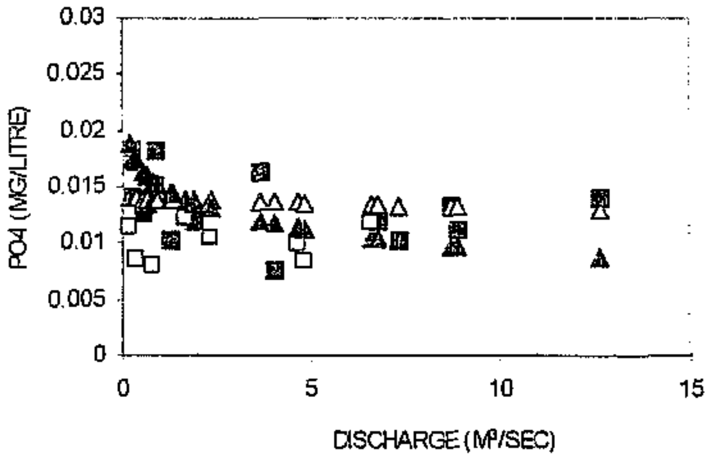
IFR1: [TDS] VS FLOW



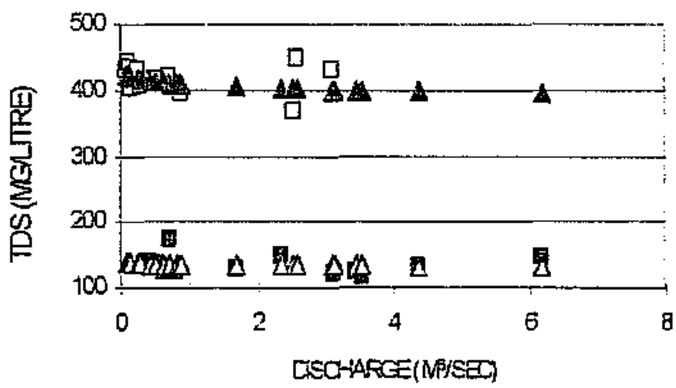
IFR1: [SO4] VS FLOW



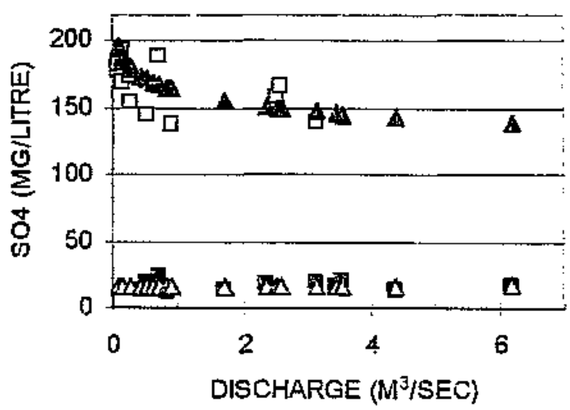
IFR1: [PO4] VS FLOW



IFR 3: TDS VS FLOW

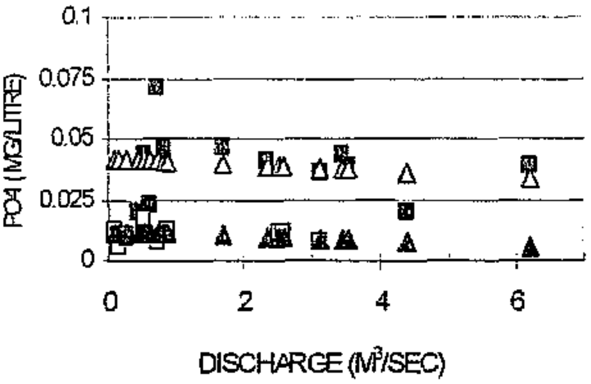


IFR 3: SO4 VS FLOW

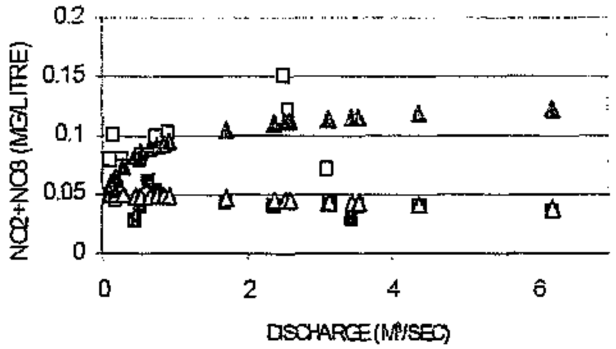


Olifants River

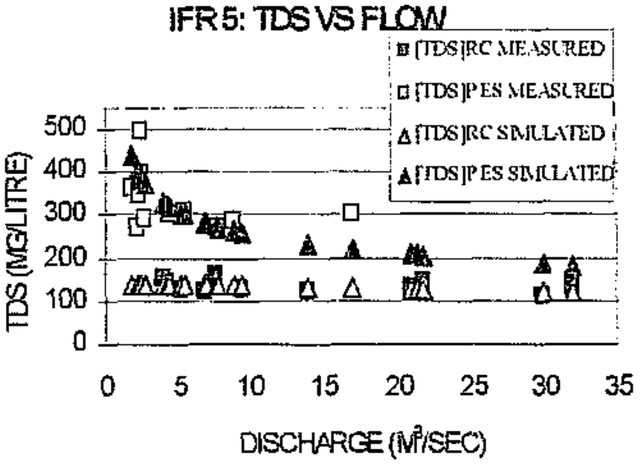
IFR 3: PO4 VS FLOW



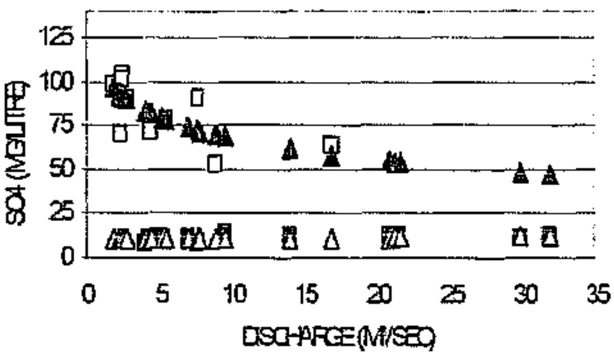
IFR 3: N02+NO3 VS FLOW



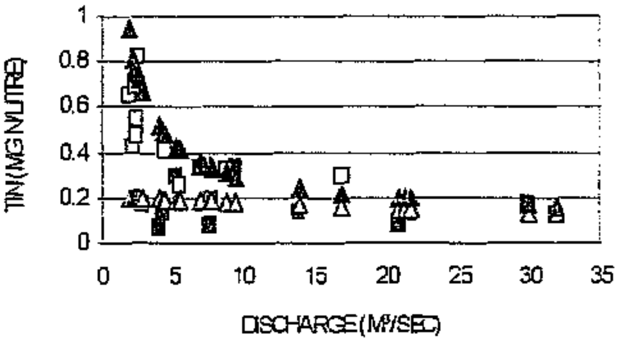
IFR 5: TDS VS FLOW



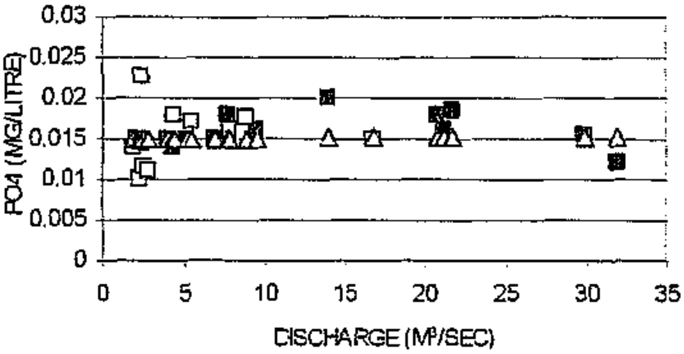
IFR 5: SO4 VS FLOW



IFR 5: TIN VS FLOW

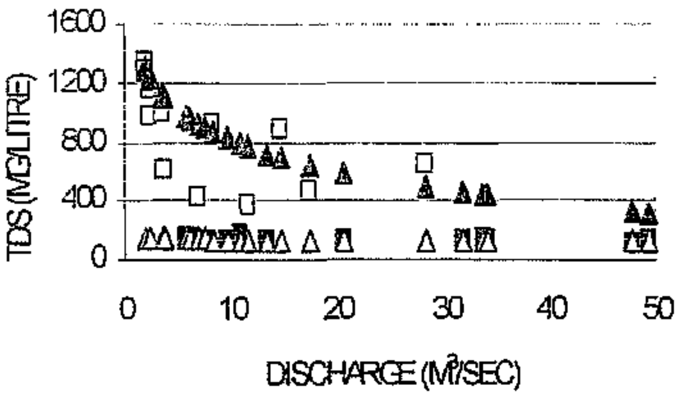


IFR 5: PO4 VS FLOW



Olifants River

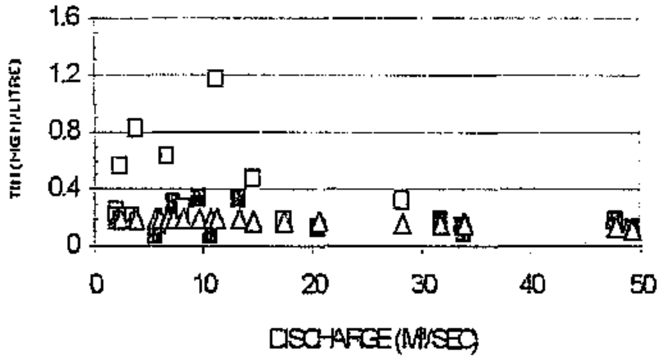
IFR7: TDS VS FLOW



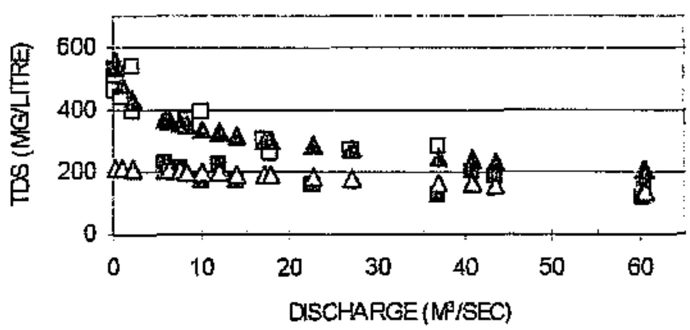
IFR7: SO4 VS FLOW

IFR7: PO4 VS FLOW

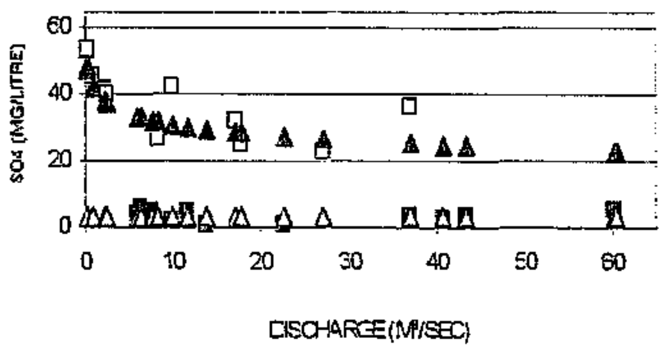
IFR7: TIN VS FLOW



IFR 8: TDS VS FLOW

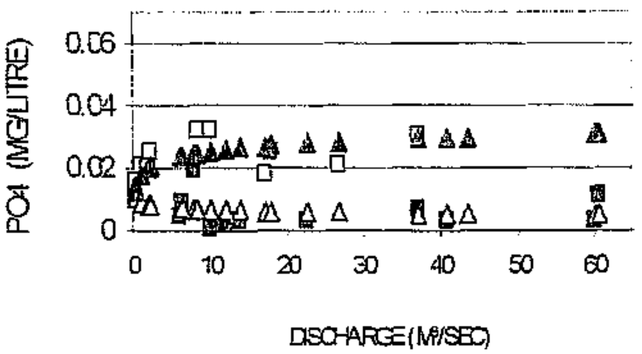


IFR 8: SO4 VS FLOW

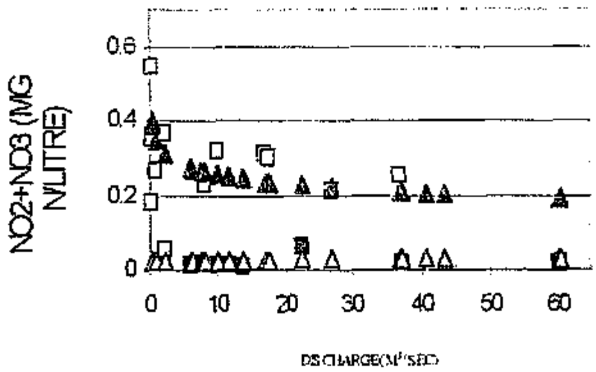


Olifants River

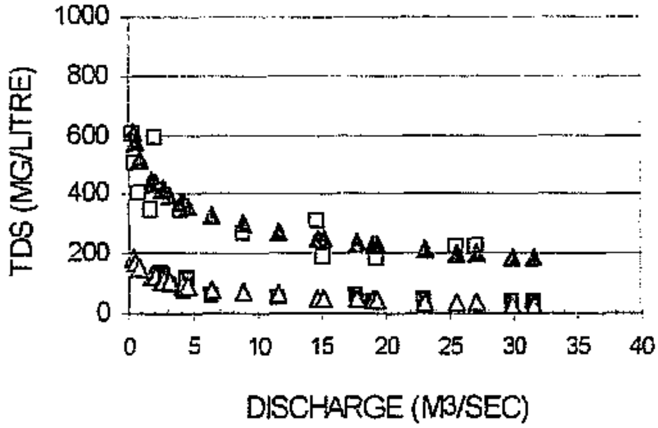
IFR 8: PO4 VS FLOW



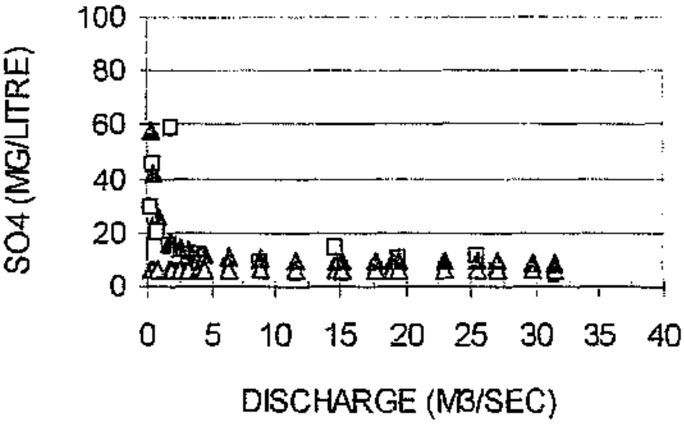
IFR 8: NO2+NO3 VS FLOW



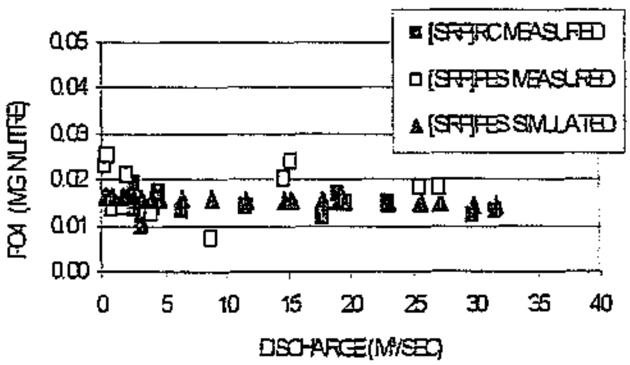
IFR 10: TDS VS FLOW



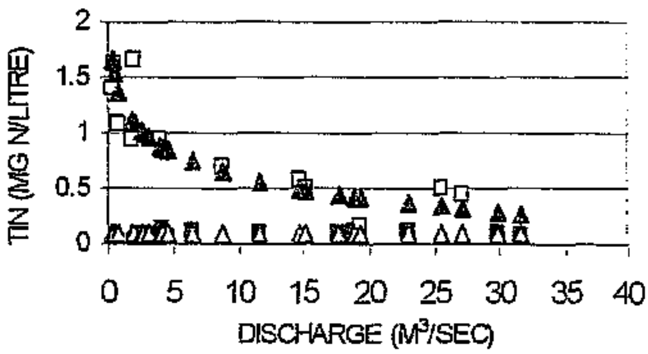
IFR 10: SO4 VS FLOW



IFR 10: PO4 VS DISCHARGE



IFR 10: TIN VS FLOW



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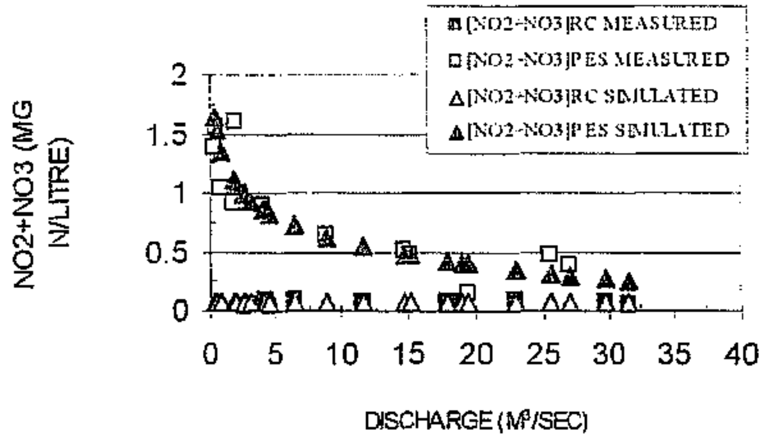
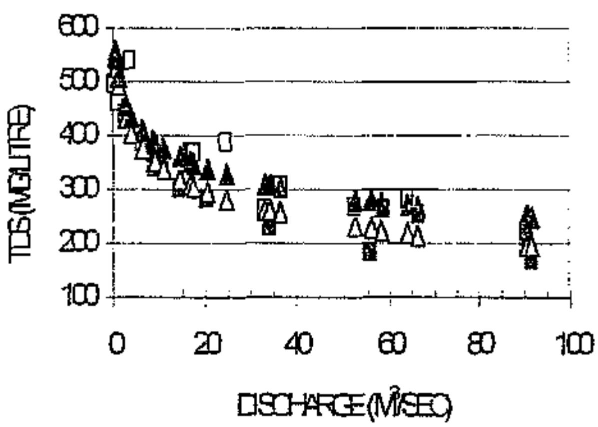
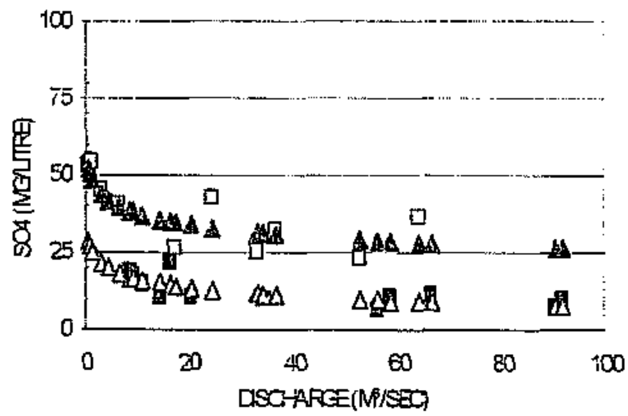
IFR 10: NO2+NO3 VS FLOW



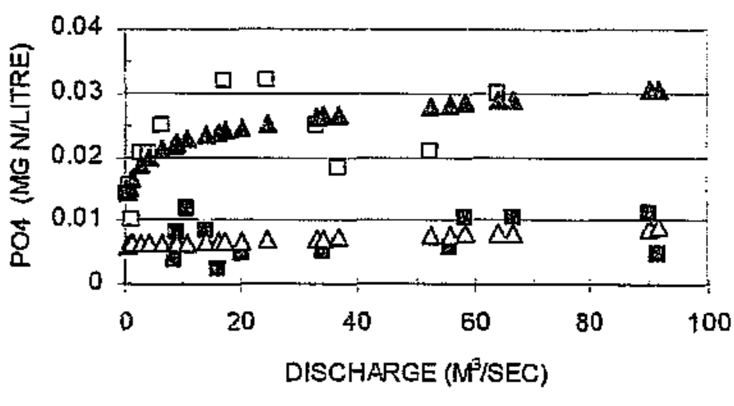
IFR11: TDS VS FLOW



IFR 11: SO4 VS FLOW



IFR 11: PO4 VS FLOW



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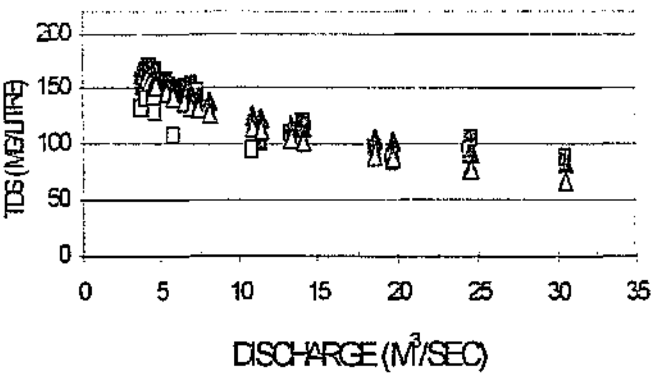
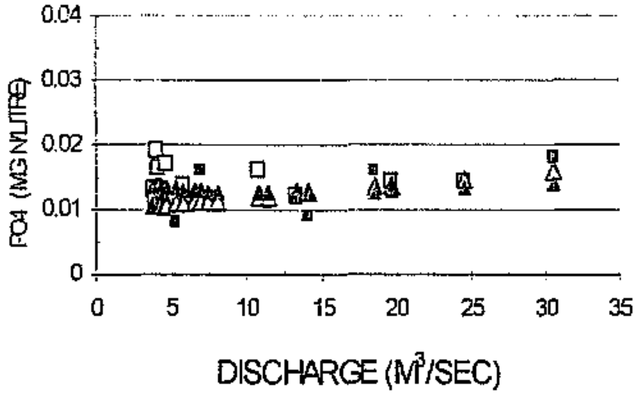
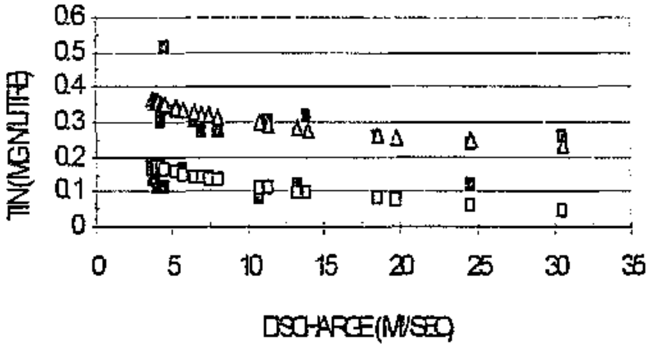
IFR 12: TDS VS FLOW



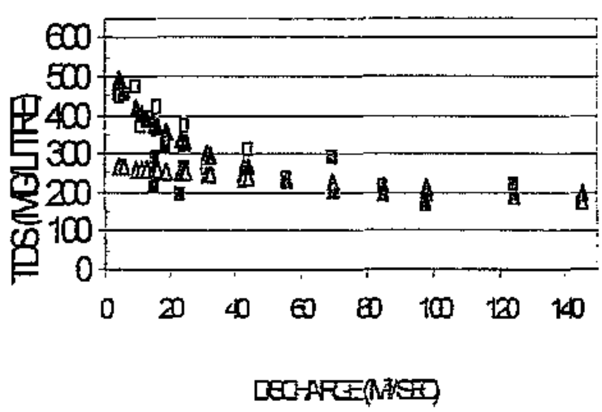
IFR 12: PO4 VS FLOW



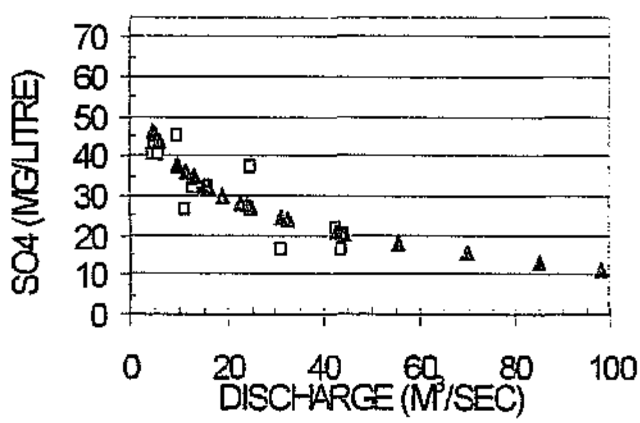
IFR 12: TIN VS FLOW



IFR 13: TDS VS FLOW



IFR 13: SO4 VS FLOW



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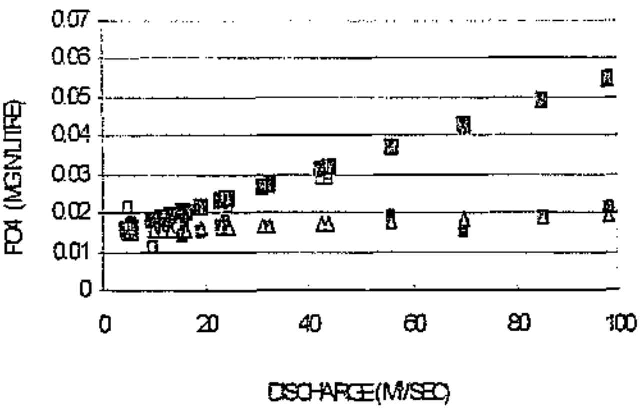
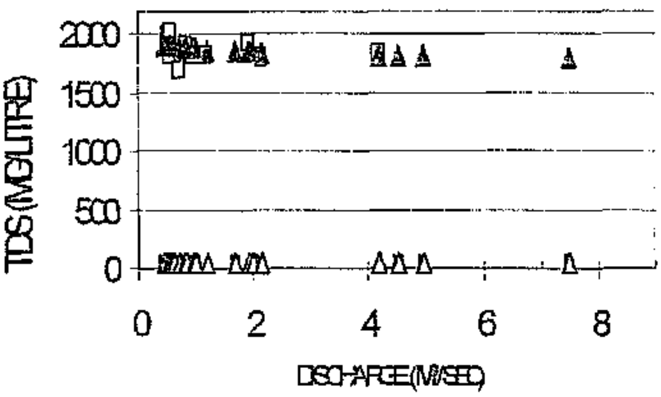
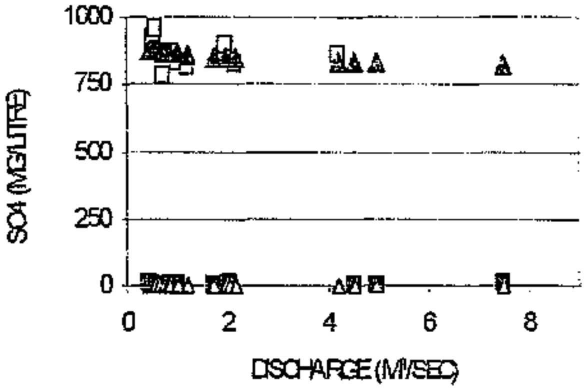
IFR 13: PO4 VS FLOW



IFR 14B: TDS VS FLOW



IFR 14B: SO4 VS FLOW



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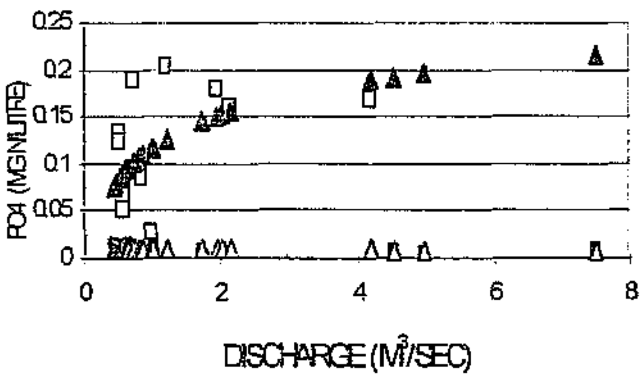
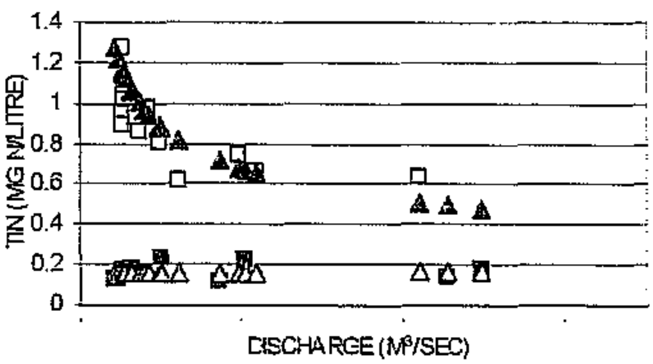
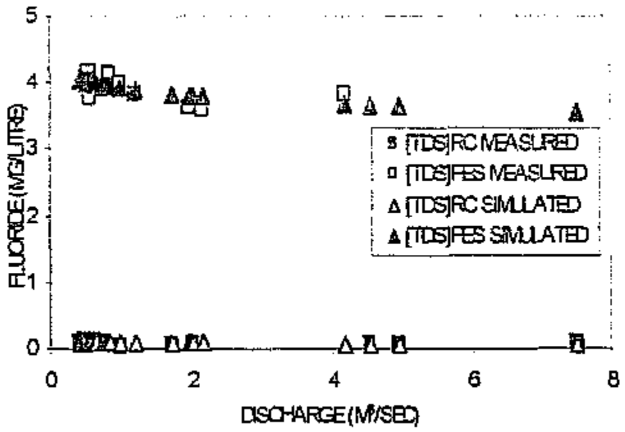


IFR 14B: TIN VS FLOW

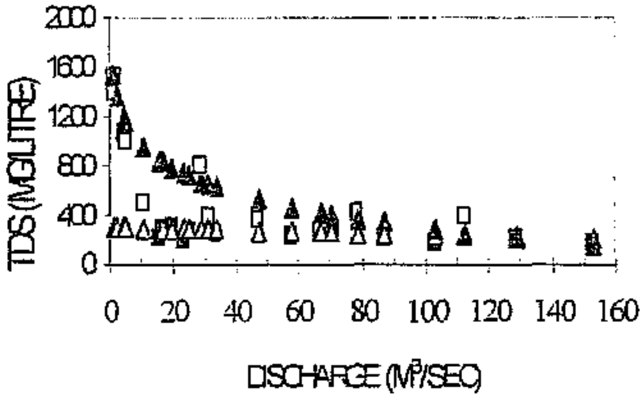


Olifants River

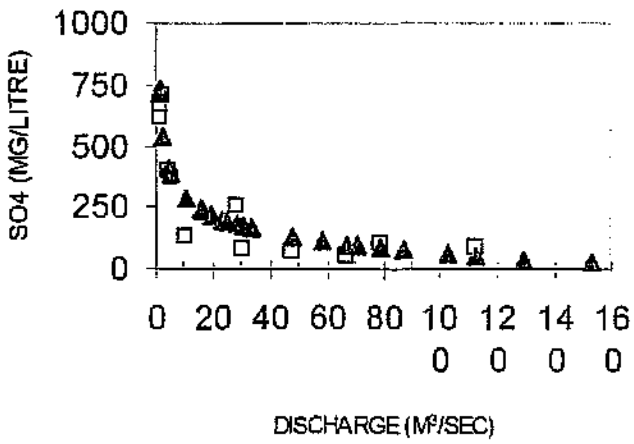
IFR 14B: FLUORIDE VS FLOW



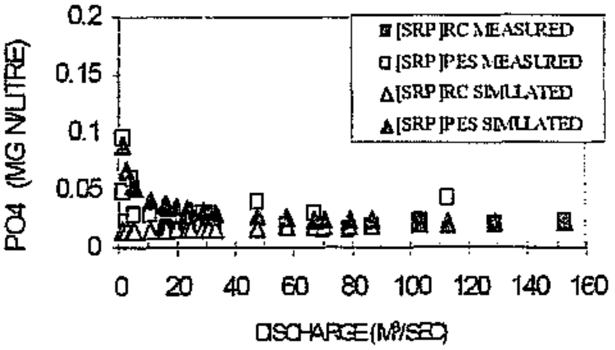
IFR 15: TDS VS FLOW



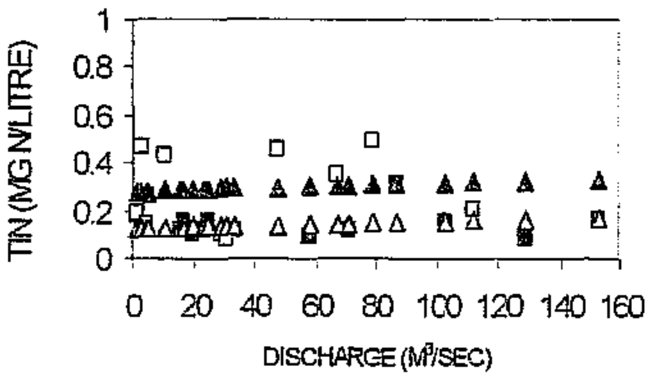
IFR 15: SO4 VS FLOW



IFR 15: PO4 VS FLOW



IFR 15: TIN VS FLOW



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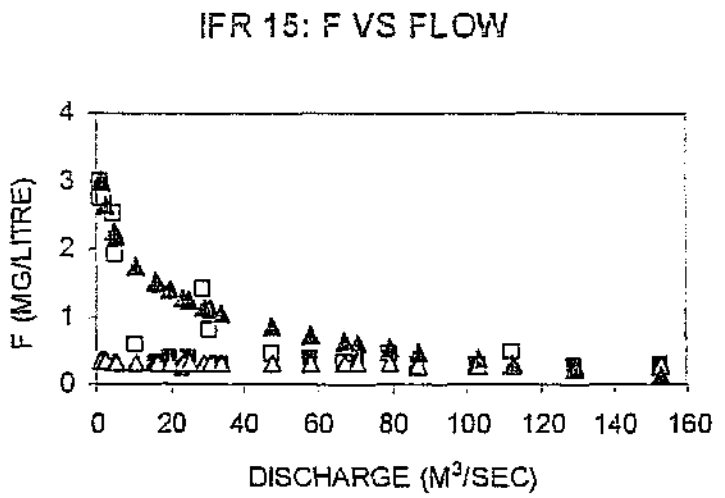

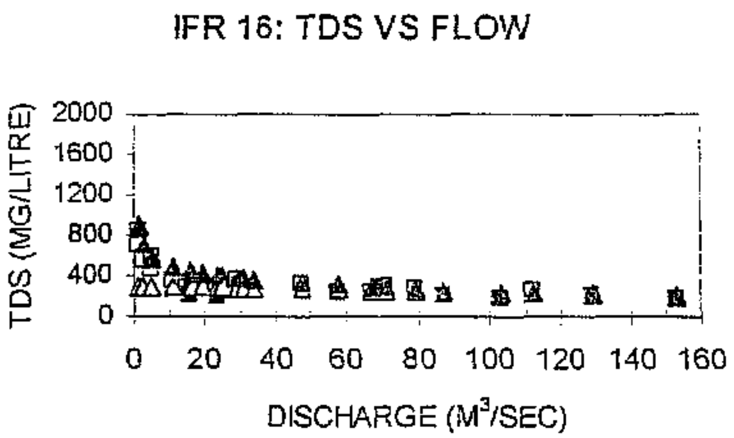

APPENDIX C

List of the gauging sites that were screened for impacted water quality. The % of observations at the site that were above the water quality criteria used for screening sites (EC ≥ 500 mSm, NO_3^- + NO_2^- ≥ 0.5 mg/l and PO_4^{-3} ≥ 0.1 mg/l see text for details) is shown. Also given is the number of records in the data-set. EC = electrical conductivity; NO = NO_3^- + NO_2^-; P = PO_4^{-3}.

Gauging station	% observations: EC ≥ 500 mSm	% observations: NO ≥ 0.5 mg/l	% observations: P ≥ 0.1 mg/l	Sample size (n) EC; NO; P
A6H018	0	0	0	159; 123; 139
A6H019	0	0	0	154; 111; 144
A6H021	0	0	0	46; 26; 34
A6H011	0	1.1	0.4	422; 284; 276
A6H020	0	1.8	0	147; 114; 120
A6H012	0	0	2.0	383; 242; 246
A9H004	0	2.3	1.2	102; 89; 85
A9H001	0	4.2	0.6	180; 167; 164
A4H008	0	4.6	0.4	471; 284; 273
A2H039	0	0	5.0	32; 16; 20
A9H003	0	7.3	0	109; 96; 97
A9H002	0	2.5	4.9	134; 122; 122
A5H004	0	9.3	0	262; 86; 102
A2H029	0	8.4	2.7	1474; 1493
A4H005	0	14.0	0	400; 222; 236
A4H002	0	19.5	3.1	371 262; 290
A2H053	0	30.0	0	277; 197; 180
A2H032	0	43.4	8.3	119; 76; 72
A2H049	0	100	2.9	503; 398; 386
A2H050	0	93	78.3	509; 400; 401
B7H004	0	0.5	0	257, 303, 297
B4H005	0	0.8	0	494, 175, 152
B6H003	0	1.0	0.7	379, 316, 330
B7H014	0	0.9	1.5	208; 108; 131
B6H006	0	0	3.6	308; 142; 139
B6H001	0	3.2	1.0	485; 316; 289
B4H009	0	3.5	1.4	603; 427; 419
B7H010	0	3.8	1.4	245; 133; 142
B7H009	0	14.4	1.9	400; 319; 312
B1H002	0	22.3	1.5	861; 471; 459
B3H001	0	25.2	2.2	454; 369; 358
B5H002	0.5	44.9	1.9	185; 118; 107
B8H010	0	30.7	19.4	490; 267; 294
B7H008	0	50	6.7	16; 16; 15
B1H004	0	64.5	0.6	980; 521; 504
C2H026	0	2.6	1.6	211; 191; 189
C2H028	0	4.6	0.8	434; 368; 377
C1H007	0	2.8	3.3	1029; 858, 847
C5H012	0	2.5	8.2	232; 162; 171
C8H012	0	6.4	4.3	273; 204; 207
C8H022	0	13.2	7.0	79; 76; 71
C8H003	0	17.1	4.3	522; 340; 346
C6H004	0	13.3	17.0	107; 60; 59

Gauging station	% observations: EC ≥ 500 mSm	% observations: NO ≥ 0.5 mg/l	% observations: P ≥ 0.1 mg/l	Sample size (n) EC; NO; P
C3H003	0	29.0	2.4	818; 334; 328
C7H003	0	3.3	32.8	196; 121; 122
C2H067	0	13.7	26.0	103; 93; 100
C5H007	0	4.8	36.8	109; 84; 87
C2H027	0	17.7	27.8	21; 17; 18
C2H065	0	43.2	8.2	565; 509; 501
C6H003	0	27.3	53.1	267; 231; 243
C8H005	0	56.9	42.8	440; 336; 346
C5H008	4.8	35.3	82.4	21; 17; 17
D8H003	0	8.0	2.1	920; 474; 476
D1H011	0	6.1	4.6	620; 360; 367
D2H012	0	7.5	5.0	414; 227; 242
D5H003	0	0	16.7	10; 7; 6
D1H009	0	26.9	3.3	562; 398; 393
D1H003	0	25.5	9.0	999; 907; 889
D7H002	0	45.2	2.8	604; 398; 392
D3H008	0	53.8	1.0	608; 301; 300
D2H001	0	43.3	13.0	110; 97; 92
E1H006	0	0.9	0	285; 118; 115
E2H002	0	0	0.9	167; 115; 112
E2H003	0	2.5	1.2	466; 245; 252
E2H007	0	3.4	3.0	343; 233; 230
G1H016	0	0	0	55; 43; 30
G1H012	0	0	1.2	383; 247; 257
G4H008	0	2.0	0	58; 51; 49
G1H017	0	2.4	0	44; 41; 20
G4H012	0	2.6	0	52; 39; 44
G1H021	0	0	2.6	287; 244; 231
G1H018	0	2.1	0.9	161; 143; 109
G1H011	0	3.1	1.0	131; 96; 101
G4H010	0	2.8	4.3	396; 284; 326
G1H015	0	3.3	4.6	40; 30; 22
G2H008	0.2	4.0	6.1	558; 455; 459
G1H014	0	5.9	5	52; 51; 20
G1H009	0.6	11.0	0.7	163; 155; 151
G5H006	0	13.5	0	41; 37; 39
G1H010	0	22.2	0	77; 72; 69
G1H019	0	21.2	2.0	876; 829; 733
G1H008	0	22.7	0.6	766; 664; 648
G4H014	0	21.9	1.6	656; 187; 185
G4H006	0	28.7	1.5	289; 192; 199
G3H001	24.1	19.7	2.1	245; 188; 190
G2H012	0	48.9	7.3	462; 184; 177
G1H007	0	54.4	14.5	78; 68; 62
G1H003	0	79.1	1.6	494; 249; 246
G5H008	91.1	40.9	14.1	247; 149; 142
H2H005	0	0	0	282; 178; 197
H3H004	0	0	0	82; 35; 34
H6H010	0	0	0	174; 102; 117
H7H005	0	0	0.6	211; 123; 163
H1H018	0	0.3	0.4	914; 791; 818
H9H004	0	0.4	0.4	313; 245; 284

Gauging station	% observations: EC ≥ 500 mSm	% observations: NO ≥ 0.5 mg/l	% observations: P ≥ 0.1 mg/l	Sample size (n) EC; NO; P
H4H012	0	0.5	0.5	316; 192; 204
H1H012	0	1.3	0	194; 78; 84
H5H003	0.3	1.1	0	401; 272; 285
H6H007	0	0.5	0.9	375; 206; 228
H6H008	0	1.0	0.8	378; 194; 238
H1H007	0	0.7	1.3	871; 739; 789
H1H017	0	0.9	1.1	699; 571; 613
H9H005	0	0.8	1.5	327; 130; 136
H9H002	0	0	3.4	150; 60; 59
H7H007	0	1.7	2.4	211; 120; 166
H7H003	0	2.9	2.1	211; 123; 163
H4H005	0	5.0	0.6	177; 160; 160
H7H004	0	4.9	2.9	324; 225; 243
H6H006	0	5.8	4.3	175; 104; 94
H1H013	0	33.5	0	346; 251; 244
H6H003	0	50.0	0	3; 2; 2
H3H005	0	54.1	1.3	145; 85; 80
H2H001	0	84.5	0.9	0; 120; 1
H2H003	0	86.5	1.2	321; 193; 168
J1H015	0	0	1.0	160; 103; 103
J3H013	0	0.9	0.8	370; 230; 247
J3H018	0	1.8	0.6	217; 171; 169
J2H007	0	1.2	1.1	204; 164; 177
J3H012	0	1.0	1.8	341; 207; 217
J2H006	0	2.5	0.6	189; 162; 172
J3H020	0	1.1	2.2	277; 180; 180
J1H016	0	2.3	1.1	226; 173; 176
J4H003	0	1.2	2.6	451; 249; 269
J3H017	0.3	1.1	3.6	340; 188; 196
J2H005	0	3.8	2.4	351; 235; 249
J4H004	0	5.6	1.8	338; 180; 169
J3H016	0	41.0	2.1	336; 244; 234
K3H001	0.3	0	1.6	388; 220; 250
K4H003	0	0.9	1.2	327; 227; 236
K5H002	0	0.9	1.2	395; 234; 251
K6H001	0	0.5	1.8	281; 209; 223
K7H001	0	0	3.3	437; 293; 332
K4H001	0.5	0.4	3.3	403; 260; 299
K4H002	0.2	2.5	1.7	571; 320; 352
K8H002	0	1.8	3.4	415; 284; 294
K3H005	0.3	0.5	4.4	374; 213; 245
K3H003	0.3	3.2	2.9	350; 222; 244
K8H001	0	1.1	5.3	416; 269; 303
K3H002	0.3	0.9	5.7	593; 459; 488
K3H004	0.3	10.7	4.3	385; 215; 231
L8H001	0	0	1.3	352; 243; 234
L8H002	0	0	0.9	323; 207; 222
L7H006	6.2	4.5	1.4	634; 528; 572
L1H001	0	31.0	11.4	38; 14; 12
L6H001	54.4	37.8	57.9	74; 42; 70
N2H009	0	48.3	46.1	771; 694; 701

Gauging station	% observations: EC ≥ 500 mSm	% observations: NO ≥ 0.5 mg/l	% observations: P ≥ 0.1 mg/l	Sample size (n) EC; NO; P
P4H001	8.6	14.4	6.9	29; 29; 15
Q9H014	0	8.4	2.9	191; 131; 140
Q9H002	0	6.1	6.4	316; 261; 264
Q9H016	0	6.0	4.5	343; 215; 221
Q9H019	0	11.3	1.5	425; 335; 335
Q6H003	0	15.5	6.6	400; 341; 347
Q1H001	0	31.8	9.2	736; 289; 293
Q4H003	0	36.5	14.6	314; 211; 212
Q2H002	0	51.2	29.4	901; 377; 395
Q3H004	0	61.2	46.9	443; 314; 326
R1H014	0	1.4	1.0	465; 289; 291
R2H001	0	2.8	1.3	386; 327; 316
R2H008	0	11.5	2.5	295; 253; 241
R2H006	0	18.6	2.0	540; 435; 403
R2H012	0	20.4	2.4	393; 339; 337
R1H005	0	35.0	4.4	310; 306; 298
R1H015	0	41.4	6.4	788; 406; 391
R1H001	0	43.5	5.8	227; 108; 103
R2H005	0	53.1	5.4	355; 305; 294
S6H001	0	2.9	2.6	453; 316; 306
S6H003	0	5.5	5.6	430; 290; 285
S3H002	0	13.0	8.7	249; 154; 161
S3H004	0	11.1	14.1	612; 270; 284
S6H002	0.2	20.4	4.8	629; 427; 415
S3H006	0	26.8	7.2	581; 235; 237
T3H009	0	0.5	0.2	574; 428; 444
T3H004	0	0.4	1.6	363; 239; 249
T5H003	0	2.8	0	407; 215; 213
T5H004	0	2.7	1.3	443; 225; 234
T3H008	0	2.7	1.7	360; 226; 241
T4H001	0	5.7	1.3	358; 246; 237
T5H002	0	22.2	0	20; 18; 19
T1H004	0	41.7	16.7	13; 12; 12
T5H005	0	63.9	1.2	344; 183; 168
U3H002	0	0	0	14; 13; 12
U4H002	0	1.9	1.8	384; 208; 225
U2H013	0	3.3	1.1	760; 491; 374
U6H002	0	5.9	1.4	334; 237; 223
U2H007	0	10.6	0	664; 498; 399
U2H001	0	9.4	2.2	719; 500; 414
U2H006	0	13.6	0.4	791; 574; 510
U7H007	0	13.0	3.6	479; 284; 253
U2H012	0	20.8	0.7	786; 573; 540
U7H001	0	21.9	3.5	203; 96; 86
U3H001	0	59.1	3.5	364; 181; 173
U2H011	0	66.0	1.0	500; 459; 413
V7H017	0	0	0	407; 215; 238
V1H041	0	0.5	1.3	412; 253; 270
V2H007	0	0.4	1.5	363; 215; 224
V3H005	0	1.6	1.4	340; 194; 209
V7H016	0	1.9	1.3	426; 213; 226

Gauging station	% observations: EC ≥ 500 mSm	% observations: NO ≥ 0.5 mg/l	% observations: P ≥ 0.1 mg/l	Sample size (n) EC; NO; P
V6H004	0	2.3	1.3	415; 218; 233
V3H009	0	3.1	0.7	915; 546; 560
V6H006	0	3.0	1.4	318; 133; 144
V1H026	0	2.2	2.3	833; 402; 400
V3H007	0	2.5	2.1	377; 235; 244
V1H029	0	1.8	3.6	241; 113; 112
V1H001	0	3.0	3.0	865; 269; 268
V2H005	0	1.9	4.8	327; 213; 228
V2H006	0	5.8	1.4	356; 206; 215
V1H010	0	3.2	4.4	413; 216; 229
V6H003	0	7.0	2.0	424; 242; 254
V1H038	0	3.2	6.5	461; 254; 260
V3H002	0	9.6	1.3	545; 304; 304
V7H018	0	14.1	1.2	436; 277; 260
V1H031	0	12.4	3.9	347; 194; 206
V1H009	0	12.2	5.2	406; 214; 231
V5H002	0	21.2	8.3	842; 372; 374
V7H012	0	27.3	10.5	394; 172; 172
W5H008	0	0	0.9	384; 199; 215
W5H006	0	0.8	0.8	351; 246; 259
W5H004	0	3.0	0	258; 101; 108
W3H014	0	3.0	1.2	255; 166; 166
W2H006	0	5.3	2.2	982; 549; 555
W4H004	0	6.2	2.2	388; 227; 233
W1H004	0	13.0	0.6	253; 165; 176
W1H010	0	13.3	1.9	381; 211; 212
W2H002	0	25.6	0	40; 39; 38
W2H009	0	44.4	33.0	393; 216; 215
W4H008	0	81.4	24.8	219; 113; 109
X2H014	0	0	0.4	423; 223; 225
X2H010	0	0	0.5	316; 196; 223
X3H006	0	0.7	0.4	446; 270; 251
X2H031	0	0	1.4	381; 195; 211
X2H011	0	0.8	0.9	286; 127; 113
X2H008	0	1.2	0.8	389; 245; 252
X1H003	0	2.5	0	898; 637; 636
X2H027	0	0	2.6	47; 26; 38
X2H024	0	0	2.8	217; 92; 109
X2H005	0	2.9	0.4	444; 277; 265
X3H002	0	3.5	0	421; 255; 215
X3H007	0	0.8	3.0	170; 130; 132
X2H028	0	3.9	0	47; 26; 39
X3H001	0	2.5	1.8	412; 238; 218
X2H026	0	0	5.1	48; 22; 39
X2H025	0	3.2	2.4	47; 31; 41
X2H012	0	2.2	4.0	266; 134; 126
X1H014	0	8.3	0.7	432; 300; 282
X3H004	0	31.7	0	410; 240; 213