LINKING DISCHARGE, WATER QUALITY AND BIOTIC RESPONSE IN RIVERS: A LITERATURE REVIEW

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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.

LINKING DISCHARGE, WATER QUALITY AND BIOTIC RESPONSE IN RIVERS: A LITERATURE REVIEW

EXECUTIVE SUMMARY

This document represents a review of the literature for 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 survey covers the wide field of issues concerned with water quality and water quantity as well as the tolerance ranges of riverine organisms with regard to the various water quality constituents. It considers links between streamflow and the instream concentration of chemical constituents and values of physical variables (in other words, water quality), including methods to make quantitative predictions of water quality from flow. The project considers such predictions in the context of environmental flow assessments (known also as instream flow requirement [IFR] determinations). An important concern is the ultimate effect of shifts in water quality on riverine ecosystem functioning. Thus the effect of changes in the concentration of chemical constituents and values of physical variables (either predicted or measured) on the biota is considered. Methods to predict such effects either quantitatively or qualitatively that have been reported in the literature are also discussed.

For many rivers in South Africa, discharge varies substantially from season to season and water quality is profoundly influenced by flow. Although the methods developed and in use in this country to determine the quantity of water required for ecological functioning are well advanced, thus far the effect of alterations in river discharge on water quality is not adequately addressed in this process. It is important that predictions of water quality resulting from a changed discharge regime be made and taken into account when setting environmental flows since the aquatic biota is profoundly influenced by the water quality to which it is exposed. Thus incorporation of flow-water quality linkage is important in order to

make certain that in addition to the water quantity Reserve, the water quality Reserve is also attained.

The following aspects are discussed in this review:

- Chapter 1 outlines the needs and objectives of the study. Also included is a brief discussion of aspects of the New Water Act of 1998, and how this makes predictions of water quality and the implications for the aquatic biota even more necessary.
- Chapter 2 looks at some of the main features of the most significant water quality variables. Most importantly it considers the interactions between them, explaining how if one is changed this may have an effect on another variable.
- Chapter 3 considers the effect of change in streamflow on water quality and the mechanisms or reasons for this. It includes an outline of some basic hydrological theory and then goes on to discuss the factors (including flow) that determine instream concentrations of chemical constituents. A table is drawn up of the trends reported in the literature between flow and chemical concentration of significant water quality variables. From this, the likely behaviour of water quality variables has been inferred and summarized. Since the flow of many of South Africa's rivers is highly regulated, discharge will often be the consequence of releases from impoundments and to a lesser extent from inter-basin transfers. Thus the effect on water quality of these two types of water resource development is discussed. An important conclusion arising from this chapter is that predictions need to be site specific. Because there are so many factors that can influence water quality it is not easy to make accurate predictions of constituent concentrations.
- Chapter 4 presents a review of the methods that have been used to predict water quality from flow, in other words, water quality modelling. An explanation of some of the terms frequently used in this field is given as well as an outline of how to chose an appropriate model. The basic steps to be used when using a water quality model are discussed. This chapter then goes on to discuss some types of models namely: simple rating curves, more complex computer models and catchment runoff models. Modelling of three specific variables, total dissolved solides (TDS), dissolved oxygen (DO) and

phosphate is discussed. Also included is an examination of the underlying assumptions that are frequently made during the modelling process. A table listing some of the characteristics and attributes of commonly used riverine water quality models is given. The major conclusion from this chapter is that water quality modelling ranges from very simple numerical methods to complex, data intensive computer models. Despite the problems, limitations and approximations in water quality modelling, such techniques can be very useful and will be increasingly important in the management of water resources that have more and more complex and insistent demands made on them.

- The overall objective of the project is to incorporate predictions of water quality (resulting from charges in streamflow) and predictions of implications for the biota (from altered water quality conditions) into environmental flow assessments. Chapter 5 therefore gives a brief overview of the two holistic environmental flow assessment methods currently used in South Africa. This is the Building Block Method (BBM) which is currently used for comprehensive Reserve determinations, and DRIFT, a developing methodology. Chapter five goes on to consider how, and to what extent, water quality is included in the assessments. A review of how water quality has been incorporated into other environmental flow assessment methods, used internationally, is also given. In many cases water quality and flow do not appear to be linked and when they are, how it is done is not explicit or well documented.
- The final chapter (Chapter 6) examines the need for and ways used in the literature to make predictions of the implications of charged water quality on aquatic biota. A summary of the effects of water quality variables on aquatic biota is presented, followed by a discussion of tolerance ranges. The role and importance of biomonitoring is reviewed, especially with regard to SASS the South African system using macroinvertebrates. Methods used in the literature namely: ecotoxicology, multivariate discriminant analysis and ecological models are reviewed with regard to strengths and weaknesses. Above all, the complexity of aquatic systems, the variability in response and the need for caution in making predictions are emphasised.

In conclusion, this review presents the background information necessary for the project. It covers relatively broad fields and several disciplines ranging from hydrology and water quality modelling to ecotoxicology and biomonitoring. The field of environmental flow assessments is growing rapidly. Many cases of the incorporation of flow-quality links probably occur, but are not formally documented or exist in the "grey literature". Although extensive use of the web has been made, it is possible that reports have been missed and the authors give apologies for this and ask that their attention be directed in the right direction. In addition, water quality modelling is a rather specialised and rapidly expanding field and there is a wealth of models available for various applications. This review has concentrated on the overall philosophy of the modelling method rather than the details of particular models. It is intended to be read by biologists and people involved in the management of water resources rather, than specialist water quality modellers. The aim is to give a broad understanding of what water quality models are, how they work and how their output can be used by ecologists. For that reason complicated equations have been omitted and an explanation of terms commonly used in water quality modelling has been included. It is hoped that presenting the literature in this way will help to promote a multidisciplinary view of the complex problem of water resource management. There is a chain of cause and effect between the construction of a proposed water development, altered flow and hence shifts in water quality and the resultant effects on the aquatic biota. It is hoped that this literature review and ultimately this WRC project will help to make people aware of such links and go some way in facilitating predictions, or at least "educated guesses", of the total impact on aquatic ecosystems. This will aid in estimations of the real cost of water resource developments. In addition, it will also ensure that environmental flow allocations can be made effectively so that the ultimate aim, protection of all aspects of the aquatic resource is attained.

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ABBREVIATIONS

BOD – biochemical oxygen demand
C – concentration
DO – dissolved oxygen
DWAF – department of water affairs and forestry
EC – electrical conductivity
EFA – environmental flow assessment
EFR – environmental flow requirement
FRU – Freshwater Research Unit
IBT – inter-basin transfer
IFIM – instream flow incremental methodology
IFR – instream flow regime
LC_{50} – the lethal concentration that corresponds to a cumulative probability of 50% for death of the test population (DWAF 1996)
MDA – multivariate discriminant analysis
Q – discharge/flow
RIVPACS - River invertebrate prediction and classification system
SASS – South African scoring system
TDS – total dissolved solids
TSS – total suspended solids
WRC – Water Research Commission

CONTENTS

CHAPTER 1

INTRODUCTION

1.1 Background

The project for which this literature survey has been commissioned is titled "Development of numerical methods for assessing water quality in rivers, with particular reference to the environmental flow requirements process". This literature survey covers a wider field of issues concerned with water quality and water quantity as well as tolerance ranges of riverine organisms with regard to the various water quality constituents. Also included are aspects of the environmental flow requirements (EFR) process that pertain to water quality. The most important topic covered by this literature synthesis however, is the relationship between water quality and water quantity and how the former changes in response to the latter. Trends that have been reported in the literature in this regard, are summarised. In addition, methods or models that have been used to predict the values of given water quality constituents in response to changes in discharge are discussed. The body of literature covering water quality modelling is extensive however and there is a bewildering array of models available. Therefore discussion of water quality models has been limited to those techniques and methods that are likely to be the most relevant to environmental flows.

This literature review is divided into the following sections:

- **Chapter 1:** General background information including, the new South African Water Law and the ecological Reserve, the variability of South African rivers with regard to flow and water quality and the need for the incorporation of quantitative predictions of water quality into the EFR process. The objectives of the project are also included in this section, as well as a discussion of instream versus catchment processes.

- **Chapter 2:** A brief description of the water quality variables under consideration (temperature, pH, dissolved oxygen, total suspended solids, total dissolved solids, nutrients and toxic substances) as well as the main interactions between them.
- **Chapter 3:** Some basic hydrological concepts and general trends that have emerged from reports of site studies with regard to discharge versus water quality. The factors influencing such trends and the extent to which they are applicable to South African rivers. Effects of impoundments and inter-basin transfer on downstream water quality are also considered briefly.
- **Chapter 4:** Description of the computer/numerical methods that have been used to predict the relationship between water quality and discharge. Definition of some of the terminology used in this field, the types of models available and the choice of an appropriate method. In addition, the quality and type of data that are needed for these methods is discussed.
- **Chapter 5:** The EFR process and water quality. The importance of including water quality in Environmental Flow Assessments (EFAs). Attempts that have been made to link water quantity and quality, both in South Africa as well as internationally.
- **Chapter 6:** The importance and difficulties in making predictions of the implications of changed water quality on the aquatic biota. Important considerations and the types of approaches that have been used in the literature.

1.2 Rationale for the project

Water is a limited resource in South Africa and protection of our aquatic resources is of prime importance. It is perhaps not yet sufficiently widely realised that rivers can perform a self-cleansing function and if protected, can replenish the resource (Dobbs and Zabel 1994; Davies and Day 1998). If rivers are subjected to the discharge of pollutants, invasion of riparian and aquatic habitats by alien plant species, or excessive water abstraction however, ecosystem functioning may be compromised. Water quality is then 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 of 1998. 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 and synergistic toxicity effects between water quality components may become of significance during such periods (Petts and Maddock 1994). For this reason it is especially important that the water quality of aquatic resources is monitored and assessed.

The new South African National Water Act (Act No. 36, August 1998) 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, supplied from a water resource.
- It is also the water *quantity* and *quality* required for maintenance of the ecological functioning of the aquatic ecosystem.

It should be noted that the Reserve has priority and this must first be calculated before the quotas for other water users can be allocated. It is therefore imperative that the Reserve be determined as quickly as possible (DWAF 1999). It is envisaged that estimation of the Reserve will be by means of a two-step process. Firstly, a preliminary assessment (Rapid Reserve) will be made. This can be used for licensing purposes but has to be reviewed after a certain time period. Secondly, a detailed determination (Intermediate or Comprehensive Reserve assessment), which includes an environmental flow assessment needs to be carried out. The ecological component of the Reserve is considered to consist of four aspects, namely; water quantity, quality, habitat (both instream and riparian) as well as the aquatic biota, all of which need to be assessed. With regard to water quality, this will be determined 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 has become degraded with regard to water quality. These values are also compared to those associated with the future Ecological Management Class (where an "A class river" represents one in a pristine condition and a "D class river", one which is highly modified) in order to identify what management actions are required. Incorporation of water quality into the determination of the ecological Reserve is discussed fully in DWAF (1999). See also the end of this document for the Institute of Water Quality Studies web site address from which the latest documentation on the Reserve methodology can be accessed.

Although the new act recognises that water quantity and quality are interdependent and should be managed in an integrated manner (Rabie 1996), the relationship between the two is not at all clear. For many rivers in South Africa, discharge (the amount of water flowing down a river) varies substantially from season to season and from year to year depending on the balance of evaporation and precipitation. River flows of arid lands, including South Africa, are naturally more stochastic than those of mesic areas and are prone to extremes of drought and flood (Davies, O'Keeffe and Snaddon 1993). Regulation of rivers can also lead to alterations in the natural hydrological patterns. Not enough is known concerning the resultant effects of such changes in discharge (either natural or anthropogenic) on water quality variables. Further, even less is known about the implications of fluctuations in water chemistry on aquatic ecosystems. Methods have been developed to determine the quantity of water required for ecological functioning of a river, commonly referred to in South Africa as the Instream or alternatively as the Environmental Flow Requirement (King and Tharme 1994). Thus far however, the effect of alterations in river discharge on water quality is not adequately addressed in the current flow determination protocols (Tharme 1996). According to King and Louw (1998) provision of a suitable flow regime will not result in optimum ecosystem health if the question of water quality is not also considered. Currently during Environmental Flow Assessments (EFA), predictions of changes in the concentrations of chemical constituents as well as of physical variables in response to altered flow are made purely on a qualitative basis. There is an urgent need to incorporate quantitative methods for predicting water quality into the protocol. It should also be noted that in addition to rather variable flow, the concentration of chemical constituents as well as the values of physical variables also show natural variations (Dallas and Day 1993). Davies *et al.* (1993) state that, with regard to the EFR process in South Africa, there is a need to manage for variability not for constancy. It seems likely that since water quality is so dependent on water quantity and is naturally variable this basic tenet may also apply to management of this aspect as well.

The first part of this project concentrates on identifying trends in concentration-flow relationships as well as in relationships between flow and some physical properties of water (e.g. temperature, dissolved oxygen). Methods will be developed in an attempt to predict such relationships quantitatively. The second part of the project is concerned with linking water quality with the requirements of the biota. Just as it is fairly pointless to compile "environmental flows" for a river if the water quality is detrimental to aquatic ecosystem functioning, it is also short-sighted to set discharges to obtain a given concentration of a water quality variable if the effect on the biota is not known. There are limited data with regard to the tolerance ranges of aquatic biota in South Africa. Work on this aspect has been aided by the compilation of a database linking macro-invertebrate species with the water quality in which they were found (Dallas, Day, Musibono and Day 1998). In addition, the use of indigenous riverine organisms for derivation of ecotoxicological parameters is currently under investigation (Palmer, C., IWR, *pers. comm. 1999;* see for example Goetsch and Palmer 1997).

1.3 <u>Instream versus catchment processes</u>

If discharge in a river is reduced, instream concentrations of water quality variables as well as values of physical variables will change (Chapter 3). This is in part due to instream processes, such as for example, a concentration effect in the case of individual salts, as well as settling of sediments in the case of total phosphates. It is however simplistic to consider only instream effects. Processes occurring in the catchment can also exert a profound effect on water quality. A decrease in discharge may result from upstream impoundment or abstraction of water, or may result from reduced precipitation in the catchment. In the latter case this is likely to mean reduced runoff from the catchment and hence reduced sediments (and other materials) ending up in the stream. Because the interaction of catchment and instream processes is complex, this project is largely limited to consideration of instream processes on water quality. Thus, although a consideration of catchment-wide processes that affect water quality is included in this literature synthesis, in the development of modelling methods for predicting water quality, only instream effects are considered. This is acknowledged to be a limitation of the study, but is necessary in the interest of simplicity. This aspect is considered in more detail in connection with modelling in Chapter 4.

1.4 Objectives of the project

The objectives of the project, of which this literature synthesis forms a part, are given below. The results of the project, including how and to what extent the objectives were attained are documented in the final report which is titled "Development of numerical methods for assessing water quality with particular reference to the *Instream Flow Requirement* process" (Malan and Day *in prep.).*

- To address the question of the relationship between water quality and discharge and to compile a synthesis of trends found in the literature.
- To provide a review of the methods developed elsewhere for predicting these relationships as well as the kinds and forms of data needed.
- To develop methods for processing water quality data in order to derive time and flow dependencies for key water quality constituents and to identify and assess methods for modelling selected water quality constituents at the feasibility level of the EFR process.
- To investigate the extent to which available data (e.g. in the Biobase database) can be used for setting environmental water quality requirements as part of an EFR assessment. In other words, to try to predict what the effect of proposed water quality will have on the biota.

- Based on the above, to produce a protocol for the assessment of water quality which can be incorporated into the EFA.

CHAPTER 2

THE WATER QUALITY CONSTITUENTS UNDER CONSIDERATION

2.1 Introduction

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 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 are examined:

- *System variables* - temperature, pH, dissolved oxygen (DO).
- *Non-toxic inorganic constituents* - total suspended solids (TSS), total dissolved solids (TDS), as well as (when relevant), individual ions such as K^+, Mg^{2+}.
- *Nutrients* - phosphates (ortho-phosphate and total P), inorganic nitrogen (nitrate, nitrite, ammonia/ammonium).
- *Toxins and pollutants* - will only be considered in this study to a limited extent. Such water quality constituents are chemically complex and diverse in nature. In addition, limited data is available concerning the levels of these water quality constituents in South African rivers. They are best managed by control at the site of entry into the aquatic system.

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.

From the point of view of water quality modelling, it is important to make a distinction between *conservative* and *non-conservative* water quality constituents. The term "conservative constituent" refers to those components such as chlorides that are essentially unchanged in their progression along a watercourse. Non-conservative constituents on the other hand, which include nutrients such as nitrates and phosphates, are altered in quantity and form as they progress downstream due to chemical inter-conversions, microbial or biotic activities and fluxes with sediments (DWAF 1996).

The nature and measurement of the water quality variables mentioned above, as well as suggested ranges in concentration and values for aquatic ecosystems, have been described in the South African Water Quality Guidelines (DWAF 1996). These aspects will not be discussed in detail therefore, except to outline some of the correlations and inter-connections that exist between the water quality constituents. Similarly, the effects of such variables on aquatic fauna have been reviewed by Dallas and Day (1993) and Dallas, Day and Reynolds (1994) and will be considered only briefly.

It has been shown that for non-impacted rivers, the natural values for system variables, non-toxic inorganic constituents and nutrients vary from region to region. This is a result of varying climate, geology, geomorphology and the biota (Dallas and Day 1993). An additional factor is the chemical composition of rainwater, which varies from place to place and can exert a considerable effect on the chemistry of streams (Weibel, Weidner, Cohen and Christianson 1965; Bishop, Grip and O'Neill 1990). Catchments have been grouped into Water Quality Management Regions (WQMR) on the basis of water chemistry (Day, Dallas and Wackernagel 1998). Concentrations of chemical constituents as well as the magnitudes of physical variables are also modified longitudinally as the character of the water course changes from mountain stream, to foothill, to transitional zone and finally to lowland river (Petts and Maddock 1994;

Dallas, Day, Musibono and Day 1998). In addition, for a given site, some system variables (e.g. dissolved oxygen) can vary with depth (DWAF 1996).

Temporal variation in discharge and water quality is also a feature of South African rivers. It was mentioned in section 1.2 that water quality can vary quite markedly depending on the season and in addition, some water quality constituents such as temperature, DO and dissolved CO_2 also vary diurnally (Dallas and Day 1993; DWAF 1996). Over and above the natural variations in water quality, anthropogenic effects in the form of land-use and effluent discharge are increasingly influencing water quality (De Villiers and Malan 1985). From the above it can be seen that establishment of baseline values for water quality variables in natural systems can be difficult and thus water quality guidelines often have to be site specific (DWAF 1996).

This raises the important question of whether changes in discharge will have the same effect on water quality in all rivers and in all reaches for a particular river, in other words how site-specific such relationships are. One of the objectives of this project is to answer this question.

2.2 System variables

2.2.1 Temperature

Natural fluctuations in water temperature occur on a daily and seasonal basis, and indigenous organisms are adapted to such variations. Anthropogenic activities such as thermal outflow from power stations or cooling waters from manufacturing plants, can result in modifications to the temperature regime (Sweeting 1994). In addition, the building of structures such as impoundments or inter-basin transfers (IBTs), that alter the velocity or volume of water in rivers can also lead to changes in water temperature. The extent to which an upstream reservoir modifies downstream thermal conditions depends on operational variables such as release depth and discharge pattern (Dallas and Day 1993; Davies and Day 1998). The effects of impoundments and inter-basin transfer (IBT) on water quality are discussed in sections 3.7 and 3.8 respectively.

According to Dallas and Day (1993), removal of riparian vegetation and its associated shading effect may also result in temperature changes.

Increases in temperature can enhance the toxic effect of some metals. In addition, oxygen solubility is depressed as temperature increases, which can lead to oxygen stress of susceptible species (Dallas and Day 1993). Temperature however has little effect on pH (Dallas, Day, Musibono and Day 1998). It is thought that the temperature value is less important than the rate of change, and sudden changes can have a significant effect on the biota (Weeks, O'Keeffe, Fourie and Davies 1996). Organisms living in areas within their thermal range may not be affected by small changes in temperature, but for those at the limit of their tolerance range, this additional environmental stress may be critical (Sweeting 1994). Temperature has also been shown to affect sediment transport. Below impoundments for example, if discharge and other variables are kept constant, TSS shows periodic variation. As a result of the higher viscosity of cold water compared to warm, the transport capacity of the water is increased. Sediment transport in such cases has been found be to be higher in winter than in summer, although this effect was noticeable only on fine sediments less than 0.3 mm in diameter (Leliavsky 1955).

2.2.2 pH

In some areas, pH may vary naturally from season to season as a result of the rainfall pattern. Streams of the Western Cape for example, tend to be more acidic during winter, due to leaching of humic substances from the soil during the winter rains (Dallas, Day Musibono and Day 1998). In productive ecosystems, pH also exhibits daily fluctuations due to the competing processes of respiration and photosynthesis as well as changing gas solubilities (Sweeting 1994).

The pH of the water can have a marked effect on the speciation (chemical form) of metal ions and nutrients and hence on bioavailability (Filella, Town and Buffle 1995). Decreasing pH frequently leads to increased levels of metal cations in the water column as a result of desorption from the surfaces of suspended particles. The metals that are most profoundly affected by pH shifts in the 7-4 range include Al, Cu, Hg, Pb and Fe (Sweeting 1994). Low levels of Ca^{+2}, Cl^{-} and Na^{+} can exacerbate the toxic effect of low pH (Dallas, Day, Musibono and Day 1998). Alkaline conditions on the other hand can result in enhanced levels of un-ionised ammonia (NH_3), which is toxic to living organisms. Casey and Farr (1982) reported that the equilibrium between phosphate bound to sediments and that dissolved in the water column was sensitive to pH. As pH increased, phosphate was displaced from the sediment and the concentration in the water increased. In general, rivers with low TDS tend to be poorly buffered and are therefore very sensitive to even slight changes in pH (Sweeting 1994).

2.2.3 Dissolved oxygen (DO)

The dissolved oxygen content of the water column is a useful indicator of the biotic integrity of an aquatic ecosystem. Diurnal variation occurs due to the competing processes of respiration and photosynthesis, as well as due to the effect of temperature on gaseous solubility. As a result of the uptake of oxygen by living organisms during respiration, DO levels in natural water bodies are at a minimum at dawn. During daylight, plants photosynthesize, releasing oxygen and resulting in maximal concentrations at mid-afternoon in eutrophic systems (Dallas and Day 1993).

Factors that can affect the concentration of DO include temperature. Increasing temperature decreases the solubility of oxygen in water (Wetzel 1983). Other factors include lake turnover, or release of anoxic benthic sediments from reservoirs which can lead to oxygen depletion downstream. Re-suspension of anoxic sediments during flood events, or the presence of excess oxidizable organic matter (either anthropogenic or natural) which lead to blooms of heterotrophic bacteria, can also lead to low oxygen levels (DWAF1996). Metals (eg. iron and manganese), as well as sulphides, can appear in solution under conditions of oxygen depletion, a situation that may be exacerbated at high temperatures (DWAF 1996). Sudden decreases in the DO content of a water body ("anoxic events") may have a severe impact on living organisms, especially animals, leading to death (Wetzel 1983), for example as fish-kills. Situations in which low flow conditions in rivers lead to pools of standing water can be especially dangerous in this regard. The diffusion rate of oxygen between air and water has a marked effect on the oxygen content of the water and this in turn is strongly influenced by turbulence, which is a function of current speed and stream bed characteristics (Campbell 1982). Special aeration weirs are sometimes built to increase turbulence in rivers and thus increase the DO concentration. Increasing salinity also reduces the solubility of oxygen (Wetzel1983; DWAF 1996).

The natural purification processes occurring in streams must be taken into account when modelling water quality. Clean stream water is usually saturated with DO but the concentration decreases as sewage or other organic matter is added. Bacteria responsible for the breakdown of organic matter consume DO in the process. Oxygen from the atmosphere dissolves into the water as it is no longer saturated with DO. Finally the organic matter is completely decomposed and the stream water becomes saturated with oxygen again (Gromiec, Loucks and Orlob 1983).

2.3 Non-toxic inorganic constituents

2.3.1 Total suspended solids (TSS)

In many rivers there is an increase in TSS with enhanced discharge (such as during a storm) due to re-suspension of sediments within the channel and run-off from the land. This is a natural phenomenon for rivers in South Africa, which tend to become turbid during the rainy season (DWAF 1996). Even the streams of the Natal foothills and the south-western Cape, which tend to carry low sediment loads, become turbid transiently after the first spate in the rainy season or during a major flood event. This effect can be exacerbated however by poor land management practices in the catchment (Kelbe and Snyman 1993; Quibell, van Vliet and van der Merwe 1997). As turbulence and mixing energy decrease (which usually results as flow velocity is reduced), suspended matter settles out and the concentration of TSS in the water column decreases (DWAF 1996). Changes in the quantity of suspended particles can have effects on the concentrations of inorganic ions dissolved in the water column (Dallas *et al.* 1998). Phosphate, for example, binds to such particles, sometimes resulting in deficiencies of this nutrient in rivers carrying unnaturally high TSS loads. In general, sediments are of major importance in determining nutrient availability in the water column (Sweeting 1994).

This ameliorating effect may be reversed however, if sediments are disturbed or other system variables such as pH are changed.

High TSS levels can also decrease water temperature. This is due to reflection of heat by the suspended particles resulting in reduced absorption by the water molecules (DWAF 1996). An important correlation is that between TSS in the water column and the extent of light penetration in a water body. Reduced light penetration limits the potential for primary production by phytoplankton and rooted macrophytes because the depth of the photic zone (the zone in which enough light is available for photosynthesis) is limited (Grobler, Toerien and Rossouw 1987). Changes in this aspect can thus have major effects on the biota (Dallas and Day 1993). Reduced flow often leads to enhanced autotrophic production resulting in increased biomass of benthic algae, floating algae and submerged angiosperms (Petts 1989). According to Grobler, Toerien and Rossouw (1987), in a system enriched with nutrients such as the Vaal Dam, high turbidities can be considered to be beneficial in limiting excessive algal growth. These authors state that increasing salinity has been shown to cause the flocculation of suspended sediment and can lead to an increase in the depth of the photic zone.

Monitoring of TSS is not carried out on a routine basis in South Africa. The availability of data on sediments (in addition to pathogens, metals and other constituents) is highly uneven and where they are available there are often missing values (Pegram and Görgens *in press*). One of the major reasons for this 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 described above) as well as on aquatic biota (Sweeting 1994). It is therefore important that this variable should be measured in order to be able to model and predict changes in water quality and ecological functioning in a comprehensive manner.

2.3.2 Conductivity and total dissolved solids (TDS)

TDS is a measure of the quantity of all compounds dissolved in the water. Conductivity on the other hand, is a measure of the ionised substances within the water and represents a subset of the dissolved components. TDS and conductivity are therefore usually strongly correlated for a particular type of water and can be interconverted (Dallas and Day 1993). Increases in TDS (due either to natural causes or to pollutants) may lead to lowered oxygen levels. In addition, an increase in TDS may be accompanied by an increase in nutrient levels with the associated risk of enhanced algal growth. It should be noted that salinization (the process whereby the concentration of TDS in inland waters is increased) has been identified as a potentially serious problem in several South African rivers (Hart and Allanson 1984, Kirchner 1995).

2.4 Nutrients

Eutrophication, or excessive accumulation of nutrients in a water body, is an increasingly serious problem world-wide (Sweeting 1994). From the point of view of water use, one of the most serious implications of eutrophication is the increased risk of excessive algal growth (Ridge, Pillinger and Walters 1995). Other characteristics of eutrophication include detritus accumulation, exhaustion of DO and fish-kills (Chen 1970). In South Africa, as in many other countries, bioavailable phosphate in unimpacted waters is usually limiting to plant growth (Wiechers 1985). Thus considerable attention has been paid to phosphate levels in streams and rivers since it is thought that by controlling this nutrient, excessive algal growth may be limited (Cahill, Imperato and Verhoff 1974; De Jong, De Oude, Smits *et al.* 1989). As mentioned previously, this nutrient tends to bind to sediments. Studies have shown that the ratio of inorganic nitrogen to phosphorus is critical for healthy functioning of aquatic ecosystems. Un-impacted streams typically have an inorganic N:P ratio ranging between 25 to 40:1 whilst most impacted streams have an inorganic N:P ratio of less than 10:1 (DWAF 1996). Changes in stream flow which affect the concentration of phosphate therefore, may also change the nutrient status of the ecosystem and hence the composition of the biota.

Nitrogen is present in aquatic ecosystems in several inter-convertible forms namely nitrate, nitrite, ammonia, ammonium and organic nitrogen. Nitrite and ammonia are both toxic in high concentrations. Formation of the latter is enhanced by conditions of elevated pH and temperature (Dallas and Day 1993).

From the point of water quality modelling, nutrients are particularly challenging – because of their ability to inter-convert between different forms due to chemical and microbial action. Various factors, notably temperature, DO, pH, as well as the biomass and activity of bacteria, affect the rate of these transformations and usually need to be taken into account during the modelling exercise (Chapter 4). In addition, nutrients are taken up by plants such as macrophytes and algae. As in the case of biological transformation, plant uptake is a function of several factors, including the levels of other essential nutrients, plant species and abundance, light, temperature and pH (Holtan, Kamp-Nielsen and Stuanes 1988).

2.5 Toxins and pollutants

2.5.1 Organic enrichment

Organic enrichment, in addition to being one of the most modelled water quality constituents (section 4.7.1), is also probably one of the most common and best documented forms of pollution occurring in rivers (Dallas and Day 1993). This type of pollution is usually due to a heterogeneous range of chemical compounds, the major effect of which, as mentioned previously, is usually a decrease in ambient oxygen levels due to microbial degradation. Organic enrichment is often quantified therefore in terms of BOD (biochemical oxygen demand) or COD (chemical oxygen demand), which are measures of the rate at which oxygen is consumed (DWAF 1996). Other potential effects of organic enrichment include increased turbidity, suspended solids and nutrients (Dallas and Day 1993).

2.5.2 Toxic substances

The term "toxic substances" represents a very diverse group, from metal pollutants to organic substances including pesticides and whole effluents. These chemicals frequently enter water bodies as a point source and as mentioned in section 2.1, such water quality constituents are best managed at source. Toxic substances, such as agricultural pesticides, that 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 is 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.

CHAPTER 3

GENERAL TRENDS IN CONCENTRATION-DISCHARGE RELATIONSHIPS

3.1 Introduction: Basic hydrological concepts

Some basic hydrological concepts are presented briefly in this section. For more comprehensive explanations, the reader is directed to works such as Gray (1962), Gregory and Walling (1973) and Gordon, McMahon and Finlayson (1992).

Precipitation falling within a catchment may travel towards a water-course by means of four different pathways, namely, as surface run-off, as throughflow and interflow in the soil and lastly as baseflow through the underlying substrate (Watson and Burnett 1993). A diagrammatic representation of the different pathways is shown in Fig. 3.1. Whether the water infiltrates into the soil or flows away from the area as surface flow depends on the intensity and duration of the rain as well as the storage capacity of the substrate (Gray 1962), which in turn is dependent on the type of substrate (e.g. sand, clay). As the soil of the catchment becomes saturated, the water draining into streams becomes increasingly derived from the upper horizons (Billett and Cresser 1992).

A hydrograph is a graphical representation of the instantaneous rate of discharge (Q) with time (T) within a watercourse and includes the integrated contributions from groundwater, sub-surface flow, surface runoff and channel precipitation (Gray 1962). For any stream, the nature of the hydrograph that is produced by a single, short-duration storm occurring over the drainage area follows a general pattern. A typical hydrograph is shown in Fig. 3.2.

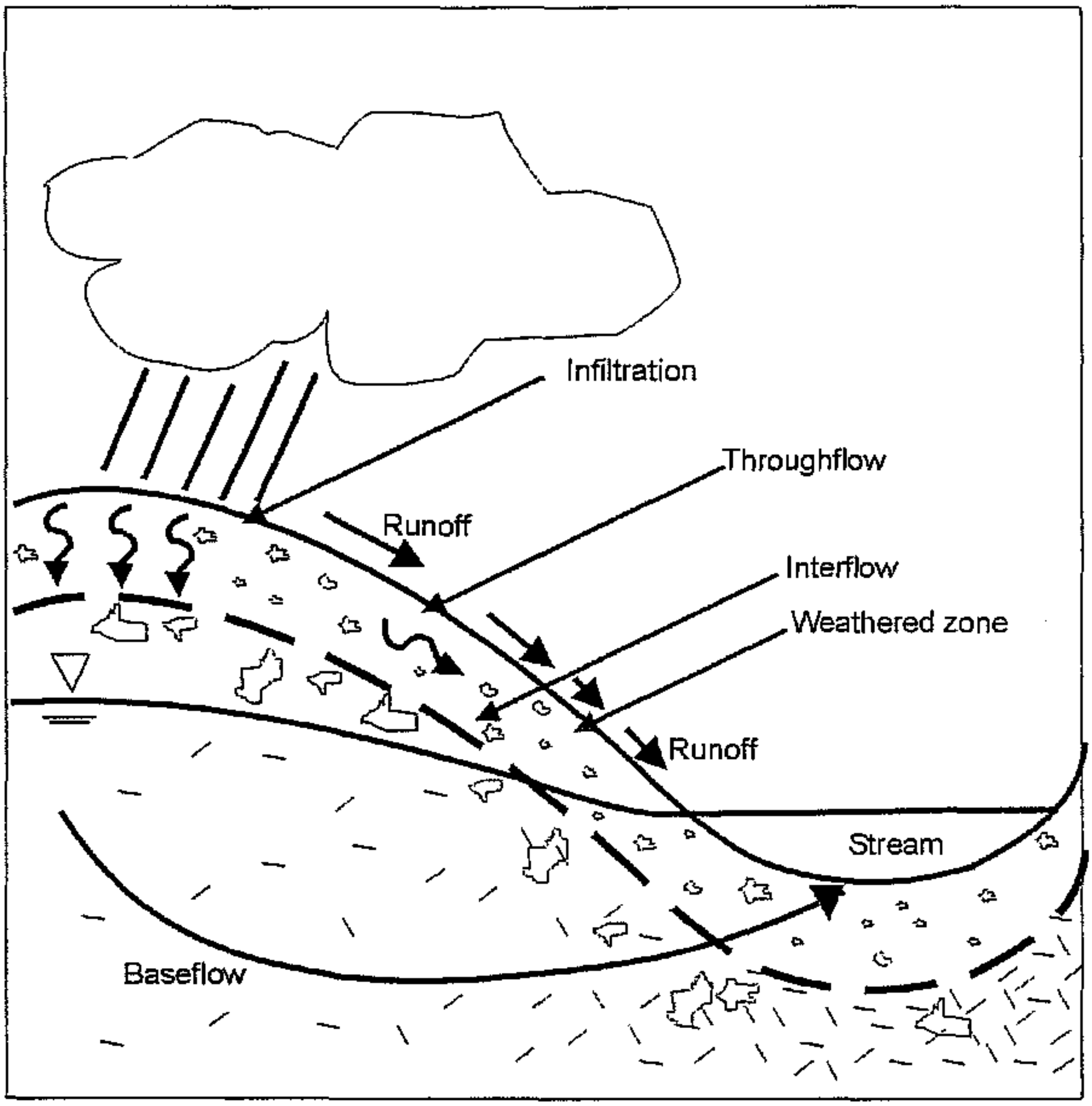

***Figure 3.1** The portion of the hydrological cycle that is pertinent to stream flow. Water present in streams is mainly a function of:*

- ***Runoff** – direct surface water inflow to the stream.*
- ***Baseflow** – ground-water discharge to the stream.*

Lesser volumes of water are derived from

- ***Throughflow** – unsaturated flow through soil*
- ***Interflow** – unsaturated flow, in a concentrated form at the base of a weathered zone or along a fault.*
- ***Direct precipitation** – rain falling directly into the stream.*

(Adapted from Watson and Burnet 1993)

The proportion of the hydrograph due to each of the flow components will vary. In small catchments, or those that are heavily urbanised (and thus have "hardened" surfaces), much of the contribution to the hydrograph will be due to surface runoff (Gray 1962). During long dry periods however, groundwater accounts for almost all the water flowing in streams (Watson and Burnett 1993). Hydrographs for a particular watercourse may be constructed for short periods, such as for a storm, for an entire season, or for an even longer period of time (Watson and Burnett 1993). The shape of the hydrograph is influenced by factors such as the duration and intensity of precipitation, antecedent moisture content of the soil as well as physical characteristics of the catchment, including size, shape and topography (Visocky 1970; Gregory and Walling 1973). In addition, the geology of the route through which water must pass before reaching the stream will be instrumental in determining the form of the hydrograph as well as exerting a profound effect on the chemical content of the water (Johnson and East 1982). There has been considerable debate concerning the most accurate method for determining the groundwater, or base flow component of storm discharge (Pinder and Jones 1969). A general approximation for graphically separating base flow from surface runoff is to draw a horizontal line across the storm hydrograph (Fig 3.2), from the point where the rising limb of the hydrograph begins, to the equivalent point on the receding limb (Visocky 1970).

In a similar vein, the change in concentration of a chemical substance with time can also be plotted for a given point on a watercourse (Fig. 3.3) and is sometimes referred to in the literature as a "chemograph" (Overton and Meadows 1976). At a given point along the length of a stream, the peak of the chemograph may be coincident with that of the hydrograph. Alternatively, maximum concentration may occur before or after the peak discharge value (Walling and Foster 1975). Extensive research has been conducted into the effect of fluctuations in discharge on the concentrations of mineral ions. One of the primary reasons (see also the next section) for such work has been to investigate hydrological pathways and to separate the hydrograph accurately into baseflow, interflow and surface flow (Cooke and Dons 1988; Pinder and Jones 1969).

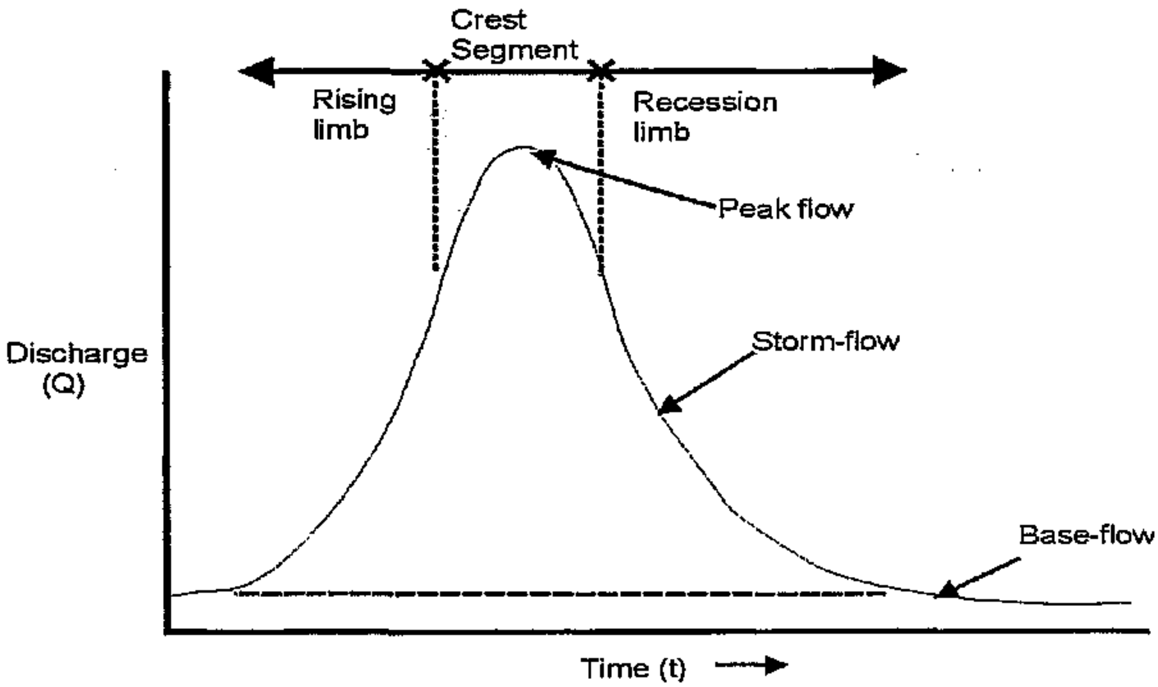

Figure 3.2 *A typical hydrograph showing the rising limb (increasing discharge), peak flow and recession limb (decreasing discharge). The portion of the hydrograph originating from baseflow is also indicated. (Adapted from Gray 1962).*

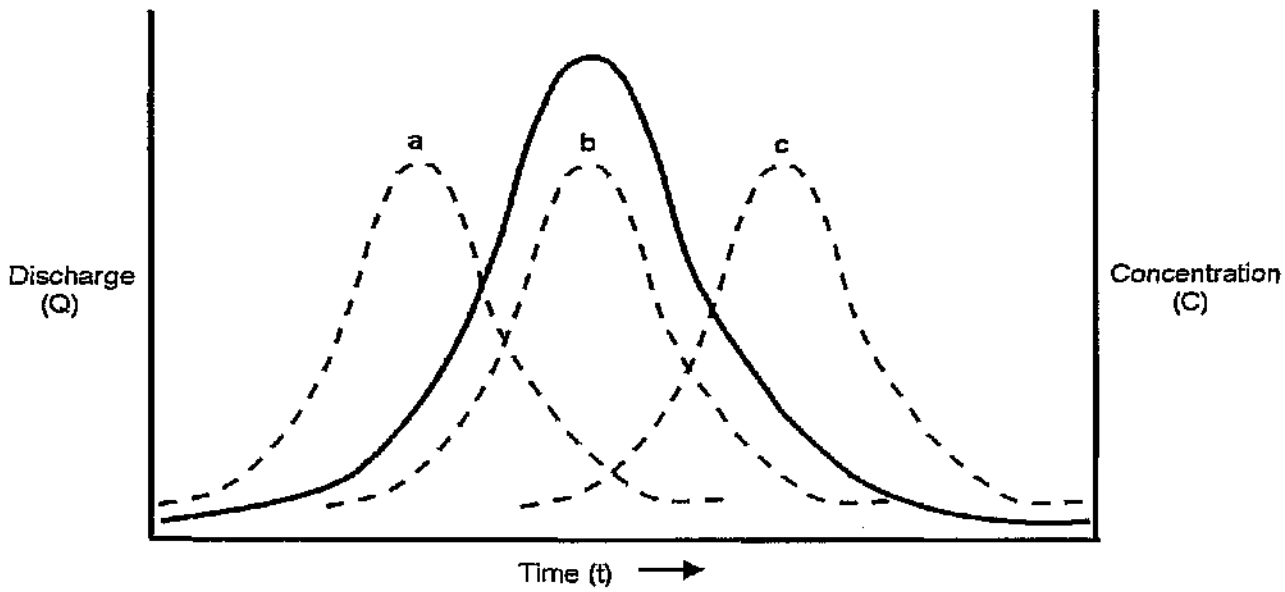

Figure 3.3 *A typical chemograph showing the change in concentration of a chemical substance at a given point on a watercourse. Note that the concentration peak may be before (a), coincident (b) or after (c) the discharge peak. Discharge ——— ; Concentration - - - - - - - (Overton and Meadows 1976).*

3.2 Generalised concentration-discharge (Q-C) relationships

For a given location on a stream, a graph can be plotted of the concentration of a particular chemical constituent (for example TSS, K^+, or TDS) over a range of discharge levels. Examples of the general form such flow-concentration (Q-C) plots may take are given in Fig. 3.4. The mathematical relationship between discharge and solute or sediment concentration is commonly called a rating curve and is site specific (Sidle 1988). Calculation of such relationships can be useful in making predictions of concentration for a given discharge, as well as for estimating solute or suspended particle loads. Additionally, in the case of sediments, such rating curves can provide an index of catchment condition as represented by the extent of erosion (Gregory and Walling 1973).

An inverse relationship between solute concentration and discharge has been described for many rivers and is to be expected with increasing discharge if all other factors affecting river chemistry are constant (Fig. 3.4 a). This is due to the fact that baseflow, which is usually relatively rich in solutes, is diluted by interflow and surface flow which have shorter residence times in the catchment and hence a shorter period in which solution reactions can take place (Toler 1965; Edwards 1973). In situations where one or more relatively constant point sources are discharged into a river, an inverse relationship with discharge can also be expected. Such a relationship may be linear (i) or be better described by a logarithmic function (ii). Other water quality constituents (Fig. 3.4 b), notably sediments, usually exhibit positive slopes, indicating an increase in concentration in response to enhanced flow (Brooker and Johnson 1984). Such increases in concentration are frequently attributed to a wash-off effect (i.e. the flushing of accumulated products from the soil into the river) which is greater than the dilution effect (Wang and Evans 1970; Britton *et al.* 1993). Churning of the riverbed resulting from increased turbulence at high discharge may also lead to an increase in suspended solids and other constituents. Rating curves have also been reported that show a tendency for the upper segment to flatten off (Fig.3.4 c). Such a trend may be explained by the fact that above a certain discharge value, sediment

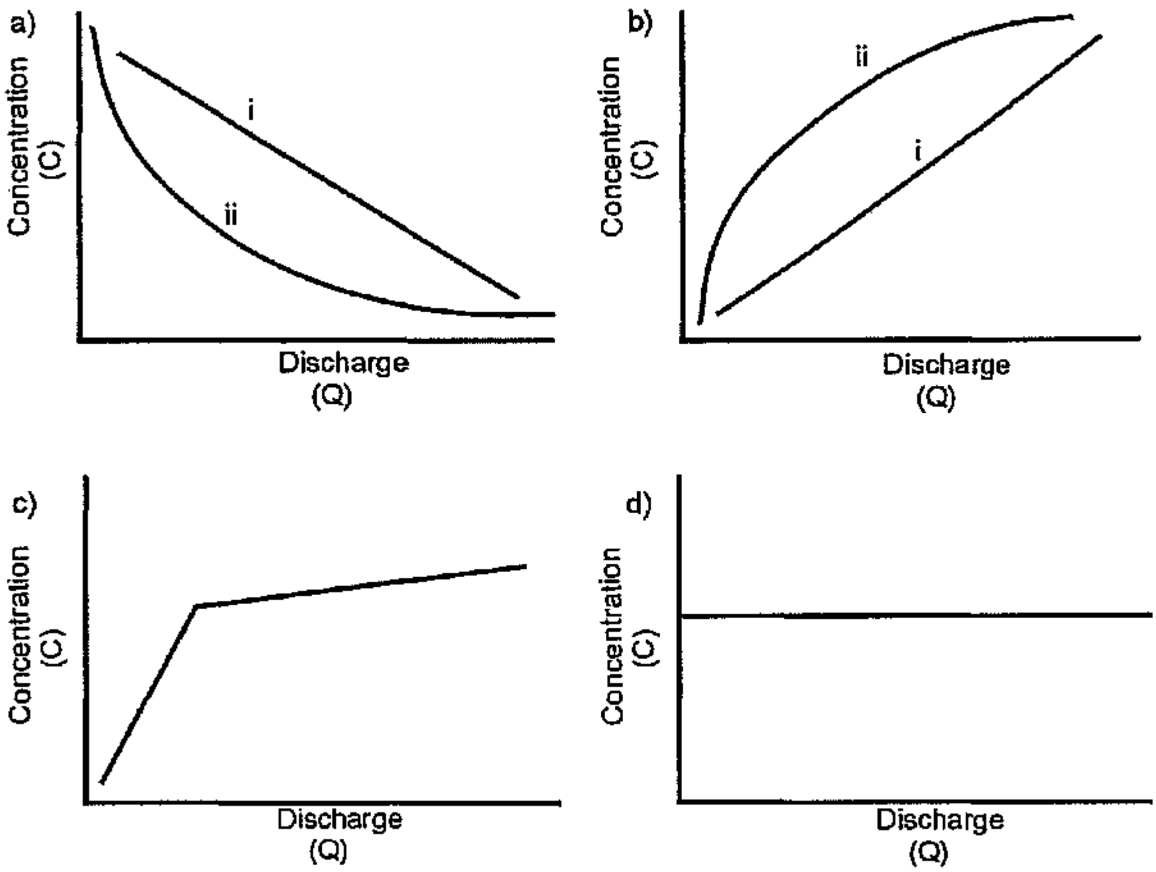

Figure 3.4 *Forms of flow-concentration (rating curves) that have been reported in the literature. As discharge (Q) increases concentration (C) may either decrease (a), increase (b) or initially increase followed by a levelling off effect (c). Alternatively, the concentration of some substances may be independent of flow (d). (Adapted from Smith et al. 1996).*

concentrations for example, do not continue to increase, since the rate of supply of this constituent becomes limiting (Gregory and Walling 1973).

The concentration of other water constituents may remain constant, despite changes in discharge (Fig. 3.4 d). Such a response indicates the action of buffering mechanisms that compensate for changes in flow (Johnson, Likens, Bormann *et al.* 1969). Finally, in streams where solute sources and runoff dynamics are complex, some water constituents may show little or no apparent correlation with discharge (Taylor, Edwards and Simpson 1971; Naiman and Sibert 1978; Karlsson, Grimvall and Lowgren 1988). At a given site, individual water constituents may exhibit different Q-C relationships (Gregory and Walling 1973). The mathematical relationships of rating curves are discussed in more detail in the next chapter.

Rating curves often exhibit considerable scattering, a consequence of the many factors that affect concentration-discharge relationships (Gregory and Walling 1973). Several authors (Johnson and East 1982; Davis and Keller 1983, amongst others) have attempted to elucidate the reasons for this variation by plotting, for a given catchment, the concentration at a given discharge with time (Q-C-T plots). Comparison of C/Q values for the rising limb of the hydrograph with those obtained for the falling limb of the hydrograph have frequently shown that such ratios do not have the same value. In other words, the relationship between concentration and discharge is frequently hysteretic. Hysteresis can be defined as the phenomenon in which two physical quantities are related in a manner that depends on whether one is increasing or decreasing in relation to the other (Oxford 1991). Various mechanisms appear to be responsible for this effect which is often exhibited between flow and concentration (Davis and Keller 1983) and are discussed below. Such plots may be expressed in the form of idealized cyclical patterns, which are thought to provide an index of the general hydrogeological characteristics of a catchment area (Johnson and East 1982). Williams (1989) identified five common classes of Q-C-T relationships exhibited by sediments, including the classical clockwise hysteresis loop (Fig. 3.5). Note that in this work, the direction of the hysteresis loops refers to graphs in which discharge is plotted on the x-

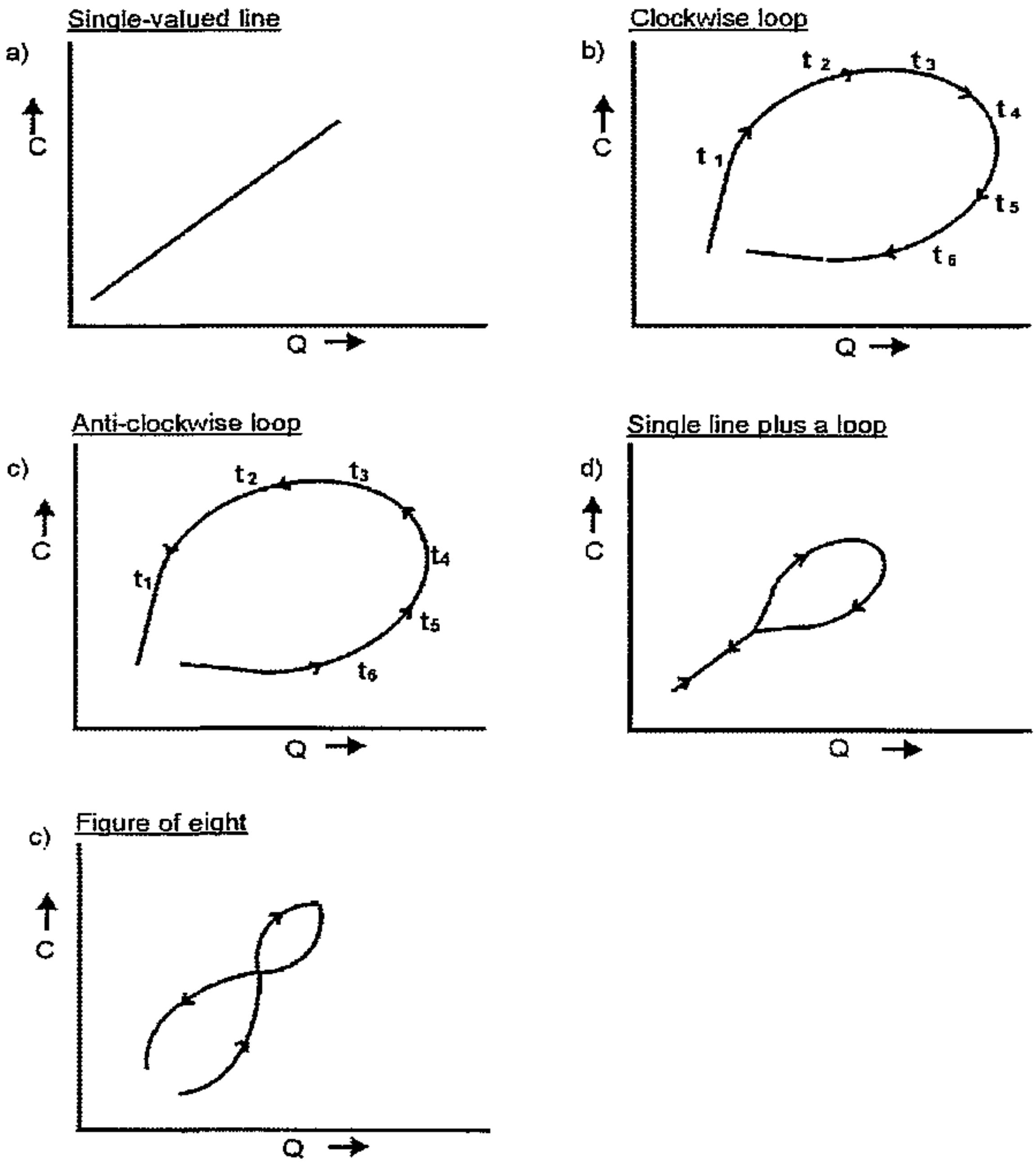

Figure 3.5 *Types of C-Q-T (concentration-discharge-time) relationships reported in the literature (e.g. Williams 1989). These may be a single-valued line (a) a clockwise (b) or anticlockwise (c) loop or may have a more complex form (d)and (e). C = concentration, Q = discharge,* t_1, t_2 *etc. = successive points in time.*

axis and concentration on the y-axis). Other water quality constituents, other than TSS, have also been found to yield similar graphs when concentration is plotted against time and flow (Davis and Keller 1983; McDiffett *et al.* 1989).

The relationship between discharge and concentration may be observed for the duration of a single pulse in stream discharge whether this spans a season or a much shorter period (Bond 1979). Thus seasonal cyclic hysteretic effects are analogous to those exhibited for a single storm event (Gregory and Walling 1973) and both long-term data and data from high-flow events may be analyzed in this manner. For example, Bruijnzeel (1983) examined the change in Si concentration during a tropical storm, whilst other authors (Jenkins 1989) have examined Q-C relationships over a season or throughout an entire hydrological year.

The five types of Q-C-T relationships (Fig. 3.5) as classified by Williams (1989) are described below. The different types are reviewed only briefly and the above work should be consulted for a more comprehensive account.

a) Single line:

This is the simplest type of Q-C relationship and is equivalent to the rating curves shown in Fig. 3.4. For each discharge (Q) value there is only one concentration value and no hysteresis is apparent. Thus C/Q values for the rising limb of the hydrograph are equal to the C/Q values for the falling limb ($C/Q_r \cong C/Q_f$). Within this group, as described previously, variations may occur in that the line may have a positive, negative or zero slope (Bond 1979). In the case of sediments, such Q-C relationships have been associated with an uninterrupted sediment supply throughout the hydrological event. Levelling off is considered to be due to limitation in the supply of this constituent.

b) Clockwise loop

If the C/Q values for the rising limb of the hydrograph are higher than for the falling limb (i.e. $C/Q_r > C/Q_f$), a plot of Q-C with time will be in the form of a generalized clockwise loop (Fig. 3.5 b). According to Williams (1989) and Miller (1997) such loops are common and result when the sediment peak at a site arrives before the discharge peak (as in Fig 3.3 a). In such cases, either sediment is depleted by the time of flood recession or conceivably, armouring (hardening) of the streambed has occurred by this stage, diminishing availability of the resource. Such hysteresis loops were obtained for suspended sediments on the Ausable River, Canada by Irvine and Drake (1987). Williams (1989) reported that clockwise loops tend to be more frequent at the beginning of a storm or runoff season when sediment availability is greater. McDiffett *et al.* (1989) constructed nutrient-discharge trajectories and found that patterns could be predicted for a first-order, nutrient-rich, Northern Hemisphere stream. Nitrate and phosphate showed an initially positive correlation with discharge, which was weak and was due to the flushing effect of the rapidly increasing discharge. The levels of these nutrients in the stream water then decreased due to dilution. Such a pattern resulted in a general clockwise trajectory.

c) Anti-clockwise loop

If the concentration peak arrives later than the discharge peak (i.e. $C/Q_r < C/Q_f$) an anti-clockwise loop will be formed when plotted against time. According to Williams (1989), although few examples have been published of this type of Q-C relationship, such a graph could conceivably result from any of at least three causes. Firstly, since the sediment flux tends to lag behind the flood wave (Fig. 3. 3 c), higher sediment concentrations during the falling limb are possible. Secondly, this type of Q-C relation has been reported in areas of high erodibility in conjunction with prolonged erosion during the flood. A third cause of anti-clockwise loops is seasonal variability of rainfall and thus of sediment production within a catchment. According to Walling and Foster (1975), lags in the chemograph behind the peak of the hydrograph are associated with dry antecedent conditions. Maximum lag values were thus measured after or during the dry season and minimum values in the wet season.

Toler (1965) reported this type of hysteresis effect for dissolved solids in Spring Creek, Georgia. When discharge increased from that which characterised base flow, the concentration of dissolved solids decreased, rapidly at first, then at a slower rate until a minimum concentration was reached, usually just before peak discharge. Thereafter dissolved solid concentration increased as discharge decreased.

d) Single line plus a loop

This type of Q-C-T graph combines class a with b or c and thus the reasons for the above are also applicable. The single line at the lower discharges indicates that at the beginning and end of the hydrograph sediment concentration varies directly with flow. At higher Q values, other factors are involved producing a hysteresis effect.

e) Figure-of-eight

At low discharge values a cyclic hysteresis effect is in operation and at higher Q values the effect is in the opposite direction (Fig 3.5 e). Such relationships arise from complex interaction of factors and do not appear to be common (Williams 1989). An example of such a complex Q-C-T graph was exhibited by Si in the Kali Mondo basin (Java). The author however (Bruijnzeel 1983) did not speculate on possible reasons for the phenomenon. Potassium (K^+) concentration in a small perennial stream in New South Wales, was also found to exhibit a variation on the figure-of-eight pattern. This ion decreased in concentration during the initial part of the rising limb of the hydrograph due to a dilution effect, but then increased around the time of peak discharge. The author (Cornish 1982) attributed this increase to high concentrations of these ions in peak surface runoff water that resulted from dissolution out of organic sources present in the forest litter and in tree canopies.

3.3 Other factors affecting concentration-discharge relationships

Concentration-discharge relationships are complex and influenced by a wide range of factors (McDiffett, Beidler, Dominick *et al.* 1989). Antecedent rainfall patterns in a catchment are of particular importance in determining the hydrogeochemical response (Johnson and East 1982). The moisture content of the soil determines the flow path of water entering a river and whether the water is primarily from surface flow, interflow or groundflow (Britton *et al.* 1993). The chemistry of the water entering into a watercourse is a consequence of the nature and geology of the flow path it has traversed. In addition, the length of time between rainfall events, the amount of precipitation and the chemical composition of rainwater, as well as seasonal climatic effects, can each influence the response of a particular constituent to flow (Neal, Christophersen, Neale *et al.* 1988; McDiffett *et al.* 1989). According to Williams (1989), in the case of sediments, other factors of importance include, the areal distribution of precipitation within a catchment, travel rates and distance of floodwater, spatial and temporal storage-mobilization-depletion processes of available sediment as well as sediment travel rates over distances.

Finally, within-stream disturbance should also be considered when studying discharge-concentration relationships. Casey and Farr (1982) examined the effect of discharge on the concentration of several water quality variables in the absence of allochthonous (i.e. exterior or off-land) inputs. This was achieved by examining Q-C relations downstream of a lock on a Dorset river. Artificial spates were caused by releasing water by means of the lock. Suspended solids, BOD, bacterial counts and K^+ all increased with discharge. Calcium and Na^+ on the other hand were not affected. Nutrients showed a variable response. The authors concluded that the observed variations in water chemistry were due to within-channel effects. They attributed such variations to disturbance of sediments, with concomitant release of interstitial water and exposure of sediment surfaces to adsorption/desorption reactions.

The wide range of factors that influence Q-C relationships, make predictions of concentration at a given discharge value unreliable (Williams 1989) and any attempt to explain them must consider processes operating over the entire catchment (Walling and Foster 1975). Furthermore it should be emphasised that whatever the relationship between concentration of a given chemical constituent and discharge, this is likely to be very site specific and in addition will change if significant changes in the catchment (eg. landuse) occur. According to Meybeck (1996) water quality in large rivers is temporally less variable than in streams. This is due to the fact that the hydrological regime also varies less as the size of the watercourse increases.

3.4 Concentration-discharge studies in the literature

A substantial body of literature exists that addresses the relationship between discharge and various water quality variables. Many of these document empirical studies concerned with changes in flow and the resulting concentrations of chemical constituents. These studies can be sub-divided according to the nature of or reason for the change in discharge:

- Seasonal changes in flow (e.g. Jha *et al.* 1988; Burrus *et al.* 1990).
- Short-term, high flow events, such as storms (e.g. McDiffett *et al.* 1989; Britton *et al.*1993).
- Pre- and post-impoundment studies (e.g. O'Keeffe, Palmer, Byren and Davies 1990).
- Experimental alteration of discharge (e.g. Tharme and King 1998).
- Inter-basin transfers (IBTs) (e.g. Snaddon, Davies and Wishart 2000).

The reason for the change in discharge (the flow modifier) is critical in determining the effect of such fluctuations on water quality. Changes in water quality due to seasonal changes in flow may be quite different to those resulting from the influence of an upstream impoundment. Construction of impoundments usually alter not only water flow, but also nutrients and sediments (see section 3.7) and the exact effect will be dependent on the type and operation of the dam (reservoir). To look at the effects of changing discharge on water quality therefore, without taking into account the nature of the change, is simplistic. In this literature synthesis, we are to a large extent interested in decreases in discharge due to abstraction/impoundments as well as to natural seasonal changes and the associated fluctuations in water quality variables. To a more limited extent, the effect of inter-basin transfers (IBT) on donor and receiver rivers is considered.

A distinction should also be made between concentration (i.e. the amount or mass of a substance per unit volume) and load. The load of a chemical substance can be defined as the integral of the volumetric flow multiplied by the concentration over the specified period of time at a given point on a river (Verhoff, Yaksich and Melfi 1980). Put more simply, it is the total amount of a chemical substance carried by a river per unit time. Naiman and Sibert (1978) found that the rise in the autumnal hydrograph of a Vancouver river did not lead to increased concentrations of total phosphate, nitrate or ammonia, although maximal loading occurred during this period because flow volume was very high. From the point of view of the biota and other users of river water, the concentration of a solute is usually more important than the load and thus the latter will be largely ignored in this review.

Studies in the literature that have examined the effect of increased discharge on specific water quality variables are summarised in Table 3.1. The location of the study site is given, as well as the duration of the time period over which parameters were monitored (i.e. storm event, seasonal, or long-term trends over several years). Where possible, an indication of the major landuse in the catchment and in particular whether it is in an unmodified (natural) state is included. To be consistent, the response of given chemical constituents and physical variables to *increased* discharge is recorded in the table. The general trend to decreasing discharge is likely to be the opposite to that reported for increasing discharge, although as reported in section 3.2 due to hysteresis the mathematical relationship may not be exactly the same.

The effect of altered discharge on each of the water quality variables is discussed below. This is followed by a summary of the general conclusions that can be drawn concerning water quantity/water quality relationships. In many studies, the authors have not stated specifically that discharge and concentration are related, although this is often implied. For example, the concentration of a given solute is often reported as being highest during low flow conditions and lowest during high flow. As discussed above, because of the many other factors that influence discharge-concentration relationships, other mechanisms may also be in operation other than an apparent dilution effect. Nonetheless, because they contain valuable information, such studies have been included, especially if they pertain to rivers in southern Africa.

*Table 3.1 The effect of an **increase** in discharge (Q) on the concentration of water quality constituents. Abbreviations given at bottom of table.*

Site	Event	TDS*	Nitrogen	Phosphorus	Other	Comments	Reference
Fynbos, mountain stream, SA	Seasonal storm	ND	NO_3^- ↓ in winter storms, but ↑ during summer storms.	oP ↑ during winter storms (ns) but ↓ during summer	pH ↓ during storm events. HCO_3^-, usually initial rise early in storm followed by decrease. Cl^- ↓ in winter storms but ↑ during summer storms. NH_4^+ ↑ for all storms.	Unimpacted, nutrient poor system. Effect of discharge on NO_3^- shows seasonal effect. Rainwater acidic. HCO_3^- conc. highest during baseflow. Cl^-, oP, NO_3^{-2}, exhibit seasonal responses. pH, HCO_3^- and NH_4^+ do not.	Britton *et al.* 1993
Upper Hennops River, SA	Seasonal and storm event	Not reported	Poor correlation with Q	Poor correlation with Q	Na^+, K^+, Cl^-, SO_4^{-2}, Ca^{+2} conc. highest during low flow (winter). Lowest during high flow (summer).	Sewage works in the headwaters contribute to mineral loading and water flow. Wetlands occur further downstream. Catchment is used extensively for agriculture. $CaCO_3$ showed an initial dilution effect during storm then rapid increase. Up to 70% of flow is treated sewage, possibly masking seasonal patterns of N and P.	Toerien and Walmsley 1979
Tributary. of Ntuze River, Natal, SA	Storm	Conductivity ↓	NO_3^- ↑	ND	pH ↓. Turbidity ↑ followed by a rapid decrease.	Catchment land use comprises indigenous and exotic forests, commercial and subsistence agriculture, as well as livestock breeding.	Kelbe and Snyman 1993
Two tributaries of Ntuze River, Natal, SA	Storm	TDS ↓	NO_3^- ↑	TP↑ oP little trend	pH ↓ but variable. Turbidity↑	Trend of increasing turbidity more distinct in disturbed catchment compared to the natural catchment.	Kelbe and Germishuyse 1999
Modder River, Free State, SA	Seasonal (22 months)	TDS ↓	NO_3^- ↑	oP ↑	Turbidity ↑	NO_3^-, oP trends not always clear but increases usually associated with rainfall events.	Koning *et al.* *in press*

***Table 3.1** The effect of an **increase** in discharge (Q) on the concentration of water quality constituents. Continued.*

Site	Event	TDS*	Nitrogen	Phosphorus	Other	Comments	Reference
Senqunyane, Senqu, and Matsoku Rivers, Lesotho.	Seasonal		Total N ↑	TP↑	TSS ↑ pH ↓	TDS, NO_3^-, DO, NH_4, TOC did not exhibit consistent trends at the various sites. Data collected 3 hourly over a period of 20 months. Relatively unimpacted catchments (no industrial development but overgrazing).	Skorozewski 1999
Three small rivers, Victoria, Australia	Seasonal (12 months)	ND	ND	ND	Na^+, Ca^{+2} very slight ↓, K^+, Cl^- no relationship with flow.	Forested catchment. Absence of marked Q-C relationships possibly because the catchment is composed of weathered rock, and therefore nutrient-poor.	Flinn *et al.* 1979
Streams in N.S. Wales, Australia	Storm	TDS ↓	NO_3^- ↑		Na^+,Ca^{+2}, Mg^{+2}, Cl^-,CO_3^- ↓	Forested catchment	Cornish 1982
Annan River, Queenslan, Australia	Storm	TDS ↓	ND	ND	TSS ↑ pH little change	Forests and small tin mines occur in the catchment. Peak in TSS concentration occurs before peak discharge.	Hart *et al.* 1988
77 River sites, New Zealand	Long term trends (5 yrs)	TDS ↓	NO_3^- and NO_2^- ↑ Total N ↑ NH_4^- ↑ or no trend	TP↑ oP↑ or no trend.	pH ↓ or no trend	The 77 river sites all varied in terms of catchment land use and degree of impact. NH_4^- increased with Q at impacted (polluted) sites. Little change with discharge at non-impacted sites. Similar pattern for ortho-phosphate.	Smith *et al.* 1996
Highly regulated river, South America	Seasonal	TDS not related to Q	NO_3^- not correlated with Q, but often ↑ during initial rising water phase.	TP and PP ↑	DOC, Fe pH, not related to flow. TSS and turbidity ↑	Main land use is intense mechanised agriculture. TP and PP correlated with TSS content.	Pedrozo and Bonetto 1989
Kali Mondo River, Indonesia	Storm	ND	ND	ND	Si ↓	Forested catchment. Tropical river. Anti-clockwise Q-C-T loop.	Bruijnzeel 1983

Site	Event	TDS*	Nitrogen	Phosphorus	Other	Comments	Reference
Hubbard brook, USA	Seasonal	ND	$NO_3^- \uparrow$	ND	Na^+, $SiO_2 \downarrow$ Mg^{+2}, SO_4^{-2} slight $\downarrow$, Cl^- no trend. Al, H^+, $K^+ \uparrow$	Forested catchment.The concentration of NO_3^- and K^+ decreased during summer due to biological uptake.	Johnson *et al.* 1969
Hubbard brook, USA	Seasonal including snow melt	ND	ND	oP $\uparrow$ but only slight increase	DOC and FPOC $\uparrow$	Loss of P (both oP and particulate P) enhanced by deforestation. P strongly conserved in undisturbed catchment. No seasonal patterns.	Hobbie and Likens 1973
Hubbard brook, USA	Storm	ND	ND	ND	FPOC, FPP $\uparrow$	Forested watershed. FPOC and FPP increase initially very rapid, then decreases before hydrograph peak due to depletion of resource.	Bilby and Likens 1979
Western Cascade Mt slopes, Oregon, USA	Seasonal	TDS $\downarrow$	Total N $\downarrow$	oP no trend	HCO_3^-, Ca^{+2} and $Mg^{+2} \downarrow$	Undisturbed forested catchment.	Martin and Harr 1988
Small Oklahoma stream, USA	Seasonal	ND	NO_3^-, $NO_2^- \uparrow$	ND	ND	Forested catchment.	Lawrence and Wigington 1987
Sulphur River, USA	Seasonal	TDS $\downarrow$	Not correlated	Not correlated	pH , Cl^-, SO_4^{-2}, TSS, DO, POC all $\downarrow$	A paper mill and sewage treatment plant occur in the catchment. Due to upstream lake, TSS not increased at higher flows. C-Q relationships derived in absence of effluent flows.	Fisher *et al.* 1988
Illinois River, USA	Seasonal (late spring – autumn)	ND	$NO_3^- \uparrow$ $NH_4^+ \uparrow$ and then levels off	oP $\downarrow$	$F^- \downarrow$	Upstream of the study area, wastewater effluents, diversion water, industrial wastes and runoff contribute to nutrient load. F^- is a conservative element in streams. F^- and oP fit an exponential equation. Both oP and F^- from constant point source.	Wang and Evans 1970

Site	Event	TDS*	Nitrogen	Phosphorus	Other	Comments	Reference
Mountain stream, Colorado, USA	Long term trends (3 yrs)	ND	NO_3^- ↓, NH_4^- no trend	oP ↑ but not all years	HCO_3^-, Ca^{+2}, Mg^{+2}, Na^+, pH ↓ K^+, SO_4^{-2} no trend DOC ↑	Undisturbed forested catchment. Hydrograph shows single peak/per year resulting from snowmelt.	Lewis and Grant 1979
Mountain stream, Colorado, USA	Long term trends (3 yrs)	ND	NO_3^- ↓, NH_4^- no trend	oP ↑ but not all years	HCO_3^-, Ca^{+2}, Mg^{+2}, Na^+, pH ↓ K^+, SO_4^{-2} no trend DOC ↑	Undisturbed forested catchment. Hydrograph shows single peak/per year resulting from snowmelt.	Lewis and Grant 1979
Matahango Creek, USA	Summer storm	TDS ↓	NO_3^- ↓ NH_4^- ↑	TP ↑	ND	Unfertilized permanent pasture.	Pionke *et al.* 1988
Brandywine River Watershed, USA	Summer and storm event	ND	ND	Steady state Q; TP ↓ Storm event: TP ↑		Two mechanisms of P transport. Steady state conditions mostly oP. Storm events mostly particulate P.	Cahill *et al.* 1974
Nutrient-rich, 1st order stream, northern hemisphere	Storm	ND	NO_3^- first ↑ then ↓	oP ↑ but variable	Ca^{+2} and Mg^{+2} ↓ i.e. ground-water dilution effect.	Factors other than discharge also involved in determining concentration. NO_3^- and P give clockwise Q-C-T plot. Plots for Ca^{+2} and Mg^{+2} variable. Agricultural land.	McDiffett *et al.* 1989
Two streams, Alberta, Canada	Seasonal	ND	ND	TDP, FPP, TP ↑	ND	Largely undisturbed forested catchments.	Munn and Prepas 1986
Ausable River, Canada	Seasonal and storm event	ND	ND	ND	TSS ↑	Agricultural land-use. Sediment peak occurs before that of discharge. Hysteresis effects.	Irvine and Drake 1987
Seven headwater streams in Quebec	Seasonal and storm event	Conductivity ↓	ND	DP	DP ↑, ↓ and remained constant depending on the stream studied. PP ↑	Catchments vary in land use. For a given stream, the trend was constant over 2 seasons and during storms. PP was directly proportional to discharge except in 2 streams where sediment limitation effects were noted.	Prairie and Kalff 1988a + b
Low gradient, limestone streams, Norfolk, UK.	Seasonal	ND	NO_3^- ↑ for all streams	oP ↓	Mg^{+2} ↑ in all streams. HCO_3^- usually ↓. SO_4^{-2} ↑. Na^+ and K^+ very little change. Ca^{+2} and Si, no correlation.	Mainly agricultural, some villages. Rivers all unpolluted (except one). Variation between individual streams. K^+ frequently shows small initial peak. Area of low gradient and therefore little surface runoff.	Edwards 1973

Site	Event	TDS*	Nitrogen	Phosphorus	Other	Comments	Reference
Devon Rivers, UK.	Storm	TDS $\downarrow$	NO_3^- $\uparrow$ *but* often a complex response	oP$\uparrow$ but variable	$K^+\uparrow$. Ca^{+2}, Mg^{+2}, $Na^+\downarrow$	Mixed land use in catchment. Lag time between flood peak and chemograph response varies. Dependent on antecedent soil moisture status and storm intensity.	Walling and Foster 1975
Slapton wood, UK	Long term trends	ND	NO_3^- $\uparrow$	ND		Antecedent rainfall conditions very important. NO_3^- concentration very high during rainfall following period of drought due to build up of NO_3 in soil.	Burt *et al.* 1988
River Thames, London	Seasonal	ND	ND	oP $\downarrow$	ND	Heavily urbanised catchment, with some agriculture in rurral areas. Many sewage-treatment works. Highest conc. in summer, during low flow. During drought years, even higher oP levels due to the increased proportion of total Q originating from sewage.	Kinniburgh *et al.* 1997
Twelve Welsh streams	Seasonal	ND	NO_3^- usually $\uparrow$, although dilution effect in some rivers	oP $\downarrow$	$CaCO_3$, Cl^- $\downarrow$	NO_3^- levels high in winter when Q maximal. $CaCO_3$, Cl^- and oP show dilution effect. NO_3^- was the major form of inorganic N, except in urbanised catchments	Brooker and Johnson 1984
Two lowland streams, Wales.	Seasonal and storm event	ND	Seasonal: NO_3^- $\downarrow$ in one river but $\uparrow$ in the other. NO_2^-, NH_4^- both $\downarrow$ Storm: NO_3^- $\uparrow$	Seasonal: oP $\downarrow$ Storm: oP $\downarrow$	Seasonal: SiO_2 $\downarrow$ Storm: SiO_2 $\uparrow$	Intensive agriculture in catchments. Variation in solute behaviour between different storm events. Principal form of inorganic N in both catchments was NO_3^-.	Houston and Brooker 1981

Site	Event	TDS*	Nitrogen	Phosphorus	Other	Comments	Reference
Morlaix River, France.	Seasonal	ND	NO_3^- $\uparrow$ NH_4^+ $\downarrow$ NO_2^- $\downarrow$	oP $\downarrow$	DOP, DON, DOC all independent of Q	Urban pollution and an excessive use of fertilizers occur in the catchment. DOC shows peak in summer due to algal exudates also peak in winter due to flushing from soil.	Wafar *et al.* 1989
Two rivers in Denmark	Seasonal	ND	ND	PIP andPOP $\uparrow$ DIP $\downarrow$	PIM and POM $\uparrow$	Agricultural activities in the catchment. PIM more strongly correlated with Q than POM. PIP and POP show same trends as particulate matter.	Kronvang 1992
Svarta River, Sweden	Seasonal	ND	Total N $\uparrow$	Not reported		Catchment land use varies between forested areas, cultivated plains and agricultural areas. A quadratic relationship gave a good fit between Q and Total N. For all other water quality variables the Q-C relationship was weak and a linear regression model appropriate.	Karlsson *et al.* 1988

Abbreviations: ns = not significant, ND = not determined, oP = ortho-phosphate (PO_4^{3}-P), PIM = particulate inorganic matter, POM = particulate organic matter, PIP = particulate inorganic phosphorus, POP = particulate organic phosphorus, DOP = dissolved organic phosphate, DON = dissolved organic nitrogen, DOC = dissolved organic carbon, PIP = particulate inorganic phosphorus, POP = particulate organic phosphorus, TDP = total dissolved P, FPP = fine particulate P, TP = total P, DOP = dissolved organic phosphate, DON = dissolved organic nitrogen, TDS = total dissolved solids, TSS = total suspended solids.
Note that TDS and conductivity have been used interchangeably.

3.4.1 Temperature

Water temperature is profoundly influenced by climate and exhibits a cyclic behaviour in response to the changing seasons (Davis and Keller 1983). Dallas (1998) reported generalised seasonal trends in water temperature for Western Cape Rivers. Not unexpectedly, highest temperatures in least impacted rivers were recorded in the summer, when discharge was at a minimum. Lowest temperatures were recorded in winter, when discharge was maximal. Stream temperature is directly proportional to surface area and solar input and inversely proportional to streamflow (Gibbons and Salo 1973, cited by Meehan, Swanson and Sedell 1977). Smith *et al.* (1996) found that water temperature either decreased with increased discharge, or showed no correlation. This effect was noted in both pristine as well regulated rivers. Note though, that these studies were carried out on rivers in New Zealand where conditions of maximum discharge usually occur in winter. Such a trend might not be expected in summer rainfall regions however, when peak flow occurs during periods of high air temperature.

3.4.2 pH

Many studies have reported that high flow rates result in increased proton concentration and thus a decrease in the pH of stream water (Billett and Cresser 1992). Lawrence *et al.* (1990) citing the work of Johnson (1969) and Bishop, Grip and O'Neill (1990) attributed this effect to the flushing of organic acids from the soil of the surrounding catchment. Several studies have been carried out on streams in the South Western Cape whose catchments are largely dominated by fynbos. Fynbos vegetation produces organic acids, which wash into rivers, lowering pH. In the case of the Eerste River just outside of Cape Town, the first floods of the rainy season lead to a noticeable decrease in pH, whereas subsequent floods decrease pH to a less marked extent and an increase in pH may even be observed (Day *pers. comm.* 1999). Such an effect is consistent with the idea of organic acids in the surrounding catchment being a limiting substrate. Flood events early in the rainy season are likely to wash off organic acids accumulated within soil during the long, dry summer. Due to depletion of these chemicals within the soil, subsequent floods will not result in such high concentrations reaching the watercourse. Consequently, during winter when discharge is high, pH has been reported to rise in such rivers due to dilution of organic acids (DWAF 1996). Britton *et al.* (1993), working on a stream in a fynbos mountain catchment, found that during storm events, pH decreased. This effect was noted during both winter and summer storms. The authors postulated that one of the reasons for this effect, was the naturally acidic precipitation in the study area.

Kelbe and Snyman (1993) monitored changes in stream water pH during a short-duration, high intensity storm event in November. The stream studied was a tributary of the Ntuze River, Natal, which is in the summer rainfall region. The pH of this system also showed an immediate drop, coincident with the first hydrograph peak and then gradually rose again over the next 25 hrs. On the other hand, De Villiers and Malan (1985) investigating the seasonal changes in water quality of the Palmiet River, Natal reported virtually no difference in pH between summer and winter. Thus it is likely that antecedent moisture conditions and the organic acid content of the soil are important in predicting pH changes for individual rivers. In addition, the extent to which the water is buffered will also affect the response. Whilst most fresh waters in South Africa are relatively well buffered, the soft waters of the Western Cape are low in carbonates and bicarbonates (Dallas and Day 1993), and are thus likely to exhibit more marked fluctuations in pH.

Investigations into the effect of discharge on pH can be complicated by the presence of anthropogenic influences on precipitation. Commonly known as "acid rain", this phenomenon has lead to the acidification of surface waters in North America, Western Europe (Hall and Likens 1984; Martin and Harr 1988; Jenkins, 1989) and other parts of the world. Before interpreting the response of stream water pH to discharge, it is important to ascertain whether rain falling over the catchment is likely to be at natural pH levels or is acidified due to the presence of pollutants. Thus Q-C patterns attributed to a pristine catchment may be deceptive due to the unperceived effect of acid precipitation.

The pH of rainwater, surface flow or streamwater can in turn exert an effect on the mobility of other ions. Johnson *et al.* (1969) found that the concentration of NO_3^-, Al^{+3}

and Ca^{+2} in streamwater all increased with discharge. These authors attributed this effect in the case of NO_3^- and Al^{+3}, at least, to decreased pH, which accompanied increased streamflow. This resulted from precipitation which was acidic (pH ≈ 4) compared to the pH of the soil water and stream. Thus during high stream discharges, rainwater made up a large fraction of stream flow, and pH decreased.

3.4.3 Dissolved oxygen (DO)

In a study of the Sabie-Sand River complex, it was found that DO levels were usually high (100% saturation) whilst the river was flowing, although concentrations in isolated pools during no-flow periods did drop (Weeks, O'Keeffe, Fourie and Davies 1996). Similar results were found in the tributaries of the River Jong, Sierra Leone (Wright 1985). The above studies highlight one of the most critical Q-C relationships, namely that of DO and discharge. The levels of DO in non-impacted running waters are usually close to saturation (Dallas 1998) and thus increases in discharge have little effect. If discharge is reduced sufficiently, due either to natural or anthropogenic causes, pools of standing water may develop. Particularly during summer months when water temperatures are high, DO levels in such pools may reach critically low levels. Salinity increases in pools due to evaporation of the water, may also exacerbate such effects (see below, section 3.4.5). Deeper pools may also exhibit thermal stratification, whereby distinct layers are formed between the warm surface water in contact with the atmosphere and the cold bottom water (Davies and Day 1998). No mixing occurs between the water strata and once oxygen is depleted in the lower section, as a result of respiration by the biota or due to chemical reactions, it is not replenished. Such anoxic conditions can persist until stratification is disrupted by mixing of the water layers once again at the end of summer or by the onset of a storm event.

Another important factor that has a controlling influence on the DO level of a water body is the ratio of surface area to volume. The larger the surface area, the greater is the potential for oxygen from the atmosphere to dissolve in the water body. Thus Skoroszewski (1999) reported that DO levels in Senqu, Senqunyane and Matsoku Rivers, Lesotho (a summer rainfall region) were highest during winter. This was attributed to the fact that lowest water temperature occurred during that part of the year. In addition it was also when discharge was at a minimum and thus the surface to volume ratio was at a maximum, allowing rapid gaseous exchange. It was noted in Chapter 2 however that turbulence is important in aeration. In rivers that become very turbulent during high flow, a different result might be expected.

3.4.4 Total suspended solids (TSS)

In general, TSS increases with discharge due to washing of sediment from the land and riverbanks as well as resuspension from the riverbed (Jha *et al.* 1988; Burrus, Thomas, Dominik *et al.* 1990). As flow decreases, solids settle out (Dallas *et al.* 1998). In South Africa all rivers, except those in the foothills of the Drakensberg mountains and in the south-western Cape, become highly turbid and laden with suspended solids during the rainy season. Bremner *et al.* (1990, cited by Davies *et al.* 1993) studied the Orange River floods of 1988. Over the three-month period of the floods, sediment transport, although substantial, was not proportional to discharge. Total suspended sediment load was highest in the first month of the flood and then gradually decreased. A pre-impoundment study of the Sabie-Sand river system also showed that the highest levels of TSS were associated with spates (Weeks *et al.* 1996). One of the tributaries of this system has a large portion of its catchment in the Kruger National Park and is therefore relatively pristine. High TSS levels during storm events were also recorded in this river. Thus high turbidity is a natural condition in many rivers, but may be exacerbated by anthropogenic activities in the catchment.

Suspended sediment was found to exhibit clockwise hysteresis loops during storm events in an Alaskan creek (Sidle 1988). Plots of total suspended solids versus stream flow showed higher concentrations in the rising limb of the hydrograph compared to the falling limb. This researcher found however, that patterns of bed load movement (the heavy sediment that rolls along the stream bed) within storms were inconsistent and attributed this to the non-steady release of sediment during armour break-up and damming effects due to large accumulations of organic debris upstream (Sidle 1988). This made modelling of such processes difficult using simple flow-dependent equations (Chapter 4).

Using filtration, Hart, Day, Sharp-Paul *et al.* (1988) distinguished between metal pollutants adsorbed to sediment and that fraction that was present in the filtrate, either as the free metal ion or complexed with organic or inorganic constituents. These authors examined the changes in concentration of the different metal fractions during a storm event in the Annan River, Queensland, Australia. Interestingly they found that most of the iron (99%), manganese (95%), lead, zinc, tin (all roughly 80%) and copper (60%) were transported adsorbed to sediments. Not surprisingly, this occurred mostly in the early part of the flood on the rising limb of the hydrograph when transport of TSS was maximal. The concentrations of the filterable forms of the metal on the other hand (free metal ion, metal-organic/inorganic complexes) were relatively low and were not correlated with flow or suspended sediments. Other studies have also found that the transport of metals in freshwater is primarily associated with suspended sediments (Miller 1997).

3.4.5 Total dissolved solids (TDS) and individual ions

In the absence of other factors affecting river chemistry, an inverse relationship between the concentration of dissolved constituents and water discharge is commonly recorded (Hem 1970, cited by Edwards 1973). Thus McDiffett *et al.* (1989) found that Mg and Ca concentrations, although variable, showed a consistent decrease due to dilution, as discharge increased. The concentrations of these ions then slowly increased as Q returned back to baseflow levels. This pattern of behaviour was opposite to that exhibited by nutrients in the same study. The different trajectory patterns (hysteresis loops) shown by the nutrients are thought to indicate the different sources of water. Nitrate and phosphate originate from surface runoff and Mg and Ca from groundwater. Magnesium and Ca are thus diluted by increased stream flow due to rainfall and surface run-off, which has a low concentration of these elements, a reflection of the short contact time between these two types of water and the underlying substrate. As the hydrograph recedes, groundwater becomes a significant proportion of stream flow once again, and the concentration of Ca and Mg rises to pre-storm levels. In contrast to nutrients, conservative ions are not sequestered (or released) by the biota to a marked extent and thus behave differently to those affected by the biota (see section 3.4.6).

De Villiers and Malan (1985) investigated the seasonal changes in levels of various cations and anions in the moderately polluted Palmiet River, Natal. They found that whilst the geochemistry of the area had a limited effect on water quality, pollution exerted a profound influence. TDS concentrations were higher in summer than in winter, despite the higher discharge in the warmer months. This was attributed to the high rainfall in summer, which washed minerals into the river that had accumulated on the land during the previous season. Thus a dilution effect on TDS was not observed in this catchment. In contrast, another author quoted by the above (Hart 1982) found that for the Buffalo River (also in the summer rainfall area), lower levels were recorded in summer due to dilution by high summer flows.

It was mentioned under the discussion of DO (section 3.4.3) that cessation of flow has marked effects on the chemistry of the resulting standing waters. Anderson (1987, cited in Water Victoria 1988) reported the dramatic physical and chemical changes occurring in isolated pools of the seasonal Wimmera River, Australia. Temperatures in the pools became high and as a result of high levels of organic material, oxygen was depleted. Where influxes of saline groundwater occurred, this depleting effect on DO was exacerbated. The salinity of the groundwater entering these pools was sufficient to create a stable chemocline, below which oxygen concentrations approached zero. Thus the volume of water available for the biota was diminished even more. Influx of saline groundwater has almost certainly increased as a result of catchment land clearance and is well demonstrated in Australia. According to Davies, O'Keeffe and Snaddon (1993), water quality in many South African rivers is also declining due to salinization and increased organic pollution. Salinization results from runoff containing high levels of dissolved ions. Whilst this is a natural process, irrigation of semi-arid land, in which high evaporation rates result in abnormally high levels of TDS in return flows greatly accelerates the process of salinization in rivers. Urban waste can also contribute to high TDS levels (Davies *et al.* 1993). A pre-impoundment study of the Sabie-Sand River system in Mpumalanga showed that during the study period, the highest levels of TDS occurred in the Sand River when the river was not flowing (Weeks *et al.* 1996).

3.4.6 Nutrients

3.4.6.1 *General considerations*

When considering the effect of increased discharge on nutrient levels, several points must be borne in mind. Firstly, although plants require many nutrients for growth, phosphate and nitrates in particular are often limiting in non-impacted aquatic ecosystems. These nutrients are frequently rapidly taken up by aquatic macrophytes as well as by algae (Dallas *et al.* 1994), thus reducing the concentration in stream water. As a result of assimilation by living organisms as well as adsorption onto particulate matter, phosphate tends to decrease downstream from an effluent point source (Wang and Evans 1970). Naiman and Sibert (1978) attributed peaks in nitrate levels during winter partly to the sparse periphyton community that at other times of the year, lowers the concentration of this nutrient in the water. In addition, sudden death of a biotic element (e.g. macrophytes, diatoms etc.) due to seasonal changes or other causes can lead to the sudden introduction of large amounts of nutrients into the system once again.

A second aspect should be taken into account when considering the effect of discharge on the nutrient concentration of stream water, and that is the differential behaviour of nitrate and phosphate in soil. Whilst nitrate is extremely mobile in soil, phosphate is nearly immobile. This, according to Wang and Evans (1970), is an explanation for the flushing effect frequently exhibited by nitrates during periods of high discharge, since this ion is easily removed from the soil of the surrounding catchment into streams. The dilution effect often reported for ortho-phosphate results from the fact that only small amounts of this chemical species are released by soils. In other words, soils act as better ion-exchange systems in the case of nitrate and better ion-absorption systems in the case of phosphate.

Finally, spate-generated foam is a common feature of winter floods in many acid, black water streams of the South Western Cape. These foams contain up to 20% N and it is claimed that they represent an important source of nutrients for invertebrates (Davies *et al.* 1993; *Koch et al.* 1994). It is thought that upwelling of groundwater through the streambed releases organic matter, resulting in formation of the foam.

3.4.6.2 *Phosphate*

In studying the relationship of phosphorus concentration with discharge, distinction should be made between the different chemical species or forms of this element. A large fraction of the total phosphorus that enters a river is adsorbed onto sediments and a smaller fraction of the total load is dissolved in water (Holtan, Kamp-Nielsen and Stuanes 1988). Thus, much of the total phosphorus is delivered during storms when discharge and velocity are high and scouring of bottom sediments and soil wash-off occur (Verhoff, Melfi and Yaksich 1982; Munn and Prepas 1986). In contrast, dissolved phosphate frequently shows a dilution effect. Cahill, Imperato and Verhoff (1974) also found a distinction in the behaviour of total phosphate and soluble orthophosphate. During steady state conditions both total and orthophosphate showed a dilution effect. During storm events however, only total P was increased by increasing flow, orthophosphate remaining constant. These authors found that peak total phosphate concentrations occurred slightly after the crest of the storm. It is important to note that changes for example, in pH and conductivity resulting from altered discharge, may also affect the absorption/desorption equilibrium and thus the proportion of dissolved to bound phosphorus.

Examination of Table 3.1 shows that the above generalisations concerning this nutrient are not always valid. Whilst total phosphorus concentration usually increases during the rising limb of the hydrograph, dilution of soluble inorganic forms of phosphate such as orthophosphate does not always occur (Munn and Prepas 1986). Prairie and Kalff (1988a) examined Q-C relationships in seven headwater streams in Quebec. They found that increasing discharge either increased, decreased or had little effect on the dissolved phosphate levels of streams. For each stream, the relationship was valid whether examined over a period of two seasons or during an individual storm event. Houston and Brooker (1981) investigated phosphate levels in two Welsh streams and found that concentrations were generally low in winter when flow was maximal, and high in summer when discharge was minimal. Peak ortho-phosphate levels however, were recorded in November and December during periods of increasing flow and may indicate a small flushing effect, followed by the normal dilution mechanism. On the other hand, significant correlation between discharge and total phosphorus concentration, as well as between discharge and dissolved phosphorus levels, were not found by Zeman and Slaymaker (1985). The authors attributed the absence of a statistically significant relationship to the regulated nature of the river flow.

A consideration of catchment land-use may be important when making predictions of phosphorus-discharge relationships. Houston and Brooker (1981) found that intensive land use (arable farming, high numbers of livestock and people) lead to increased phosphate runoff compared to a similar Welsh stream in which the major activity was sheep grazing coupled with small human populations. Hobbie and Likens (1973) found that there was little seasonal change in total phosphate concentration in an undisturbed forested catchment and concluded that the ecosystem was strongly conserving phosphorus. It is likely that in nutrient-poor ecosystems, phosphate will be strongly bound either to soil or within living organisms and therefore stream water concentrations may not vary greatly. Prairie and Kalff (1988a), in their study of seven Quebec streams, found that nutrient-rich streams were temporally more variable than nutrient-poor ones.

The origin of phosphates entering stream water, i.e. whether of point or non-point source, also appears to be important. Burrus *et al.* (1990) analysed the chemical composition of suspended sediments from the upper reaches of the River Rhine. Their findings indicate that total phosphate concentration decreased during periods of high flow. The portion of phosphate associated with minerals (apatite phosphorus) was more or less constant throughout the year, but organic phosphorus and non-apatite inorganic phosphorus showed the normal dilution effect. The authors postulated that organic and non-apatite inorganic phosphorus originated from point sources that produced a constant volume of effluent throughout the year and consequently exhibited concentration during periods of low flow. Kinniburgh, Tinsley and Bennett (1997) found that orthophosphate levels (approximately 91% of the total phosphorus load) in the River Thames fluctuated seasonally, reaching peak levels in summer. This was as a result of the increasing proportion of flow made up of sewage effluent, such an effect being exacerbated during drought years.

3.4.6.3 *Nitrogen (nitrates, nitrites, ionised and un-ionised ammonia)*

Houston and Brooker (1981) found that nitrate-N was the principal form of soluble inorganic nitrogen found in stream water analyses of two lowland Welsh streams. Ammonium-N and nitrite-N generally accounted for only a small proportion (5 and 0,2%) respectively of the mean total soluble nitrogen concentration. Similar results have been reported for a highly regulated South American river (Pedrozo and Bonetto 1989). Significant levels of ammonium-N were found only in urbanised catchments (Brooker and Johnson 1984).

As mentioned above, in general, nitrate levels show a positive correlation with stream discharge. Brooker and Johnson (1984) found in a survey of twelve Welsh streams that nitrate levels were generally highest in the winter (discharge high) and lowest in the summer. In these same streams phosphate showed an opposite pattern, being high in summer and low in winter. Pedrozo and Bonetto (1989) could not find a strong correlation between discharge and nitrate levels in the River Paraná, South America, possibly because the river is highly regulated. Nonetheless, they did note that high levels of this nutrient in the river were associated with high rainfall events following a period of drought. These authors ascribed such high levels to a diffuse source of nitrates, most likely associated with soil leaching. Wang and Evans (1970) also found that nitrate levels, in the Illinois River at Peoria Lake, increased with discharge and also ascribed this to runoff effects. It would appear that whilst nitrate levels are positively correlated with discharge, the amount of antecedent rainfall has a very marked effect. According to Burt *et al.* (1988), during dry periods especially during the summer, there is a build up of mineralised nitrate from organic nitrogen in the soil. This is released from the catchment during the next rainfall event. Nitrates appearing in streamflow during storms may also originate from leaching of biotic material such as forest litter, tree canopies and stems (Cornish 1982). The rate of loss from soil and other sources may well decline towards the end of the rainy season however, due to depletion of soluble nitrate reserves (Mason 1991).

Local geological conditions may also have an effect on nitrate-discharge relationships. In their study of two Welsh streams, Houston and Brooker (1981) found different seasonal patterns in nitrate levels. In the case of both streams, peak nitrate levels were measured during periods of maximum flow in December. But whereas nitrate levels decreased in the Trothy during summer (low flow) they increased in the Frome. The authors concluded that this was most likely a consequence of differing levels of nitrate in groundwater. Groundwater supplying the Frome contained higher concentrations of nitrate and, in addition, made up a higher proportion of the total summer flow in the Frome, than in the River Trothy.

Wang and Evans (1970), examined changes in ammonium (ionised ammonia) concentration in the Illinois River at Peoria Lake. They found that ammonium-N increased with discharge due to runoff effects and then decreased at the highest flow levels due to a dilution effect. Houston and Brooker (1981) reported that the observed ammonium levels in two Welsh streams followed a trend fairly similar to that of phosphorus, namely low in winter, higher in summer and highest in November to December coincident with the onset of winter rains. According to Wang and Evans (1970), the ammonium ion is moderately mobile in soils, which is consistent with the trends described above. Thus in the absence of point sources, a limited wash-off effect would be expected for ammonium as discharge increases, followed by a levelling off as the resource becomes limiting. At very high discharges, a dilution effect may be observed.

In the case of inorganic nitrogen (both nitrates and ammonia/ammonium) as in the case of phosphates, landuse and anthropogenic activities within the catchment may be expected to profoundly influence the Q-C relationship. This is well illustrated by the work of Smith et al. (1996), summarised in Table 3.1. In a survey of 77 sites, these authors found that whilst NH_4^+ increased with discharge at impacted (polluted) sites, there was little change with discharge at non-impacted sites. Similar patterns were exhibited by ortho-phosphate. Such effects are likely to reflect the availability of nutrient ions, which would be expected to be abundant at impacted sites but possibly limiting at natural sites, due to uptake by the biota.

3.5 Extrapolation to arid lands

There is a need for caution when extrapolating from studies carried out in temperate, mesic areas, to hot, arid countries such as South Africa. Pinol, Avila and Roda (1992) examined seasonal variation in stream water chemistry in three forested Mediterranean catchments. Two streams were non-flowing during the summer, whilst the other was located in a wetter area and was perennial. The authors found that ion concentrations were closely dependent on instantaneous discharge in the perennial stream. Stream chemistry in the two intermittent systems showed seasonal cyclic effects and was strongly affected by the onset of flow with the autumn rains. This caused a wash-off effect, particularly of SO_4^{2-} and Cl^- that had accumulated in the soil during the dry summer. Rivers situated in areas where precipitation is more evenly spaced throughout the year, and where a large proportion of rainfall ends up as run-off, are less likely to exhibit seasonal wash-off effects. Pinol *et al.* (1992) also highlighted the cyclic variation in the concentration of K^+ in the stream water. Potassium levels were reported as being high in summer in Mediterranean catchments but low during the same period in cool-temperate catchments. They attributed this difference to the fact that in the former, chemical weathering (which liberates K^+) is active especially during warm weather and biological uptake (which results in a lowering of stream K^+ concentration) is in operation the whole year round. In the case of cool-temperate catchments, whilst weathering occurs mostly during the summer months, biological growth and uptake of K^+ are also limited to this time-period, resulting in a lowering in K^+ concentration during the warm part of the year.

3.6 Summary of discharge-concentration trends

An attempt has been made to summarise the various trends exhibited by chemical constituents and physical variables in response to increased discharge. Predictions for individual ions may be made for a given catchment from a consideration of climatic region and catchment landuse as well as the underlying geochemistry. As mentioned in the introduction to this chapter however, responses of stream water chemistry to discharge can be extremely complex and site-specific. This is confirmed by the data summarised in Table 3.1, where it is evident that clear patterns and trends are not always easy to discern. Thus predictions of stream chemistry in response to 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 discharges as substrate becomes limiting. Storms occurring early during the wet season are likely to carry heavier loads of sediments compared to storms later on in the season. This once again is due to limitation in the supply of this material.

- Dissolved minerals derived from the underlying substrate are likely to decrease as discharge increases due to dilution by rainfall and surface run-off containing low solute loads.

- Due to the 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 of 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 and followed by rapid assimilation of nitrates by aquatic organisms.

- pH is likely to decrease during storm events, especially in the South Western Cape. This variable is likely to decrease in Cape rivers in autumn but may increase during the latter part of winter due to a dilution effect. Increased acidity during high flow events is also to be expected in other parts of South Africa although the effect may not be so pronounced.

- Particulate phosphate is likely to increase during spates due to enhanced sediment loads. In the absence of point sources of pollutants, dissolved phosphate (orthophosphate) is likely to decrease or remain constant in nutrient poor areas in response to increased discharge. 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.

- The resultant effect of discharge increases on TDS is difficult to predict, reflecting as it does the sum of effects on pH, nitrates, phosphates as well as other chemical constituents. Due to the high rate of evaporation in SA, in non-impacted catchments, TDS is likely to be at a maximum during periods of low flow, and at a minimum during high flow. In urban, or polluted, areas however, or where surface wash-off of ions is likely to be substantial, such a response may be obscured.

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

3.7 The effect of impoundments on water chemistry

The primary aim of this project is to predict, for a given stream, the effect of changes in discharge on the resultant concentrations of chemical constituents and the values of physical variables. In South Africa, the flow from a large proportion of our rivers is already regulated and this is likely to increase in the future as the demand for this limited resource expands. Thus an increasing proportion of the mean annual runoff (MAR) for a given river will be stored in impoundments (colloquially known in this country as "dams") and discharge will be controlled by means of these structures. Regulation of running waters by impoundment has many diverse manifestations, which are unique for each system, although it is possible to generalize the responses (Hart and Allanson 1984). The chemistry of 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 (Davies and Day 1998; Webb and Walling 1997).

In section 3.4.3 it was mentioned that in deep stagnant pools during summer, thermal stratification occurs. A similar process is also operational in reservoirs and in South Africa the combined effect of local climate and morphology results in many water bodies 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). The chemical composition, temperature and oxygen content of the different water layers can differ markedly (Dortch 1997). The release mechanism and management of the impoundment is therefore pivotal in determining the effect 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). In addition to altering the chemistry of water flowing out of the impoundment, altered discharge regimes may lead to changes in downstream processes (e.g. increased sediment transport) which themselves may have a profound effect on water quality (Hart and Allanson 1984).

A comprehensive comparison was made of the effects caused by impoundments on the water quality of two rivers, namely the Palmiet River (Western Cape) and the Buffalo River (Eastern Cape). These findings are reported in O'Keeffe, Palmer, Byren and Davies (1990) and Davies *et al.* (1993). A wide range of water quality variables was examined, but none were affected consistently by all six impoundments in the same manner. Whilst differences between the chemistry and character of the two rivers appeared to have little influence on the type of effects noted, the position of the impoundment on the river was important. Despite different operating mechanisms, upper-river impoundments all exhibited increases in median spot temperature and only weak effects on water chemistry. Such effects showed recovery within 3 km downstream of the impoundment. Reservoirs further down the river caused more severe changes and typically took up to 30 km to reach pre-impoundment values. Although it was difficult to separate reservoir size from position on the river, since all larger impoundments tended to be in the middle and lower reaches, larger impoundments did appear to cause more intense impacts on water chemistry downstream and longer distances were required for the river to recover from these impacts. Because effects on water quality variables were not consistent, predictions regarding the effect of impoundments on water quality variables are therefore difficult to make. The above authors emphasise that generalisations regarding regulatory effects of impoundments should be made with caution.

3.7.1 Temperature

Pre- and post-impoundment studies have shown that a narrowing in the thermal range associated with discharged water is frequently observed. Pitchford and Visser (1975) cited by Davies *et al.* (1993) noted that after completion of the former Verwoerd Dam on the Orange River, because of thermal inertia in large water bodies, discharges tended to be cooler in summer and warmer in winter. As a result, the seasonal temperature range was decreased from approximately 20° C to 13° C. Reduction of annual temperature ranges was also found in five out of six impoundments investigated by O'Keeffe, Palmer *et al.* (1990). The reservoir management strategy, in addition to offtake depth (i.e. whether the discharge is hypolimnetic, epilimnetic or from multi-point discharges) exert a great influence on the resulting water temperature in down stream reaches. Hypolimnetic discharge of cold, but thermally constant water can lead to disruption of thermal cues and have major impacts on the lifecycles of downstream biota (Gore 1994). King, Cambray and Impson (1998) found that epilimnetic releases of water from Clanwilliam dam on the Oliphants River, South Africa (mean temperature 19°C) were effective in encouraging spawning of the rare *Barbus capensis*. Hypolimnetic releases on the other hand were too cold, causing fish to move away from spawn beds and caused deformities in free embryos and larva spawned earlier during a warm spell. Alavian (1997; cited by Skoroszewski 1999) as part of the environmental flow assessment for the Lesotho Highlands Development Project, made predictions of the likely water temperature of releases from Mohale Dam. The predictions were of the probable temperatures to be expected since the Dam had not been completed and the way it would be managed had not been completely defined. It was predicted (using a reservoir temperature simulation model) that water temperature downstream of the dam would depend on the drawoff level as well as the volume of water released. Under certain conditions, significant impacts to the seasonal thermal regime could occur downstream of the dam.

The effects of release water on temperatures downstream can be ameliorated by attention to the level in the impoundment water column from which release flows are drawn (Bluhdorn and Arthington 1994). Considerable research in this field has been undertaken in USA as a result of the collapse in the salmon industry. Construction of dams, largely for hydroelectric power generation, has resulted in changes in temperature regimes. This in turn has resulted in decreased spawning and recruitment of juveniles (Anderson, Orlob and King 1997). Because of the financial implications, much research effort has been directed towards constructing models that can predict temperature regimes in the tail waters below hydroelectric power schemes.

3.7.2 pH
Little could be found in the literature concerning the effect of impoundment on the pH of the water. In the study of O'Keeffe *et al.* (1990) detailed above, the pH of out-flowing water was found to increase, decrease or remain unaltered depending on the particular impoundment. It is likely that in eutrophic reservoirs, the pH of discharged water may be higher than that flowing into the reservoir as a result of increased photosynthetic activities of phytoplankton in the system.

3.7.3 Dissolved oxygen (DO)
Due to biological activity and the absence of mixing with other water layers, the hypolimnion of stratified reservoirs tends to become depleted in oxygen. Release of such waters, especially if accompanied by a high silt load or high H_2S levels, can lead to anoxic conditions down stream and possible death of susceptible biota (Armitage 1984). The DO level of water can be increased rapidly however by means of special aeration structures which cause the water to become turbulent (Bluhdorn and Arthington 1994).

3.7.4 Total suspended solids (TSS)
One of the most important effects of impoundments on river chemistry is the reduction in transport of suspended sediments, which may be especially marked in the absence of hypolimnetic releases. Cessation of silt transport can have profound effects on riverine systems. Firstly, silt may be deposited within the lower reaches of the in-flowing river, increasing the danger of flooding in those areas. Secondly, suspended sediments settle out within the body of the reservoir, which shortens the life span of such impoundments and necessitates costly excavations. In addition, increased water clarity can lead to an increase in algal growth, both within the reservoir and in the river reach below the outlet. This is especially likely if accompanied by elevated nutrient concentrations (Bath *et al.* 1997). Washing away of lighter sediments may occur immediately below the dam wall and can lead to erosion in this area (Leliavsky 1955; Davies and Day 1998). Finally, inhibition of sediment transport may have catastrophic effects on downstream flood plains and deltas (Davies, O'Keeffe and Snaddon 1993). On the other hand, a common result of the construction of many impoundments is that

overall, flow is reduced compared to the pre-impoundment situation. This can result in enhanced siltation of the river channel further downstream, a problem frequently exacerbated by encroachment of aquatic macrophytes and riparian vegetation.

Hypolimnetic releases, although potentially supplying some sediment down stream, tend to be connected with other problems. Such sediments may carry high loads of metal pollutants, nutrients, and organic compounds (Bath *et al.* 1997).

3.7.5 Conductivity and total dissolved solids (TDS)

Impoundment of rivers tends to result in a reduction in seasonal amplitudes not only of temperature but also of other water quality variables (Hart and Allanson 1984). Toerien and Walmsley (1979) found that impoundment of water in Rietvlei Dam tended to smooth out variations in the chemical quality of the water with regard to sodium, potassium, chloride and sulphate. Levels of these chemical constituents were lower in the dam outflow than in water just above the impoundment during the dry weather and higher during the wet season. Calcium levels were lowered in the impoundment. The authors postulate that this may be due to precipitation of calcium compounds, such as $CaCO_3$ formed during high pH conditions created by algal growth. Silicate was also retained within the impoundment as a result of diatom growth or chemical precipitation or both. Sodium, potassium, magnesium and sulphate on the other hand did not appear to be retained within the system.

Bluhdorn and Arthington (1994) also observed an attenuating effect on natural salinity ranges due to impoundment. The bulk of the water in Lake Barambah impoundment, Australia, is due to high volume discharge that is typically of low conductivity. The overall effect is to dampen the normal range of electrical conductivity so that the high 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.

In other cases, excessive evaporation from the water surface can lead to elevated levels of TDS, especially in shallow reservoirs with a large surface area. This problem is obviously more important in hot, arid parts of the world.

3.7.6 Nutrients

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). Toerien and Walmsley (1979) found that the load of ortho-phosphate leaving Rietvlei dam was also reduced.

Nitrates on the other hand may or may not be retained in a reservoir. Toerien and Walmsley (1979) found that nitrate concentrations in water flowing out or the Rietvlei dam, SA, were higher than in in-flowing water. They attributed this to contributions form nitrogen-fixing algae within the reservoir. Pedrozo and Bonetto (1989) on the other hand found that inorganic nitrogen transport in a highly regulated South American river was lower than expected. They postulated that this was due to denitrification within the reservoirs, which acted as important nitrogen sinks.

In conclusion, 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 water quality. In this regard, O'Keeffe (1995) cites the Buffalo River (Eastern Cape) and Sabie River (Mpumalanga) as examples. In the case of the former, water quality is improved downstream of the Laing Dam due to the impoundment acting as a "settling pond". As a result water downstream of the dam carries substantially lower loads of nutrients and salts than are found in the river upstream. In the case of the Sabie River, poor water quality (arising from the formation of stagnant pools) could be improved during periods of low flow by supplementation with water from off-channel impoundments. As noted by O'Keeffe (1995), it is important for ecologists to work with engineers to develop optimum management strategies for operation of these structures. According to King, Tharme and Brown (1999) it is imperative that release mechanisms for dams be designed that do as little damage as possible to rivers rather than relying on the tried and tested methods. These authors list several features that should be incorporated in dam design. These include structures that allow multiple-level releases as well as a linkage between the release structure and a facility recording natural flow higher in the catchment. Thus it would be possible to match releases from the impoundment with climate and therefore incorporate much of the natural flow variability in the downstream reaches.

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

Transfer of water from regions of surplus to those in deficit has become an increasing frequent phenomenon and yet up until recently, little research has been carried out concerning the ecological implications of such schemes (Davies *et al.* 1993). 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). The effects of IBTs on water quality have been summarised recently by Snaddon, Davies and Wishart (2000) and therefore will only be considered briefly. Most of the following is taken from the above document. 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 flow in the recipient system is also important. Water temperature and TDS appear to be water quality variables of especial concern in connection with IBTs. In addition, transfer of pollutants from one area to another can be problematic.

In regard to water quality issues, several other points are of interest. Firstly, during times when water is not being actively pumped, water can stagnate in connecting pipes and canals. This may result in blooms of filamentous algae that can cause blockages and water quality problems when the algae decompose. Secondly, the donor water source is frequently subjected to impoundment before transfer to the recipient water body. Thus the changes in chemical constituents and physical variables associated with reservoirs may also affect the transferred water. Finally, it should be noted that as in the case of impoundments, IBTs can also be used to improve water quality, usually by means of exerting a dilution effect on point-source pollution or on river reaches with high salinities.

CHAPTER 4

METHODS USED IN THE LITERATURE TO MAKE QUANTITATIVE PREDICTIONS OF WATER QUALITY

4.1 Introduction

Several methods have been used in the literature to predict the effect of change in discharge on the concentrations of chemical constituents and values of physical variables in streams. These range from simple, statistical or numerical techniques, through to complex, data-intensive computer models. Indeed, all these techniques can be considered to be "models" of increasing complexity (Martin 1987; Crockett 1994). Many of the complex computer models (specifically the empirical or so-called "black box" type - see below) are underpinned by regression relationships between flow and the concentrations of relevant constituents. According to Dortch and Martin (1989), models may be defined as formal expressions of relationships between defined entities expressed in mathematical or quantitative terms (a definition that is applicable to models in general and not just those pertaining to water quality). Because they are easier to understand and manipulate than the real system that they represent, different options and scenarios can easily be examined (Dortch and Martin 1989). Modelling can therefore be used as a replacement for experimentation in a real system which is often too inconvenient, time consuming or costly, or because it is physically not possible to create the test conditions in the real system (Grantham, Schaake and Pyatt 1971). For example, a modelling exercise might be performed in order to identify the set of conditions (discharge, season, temperature, effluent loads etc.) that would lead to critically low DO levels in a given river. Such an exercise may be useful in identifying critical processes and conditions, as well as yielding other information of potential value in management of that system. Manipulation of the above factors in an actual river however, in order to determine these critical conditions, may well have disastrous effects on the aquatic ecosystem!

Increasing sophistication of water quality management practices and greater demand on water resources, coupled to rapidly escalating computational capacity, has resulted in the wide use of computer models (Crockett 1994). Consequently, the behaviour of many water quality constituents can now be predicted, not only in response to changes in flow but also in response to other factors, such as changes in catchment landuse and in acid rain (Tarboton and Cluer 1993; Crockett 1994; Arnell 1994). An increasingly important aspect is the use of models to predict the effect of global climate change on water quality (Avila, Neal and Terradas 1996). Modelling exercises can also be extended to examine the effects of changes in discharge and/or water quality constituents on ecological processes such as the growth of fish, phytoplankton or aquatic macrophytes (Chen and Wells 1976; Rossouw and Quibell 1993; Armitage 1994; Pieterse and Janse van Vuuren 1997). This aspect is discussed more fully in Chapter 6.

Water quality modelling, in the context of water 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 flow value. For accurate results, the simulated data that are produced should incorporate seasonal variation as well as the inherent randomness of natural systems, so that the statistical properties of the simulated data mirror closely those of the original data set (Martin 1987). Results are often expressed as changes in water quality along the length of a stream, for a given point in time. Alternately, predictions may be represented as changes in water quality with time, at a given monitoring station or point on the river. These two different ways of expressing water quality results are illustrated in Fig. 4.1. This figure also introduces the concept of steady state versus dynamic systems, which is discussed in section 4.2.

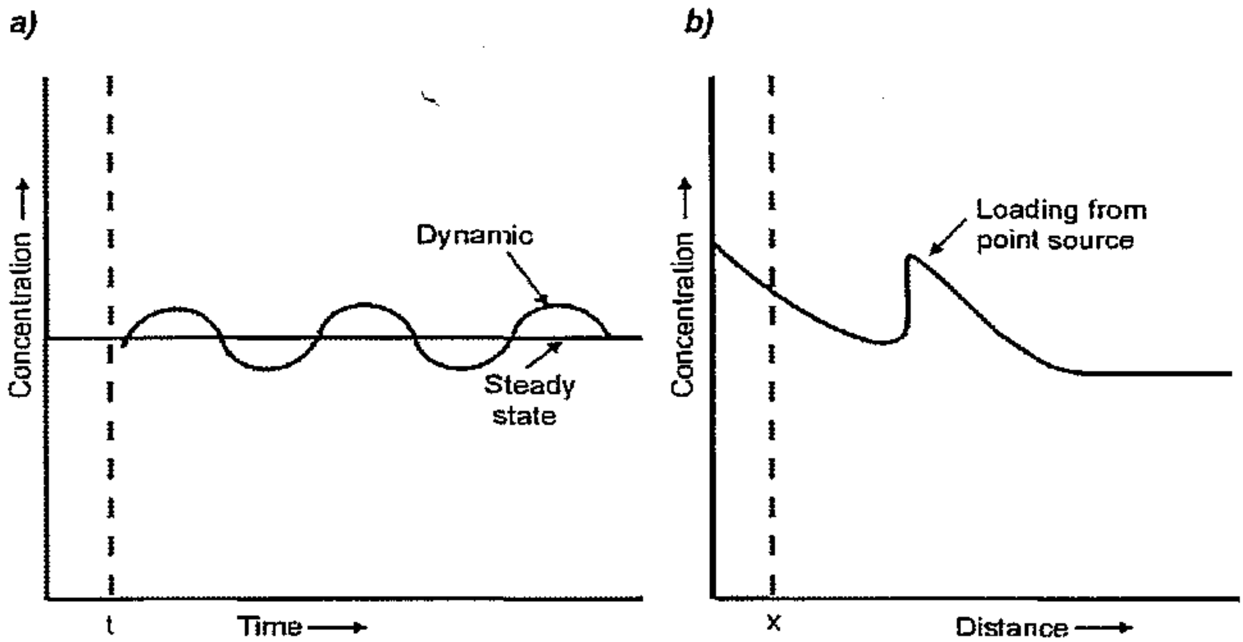

***Fig 4.1** Example of concentration distribution with time and space for steady-state and dynamic conditions. Where a) represents changes in concentration with time at point x on a river, and b) changes along the length of the river at time t. (adapted from Dortch and Martin 1989).*

4.2 Some definitions and terms used in water quality modelling

General classes of water quality models have been reviewed in Dortch and Martin (1989) as well as Crockett (1994) although, as noted by the last author, the different types are not always clear-cut and there may be a graduation between classes. Brief descriptions of the different types of water quality models, as well as explanations for some of the most common terms used, are given below:

System:

The assemblage of elements which are tied together by common flows of materials or information (Himmelblau and Bischoff 1968).

Simulation:

The study of a system, or its parts, by manipulation of a mathematical representation (Himmelblau and Bischoff 1968). Put more simply, simulation involves development of a model of a system and subjection of the model to various environmental situations to explore the nature of the results which are equivalent to, or in some way representative of, the system (Grantham *et al.* 1971).

***Parameters*:**

A parameter is a property of the process or its environment that can be assigned arbitrary numerical values. It is also a constant or coefficient in an equation describing that process (Himmelblau and Bischoff 1968). It is necessary to distinguish between parameters that are constant for all simulations in a system and those that might differ from reach to reach. For example, the number of daylight hours per day would be constant for all simulations carried out during the same time of year. The settling rate of algae or organic phosphorus on the other hand, might differ depending on the velocity of the water flow in each reach.

Variables:

Forcing variables are those that drive the system and include discharge from headwaters, tributaries or point sources as well as climate variables such as temperature. Another climatic forcing variable is rainfall, which is responsible for driving the system in catchment runoff models (see section 4.3). Whilst the instream concentrations of water quality constituents is also variables, they are dependent on discharge and are therefore not forcing variables.

Conservative or non-conservative:

The terms "conservative" and "non-conservative" chemical constituents were defined in section 2.1. Conservative chemical constituents (e.g. TDS, Cl^-) do not undergo chemical transformation in their progress along a watercourse. Non-conservative constituents on the other hand (e.g. nutrients, organic carbon, 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 of non-conservative chemical constituents is therefore more challenging than of conservative constituents (Dortch and Martin 1989).

Multi-constituent:

Some modern water quality models are able to predict physical, chemical and biological interactions between many of the chemical constituents and living organisms found in natural waters (Gromiec, Loucks, and Orlob 1983). Other models are restricted to simulation of a limited number of water quality variables, such as temperature and DO. In many multi-constituent models, simulation of some variables can be "turned off" if they are not required, or if data are unavailable. Other water quality variables, notably temperature, frequently need to be specified in order to model other temperature-regulated processes such as growth, respiration, nitrification and decomposition rates (Dortch and Martin 1989).

Steady state or dynamic:

A water-quality determinand or process may be simulated in a steady state condition, in which case the quality and quantity variables do not change with time. Alternately, they may be dynamic and thus vary with time (Gromiec *et al.* 1983; Crockett 1994). This concept was illustrated in Fig. 4.1. Note that the concentrations of chemical constituents, or values of physical variables, may be steady state or dynamic. In addition, flow may also be either steady state or dynamic. The model QUAL2E, for example, is semi-dynamic in that discharge is presumed to remain steady with time, but some water quality variables are allowed to vary diurnally (Rossouw and Quibell 1993). Dynamic models can also be used to obtain steady-state solutions by simply holding their boundary conditions constant (Dortch and Martin 1989). Although dynamic models may yield more realistic simulations, they are relatively demanding in their data and computational requirements (Martin 1987).

Zero, one-, two- or three-dimensional:

In setting up a model, assumptions are made concerning the mixing of pollutants or chemical constituents. A zero-dimensional model is a representation of a fully-mixed tank reactor. In the case of one-dimensional, longitudinal models for rivers, it is assumed that vertical and lateral mixing has occurred and that concentration varies only along the length (Gromiec *et al.* 1983). Such models should not be confused with one-dimensional, *vertical* models that may be employed for simulations of water quality within impoundments (Dortch and Martin 1989). Two-dimensional models take into account layering of water bodies and should be used for instance in the case of stratified lakes, where incomplete vertical mixing may occur. Three-dimensional water quality models, in addition to the above, also take into account any lack of lateral dispersion or mixing. Considerable complexity and computational expense are added to the application as the number of dimensions is increased and water quality models used for simulating streams or rivers are usually one-dimensional, longitudinal (Dortch and Martin 1989). In the case of very shallow, wide rivers, or those that exhibit considerable meandering, it may be inaccurate to assume lateral homogeneity (Crockett 1994; Gu 1998). In such a river for example, water temperature may well be lower in the middle than in the shallows at the side. A poor correspondence between simulated and measured values of water quality variables would then be obtained if a one-dimensional, longitudinal model were to be employed.

Deterministic or stochastic:

It was explained in Chapter 3 that even in the absence of fluctuations in discharge, some water quality variables tend to exhibit seasonal and daily variation. Because of the inherent randomness of natural systems, the concentration of a chemical constituent or value of a physical variable will fluctuate around a mean value, even if these variations are accounted for. Deterministic models have a fixed relationship between input and output and it is assumed that neither is subject to random variations (Crockett 1994). Stochastic (probabilistic) models, on the other hand, take into account the randomness (uncertainty) of biological, physical and chemical processes (Himmelblau and Bischoff 1968; Gromiec *et al.* 1983). According to Martin (1987), the use of stochastic models can be visualised as being similar to taking a relatively straightforward deterministic model and running it a large number of times (e.g. several hundred times) with the input to the model being generated statistically. If at the end of all the runs, the input data are examined, the statistics of the simulated input data should be very similar to those of the real world counterpart. The distribution of water quality variables in the model should thus be similar to that in the real river under the conditions modelled.

Empirical or conceptual:

The terms "empirical" and "conceptual" appear to be used interchangeably with the terms "black-box" and "mechanistic" in the literature (see Fawthrop 1994 and Crockett 1994). Conceptual (mechanistic, also termed descriptive) models attempt to represent the processes occurring within the river system. Empirical or "black-box" models on the other hand, simulate the relationship between input and output without simulating the internal processes (Crockett 1994; Fawthrop 1994). The chemical constituent rating curves described in Chapter 3 are simple examples of empirical models. Dortch and Martin (1989) point out that no water quality model is completely mechanistic, simply because not all of the cause-effect relationships are quantified, or even known. According to these authors, it may not be necessary to describe all the mechanisms in order to resolve a process adequately and thus be able to produce accurate predictions. Decomposition reactions, for instance, are usually described as the product of concentration and a decay rate. The decay rate implicitly includes the effects of several mechanisms, including microbial activity. Although not all the mechanisms involved are thoroughly understood, the overall process can be quantified accurately enough for the requirements of most water quality models.

Conservation of mass:

The principle of conservation of mass is embodied in the mass conservation equations, which are a feature of many water quality models (Linders 1980). Such equations represent a type of accounting system, and help to keep track of all sources and sinks of each chemical constituent. According to Dortch and Martin (1989), a generalised mass balance equation for a chemical constituent can be described by:

$$\text{Accumulation of mass} = + \text{Loadings} \pm \text{transport} \pm \text{reactions}$$

where accumulation refers to the rate of change of mass or energy within a control volume. In the case of a conservative constituent for example, the rate of change will be dependent on the amount and rate of that material entering the system (i.e. loadings, or sources). Input sources may include transport from upstream reaches, tributaries, point and non-point sources. The rate of change is also dependent on the rate and amount of materials leaving the system due to transport down-stream. In the case of non-conservative constituents, the mass balance equations are complicated by the fact that chemical reactions, uptake by aquatic organisms, binding to sediments, and other process affect the concentration of that component and must also be taken into account (they are often referred to as "sinks"). In addition, chemical transformations or release by sediments and aquatic biota may lead to input of the constituent under consideration (referred to as "sources"). In some catchments, contributions from non-point sources of pollutants may be greater than point sources (DWAF 1995). In order to produce accurate simulations, good estimates of the loadings, irrespective of their origin (point or non-point source) need to be made. According to Dortch and Martin (1989), the complexity of the mass balance equations can vary tremendously from simple calculations that can be done on a hand-held calculator to complex numerical models.

4.3 Distinction between instream and catchment runoff models

Instream models provide a means to predict the impacts of natural processes and human activities on the physical, chemical and biological characteristics of water in a river. Catchment runoff models (also termed watershed models) on the other hand, simulate the hydrologic processes by which precipitation is converted to streamflow and the resultant water quality (Wurbs 1995). In the case of the former, only processes occurring in the river are considered whereas in the latter, factors arising in the catchment are also taken into account. Catchment runoff models are especially useful in assessments of the impact of nonpoint sources on water quality. A comprehensive review of methods for assessing the impacts of nonpoint sources is given in (Pegram and Görgens *in press*). The techniques reviewed include simple knowledge based and data analysis techniques as well as process models of varying complexity. In addition, catchment runoff models are useful for assessing the effects of change in landuse on water quality. The difference between instream and catchment runoff models is considered further in section 4.6.3.

4.4 Selection of a water quality model

Many different types of water quality models exist and the choice as to which is the most appropriate, depends on the purpose of the specific study. In addition to the generic, commercial type of models (e.g. QUAL2E, MIKE-11), many have been developed for specific riverine systems or projects. Examples of this latter type of model include computer software designed to predict water quality, algal growth and fish production in the Boise River, USA (Chen and Wells 1976), simple mathematical relationships used to describe salinisation of the Barker and Barambah Creeks, Australia (Bluhdorn and Arthington 1995), as well as TOMCAT, a model originally designed to simulate ortho-phosphate concentrations in the River Thames, UK, but which is now available for use in other catchments (Brown 1986; Kinniburgh, Tinsley and Bennett 1997). There is no single, best, water quality model for all streams and all planning situations (Gromiec *et al.* 1983). In addition, it is not essential for the model to be a completely realistic representation of the system. Indeed, over-realistic models may produce a mass of output from which it is difficult to assess what is meaningful (Grantham *et al.* 1971).

Crockett (1994; citing Grimsrud *et al.* 1976) presents a very clear and useful account of how to chose the most suitable model for a given task. In addition, this author gives an outline of the steps that must be taken during the modelling process. Most of following discussion is taken from the above work. A summary of the process of selection and application of the model is given in Table 4.1.

4.4.1 Model capabilities

The chosen model must be suitable for the problem under investigation. For example, a simple steady-state model may be inadequate for simulating the effects of intermittent storm overflows on a river where sudden, rapid changes in discharge occur. Frequently, detailed information on the transient behaviour of a water quality determinand is required with a description of the stochastic aspects of water quality. In such cases, the added effort that is required in using a dynamic model may be justified (Whitehead, Beck and O'Connell 1981). In general, the simplest model should be employed that can produce the required results.

Table 4.1 *Summary of steps for the selection and application of a water quality model. (Adapted from Crabtree et al. (1987) and Crockett (1994)).*

A. Define the problem and determine:

i) What questions need to be answered?

ii) What information is required?

iii) What information is available (and hence what, if any additional data must be obtained by field measurements)?

iv) What is the required level of precision and accuracy or degree of confidence in the results?

v) Identify what control options are available (i.e. what variables can be manipulated in order to produce a solution to the problem).

B. Apply an appropriate model (the simplest that can provide the answers):

i) Select or develop a model that fits the problem, not a problem that fits the model.

ii) Use the least sophisticated model that will provide the required level of accuracy.

iii) Do not confuse model complexity with accuracy or precision.

iv) Question whether increased accuracy is worth the commensurate increase in effort and cost.

v) Assess model sensitivity.

C. Evaluate the results and implications of the predictions produced by the model:

i) Consider the implications of any modelling assumptions.

ii) Consider the implications of the degree of confidence in the results.

iii) Assess the value of the results.

iv) Reassess, in the light of the results, the suitability and relative significance of the available options (e.g. different flow regimes).

v) Identify any further possible course of action or new options.

D. Make recommendations or decisions.

Figure 4.2 illustrates the increasing complexity and data requirements of different modelling projects. It can be seen from the figure that, whilst predictions of the concentration of conservative constituents are relatively straightforward, simulation of eutrophication and toxic substances is not.

Models are usually able to simulate a limited number of water quality determinands only and care should be therefore exercised in selection of the appropriate ones. The chemical constituents and physical variables which are of specific concern in themselves should be chosen, as well as those which are also representative of the broader set of substances which cannot all be modelled in detail (World Bank 1997). For example, a particular conservative substance may be selected and modelled and the results extrapolated to other conservative constituents. Measurements of Na^+ (unlike Cl^-) were very scarce for the River Rhine. By making a linear regression between the two water quality constituents, the concentration of Na^+ could be derived accurately from that of Cl^- for a modelling exercise on this river (Linders 1980).

4.4.2 <u>Data requirements</u>

Before attempting any modelling exercise, the data requirements of a proposed model should be assessed. In addition, all existing data relevant to the river and the problem under consideration, should be assembled and appraised with regard to accuracy and completeness. The possible need for monitoring programs to obtain additional water quality data should be considered. Frequently there is the situation of adequate discharge data, but sparse or inaccurate water chemistry data (Farrimond and Nelson 1980). The data requirements for different models increase with the complexity and scope of application (Linders 1980) and this is illustrated in Figure 4.2. Water quality and ecological models frequently require hydrological input and the use of flow duration statistics (i.e. data relating the magnitude of flows, how often they occur and for what duration) is a common interface between hydrological and water quality models (Fawthrop 1994). It must always be kept in mind that the accuracy of model projections is severely constrained by the quality and quantity of the available data used to calibrate and test (validate) the models (Crabtree, Cluckie, Forster *et al.* 1987).

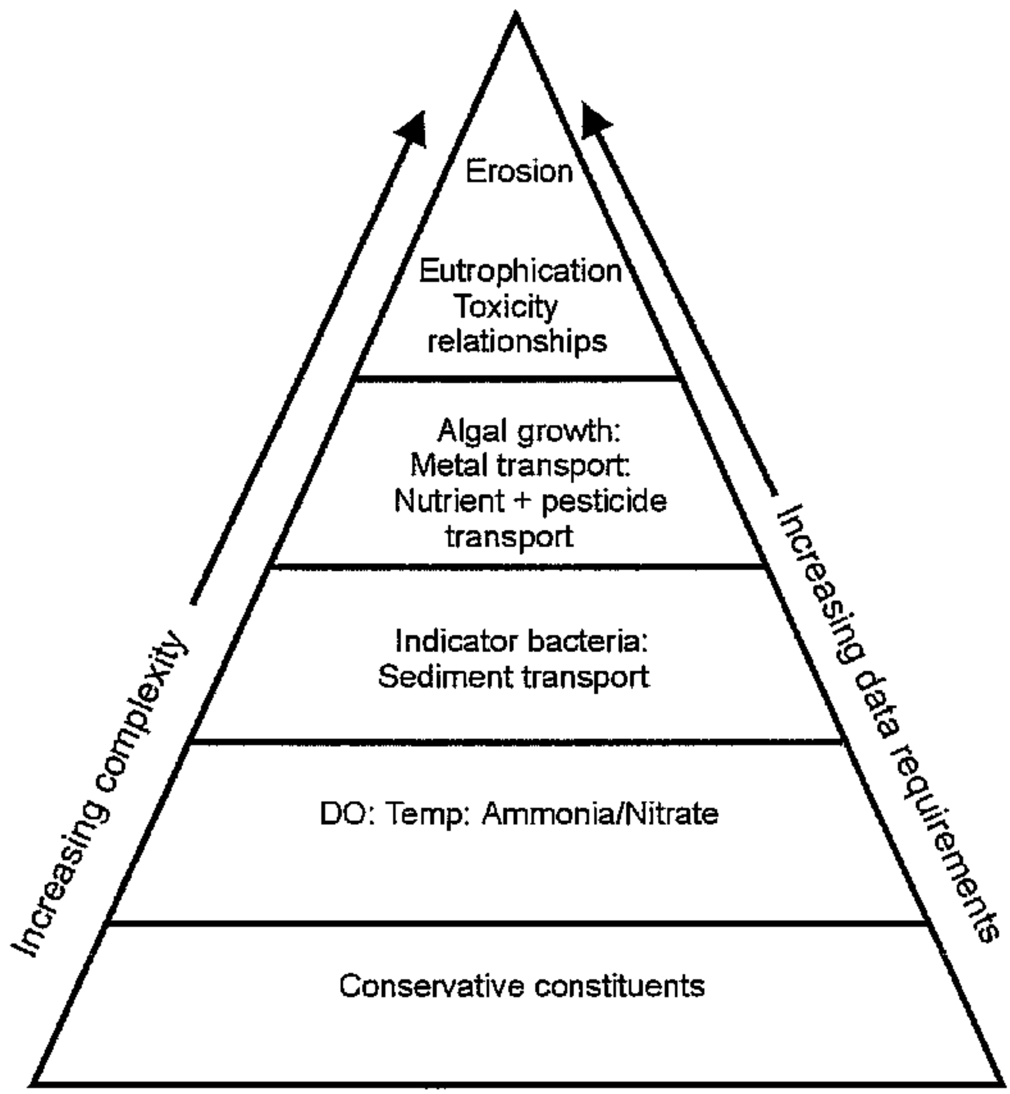

Figure 4.2 *Relative difficulty and the data requirements of applied modelling. (from Hines et al. 1975, cited by Crockett 1994).*

4.4.3 Modelling costs

There are several costs associated with projects aimed at water quality modelling. Firstly, an expense is incurred in acquiring the actual model, the necessary documentation (User Guides etc.) and the computer equipment required to run it. It is usually cheaper to use, or adapt, a ready-made model than to develop a new one, although commercial, proprietary software packages (e.g. MIKE 11) can be expensive to acquire. The Danish Hydrological Institute quoted the present authors the equivalent of R168 000 in October 1998 for MIKE 11! Public-domain versions of some models on the other hand, are available. Examples of these include WQAM, QUAL2E and WASP (World Bank 1997). Many public-domain models, including QUAL2E, are supported by the United States Environmental Protection Agency (US EPA) and are freely available (see in References, under US EPA, for web site address). There is however, a need to balance the cost of purchasing a commercial model against the costs in setting up a public-domain model. The former, are usually fully supported with user manuals, instruction courses and advice is readily available. Much time may be wasted (and therefore expense incurred) in learning public-domain models and troubleshooting any problems that arise.

Having acquired, or developed a model it must then be applied to the problem. Costs involved with this stage include the labour needed to set up, calibrate and verify the software. Additionally, labour may be required for collating of data and possibly for collection and analysis of supplementary water quality samples. The more complex a model is, the more difficult and expensive it will be to apply to a given situation (World Bank 1997). Consequently, as noted in section 4.4.1, the simplest model that can adequately address the problem should be employed.

4.4.4 Accuracy of simulations and the assumptions made

Inaccuracies between simulations of water quality variables and actual measured values can arise as a result of simplifications concerning the river system that are made during the modelling exercise. Such simplifications are necessary in order to reduce data and computational requirements. The nature of the simplifications or assumptions will obviously depend on the model employed. The following assumptions are frequently made when modelling water quality in streams (Dortch and Martin 1989):

- there is complete lateral and vertical homogeneity in water quality (i.e. as in one-dimensional water quality models).
- any inflows from point sources or tributaries mix instantaneously with the water in the main stream.
- flow and water quality conditions do not change with time (i.e. as in steady state models).
- flow is assumed to be uniform within a sub-reach. (Note that during the application of many models, the system is divided into several small units or sub-reaches. Although each unit is allowed to have different conditions of water quality and discharge, within a single unit these variables are assumed to be homogeneous).

Values are assigned to the various parameters in the equations set up to describe the most important processes affecting water quality in the system. Inaccurate values of these parameters can obviously result in poor predictions. In addition, inaccuracies in the original data used to calibrate the model may lead to errors in simulated results, as can the numerical solution technique used by the model. According to Dortch and Martin (1989), assumptions are made in order to simplify the modelling process and so that the mass balance equations can be solved. All water-quality models will have inaccuracies, however, and results should be interpreted with this in mind.

4.4.5 Ease of application

It is important that models be "user-friendly" and well documented. Preparation of input data to run the model should be as simple. Most modelling exercises require the question "What if?" to be asked. Consequently the input data file should be easy to adjust to allow repetition of this sort of option-effect to be carried out. In addition, the results or output should be in a form that is easily understandable.

4.5 General outline of the modelling process

A general outline of the modelling process is shown in Figure 4.3. After selection of the appropriate model, the next step is calibration. Ideally, environmental models are calibrated with a given data set and then verified using an independent data set. Calibration usually involves the adjustment of various parameters over likely ranges until calculated and actual values agree to within a given tolerance (Walker, Pickard and Sonzogni 1989). For instance, using a set of discharge-water quality data (i.e. discharge with corresponding concentrations of chemical constituents or values of physical variables) obtained from field measurements, the model parameters and equation coefficients are adjusted, so that when simulations are made for the input Q values, the predicted concentrations are close to the actual values. Several references, for example Bowie *et al.* (1985) contain comprehensive lists of experimentally derived values of coefficients, or parameters. Thus, before changing the values of parameters in a model it is useful to refer to the above references, to check that the changes fall within the normal measured range reported for that parameter.

In addition, many models give a default set of parameter values, which provide a basis for the initial calibration (Hogarth, Walden and McAlister 1995). Gonenc and Orhon (1986) used the model MODQUAL (an extension of QUALII, the forerunner of QUAL2E) to simulate water quality for the Porsuk River, Turkey. Values of the various parameters used by these authors in the calibrated model are shown in Table 4.2. Although the list of parameters will be specific to the processes that are simulated, this table gives an indication of the type of parameters that are required when using computer software for water quality simulations. These authors noted that the values of some parameters that were eventually used in the simulations fell outside the range reported in the literature. They stated however that no parameters were adjusted without a valid reason that could be related to an understanding of the physical system being modelled.

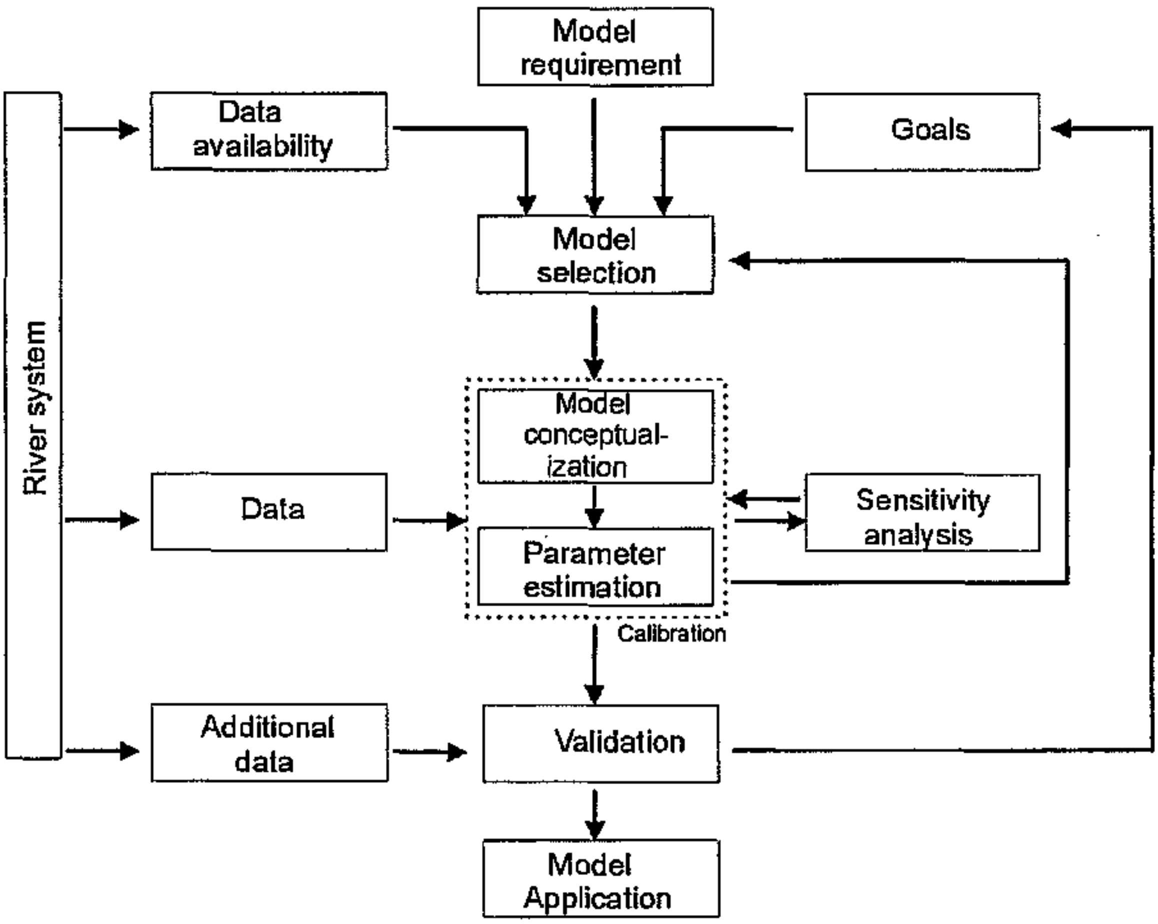

Figure 4.3 *Illustration of the modelling procedure. An appropriate model is selected after initial consideration of the goals of the project, the processes that need to be simulated and the availability of data. The model is then set up. Initial values for parameters are estimated and are refined using sensitivity analysis to examine the effect on output that changes in parameter values may have. Once the model is calibrated and simulated water quality is in agreement with the measured data, the model is validated using an additional set of measured data from the river system. Only after close agreement is obtained between the additional data set and the predicted values is the model ready for application. (Adapted from Linders 1980).*

Sensitivity tests are usually carried out. In these, all variables or parameter values are kept constant except one, which is then changed incrementally. The effect of such changes on simulation output, in other words, on the predictions of water quality, is examined. The aim of sensitivity analysis is to establish the relative sensitivity of model predictions to uncertainty in the model parameters or to errors in the input database (DWAF 1995). Sensitivity analysis enables the modeller to identify the parameters that have a marked effect on the simulated output. Thus time is not wasted finding good estimates for insensitive parameters.

Finally, the model must be validated. The model is run again using a completely independent (i.e. a new) set of discharge-water quality data from that used to calibrate the model. Predicted simulations of water quality variables are then checked against the field values. Quite often a suitable data set for validation is not available. According to Crockett (1994), this does not make the model invalid, but it will reduce the degree of confidence that can be placed in the results.

The application of various water quality models is described *inter alia,* in Brown (1986), Gonenc and Orhon (1986) and Kienzle, Lorentz and Schulze (1997). Simulations from the work of Gonenc and Orhon (1986) are shown in Figure 4.4, in which the model MODQUAL was used to predict the monthly concentration of NH_3 (amongst several other variables) at Calca, a monitoring station on the Porsuk River. Values of some of the parameters used in the application of this model were given in Table 4.2. The graph shows predicted (calculated) as well as measured concentrations. In Figure 4.5, which was taken from the same work, the concentration of DO (both simulated and measured) is shown along the length of the river. Although a reasonably close agreement was obtained between simulated and measured values, this could possibly (depending on the availability of data) have been improved by running the model at shorter intervals than one month (Fig. 4.4), or making the reach length shorter (Fig. 4.5).

Table 4.2 Process parameters used in the simulation of water quality of the Porsuk River, Turkey, using MODQUAL (from Gonenc and Orhon 1986)

		Upstream of the dam			Downstream of the dam		
Parameters to be calibrated	**Range from literature**	**Reach 1**	**Reach 2**	**Reach 3**	**Reach 1**	**Reach 2**	**Reach 3**
BOD – decay rate (K^{20})	0.10-0.9	0.10	0.47	0.25	0.47	0.25	0.25
BOD – removal rate by settling (K_3)	0-2.0	0.0	1.5	0.20	0.60	0.60	0.60
Ammonification rate (β)	0.05-0.5	0.15	0.10	0.10	0.15	0.10	0.10
Nitrification (1st step) rate (β)	0.01-0.25	0.15	0.25	0.65	0.25	0.25	0.35
Nitrification (2nd step) rate (β)	0.10-2.0	2.0	2.0	2.0	2.0	2.0	2.0
Denitrification (β)	0-2.0	0.2	0.0	0.0	0.3	0.3	0.2
Settling rate of org + part-P (σ_2)	0-2.0	0.0	0.0	0.0	0.0	0.0	0.0
Nett adsorption rate for ortho-P(f)	0.01-1.0	0.0	0.0	0.0	0.0	0.0	0.0
Maximal algal growth rate (μ)	1.5-2.5	2.0	2.0	2.0	2.0	2.0	2.0
Parameter for algal respiration (f_r)	0.05-0.2	0.05	0.05	0.05	0.05	0.05	0.05
Parameter for algal mortality (f_d)	0.05-0.2	0.07	0.07	0.07	0.07	0.07	0.07
Decay rate detritus (δ^{20})	0.05-0.5	0.1	0.1	0.1	0.1	0.1	0.1
Settling rate algae (σ_5)	0-2.0	0	0	0	0	0	0
Settling rate detritus (σ_6)	0-2.0	0.2	0.2	0.2	0.2	0.2	0.2

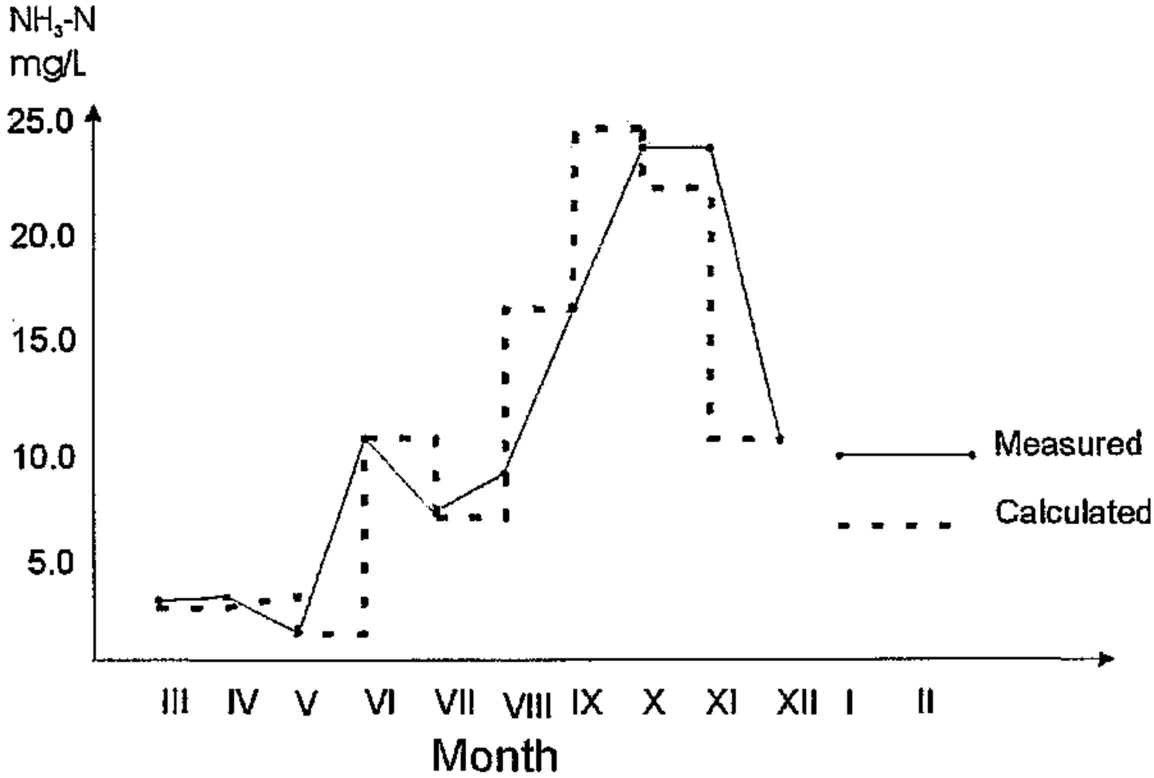

Figure 4.4 *Monthly average simulations of NH_3 (compared to measured values) for the period March 1994-February 1985, at Calca on the Porsuk River, Turkey as predicted using MODQUAL (from Gonenc and Orhon 1986).*

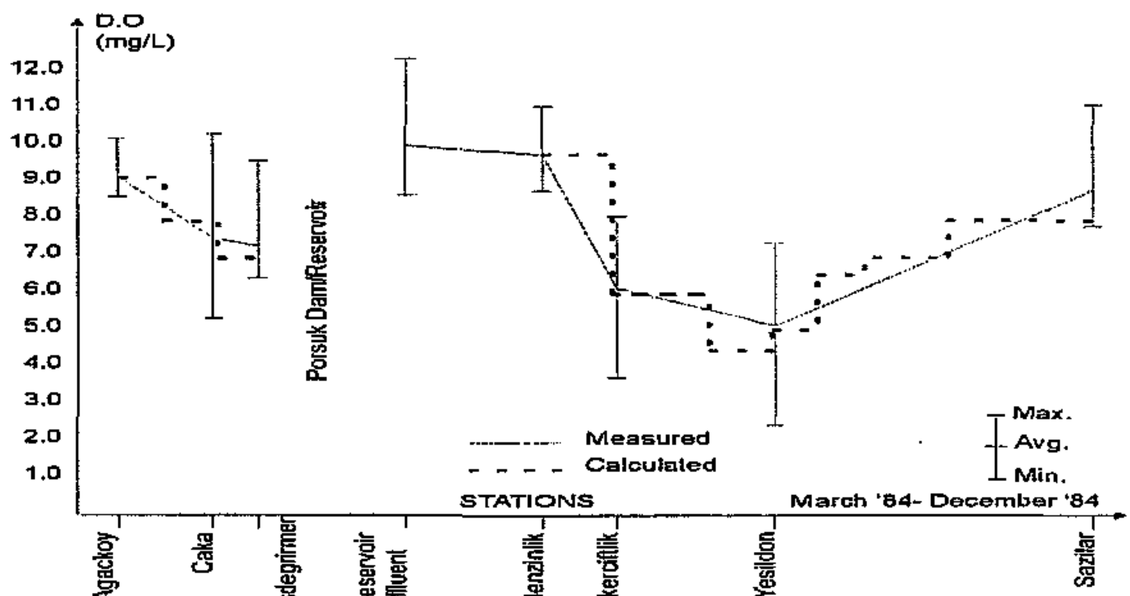

Figure 4.5 *A comparison of measured and predicted DO values (mg/l) along the length of the Porsuk River, Turkey, using MODQUAL (from Gonenc and Orhon 1986).*

4.6 Methods for predicting water quality in response to changes in discharge

4.6.1 Simple mathematical relationships

Many authors have used simple mathematical relationships relating concentration to discharge. Some have been derived from theoretical considerations, as in the case of Hem (1970; cited by Edwards 1973) and Johnson *et al.* (1969). Other authors (for example, Brooker and Johnson 1984; Bluhdorn and Arthington 1995) have performed regression analyses on field data in order to describe the Q-C relationships empirically. Hem's model, as cited by Edwards (1973) is given by:

$$\text{Log } C_{ij} = \text{Log } a - b \log Q_j$$

Where, C_i is the concentration of constituent (*i*) in a sample *j*, Q is the discharge at the time of collecting j, a and b are empirical constants.

The model derived by Johnson *et al.* (1969) is given by:

$$C = C_\delta[1/(1+\beta Q)]+C_a$$

Where, $C_\delta = C_o - C_a$, and C_o is the concentration of a solute in soil water, C_a is the concentration in rainwater. β is a constant, which is directly related to the residence time of the water moving through the system. Pinol *et al.* (1992) when deriving Q-C relationships for solutes in Mediterranean streams, used the above model but set C_o = solute concentration in groundwater and C_a = solute concentration in soil water.

Table 4.3 shows the general form of empirically derived equations that have been reported in the literature. These relationships are usually site-specific. Hence, the values of the parameters a, b and β are different for each chemical constituent, for each river and frequently for different monitoring sites on the same river. In addition, in order for the equation to accurately predict concentration for a given discharge value, the value of the parameters may need to be adjusted for different data sets. For example, Houston and Brooker (1981) found that the equation given in the table for reference 2, described the Q-C relationship for SiO_2 in the Rivers Trothy and Frome for the months January-March and November-December very well (r^2 = 0.922). If the equation was fitted to data for the whole year, however, a very poor fit was obtained (r^2 = 0.149). This is not surprising considering the large number of factors that have been found to affect Q-C relationships (Chapter 3). Irvine and Drake (1987) also found that separating data into separate seasons improved the predictive capabilities of a solute rating curve derived for sediment concentration. They suggested that hysteretic effects could be dealt with in two ways. Firstly, if enough data are available, the rising and falling limbs of the hydrograph can be modelled separately. Alternatively, an antecedent discharge term can be incorporated into the model (Table 4.3, reference 6). These authors also found that incorporating a water temperature term also gave better predictive results and they attributed this to the fact that erosion increases with temperature. In some cases, simple empirical equations were not adequate for predicting the concentration of chemical constituents. Table 4.3 shows only those water quality variables and equations for which the correlation coefficient ("goodness of fit") was greater than 0.5.

Prairie and Kalff (1988a) were able to model conductivity successfully in seven headwater streams in Canada, by using a simple numerical procedure. In their model, streamwater chemistry was considered to be the result of the mixing of two "reservoirs" (i.e. near-surface and deep-seepage) each of which contributed differentially depending on streamflow (see Table 4.3, reference 7). This model was not as successful for predicting phosphate concentrations however, and the correlation coefficient varied considerably from stream to stream. According to the authors, this was to be expected considering phosphorus-sediment interactions and the variable nature of the processes associated with changes in dissolved phosphorus in different streams. In other words, electrical conductivity, being a conservative constituent was more easily simulated than phosphate, a non-conservative variable. Inclusion of a temperature-dependent parameter was found to improve the accuracy of the model predictions for dissolved phosphorus.

***Table 4.3** Examples of simple models (solute rating curves) used in the literature to describe the relationship between concentration (C) of a chemical constituent (mg/litre) and stream flow (Q in m^3/sec). All models gave correlation coefficients higher than 0.5 for most water quality variables at most sites.*

Site	WQ variables	Regression equation	Comments	Reference
1. Three Norfolk rivers, UK	Na^+, K^+, Mg^{+2}, HCO_3^-, NO_3-N, PO_4-P, SO_4^{-2}, Ca^{+2},	Log C = a + b Log Q (Hem's model)	Si and Cl^- could not be predicted using this model. Note, not all water quality variables could be predicted accurately in all catchments.	Edwards (1973)
2. Rivers Trothy and Frome, Wales, UK.	SiO_2 PO_4-P NO_3-N	Log C = a + b Log Q	Valid for long time periods as well as for storm events. NO_2-N, NH_4-N could not be predicted. Simulation of SiO_2 was poor.	Houston and Brooker (1981)
3. Twelve Welsh rivers, UK	NO_3-N PO_4-P Cl^-	Log C = a + b Log Q		Brooker and Johnson (1984)
4. Two Danish rivers.	PIM POM	Log C = a + b Log Q	PIM = particulate inorganic matter. POM = particulate organic matter.	Kronvang (1992)
5. Illinois River, USA	F, PO_4-P	Log C = a + b Log Q		Wang and Evans (1970)
6. Ausable River, Canada	TSS	$C = 49.99 + 3.09\,(Q_i)$ or $\ln C = 4.02 + 0.0096Q_i + 0.773(dQ) + 0.044\,(\text{temp }°C)$	Where: dQ describes antecedent discharge. i.e. $dQ = 2(Q_i - Q_{i-1}) / Q_i + Q_{I-1}$ For the same data set, the correlation coefficient = 0.611 for the first equation and 0.765 for the second.	Irvine and Drake (1987)

Site	WQ variable	Regression equation	Comments	Reference
7. Seven streams, Canada	EC	$C_{st} = (\alpha C_D + QC_s/Q + \alpha)$	Where: C_{st} = observed stream EC; C_D, C_s and α are fitted constants representing the EC of deep, and near-surface reservoirs and a mixing parameter respectively.	Prairie and Kalff (1988a)
	DP	$DP = (\alpha DP_D + Q(\beta_0 + \beta_1 X\ TEMP)/Q + \alpha)$	Where: Temp = water temperature (°C); α, β_0, β_1, and DP_D = fitted constants; DP = observed concentration of dissolved phosphate in stream.	
8. Barambah Creek, Australia.	EC	$EC = 1837.4\ Q^{-0.231}$	Barambah creek is unregulated. Other factors are more important than discharge in regulated parts of the system. ($Q = 10^3 m^3 d^{-1}$).	Bluhdorn and Arthington (1995)
9. Three Mediterr-anean streams	HCO_3^-, K^+, Ca^{+2} Mg^{+2} Na^+, Cl^-, SO_4^{-2}, H^+.	$C=C_\delta[1/(1+\beta Q)]+C_a$ (Johnson's model)	Where: $C_\delta = C_0 - C_a$; C_0 = concentration in groundwater, C_a = concentration in soil water, β = a constant, proportional to the residence time of the water moving through the catchment. Note, not all water quality variables gave high correlation coefficients in all catchments.	Pinol *et al.* (1992)

4.6.1.1 A case study

Bluhdorn and Arthington (1995) derived simple mathematical models to predict EC (electrical conductivity) in Barambah and Barker Creeks, Australia. This study illustrates some interesting points relevant to modelling of regulated streams and is discussed in some detail. Due to the variable climate, the system is naturally stochastic with regard to discharge and salinity. The aquatic biota is adapted to conductivity values that vary naturally from 300 – 2 300 $\mu S.cm^{-1}$.

The upper Barambah Creek is currently unregulated. It was found that EC in this stream could be predicted from the following equation:

$$EC = 1837.4\ Q^{-0.231}$$

where:

EC = electrical conductivity ($\mu S.cm^{-1}$)

Q = discharge (10^6 litres.day^{-1})

and: r^2 (correlation coefficient) = 0.782

Thus, it was concluded that instream salinity levels were largely dependent on discharge in this part of the river system. This was termed the "intrinsic flow effect" by the authors.

Downstream of the Bjelke-Petersen Dam, models for EC predictions were found to be dependent on the origin and range of discharge:-

- During periods of low flow (i.e. in the absence of any discharge from the reservoir), all flow originated from the unregulated Barambah Creek. Conductivity could therefore be predicted from the intrinsic flow effect equation given above.
- During high volume discharge from the reservoir, downstream EC was found to be constant along the length of the creek and was equal to that of the release water (800-1 000 $\mu S.\ cm^{-1}$).
- During normal release flows, the following equation was found to predict resulting downstream EC values:

$$EC = 985 + 5.734 \times \text{distance downstream from dam (km)}$$
$$r^2 = 0.616$$

Thus EC was found to increase with distance from the reservoir. This was attributed to the increasing proportion of discharge originating from irrigation return flows (of high salinity) with distance downstream. In this middle range of discharge values, flow appeared to have little effect on EC levels downstream of the dam.

4.6.2 More complex, statistical methods

Several authors have used Box-Jenkins analysis to predict water quality. Irvine and Drake (1987) used ARIMA models to predict sediment concentrations in the Ausable River and Jayawardena and Lai (1989) to predict COD (chemical oxygen demand). This is a statistical technique that is specifically suited to time series that has been developed by Box and Jenkins (1976). According to De Jong *et al* (1989), a conceptual model is used to compare effects occurring during corresponding periods in different years. The model is considered satisfactory if it explains most of the changes, i.e. if the unexplained part (the residual) is due to random events. Irvine and Drake (1987) found that whilst the technique could be used successfully to predict sediment concentrations, it was data intensive and coefficients tended to be unstable.

4.6.3 Complex computer models

The body of literature covering water quality modelling is extensive and there is a bewildering array of computer models available that are concerned not only with rivers but with stormflow in urban regions and sewers, reservoirs, lakes etc. Similarly, a range of models is available to solve different management problems, such as estimation of non-point source loading from different landuses, or calculation of the cumulative effect of effluents discharged into a river. Only the salient points are discussed in this work and in particular those that are important or relevant to environmental flows in rivers. An inventory of computer models is given in Appendix A. This list is by no means complete, but serves to illustrate the wide range of software tools available. As mentioned earlier in this review, some models were designed for specific river systems, whilst others are generic in nature. An example of the former is the model WITSIM which was developed specifically to simulate salinity in the Upper Olifants River, South Africa. In Table 4.4, the attributes and characteristics of some of the most commonly used computer models are summarized. Also included are apparent advantages and disadvantages, and as far as can be established, the availability in South Africa and whether previously used here. Wurbs (1995) gives a detailed guide to water management models and software including many of those listed in the table. In addition, a three-part review has been recently published by the IAWQ Task Group on River Water Quality Modelling (Rauch, Henze, Koncsos *et al.* 1998; Shanahan, Henze, Koncsos *et al.* 1998; Somlyody, Henze, Koncsos *et al.* 1998). The above review discusses water quality modelling as currently in use, including the attributes of some of the most commonly used models. Problems and inaccuracies in the methods as well as future developments are also discussed. The review also provides a list of important modelling software. In addition to many of those in Table 4.4, also included are the models ATV, Salmon-Q, DUFLOW, AQUASIM and DESERT (Rauch *et al.* 1998). According to Somlyody *et al.* (1998), an important future development in the realm of water quality modelling is the incorporation of decision support systems that will guide users in model construction and usage.

***Table 4.4** Characteristics and attributes of some popular riverine water quality models documented in the literature.*

NAME	AUTHORS/ DEVELOPER	CHARACTERISTICS AND APPLICATIONS	ADVANTAGES	DISADVANTAGES	AVAILABILITY	COMMENTS	REFERENCE
*CE-QUAL-RIV1	US Army Corps of Engineers. (USACE) Waterways Experimental station	Applicable to rivers. 1-D dynamic model **Variables** = DO, IP, N (3 types) CBOD, Fe, Mn, TP, SS, *E. coli*, conservative substances.	Designed for highly unsteady streamflow conditions. Simulation of dams, locks, other control structures. Can simulate "sharp flow and quality gradients"	High level of user expertise required.	Public domain	RIV1H= hydrology model. RIV1Q= water quality model. "shows scope for application in IFA" (Tharme 1996). Specially designed for regulated rivers.	Wurbs (1995), Tharme (1996)
CE-QUAL-W2	USACE Waterways Experimental station	Applicable to complex reservoirs, lakes, rivers and estuaries. 2-D, laterally averaged. **Variables** = Chl a; DO, nutrients, Carbon, pH.	Can be used for both vertically stratified impoundments and rivers. Has been used in SA.	High degree of computer literacy required (FORTRAN) for each application.	Public domain. Local expertise available	Will be possibly one of the models used to manage reservoirs in SA.	Bath et al. (1997); US EPA 1998
Extended ACRU	Dept. Agricultural Engineering, Natal Univ. and WRC	Physical conceptual distributed catchment process model. Based on daily time steps. **Variables** = *E. coli*, sediments and P	Developed specifically for SA. Hydrological module is well researched.	Limited to simulation of *E. coli*, sediments and P. Short history of WQ modelling	Easily obtained. Expertise for hydrology component available locally	Applied to Mgeni catchment.	Kienzle et al. (1997)
HEC-5Q	USACE Hydrologic engineering centre	Rivers and reservoirs. Simulation of floods. Dynamic. **Variables** = temp, 3 conservative, 3 non-conserververvatives. DO. OR TDS, NO_3^{-2}, PO_4^{-2}, BOD, NH_3, ,DO.	-can include module that models selective withdrawal from reservoirs and effect downstream. -constituents include: temp, 3 conservative, 3non-conserv. DO. OR TDS, NO_3^{-2}, PO_4^{-2}, BOD, NH_3, ,DO.			Used in one of three modes: calibration annual simulation long-term mode. Two alternative simulation options for quality constituents.	Wurbs (1995); Dortch and Martin (1989)

***Table 4.4** Characteristics and attributes of some popular riverine water quality models documented in the literature (continued).*

NAME	AUTHORS/ DEVELOPER	CHARACTERISTICS AND APPLICATIONS	ADVANTAGES	DISADVANTAGES	AVAILABILITY	COMMENTS	REFERENCE
HSPF	Hydrocomp/ US EPA	Catchment run-off, well mixed reservoirs, instream water quality. 1D, dynamic. "Suitable for both urban and agricultural catchments "(Hogarth et al. 1995). **Variables** = "most system variables".	Good time-series management capabilities. Modules can be added. ACRU (*developed* specifically for SA) can be used as the hydrology module. Originally developed for semi-arid climatic conditions.	Complex, difficult to learn. Needs expert input.	Public domain. Local expertise available.	Although fairly data-intensive can also be used in a simplified form if data is limited. Used in ICIS (a catchment information system in use in SA). Updated continually.	Van Rensburg *et al.* (1997); Hogarth *et al.* (1995)
MIKE11 and MIKE 21	Danish Hydraulic Institute	Dynamic hydraulic model with water quality simulation abilities. *1 D (11), 2 D (21)* and *1* D two layer model (12). **Variables** = DO, BOD, nitrification, sediments.	Integrated modular structure. Add on modules eg heavy *metals, eutrophication*, rainfall-runoff. Also quasi-steady state. Relatively rapid to set up.	Relatively expensive.	Can be purchased from the suppliers in Denmark.		Used in Australia and other countries (DHI 1998). Used for modelling of estuaries in SA (Slinger *et al.* 1998).
QUAL2E And QUAL2E-UNCAS	US EPA CEAM	Well mixed streams. Up to 15 constituents. 1D, steady-state, with some dynamic features, deterministic. QUAL2E-UNCAS takes into account stochasticly of natural systems. **Variables** = TDS, N, P, DO, *E. coli*, BOD, Chl.	Well known and used. Reasonably user friendly. Good for identifying non-point loads. Models TDS, N, P, DO, *E. coli*,	Relatively simple but needs expert input. No catchment run-off module Steady state, therefore 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). Used to model eutrophication on Vaal river, SA.	Wurbs (1995); Rossouw and Quibell (1993).

NAME	AUTHORS/ DEVELOPER	CHARACTERISTICS AND APPLICATIONS	ADVANTAGES	DISADVANTAGES	AVAILABILITY	COMMENTS	REFERENCE
Riverware	CADSWES and US BR	Multipurpose, applicable to wide range of water systems eg. linked reservoirs, rivers, canals.	Can be upgraded as new features built on water system. Can be used in optimization mode.	Very expensive (Bath, pers. comm. 1998)		Simulates DO, temperature, salinity.	http://cadswes.colorado.edu/riverware.
WASP5	US EPA	For streams and reservoirs, estuaries, lakes. Water column and benthic layer modelled. 1, 2, or 3D. Dynamic, deterministic. **Variables** = TSS, TDS, DO,BOD, nutrients, toxic pollutants, and algae. Two modules TOXI4 and EUTRO4.	Water quality sub-routines easily changed or re-written by user, to make it problem-specific. Very flexible since 1,2 or 3 D. Can model water column as well as sediment layers.	High degree of computer literacy required for each application. No catchment run-off module.	Public domain.	Strong focus on organic contaminants. Can be used with DYNHYD5 as underlying hydrology model or others eg. RIVMOD, SED3D. Can be used at differing levels of complexity.	EPA (1998); Wurbs (1995); Hogarth (1995).
WQRRS	USACE Hydrologic Engineering Centre.	Applicable to streams and reservoirs. 1-D, dynamic. **Variables** = up to 18 constituents including TDS. Also fish, zooplanktom, invertebrates.			Public domain.	Package of different models. Represents "most complete set of ecological and water quality variables"	Wurbs (1995); Miller and Healon (1998)

Abbreviations: CADSWES = Centre for advanced decission support for water and environmental systems, Colorado University. US EPA = United States Environmental Protection Agency. US ACE = United States Army Corps of Engineers.

4.6.3.1 Instream and catchment runoff models

When considering computer software for simulation of water quality, a distinction should be made between catchment runoff models, (e.g. HSPF and SWMM) and instream water quality models such as QUAL2E or WASP (Wurbs 1995). Catchment runoff models simulate the hydrological process whereby precipitation is converted to stream flow. The input data for this type of model are therefore rainfall records. Some runoff models simulate only discharge, whilst others are able to simulate water quality as well. According to Pegram and Görgens (*in press*) the latter type of models usually consist of separate hydrological, sediment and water quality modules that are simulated in a sequential manner. Since pollutant and sediment fluxes are dependent on water flow, hydrology is first simulated, followed by sediment transport and finally pollutant transport. The important processes that govern non-point source impacts differ considerably between rural and urban catchments. Thus many models are aimed at use in either one or the other. Some models however, have urban and rural modules and can be applied in mixed landuse catchments.

According to Wurbs (1995), although catchment runoff models linked to water quality models can both be used to predict solute concentrations in streams, there are certain applications in which they are of particular use. Firstly, in situations where streamflow and water quality data are limited, catchment runoff models can be used to simulate discharge values from precipitation data, which are usually more complete. Secondly, due to changes in landuse within a catchment, historical discharge and water quality data may not reflect the present day situation. Catchment runoff models can be used to take such landuse changes into account and simulate appropriate water quality and discharge values. Thirdly, in a similar manner, such models can be used to predict the effects that changes in landuse (e.g. an increase in impermeable land cover, due to building and development) might have on stream discharge and water quality. In section 4.7.2, simulation of phosphorus is described firstly using an instream model (TOMCAT) and secondly using the catchment runoff model ACRU. Since it is considered by the authors that with regard to environmental flows, instream models are more appropriate, catchment runoff models are discussed to only a limited extent in this literature review.

4.7 Modelling of specific water quality variables

In order to illustrate some general points that must be considered when modelling physical variables and non-conservative constituents, simulation of DO and phosphate is discussed below.

4.7.1 Dissolved oxygen (DO)

Streeter and Phelps were responsible for development of the first water quality model in 1925, which described the BOD-DO relationship in streams (Linders 1980). Many of the riverine water quality models available today include extensions of their original equations for simulating dissolved oxygen (e.g. the model TOMCAT, (Brown 1986)). Predictions can be made of the biochemical oxygen demand (BOD) of various biodegradable constituents and the resulting DO in rivers (Grantham *et al.* 1971).

It was mentioned previously that natural purification processes occur in streams. Clean stream water is usually saturated with DO, but as sewage effluent is discharged into a stream, DO is consumed as the organic matter is degraded by bacteria. Simultaneously, oxygen from the atmosphere dissolves into the water since it is no longer saturated with DO. Finally, the organic matter is completely decomposed and the stream water becomes saturated with oxygen again. Thus two different opposing processes are in operation, which can be modelled and the resultant DO predicted. This model is given by the following equation (Gromiec *et al.* 1983):

$$\frac{dD}{dt} = K_d\, L - K_a\, D$$

Where:

D = the DO saturation deficit, or the difference between the DO saturation concentration and the concentration at time t.

L = the biochemical oxygen demand (BOD).

K_d = the deoxygenation rate coefficient.

K_a = the reaeration rate coefficient.

Dissolved oxygen sag curves can be predicted at various points downstream of an effluent point source. In many cases the main interest is the critical oxygen deficit (D_c), or lowest DO concentration, which occurs at critical time t_c. Changes in DO with time are illustrated in Figure 4.6.

In reality, the situation is not as simple as described above, and several other processes also influence stream DO levels. Goodman and Tucker (1971) and Gromiec *et al.* (1983), list the following parameters (amongst others) as being of importance when considering stream DO balance:

- The reduction of BOD due to sedimentation or adsorption.
- The increase of BOD due to resuspension of sediments, or diffusion of partly decomposed organic products from the benthic layers.
- Removal of DO due to diffusion into the benthic layer.
- The increase in DO due to photosynthesis by aquatic macrophytes and algae.
- The removal of DO due to respiration by aquatic organisms.
- The continuous redistribution of BOD and DO downstream with discharge, changes in discharge, as well as turbulence in the channel due to altered stream bed topography.
- Diurnal changes in temperature and sunlight.

In addition, nitrogenous oxygen demand, or the oxygen required to oxidise inorganic and organic nitrogenous compounds can have a significant effect on DO (Gowda 1983). According to Dortch and Martin (1989), reasonably accurate predictions of DO can now be made since the majority of processes affecting DO are well known. The processes affecting some other constituents which may in turn impact on DO are not as well studied however.

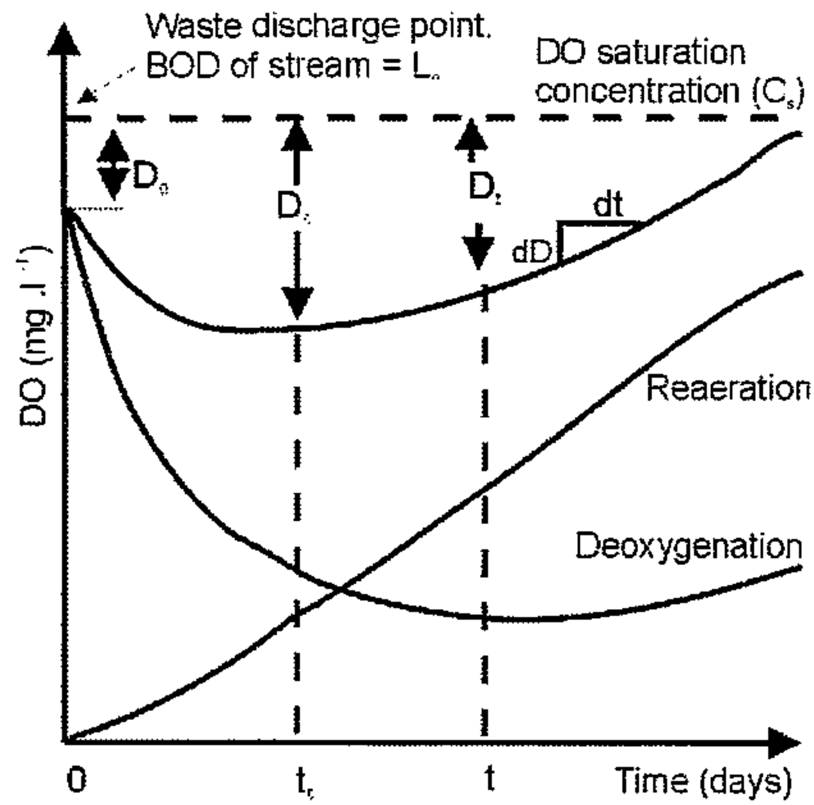

Figure 4.6 The dissolved oxygen (DO) sag curve. The Streeter-Phelps model predicts the DO concentration resulting from two opposing processes –deoxygenation due to decomposition of organic matter and reaeration from the atmosphere. D_0 = DO deficit at t=0, D_C = critical (lowest) DO level at time =t_c, D_t = DO deficit at time=t, C_S = saturated DO concentration. (Gromiec et al. 1983).

4.7.2 Phosphorus

4.7.2.1 *Point sources of phosphate*

Modelling of point sources of phosphate can be exemplified by an examination of the model "TOMCAT", which was originally developed by the Thames River Authority, but has now been extended to other rivers in the UK (Crockett, Crabtree and Cluckie 1989). This model was used by Kinniburgh *et al.* (1997) to model the average orthophosphate loads along the length of the River Thames. According to these authors, the model assumes that orthophosphate concentrations in the river decrease exponentially with distance downstream from the input source and that the rate of loss of ortho-phosphate by adsorption to sediments and uptake by plants is given by:

$$\frac{dP}{dx} = -k\,P\,(x)$$

where P(x) = the orthophosphate concentration at distance x from the source.

k = first order rate constant

The value of the rate constant (k) varies from tributary to tributary depending on the amount of sediment, shading and type of vegetation.

4.7.2.2 *Non-point source phosphorus*

Simulation of phosphorus from non-point sources with ACRU is described in Kienzle *et al.* (1997) which discusses the application of the model to the Mgeni catchment. The major sources of phosphorus are quantified in the model and include, wet and dry atmospheric deposition, agricultural input from fertilizer and livestock, as well as input from human sources such as the use of detergents in streams and the seepage from pit latrines. In the model this constituent is simulated in two separate interacting states, namely, as adsorbed and dissolved phosphorus. Phosphorus added to the soil from the various sources is distributed between the adsorbed and dissolved state. The proportion in each state is calculated by the model and is dependent on several factors including soil composition (per cent clay and organic matter), pH and soil moisture status. The final concentrations in the adsorbed and dissolved states are applied to the quantity of sediment and stormflow respectively to yield the total phosphorus from these sources.

It is important that simulations of runoff from rural areas in South Africa consider informal urban settlements and the frequent lack of adequate sanitation in these areas. In the ACRU model a distinction is made between such settlements and pit latrines within 250m of rivers and those further away from watercourses. The impact of those within the buffer zone is greater and this factor is incorporated into the model. Examples of output from the model given by Kienzle *et al.* (1997) include simulated loads of phosphorus and comparisons of observed and simulated total phosphorus concentration at a given point on a watercourse.

4.8 Modelling of solute loads

In section 3.4 it was explained that to a large extent this study is concerned with concentration rather than load, since it is the former that is more important to aquatic organisms. Consideration of total load is important when assessing changes in discharge-concentration relationships over long time periods, however. Changes in catchment land use or land cover, may well lead to alterations in solute loads and also frequently to the amount of runoff (Tarboten and Cluer 1993). Much research has been carried out to determine the loads for a given solute derived from a unit area of land and the effect of varying land use thereon (Simpson and Coleman 1993; Johnes 1996; Tufford, McKellar and Hussey 1998). Predictions of solute loads can be made using simple mathematical models in a manner similar to concentration. In addition, catchment runoff computer models (section 4.5.3) frequently simulate loadings of pollutants, nutrients and chemical constituents.

Smith and Stewart (1977) tried several mathematical relationships to relate discharge with nutrient loads in rivers draining into Lough Neagh, N. Ireland. These were derived empirically and then tested for accuracy on independent data. These authors chose a log load/log flow linear regression to describe the relationship, because it was simple, and gave steady and reliable results with highest correlations between observed and predicted values. In the Lough Neagh analysis the mathematical relationship was derived separately for each river/chemical/year combination as there were marked differences between rivers in catchment land use as well as river profile. The following general mathematical relationship was found to be useful for linking chemical constituent load and discharge (Smith and Stewart 1977; Houston and Brooker 1981):

$$\mathrm{Log}\,L = b\,\mathrm{Log}\,Q + a$$

where: L = mass flow (g/s)

Q = discharge (m^3/s)

a, *b* = empirical constants

De Jong *et al.* (1989) used the following to predict phosphate levels in the Rhine, in which point and diffuse sources of this pollutant were incorporated.

$$P_t = P_p + C_d\,(Q_t - Q_p)$$

where: P = phosphorus load

Q = discharge

C = concentration

and the subscripts refer to:

d = diffuse sources

p = point sources

t = total, sum of diffuse and point sources.

The calculation of loads especially for suspended sediments and for sediment-associated metal concentrations (i.e. determinands that tend to increase with flow) can be difficult (Jarvie, Robson and Neal 1999). Frequently there is the situation where continuous flow data is available but only intermittent concentration data. Various statistical methods have been developed to estimate loads in such a situation (Grobler, Bruwer, Kemp *et al.* 1982).

4.9 Conclusion

In conclusion, it can be said that water quality modelling in all its various forms can be a powerful tool for management of surface water resources. Such tools however, do have considerable drawbacks with regard to the extensive amount of time, money and effort that frequently must be invested in order to obtain adequate simulations. Despite this, in the words of Thomann and Mueller (1987) "for all the apparent and obvious shortcomings and difficulties of water quality modelling, there is no viable alternative." It is likely then that in the foreseeable future, water quality modelling will play an even greater role in the management of water resources.

CHAPTER 5

WATER QUALITY AND ENVIRONMENTAL FLOW ASSESSMENTS

5.1 Introduction

A considerable body of literature exists which describes the various types of methodologies used world wide to determine environmental flows, including those that have been developed in South Africa. In addition, the necessity for such assessments has been documented (Tharme 1996; King and Louw 1998; King, Tharme and Brown 1999). Consequently, this chapter is largely restricted to the incorporation of water quality considerations into environmental flow assessments. A brief discussion of what is meant by environmental flows, and why it is important to determine and implement them, is presented. This is followed by an overview of the methodology commonly used in South Africa (The Building Block Methodology; BBM) as well as a developing methodology, DRIFT (Downstream Response to Imposed Flow Transformation). The second half of this chapter is concerned with a review of literature concerning water quality and how this has been incorporated into the concept of "environmental flows".

It might be useful at this point to clarify some of the terminology used in this field. In the literature (Docampo and de Bikuña 1995; Tharme 1997; King *et al.* 1999), flow regimes specifically designed to maintain some aspect of the riverine environment have variously been called *instream, ecological, biophysical* or *environmental flows.* Although they are sometimes termed "instream flows" in South Africa (especially in connection with Reserve assessments), in this document they will be referred to as "environmental flows". This is appropriate when used in the context of holistic methodologies such as the Building Block Methodology (BBM), which considers the entire riverine ecosystem rather than just the instream component (R. Tharme, UCT, *pers. comm.*). Thus an environmental flow assessment (EFA) is performed in order to determine the environmental flow requirement (EFR) for an entire river, another type of aquatic ecosystem e.g. a wetland, estuary, or a subsection thereof.

5.2 What is an environmental flow assessment?

As human populations increase, so does the demand for water. This inevitably means that sooner or later, conflicts arise between abstraction of water from rivers for human consumption or use and the need in aquatic ecosystems for water to maintain biological integrity. It is in an attempt to resolve these two opposing demands on limited freshwater resources that the science of environmental flow assessment has developed. An EFA 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.* 1999). The riverine features of concern may be as specific as a species of fish or tree that should be retained in the system, or the goal may be as general as "maintaining the ecosystem in its current state". In other words, an EFA is carried out in order to obtain an estimate of how much water can be abstracted without an unacceptable level of degradation of the riverine ecosystem. In the case of a highly modified river from which excessive abstraction is taking place, the EFA might specify that much of the original flow should be reinstated in order to rehabilitate the ecosystem to some required condition (King *et al.* 1999). In South Africa, EFAs are now usually linked with the resource classification system (A-F) as outlined in the Resource Directed Measures for protection of water resources (DWAF 1999) for determination of the ecological Reserve. For example, if the river reach in question is currently categorised as a "C" class river, the aim of the EFA may well be to prescribe flows that will result in an improvement to a "B" class.

Interestingly, environmental flows are not a new concept. Petts, Maddock, Bickerton *et al.* (1995) cite the case of the Halifax Corporation Waterworks Act of 1888. This act set a precedent by stipulating that compensation water should be released when required in order to maintain the aesthetic quality of a local beauty spot downstream of an impoundment.

5.3 The necessity for environmental flow assessments

The need for implementation of environmental flows can be justified from both the point of view of the environment, as well as from that of resource economics. It is becoming increasingly clear that flow manipulation and regulation of rivers is linked to serious degradation of riverine ecosystems (Petts, 1989; Docampo and Bikuña 1995). Symptoms of such ecological degradation include the loss of species (invertebrates, fish, vegetation), poor water quality, loss of aesthetic appeal and general impairment of ecosystem integrity. From a resource-economics point of view, current thinking in South Africa no longer considers the river as a competing user of water, but rather as the base of the resource (DWAF 1999). If too much water is abstracted, or excessive volumes are removed at the wrong time of the year, the ability of the river to provide "goods and services" will be impaired. The term "goods" in this context includes economically exploitable fish, vegetation, sand, etc., and "services" includes removal and assimilation of waste, and recreational and tourism potential. According to King *et al.* (1999) the costs associated with flow-related degradation such as soil erosion, loss of land, loss of valued aquatic/riparian species, blooms of pest species, collapse of fisheries etc., have profound economic and social implications. These authors also point out that such impacts are often far removed spatially or temporally from the water development scheme (such as a dam) that caused the altered flow and because of this the full cost of such schemes is not always calculated. Thus whether considered from the point of view of ecosystem integrity, or from that of resource economics, there is a growing recognition that environmental flows need to be both determined and implemented as part of a rational management plan for water resources.

5.4 An outline of the EFR process in South Africa

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).

5.4.1 The Building Block Methodology

This outline is largely taken from King and Louw (1998) and Tharme and King (1998). These works, in conjunction with the BBM manual (King, Tharme and de Villiers *in press*) should be consulted for a detailed presentation of the subject.

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 events. The different flow events are the building blocks of the new hydrological regime and each are considered to perform distinct functions in maintaining ecological integrity. A modified, reduced flow regime can be constructed, representing the EFR, which incorporates the flows that are perceived to be essential or important. This concept is illustrated in Fig. 5.1. The discharge features that are considered include low flows, during both the wet and the dry seasons, and small floods (freshes) also during both wet and dry seasons, as well as larger floods. In this context the term "low flow" is defined as "the flow residing in the river if all the high flow events were removed" (Brown and King 2000). The modified flow regime (B) shown in Fig. 5.1 is perennial and exhibits the lowest and highest recommended discharge values during summer and winter respectively. These features are also characteristic of the natural hydrological regime (A). In addition, the modified flow regime also prescribes that the first major floods of the wet season should still occur in March-April. Water that is not perceived to play a specific role (such as that required for shifting of any sediments, provision of enough depth for fish, or floods required to act as

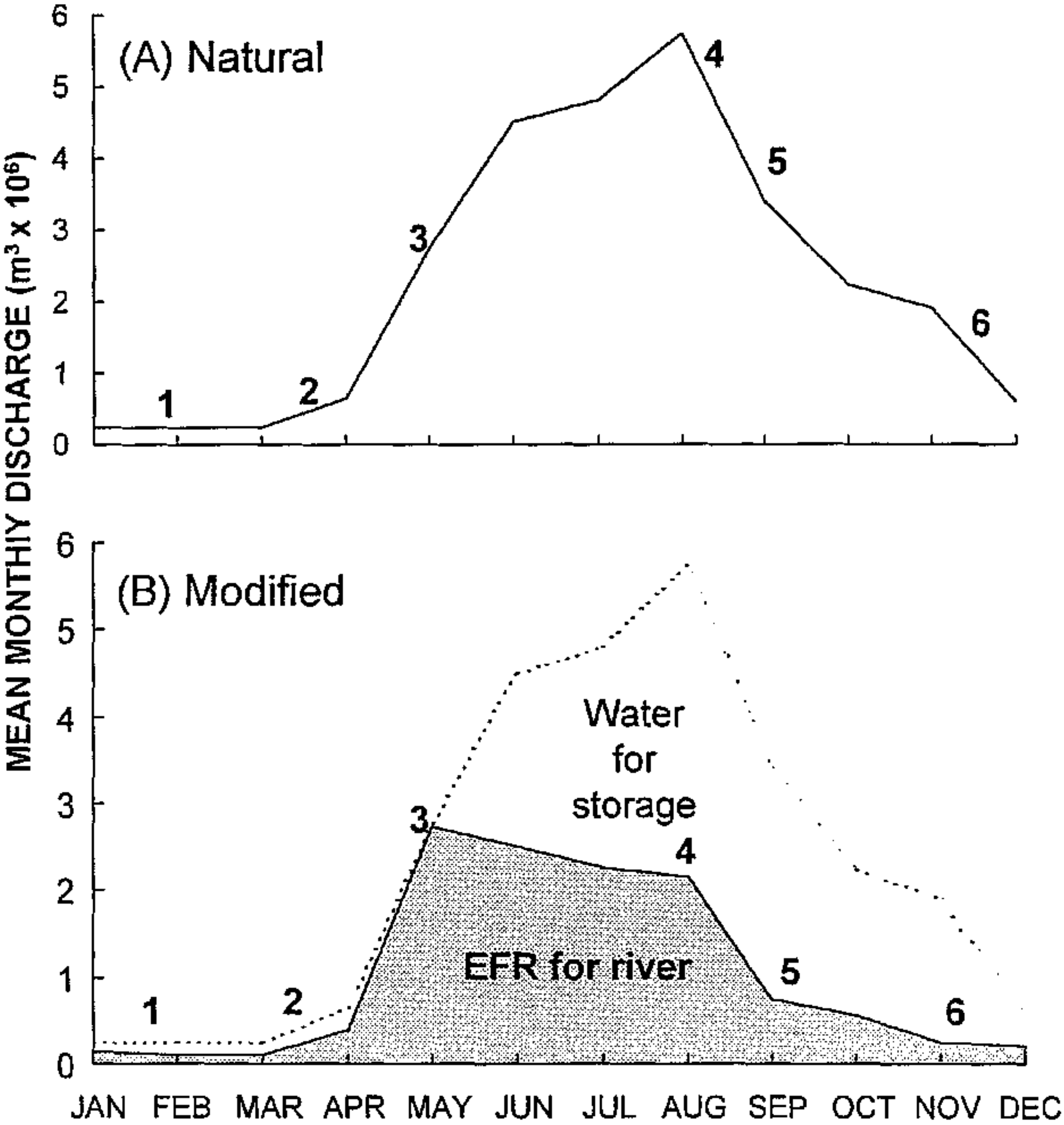

Figure. 5.1 *The perceived important features of a river's natural flow regime (A) and which of these should be retained in an EFR created using the BBM (B). For instance, features 1 and 6 recognise the perenniality of the river and the need to retain this. Features 2, 4 and 5 may recognise the need to retain the fundamental difference between wet season and dry season baseflows, and feature 3 may recognise the timing of the first major flood of the wet season and the need to retain this (from Tharme & King 1998).*

environmental cues for fish spawing etc.) is potentially available for abstraction. It can be seen (B) that most of the "excess" water that can be harvested in the case illustrated, is from the latter part of the wet season. 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. A social anthropologist may also be involved to assess the dependence of local people on the river. 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 describing the present state of various aspects of the river system. Financial constraints frequently limit the collection of new data and one of the advantages of the BBM is that it gives scientific guidance on environmental flows required for a river when biological data and understanding of the functioning of the river are limited (King and Louw 1998). One exception to the collection of new data 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 (e.g. Fig. 5.2). 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. As an example, the riparian botanist might consider that in order to maintain good growth of the reed species growing on the left hand bank of the river, the roots of these plants should be inundated with water at all times. From the figure this indicates that water depth should be no less than approximately 1 m. Thus, the lowest flows during the dry season should not be less than 23.7 $m^3.s^{-1}$. In a similar manner, 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.

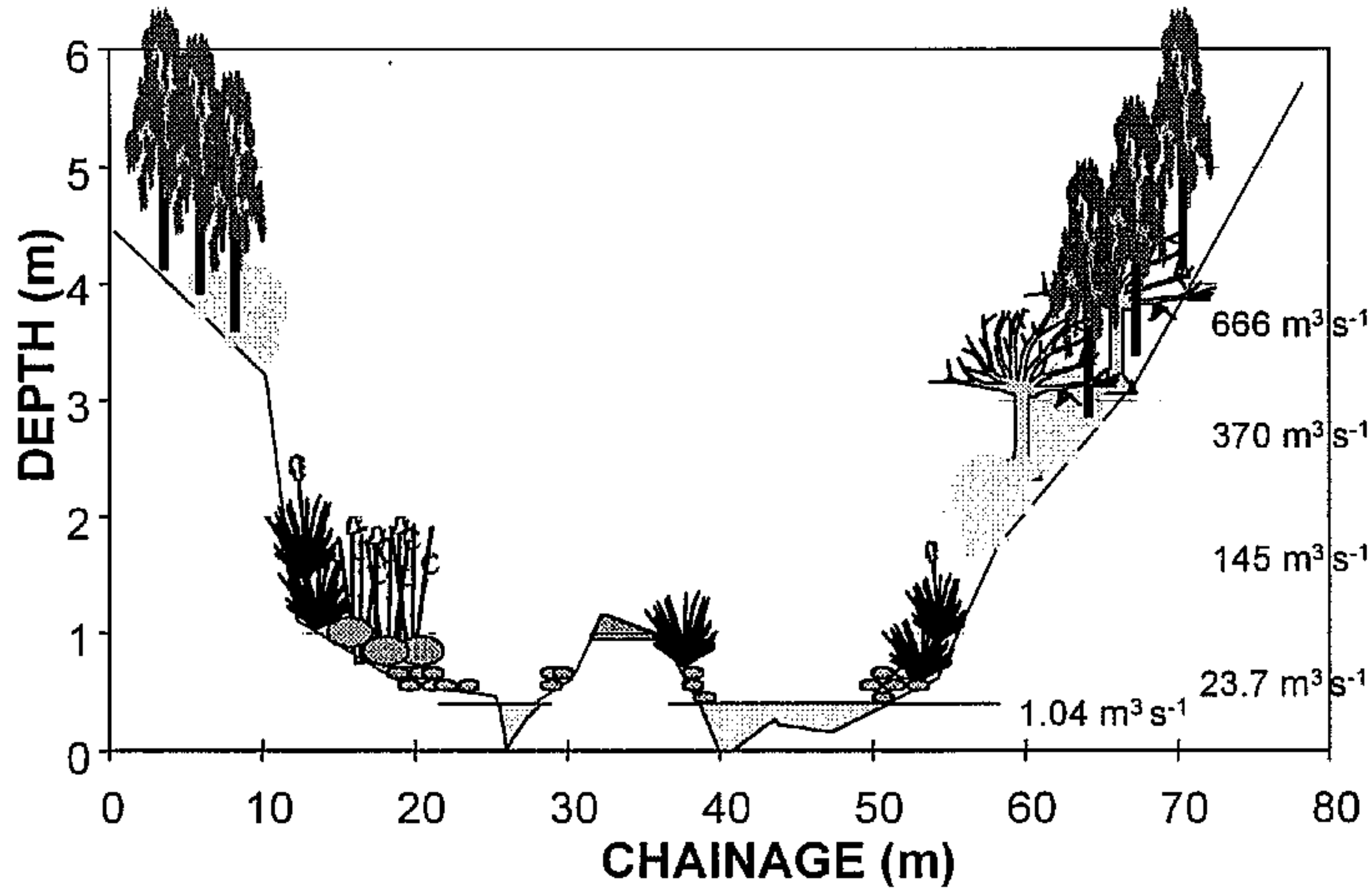

***Figure 5.2** Example of a cross-section used in a Building Block Methodology Workshop to link ecological knowledge to discharge (from King & Louw 1998).*

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.

5.4.2 The DRIFT Method

Downstream Response to Imposed Flow Transformations (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). At the moment however, it is not registered for use as a Resource Directed Methodology for determination of the ecological Reserve for riverine ecosystems. According to the above 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, 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 (Comprehensive Ecological Reserve determinations), however, flows and their associated consequences have been designed to comply with a specific future ecological management class (EMC). Using a Decision Support System (Hughes and Münster 1999), these results can be extrapolated to give low confidence estimates of any other class required. Further development of a Flow Stress-Response method (O'Keeffe, Hughes and Tharme, *in press*) may help to formalise the procedure of motivating for flows in the BBM process and facilitate the process of designing a flow regime for each class (Louw IWR, Rhodes University, *pers. comm.* 2000).

- 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 defined.

5.5 Water quality and environmental flows

Aquatic organisms respond to changes in water quality and each species exhibits specific tolerance ranges and preferences (Dallas and Day 1993). Thus it has been recognised that the provision of an appropriate flow regime for maintenance of ecological integrity is unlikely to have the required effect, unless the quality of that water is also suitable (King and Louw 1998). Indeed, water quality is one of the disciplines routinely included during an EFA using the BBM. Despite a general recognition of the importance of water quality in the determination of environmental flows however, this aspect is currently rather poorly integrated into the BBM (Tharme and King 1998). At the moment only qualitative predictions of future water quality are usually made (section 5.6). The need for quantitative linking of flow and water quality, and inclusion of predictions of the effects that changes in water quality might exert on the biota, has been recognised (King and Tharme 1994; Tharme and King 1998). It is in response to this need that Water Research Commission project K5/956 (for which this present document forms the literature synthesis) has been initiated.

The limited linking of potential changes in water quality to recommended environmental flows that is found in the BBM is not unique to this assessment technique, however. Water quality is either not included, or does not appear to be incorporated explicitly, in most environmental flow methodologies (Tharme 1996). The fact that, in general, EFA techniques tend to be poorly documented, also makes it difficult to discern exactly how water quality is incorporated into a given methodology. Possible reasons for the neglect of this aspect of environmental flow determinations have been presented (Tharme 1996; King, Tharme and Brown 1999). The principle reason appears to be the fact that incorporation of water quality in a meaningful manner involves two phases. In the first phase, predictions need to be made concerning the complex relationship between discharge and the concentration of a chemical component or value of a physical variable. In Chapter 3 it was shown that this is not always a straightforward process. Such predictions alone may be useful in considerations of water quality for drinking water, irrigation, watering of livestock etc. For maintenance of ecological integrity however, it is necessary also to have some knowledge as to what effect the predicted water quality might have on the biota. Although many sophisticated water quality models are available (Chapter 4), data on the second phase, namely, chemical tolerance ranges and preferences of indigenous aquatic biota, are lacking in South Africa (Tharme, 1996; Skoroszewski 1999) and in most other places (J. Day, UCT, *pers. comm.* 2000). Furthermore, biotic responses to changes in water quality are complex (Chapter 6). Tharme (1996) also notes that in the case of most water quality models, guidelines are seldom provided on how the output can be used to generate EFRs for the maintenance of downstream water quality.

The current manner in which water quality concerns are incorporated into EFAs can be roughly divided into qualitative and quantitative applications. These are discussed below.

5.6 Qualitative incorporation of water quality

Consideration of water quality is currently incorporated into the BBM in a largely qualitative manner (King and Louw 1998). Palmer, Malan and Day (2001) discuss in detail the protocol to be followed by water quality specialists involved in an environmental flow determination using the BBM. The protocol is in line with the Resource Directed Measures (RDM) as prescribed in DWAF (1999). The same work also gives information as to which water quality variables should be considered, sources of data, and advice as to what should be included in the Water Quality Starter document (the preliminary report for the BBM Workshop). An outline of the steps and activities involved in the protocol is given in Table 5.1. Basically, the method requires dividing the river system into reaches of homogenous water quality. The natural, or reference, water quality condition for each reach is described, either from an examination of historical data for that reach, or by extrapolating from an adjacent non-impacted river expected to exhibit similar water chemistry. The present water quality status is examined in a similar fashion. Point sources of pollutants are noted. In addition, from an examination of land use in the catchment an idea of the likely diffuse sources of pollutants can be estimated. Especial attention is paid to water quality reaches in which BBM sites are located, since it is for these sites that the environmental flow requirement will be determined. In addition, a comparison is made between the present day water quality and the results of biomonitoring. For instance, if the results of biomonitoring (presented as SASS Scores – see section 6.3.2) are lower than would be expected from chemically derived analyses of present water quality, this might indicate that toxic substances, not routinely determined under the current monitoring program, are present in the system.

At the EFA workshop, in the absence of water quality modelling, only general predictions can be made concerning the effects that changes in discharge may have on the concentrations of chemical constituents and levels of physical variables. Examination of the cross sections and consideration of the hydraulic characteristics for each BBM site can be useful. For example, the cross section for a selected site may show that at a given discharge, pools of standing water will form. If, in addition, the levels of nutrients in this reach are elevated, the specialist can predict an increased risk of eutrophication in these pools during summer. If the flow is reduced so that water no longer flows over riffle areas, and shallow pools are likely to form, there may be danger of elevated temperatures in these pools during summer. If, due to organic pollution, biological oxygen demand (BOD) is high, there is an additional risk of depressed DO concentrations and, possibly, of fish kills.

Table 5.1 *Protocol for incorporation of water quality into environmental flow assessments using the Building Block Method (summarized from Palmer, Malan and Day 2001).*

A. IN PREPARATION FOR THE BBM WORKSHOP

1. River zonation

i) Identify the broad climatic region and the ecoregion in which the river is situated.
ii) Further divide into geomorphological reaches (mountain stream, foothill, lowland river etc.).
iii) Delineate reaches expected to exhibit similar water quality.
iv) Locate the designated BBM sites as well as DWAF monitoring sites.
v) Identify significant hydrological features (weirs, tributaries and impoundments), that may have an effect on water quality.

2. Establish Reference conditions

Derive median monthly values representative of the natural (un-impacted) state for each variable in each water quality reach.

3. Determine present water quality status

i) Use the water chemistry data to derive median monthly values for present water quality in each reach.
ii) Identify current and future threats to water quality including serious point sources of pollution.
iii) Identify major landuse areas in the catchment and hence sources of diffuse pollutants.

4. Assess the completeness and accuracy of the data.

5. Use SASS4 scores to derive additional insight into water quality.

6. Prepare a report detailing the above information.

B. AT THE BBM WORKSHOP

1. Present results of Steps 1-5.

2. Consider the implications of proposed changes in discharge on water quality.

3. Highlight potential problems in water quality so that alternative management options can be considered.

C. AFTER THE BBM WORKSHOP

Scenario modelling

Investigate the effects of source-directed control of pollutants, management options and changes in catchment landuse.

The water quality specialist should also consult the geomorphology expert, who would be able to give information regarding the discharge and water velocity required for the movement of sediment and bed material. Particularly in systems with elevated nutrient levels, it is considered important that high flow events be included in the recommended flow regime. This ensures that water velocity is great enough to move the cobbles and for interstitial water, sediments and adsorbed nutrients to be flushed out (Palmer and Skoroszewski 2000).

The final stage of the BBM in which water quality is a concern, is the Scenario Modelling stage. In this phase, the consequences of combinations of recommendations for water quality or discharge, together with feasible management options (including source directed control of pollutants, catchment management activities, dam operation rules) are considered. According to Palmer, Malan and Day (2001) this aspect is complex and still requires extensive development.

Examples of water quality reports for various recent EFAs carried out in South Africa include, O'Keeffe for the Bivane and Pongolo Rivers (1996), Dallas for the Palmiet River (1998) and Wepener and Vermeulen (1998) for the Mhlathuze River. Thus far although many environmental flows have been set in South Africa, no flow recommendations have been implemented (Hughes 2001).

5.7 Quantitative incorporation of water quality

One of the most common techniques used for the determination of environmental flows, at least in the Northern Hemisphere, is the Instream Flow Incremental Methodology (Tharme 1996; King, Tharme and Brown 1999). Devised by the United States Fish and Wildlife Service, IFIM comprises a collection of analytical procedures and computer programs, of which PHABSIM, the Physical Habitat Simulation Model is the best known and most utilised (Milhous *et al.* 1989). This methodology was assessed by King and Tharme (1994) with regard to its suitability for use in South

Africa, but was found to be unsatisfactory for a variety of reasons. According to Tharme (1996) there are two ways in which IFIM can be used to incorporate water quality into environmental flows. Firstly, use can be made of a water quality model, either one developed specifically for use in IFIM, or one that is compatible. The temperature models SRTEMP, SRSHADE, SNTEMP, AND SNSHADE are components of the methodology, and QUAL2E is also considered to be suitable (Armour and Taylor 1991). According to these authors the "Total Habitat Model" integrates output from PHABSIM with output from the temperature or water quality models. Additionally, a Stream Simulation and Assessment Model (SSAM IV) for water quality has been introduced for use in IFIM (Grenney and Kraszewski 1981, cited in Tharme 1996). Despite the incorporation of various models into the methodology however, according to Tharme (1996) there are few examples in the literature where PHABSIM output has explicitly been linked with water quality or temperature modelling.

The second way in which water quality can be incorporated into environmental flow determinations using IFIM, is by means of Suitability Index (SI) curves (Herricks and Braga 1987, cited by Tharme 1996). These are plots of concentration (in the case of a chemical component) or value (in the case of a physical variable) against preference/tolerance for key aquatic species. According to Tharme (1996), since the SI curves for the various water quality variables have the same form as habitat SI curves, it is possible to use them in a similar way in calculations of overall habitat suitability in PHABSIM. This second way in which water quality can be incorporated into IFIM represents predictions of biotic responses to changed water quality and will be discussed more fully in the next chapter.

Use of water temperature modelling was made in the EFA for the Senqu, Senqunyane, Malibamatso and lower Matsoku rivers during the Lesotho Highlands Development Project (Palmer and Skoroszewski 2000). Likely changes in temperature following the release of water from a dam were modelled for the Senqunyane River, downstream of Mohale Dam. Use was made of the temperature model BETTER for this purpose (Skoroszewski 1997). For each flow-reduction scenario, the expected range of temperatures was predicted at the downstream site. From knowledge of the temperature requirements of the fish species in the river, specialists were able to predict the consequences of each recommended flow (Brown and King 2000).

Water quality has been incorporated into the Integrated Quantity Quality Model (IQQM), an Australian hydrological modelling package. Developed by the New South Wales Department of Land and Water Conservation, this model has apparently been designed to provide more reliable information relating to issues such as environmental flows and water quality (Podger, Sharma and Black 1994; Simons, Podger and Cooke 1996). It is comprised of several components designed to evaluate instream and groundwater quantity and quality as well as rainfall runoff and pollutant export. The water quality model, which forms a module of IQQM, is QUAL2E (see Chapter 4, Table 4.4). To date, only salinity modelling has been undertaken (Young, Davis, Bowmer and Fairweather 1995; D. Black New South Wales Dept. Land and Water Conservation, *pers. comm.* 1998). Exactly how and at what stage predictions of salinity are incorporated into the EFA process is not clear.

5.8 Conclusion

Whilst it is generally recognised that quantitative relationships between potential changes in water quality and recommended discharge need to be taken into account, this is not a formalised step in most EFAs. Furthermore, the need to make predictions concerning possible effects such changes may exert on the biota also needs to be incorporated. As a result, and not surprisingly, there is a general paucity of documented accounts of quantitative linkages of water quality and discharge. According to Ongley (1999) declining water quality is an emerging global crisis. In the future, supplying water of suitable quality is likely to be as problematic as supplying sufficient volumes. Thus in order to tackle this problem, it is important that attention be directed towards linking water quality and environmental flows.

CHAPTER 6

PREDICTING THE IMPLICATIONS OF ALTERED WATER QUALITY FOR THE AQUATIC BIOTA

6.1 Introduction

It is only in the last decade or so in South Africa that serious attention has been given to the subject of water quality and its effects on the aquatic biota. A Water Research Commission project examining the effects of water quality variables on riverine ecosystems was undertaken during the early 1990's (Dallas and Day 1993; Dallas, Day and Reynolds 1994). This was followed, a few years later, by the publication of the South African water quality guidelines (DWAF 1996), which gives the recommended ranges of concentrations and values of water quality variables for (amongst other water users) aquatic ecosystems. 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, as well as micro-organisms. Animals which are closely associated with a water source may also be included e.g. hippopotamus (Gore, Layzer and Russell 1992). Although all communities are likely to play an important role in ecosystem functioning, the effects of impacts brought about by pollution have been studied in some groups more than in others. For example, extensive work has been carried out on the water quality and habitat requirements of commercially important fish species such as salmon and trout (Biggs, Duncan, Jowett *et al.* 1990; Anderson, Orlob and King 1997). Because macroinvertebrates occupy a key role in the food chain (Petts and Maddock 1994) and possess several features that make them useful in biomonitoring (see section 6.3), attention has been directed towards examination of water quality effects on this community (Kovalak 1981; Smith, Kay, Edward *et al.* 1999). The nuisance value of algal blooms has ensured that the factors contributing to eutrophication, which include water quality (Elser 1999), have also received considerable attention.

In order to make predictions of the impact pollution may have on an aquatic ecosystem, it is first necessary to examine effects that have been observed in the field and in experimental work (Armitage 1994). Consideration of the possible and published effects of water quality variables on the various families of aquatic biota is an extensive field and has been documented in the work referred to above (Dallas and Day 1993). Furthermore, since some aspects have already been mentioned in previous chapters, only a short summary of the most salient points is given.

6.2 Summary of the effects of water quality variables on aquatic ecosystems

The effects of the major water quality variables on aquatic ecosystems are summarized in Table 6.1. Note that the effects of some variables are direct and that the impact is on the organism itself, whilst others are more indirect, in that they influence one or more other water quality variables, which turn affect the aquatic biota. The second (indirect) type of effect, has already been mentioned in Chapter 2 under the interactions of the various water quality determinands. It should be emphasised that not only does water quality influence the aquatic biota, but the biota can also affect water quality (Peters and Meybeck 2000). Several of these mechanisms have been mentioned already. Some of the most obvious of these interactions include the effect of eutrophication (via respiration/photosynthesis) on the concentration of DO, the marked uptake (and release) by macrophytes of various ions at certain times of the year, and the exudation of organic acids by vegetation (for example, in the fynbos biome of the Western Cape).

Flow within rivers exerts a profound influence on aquatic communities through its impact on water velocity (or its reciprocal, retention time) as well as river depth and width. In addition, flow affects water quality (Edwards 1995). The importance of catchment geology, climate and topography on the hydrological regime and instream water quality was mentioned in previous chapters. Such physical factors are important in determining the nature of the habitat available to aquatic biota through control of aspects such as channel morphology, substratum, current speed etc (Biggs, Ducan, Jowett *et al.* 1990). Thus it is often difficult to differentiate between the impact of water quality and habitat on biota. The complex web of interactions between water quality variables and riverine biota is discussed more fully in Dallas and Day (1993).

According to Dallas and Day (1993)..."each (water quality) variable has an effect, either beneficial or detrimental, on aquatic organisms and the overall effect when more than one variable is involved is dependent on whether they act synergistically or antagonistically. The effect of each variable on individual organisms is also influenced by the tolerance limits of the organism". When two or more variables interact, the resulting effect on biota may be additive (i.e. equal to the sum of the effects of individual variables), synergistic or antagonistic. In the case of synergistic effects the composite impact is greater than the sum of the individual effects and in the case of antagonistic effects it is lessened (Mason 1991). Tolerance limits are defined as being the range of an environmental variable over which a species can survive and the upper and lower values (e.g. the concentration of a chemical constituent, or temperature) are known as the tolerance limits. Within the tolerance limits, the optimal range is that to which organisms are most ideally suited and in which processes such as growth and fecundity are maximal. Outside of the tolerance limits, whilst organisms can survive, increasingly severe abnormalities will occur. Juveniles or organisms adapted to mountain streams are frequently more sensitive than adults of the same species or those from the middle, or lower river reaches (Dallas and Day 1993).

The above authors also describe how, because the species forming an assemblage in a stream possess differing tolerances, 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 species which are more tolerant will start to establish themselves. The resultant effect is a gradual shift in species composition and abundance. In addition, in stressed ecosystems, the disappearance of individual stress-sensitive species, along with increased disease and parasitism, generally leads to a decrease in species richness and dominance by a few species (Perry, Vanderklein and Lemmons 1996).

Table 6.1 *Effects of the major water quality variables on aquatic ecosystems (adapted from Dallas and Day 1993). Direct effects on aquatic organisms (as opposed to those arising from interactions with other water quality variables) are given in italics.*

Water quality variable	Major effect
Temperature	• *Determines metabolic rate (higher temperatures increase metabolic rate leading to e.g. enhanced growth)* • *Changes provide cues for life-cycle events e.g. breeding, emergence, migration etc.* • Influences bioavailability of nutrients and toxins • Influences concentration of DO (and other gases)
pH	• *Affects gill functioning* • Influences ionic balance • Influences speciation of chemical constituents
Dissolved oxygen	• *Respiration*
TSS and turbidity	• *Determines the degree of penetration of light, hence affects vision, photosynthesis* • *Smother and clog surfaces* *e.g. gills, leading to reduced functioning* *e.g. rocks/cobbles, leading to reduced habitat availability* • Adsorption of nutrients, metals, toxins
Conductivity, TDS and individual ions	• *Osmotic balance* • *Ionic balance*
Nutrients	• *Nitrite and ammonia toxic to most aquatic fauna* • Most are not toxic *per se*, but cause eutrophication and thus affect community structure
Organic enrichment	• Reduction in DO concentration and thus affects respiration
Biocides	• *Usually target specific groups (e.g. molluscs, insects, plants) and thus alter community structure*
Trace metals	• *Some are mutagenic, teratogenic, or carcinogenic* • *Some are metabolic inhibitors* • *Many are essential in low concentrations*

6.3 Biomonitoring

6.3.1 The role of biomonitoring

Increasingly, in the management of aquatic resources, use is being made of the biota themselves as indicators of the biological integrity or "health" of the resource (Uys 1994). Such biological assessment techniques are frequently referred to as "biomonitoring". In the operational context, this term is used to refer to the gathering of biological data in both the laboratory and the field for the purposes of making some sort of assessment, or in determining whether regulatory standards and criteria are being met in aquatic ecosystems (Holhs 1996).

Because ecosystems are so complex, it is difficult to know what to measure when assessing the ecological health of the system and how, for example, to determine the extent of impact of pollution. According to Dallas and Day (1993) biotic indicators that have been used to estimate the effects of impaired water quality on aquatic ecosystems include attributes of:

- individual species (e.g. survival, growth rate, behaviour)
- biotic communities (e.g. species composition, biodiversity)
- ecosystem functioning (e.g. rate of photosynthesis, decomposition)

Measurement of the attributes of individual species in response to changes in water quality, falls within the realm of toxicity testing and is discussed below in section 6.4.1. Many of the common biomonitoring techniques that are in use world-wide look at species composition and abundance and several biotic indices have been developed to assess water quality (Metcalf 1989). An example of this is the South African Scoring System (SASS), discussed in more detail in section 6.3.2. Measurements of ecosystem functioning, such as carbon isotope studies or rates of gas exchange, have not been so frequently used. They sometimes suffer from the disadvantages of being difficult to explain to non-scientists and of requiring expensive equipment (Uys 1994). Nevertheless, direct measurement of key processes can be very useful as it is considered to be a more direct measurement of river "health". Furthermore, since changes in rates of metabolism may well occur before they result in shifts in the composition of an aquatic community, they may be used as "early warning systems" (Schofield and Davies 1996).

According to Mason (1991), Roux and Everett (1994) and Chutter (1994), the inclusion of biological indicators within a water quality monitoring programme offers the following advantages compared to traditional chemical monitoring alone:

- increased overall accuracy in detecting threats to the ecosystem because the *effects* of the threats are examined rather than the *stressor or stressors* (i.e. some aspect of ecosystem structure or functioning is determined rather than the concentration of a chemical constituent, or value of a physical variable).
- integration by the biota of water quality impacts over time (e.g. transient pulses of pollutants may not be measured by tradition chemical sampling, which is usually intermittent. The effects though, if severe enough, will be shown by the biota).
- detection of impacts caused by chemical constituents not measured in the chemical monitoring programme
- evaluation of ecological effects of pollutants and not just their toxicological effects.
- possible detection of contaminants normally present in minute concentrations due to biomagnification and bioaccumulation in the tissues of some species.

A variety of different organisms have been used as bio-indicators in the field ranging from protozoans (Joska, Day, Boulle *et al.* 2001), diatoms and other algae (Whitton, Rott and Friedrich 1991), to macrophytes, fish and macroinvertebrates (Roux and Everett 1994; Schofield and Davies 1996). It is the last group however, that is most commonly used, certainly in South Africa and Australia (Uys 1994; Chutter 1995). Included in this faunal group are insects, crustaceans, worms and molluscs. Some of the advantages in using macroinvertebrates are that they are largely sedentary (and therefore can't easily avoid poor water quality), their forms, life histories, and habitat requirements are diverse, they are small and easy to collect, and they also contain families that show wide variations in tolerance to pollution. Some species are also commonly believed to be especially sensitive to water quality (Dallas and Day 1993; O'Keeffe and Dickens 2000). Disadvantages are that they can be difficult to identify by untrained personnel and that very little is known about the life histories of most indigenous macroinvertebrates. Fish have also been used as bioindicators (for example in the South African River Health Programme, a national biomonitoring initiative). They are useful in that they are at the top of the food chain and may reflect changes in the community as a whole (Mason 1991), often accumulating toxins within their tissues, which can be measured using chemical analytical techniques. They are however more difficult to collect than invertebrates and tend to move away from localised pollution sources. There is a considerable database of toxicity data for fish Wepener, Euler, van Vuren *et al.* (1992), although the number of southern African species included is likely to be limited.

6.3.2 The South African Scoring System (SASS)

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). Modifications have been made during the initial trials and implementation of the technique and currently version 4 (SASS4) is employed.

Basically, the technique involves collecting aquatic invertebrates following a set method, identifying to family level and assigning pre-defined scores for each taxon (Chutter 1995). The scores that are assigned to each taxon vary from 1 to15 depending on the sensitivity to impaired water quality (very sensitive taxa such as amphipods have high scores, whilst more tolerant taxa such as oligochaetes are assigned lower scores). Individual scores are added to obtain the total SASS score for the sample. An estimate of abundance is also recorded (Davies and Day 1998).

An important factor to remember in the use, not only of SASS, but of biotic indices in general, is that the presence or absence of pollutants is not the only factor that affects the distribution and abundance of macroinvertebrates (Parsons and Norris 1996). As mentioned previously, the availability and condition of suitable habitat must also be taken into account, as well as natural variation in temporal and geographic distribution of species (Moss, Furse, Wright *et al.* 1987; Weatherley and Ormerod 1990). Where habitat diversity is restricted, fewer families of macroinvertebrates will be found, even if water quality is unimpaired, than at a site with many different types of habitat and similar water quality. Thus scores for the former site will be lower than for the latter (Chutter 1995). The concept of Average Score Per Taxon (ASPT) has been introduced to try to minimise this effect. Thus one would expect low SASS score and high ASPT for sites where habitat diversity was poor but not water quality. Dallas (1997) reported on the effect of regional and longitudinal (i.e. down the length of a given river) differences on SASS scores and ASPT, as well as the effect of habitat availability and seasonal variation. There appears to be variation in SASS4 scores and ASPT values depending on the region of the country as well as the river type (e.g. foothill or lowland river). It is therefore important that within the River Health Programme, currently being initiated in this country, Reference, or minimally-impacted sites be identified and used to delineate reference SASS scores. Other, possibly impacted sites could be then assessed by comparing their scores with that of the reference condition within the same region and river type.

There is also a general agreement that whilst SASS4 accurately reflects water quality, its sensitivity to individual causes of water quality deterioration has not yet been thoroughly tested. According to Chutter (1994; 1995) this biomonitoring method is relatively insensitive to sulphate as well as to TDS up to concentrations of around 1000 mg/l.

6.4 Methods for predicting the implications of shifts in water quality

It was mentioned in the previous section on biomonitoring that because aquatic ecosystems tend to be complex, deciding on what aspect to monitor is difficult. In a similar manner, in making predictions of potential shifts in ecosystem functioning and structure resulting from impaired water quality, many different facets, such as community structure and abundance, rates of ecosystem processes etc. could and should, be considered. It was noted in the previous section that biotic indices represent useful indicators of ecosystem health. Some of the most useful predictive methods in the literature therefore employ the organisms and biotic indices used in biomonitoring. Thus predictions of the effects of water quality on the biota and biomonitoring are closely linked.

According to Kovalak (1981), in making predictions about the impacts of effluents on macroinvertebrates, the one certain thing is that anything added to the water in an aquatic ecosystem is apt to have some impact on the organisms living there. Thus the problem is not so much predicting whether or not there will be a change but, rather, the magnitude of that change.

Several approaches 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. Armitage (1994) presents a very useful discussion on the prediction of biological responses in rivers, including a survey of many of the papers published in this field.

6.4.1 Ecotoxicology

Ecotoxicology can be defined as the "science of how chemicals at toxic concentrations influence basic ecological relationships and processes" (Chapman 1995, citing Brown 1986). It should be noted that ecotoxicology is not necessarily limited to chemicals and includes the study of manufactured substances and other anthropogenic and natural materials and activities. It should also be noted that ecotoxicology is multidisciplinary in scope and includes the interaction of these substances with the physical environment in which organisms live (Rand, Wells and McCarty 1995).

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 (section 6.2) 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). According to Edwards (1995), bridging the gap between short-term laboratory dose-response tests and understanding the impact of water quality on fish populations so as to manage and develop fisheries more effectively is a process that is still continuing. This is a situation that is also applicable to other communities in aquatic ecosystems. Because of these difficulties, it is necessary to conduct experiments in which data from laboratory tests are combined with data from field observations. Such studies provide the strongest link between a chemical or effluent and any observed effects in the ecosystem (Chapman 1995).

Some experiments have been carried out in artificial ecosystems that attempt to mimic the situation in the field and yet allow the control of environmental parameters. For example, Goetsch and Palmer (1997), using artificial streams, conducted experiments to determine the toxicity of TDS to an indigenous mayfly. Breneman and Pontasch (1994) determined the effect of the insecticide fenvalerate to riffle insect communities, by constructing synthetic streams and examining macroinvertebrate drift within these microcosms.

Toxicity tests have been carried out using single (Suter 1996), or two or more chemical entities in order to study combined toxic effects (Musibono and Day 1999). Increasingly, whole effluents are being studied, despite the inherent problems with variability in chemical composition, in order to make assessments more accurate with regard to toxicity effects in the field. Such toxicity testing plays an important role in monitoring and setting guidelines for effluent discharges (Eagleson *et al.* 1990). In a similar vein, although the majority of toxicity tests have been carried out on single biotic species, there is a move to utilising entire communities, a situation that is more representative of the natural condition (Breneman and Ponasch 1994). Although the details vary from country to country, toxicity testing for water quality criteria (i.e. the setting of guidelines or regulations for maximum concentrations) requires the use of organisms from several taxonomic groups usually several species of fish, invertebrates and aquatic plants (Chapman 1995). Musibono and Day (1999) studied the effect of Mn on the mortality and growth of the freshwater amphipod *Paramalita nigroculus* exposed to Al and Cu in acidic waters. They found that moulting individuals, even in control solutions, frequently died. In addition, juveniles were more sensitive than adults, possibly because the latter had become adapted to the presence of the metals in their natural habitat. Thus it is also important to conduct toxicity tests on organisms at different stages of their life cycle (Mason 1991).

Toxicological parameters such as the LC_{50} (the lethal concentration that corresponds to a cumulative probability of 50% for death of the test population can be determined relatively easily and accurately in the laboratory (DWAF 2000). Mortality is a gross environmental effect, however, although often used as a simple indirect indicator of sublethal effects in the field (Chapman 1995, Goetsch and Palmer 1997). Sublethal effects, such as invertebrate drift, or impairment of growth or reproduction, investigated using chronic tests (long-term experiments with lower concentrations of toxicant) are more indicative of the real situation. Much less attention has been directed towards the investigation of sublethal effects, however, especially those conducted in field situations (Mason 1991, Dallas and Day 1993).

Despite the limitations in toxicity testing, such studies are useful in the following applications (Chapman 1995):

- assessing the toxicity of one chemical relative to another;
- assessing the sensitivity of a given species of plant or animal (including bioindicators) relative to other taxa;
- setting resource water quality guidelines (see for example, the South African Water Quality Guidelines – DWAF 1996);
- setting limits and monitoring compliance of effluent discharges;

- determining cause-effect relationships in post-impact studies.

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.

6.4.2 Suitability curves

Suitability Index (SI) curves were mentioned in the previous chapter in connection with the setting of environmental flows. These curves have been used extensively in IFIM, although more usually for physical habitat rather than water quality (Herricks and Braga 1987, cited by Tharme 1996). They are plots of concentration (in the case of a chemical component) or value (in the case of a physical variable) against the preference for a given species. Ecotoxicological data are used to derive the curves individually, for each water quality variable and for key species of biota (e.g. a fish species). Preference (or suitability) is expressed in relative terms on a scale ranging from 0 to 1, where a value of 0 indicates that conditions are completely unsuitable and a value of 1 indicates optimal conditions. For each water quality variable, there will be a range of concentrations (or values) that can be tolerated. In addition, plots are constructed for different life stages of the key species e.g. spawning, juvenile stages (King and Tharme 1994; Milhous 1998). Similar curves were constructed by Wepener *et al.* (1992) as an intermediate step in the derivation of water quality index (WQI) values, which indicated overall toxicity/water quality suitability to fish. Bell-shaped curves were reported for pH by these authors (indicating that an optimum concentration exists). Descending curves were obtained for TDS, turbidity and other chemical constituents such as ortho-phosphate, potassium, ammonium, fluoride and various metal ions (indicating that increasing concentration results in decreased suitability). Finally an ascending curve was obtained for DO. If the importance of the various water quality attributes to aquatic biota is considered, DO is probably one of the most important determinands of the "well-being" of an aquatic organism (Ambrose, Wool and Martin 1993).

6.4.3 Multivariate studies in the field

Toxicological experiments carried out in the laboratory have the advantage that it is possible to control most of the variables likely to influence the toxic effect. They suffer from the disadvantage that it is difficult to extrapolate the results from such experiments into the field. For this reason, 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 Determinant Analysis (MDA) to identify the most important variables. Studies were carried out on the physical and chemical characteristics (28 variables) and the presence of macroinvertebrate species in over two hundred sites on unpolluted British rivers (Wright, Moss, Armitage *et al.* 1984). From this study it was possible to predict with reasonable accuracy (except for rare species in low abundance) the probabilities of macroinvertebrate taxa occurring at an unpolluted sites in British rivers using five environmental variables. The five variables were: distance from source, mean substrate particle size, total oxidised nitrogen, alkalinity and chloride (Edwards 1995). Rutt, Weatherley and Ormerod (1990) also used Multiple Discriminant Analysis (MDA) to identify key variables controlling the distribution of macroinvertebrate riffle fauna in upland streams of Wales, Scotland and England. These authors found that pH, or aluminium and calcium concentrations, could be used to predict what species would be present. They concluded that MDA models could be valuable for predicting the effects of controlling factors such as sulphur emissions and conifer afforestation that influence surface water acidification. A further conclusion was that data sets from which empirical relationships between invertebrate fauna and stream-water chemistry can be derived need to be extensive. Rutt *et al.* (1990) also noted that, although the precise mechanisms that produce the faunal differences between streams of contrasting acidity had not been fully elucidated, this does not detract from the validity and usefulness of empirical relationships relating fauna and physicochemistry. On the other hand, Rundle, Weatherley and Ormerod (1995) found that after liming of three acidified streams in Wales there were marked changes in stream chemistry. Calcium and pH values increased and aluminium concentrations decreased to levels similar to those in unimpacted streams. It was predicted, using models derived from the multivariate studies described above, that stream invertebrates would respond with the re-establishment of acid-sensitive species. Although this did occur to a limited extent, contrary to the model predictions, wholesale changes in the structure of macroinvertebrate communities did not occur. In another paper, Ormerod, Weatherley, Varallo *et al.* (1988), also in connection with acidification of Welsh streams, mention the importance of "bottom-up" (i.e. food availability) and "top-down" (i.e. predation) factors. They conclude that, whilst direct effects of low pH and high aluminium concentrations have been demonstrated, it is also possible that indirect chemical influences could occur through food availability or predation. The results from these last two studies serve to illustrate the complexity of abiotic-biotic responses and the difficulty in making accurate predictions of biological responses to changes in water quality. These results also show that rehabilitation of streams subjected to chemical (as well as physical) disturbance may take some time to occur.

As a result of the type of experiments detailed above, considerable data are available for the United Kingdom that relate the distribution of riverine macroinvertebrates to instream concentrations of chemical constituents (Wright, Moss, Armitage *et al.* 1984; Rutt, Weatherley and Ormerod 1990). This has led to the development of RIVPACS (River InVertebrate Prediction and Classification System) which is used extensively in the United Kingdom and parts of Europe, and in an amended form in Australia (Alba-Tercedor and Pujante 2000; Furse 2000; Smith et al. 1999). RIVPACS is a computerized, multivariate statistical procedure that explores the relationship between community structure, habitat and water quality. It can generate site-specific predictions of the macroinvertebrate fauna that can be expected in the *absence* of environmental stress. These predictions (and a calculated biotic score) can be compared with the observed score at the site to obtain an estimate of the degree of environmental impact (Wright, Furse and Armitage 1993). The potential of RIVPACS for predicting the effects of environmental change, including those brought about by pollution, have been considered by Armitage (2000) and De Pauw (2000). Armitage concluded that RIVPACS can only be used to test those disturbances that are likely to affect wetted area and substratum characteristics. Organic and heavy metal pollution will not affect the variables used in the method. Although total alkalinity is also one of the variables used in RIVPACS, the value of this parameter is only likely to change significantly if there is a major pollution incident in the river, or in the case of the introduction of "new water" from an IBT. Thus, although this method can be used to assess sites that are impacted by pollutants by comparing the observed faunal community with that predicted for a site with similar environmental conditions, it cannot be used to predict the impacts of pollution on the biota.

Maddock (1992, cited by Petts and Maddock 1994) developed an empirical relationship between an invertebrate-based score (Biological Monitoring Working Party: BMWP) and several environmental variables. This model was developed using data from 28 streams during summer low-flow in East Anglia, UK. Four primary variables were found to be important, including an indicator of water quality (the most influential variable), flow and physical-habitat descriptors (Petts, Maddock, Bickerton *et al.* 1995). The derived empirical equation is given in Table 6.2.

The model was validated using data collected on the river Glen. The r^2 value for the combined data set was high (0.88). According to the authors, the cover variable (see Table 6.2 for an explanation of this factor) describes macroscale differences between sites, differentiating especially between reaches with natural, vegetated margins and maintained, or canalised reaches. Flow type gives an index of river-bed configuration as well as a flow index to assess habitat quality at a particular point in time. No reference is made in the above papers to further experiments using this model in which water quality (and hence chemical score) was altered and the predicted and observed invertebrate scores compared. It is also not clear if this relationship is valid for streams in other areas of the UK, or in other countries, such as South Africa.

Table 6.2 *Relationship between Invertebrate Score (BMWP) and habitat attributes for streams in the Anglian Region, UK. (Adapted from Petts et al. 1995).*

Invert Score	**= ((Chemical ScoreX 20)- 39)+((Q_{95}/W)X467)+(CoverX 0.9)+(flow typeX 53)**

- Chemical score is derived from the DO, BOD and NH_3 concentrations.
- Q_{95}/W is the 95^{th} percentile flow duration statistic divided by mean channel width.
- Cover is a measure (%) of wetted cross-sectional area with cover objects (instream and overhanging vegetation, cobbles and boulders).
- Flow type = a composite variable incorporating mean wet width as % of channel width, and the proportion of the reach with visible flow.

6.4.4 Complex ecological models

In a manner similar to the water quality models discussed in Chapter 4, ecological models can also range from a simple equation linking, for example, invertebrate score and an environmental variable, to complex, computer-based models simulating and requiring data on many variables. Furthermore, ecological models also range from those based on purely empirical relationships (equivalent to the "black-box" models in Chapter 4) to those in which the process is conceptualized (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 that control those events Kovalak (1981).

6.4.4.1 Eutrophication models

Predictive models are particularly well developed with respect to the effects of nutrient enrichment in aquatic ecosystems (Armitage 2000). Rossouw and Quibell (1993), for instance, used the steady state water quality model QUAL2E to simulate nutrient and algal growth dynamics in the middle Vaal River. Algal abundance was inferred from the predicted chlorophyll concentration. A threshold instream value of 20µg/l chlorophyll was considered to mark the onset of water quality problems, caused by algal blooms, for the water treatment works in the area. Variables that were found to be important in the simulation of algal biomass were: nutrient concentrations, light penetration, and flow (in other words, water residence time). The authors concluded that the lack of data relating to instream processes resulted in uncertainties in the model predictions. For example, four possible reasons were postulated for the observed concentrations of phosphate being much lower than those predicted by the model. Which mechanism was actually responsible for losses in the system could not, however, be determined without studies in the field.

Another model has also been developed to simulate algal blooms in the Vaal River (Cloot, Schoombie, Pieterse *et al.* 1997). Light, temperature and turbidity were taken into account, as well as dissolved silicon concentration. According to the authors, although reasonable simulations were obtained, some discrepancies between measured and predicted chlorophyll still remain. In order to improve predictions, other factors which also impact on chlorophyll would need to be taken into account. The model would be developed further to include the effect of salinity, pH, phosphate, nitrate, DO and carbon dioxide. More information on the ecological behaviour of the river as well as the growth requirements of individual algal species are also required to develop the model further (Pieterse and Janse van Vuuren 1997).

Ruse and Hutchings (1996) modelled phytoplankton composition in the river Thames and concluded that predicting abundance was easier than predicting species composition. They also found that whilst the dynamics could be modelled, the high percentage of unexplained variation made predictions unsafe.

6.4.4.2 Mechanistic models

Kovalak (1981) describes a simple model that predicts shifts in macroinvertebrate species richness resulting from changes in DO. This model can be linked with the classic "oxygen depletion" model of Streeter and Phelps that was discussed in Chapter 4 (section 4.7.1). It is interesting because the model is mechanistic and the author has used basic principles in an attempt to derive the underlying mathematical relationship between water quality (DO) and biological response (species richness). The model is discussed below.

According to Kovalak (1981), the larvae of most stream insects depend on passive diffusion of DO through the body surface or the gills in order to satisfy their oxygen requirements. Oxygen supply therefore depends on the rate of renewal at these surfaces and therefore on stream current or turbulence. Because of spatial variation at the micro- and macro-scale, DO supply can vary widely. For example, Kovalak cites other studies, in which it has been shown that DO supply at the top of stones was 19 times higher than underneath the stones and 4 times that within the interstitial spaces of gravel. Yet many invertebrates are photonegative and prefer to be away from light. In addition, as temperature is raised, insects move into faster currents because the DO content is decreased and at the same time metabolic rate is increased. Even at the micro-scale, most insect larvae show preferences for specific faces of stones in a river, which is related to the respiratory requirements of different species. For example, tests on one species showed that at high temperatures and at lower current speeds, individuals aggregated at the upstream face of bricks placed in the water. When current speed was greater than 0.7 $m.sec^{-1}$, however, a standing wave was created, resulting in lower turbulence, lower DO and a reduced preference for that micro-habitat.

Kovalak (1981) then goes on to postulate that macroinvertebrate species richness and population density is determined by distribution on a micro- and macro-scale, which is in turn determined by the variation in oxygen supply and demand. An oxygen index was defined as:

$$\mathbf{I_0} = \frac{\mathbf{CO}}{\mathbf{M}}$$

Where: $\mathbf{I_0}$ = oxygen index

C = mean current velocity

O = DO concentration (mg/l)

M = index of metabolic rate

M is determined by assuming that the metabolic rate of insects at 0°C is 1.0 and that the respiratory quotient (Q_{10}) is 2.0.

I_0 and species richness were determined for a series of stations along a creek in Colorado, USA. It was found that a linear relationship existed between I_0 and species richness, and that the latter could be predicted from the former using the following model:

$$\textbf{Species richness} = -11.3 + (7.6 \times I_0)$$

The author concluded that this model, coupled with models of effluent dispersal or oxygen depletion, should provide realistic predictions of changes in species richness. It was noted, however, that more information would be required in order to predict which species would be amongst the survivors and what the likely changes in population density would be.

Anderson, Orlob and King (1997) have developed a mechanistic ecosystem-response model by incorporating an ecological model into an existing hydrodynamic and water quality modelling framework. The model has been used to examine the impacts of flow regime, water temperature and salinity on juvenile chinook salmon in the Sacramento River and San Francisco Bay area. The approach is to first simulate the hydrodynamic and water quality characteristics of the system, the results of which form the input for the ecological model. The models can be used to examine the effects of changing water quality characteristics such as water temperature on population dynamics. Fish, in particular those of commercial importance, have been the subject of several modelling attempts. One of the earliest was by Chen and Wells (1976) in the Boise River, Idaho, USA. Simulations for fish biomass production in this model were based on the availability of food (insects, benthic fauna and detritus) and the suitability of environmental conditions (temperature, toxicity, TSS and DO). No attempt was made to consider the availability of suitable habitat for spawning. Several predictive models for fish are described in Armitage (1994).

6.4.4.3 Rule-based models

According to O'Keeffe and Dickens (2000), in the context of the BBM for determination of riverine environmental flows, there is a great need for ecological modelling in order to make the assessment of flow requirements for macroinvertebrates less subjective. They suggest that rule-based models may be a useful approach in this regard. Jewitt and Gorgens (1995) also noted that since the relationships between abiotic and biotic components of an ecosystem are usually understood in a rough and qualitative manner, rather than detailed and quantitative, it is difficult to build conventional numeric models. Expert (also known as knowledge-based) systems which are largely logical is a more feasible approach. Such systems work on a fundamental model of cause and effect implemented in a series of IF-THEN rules. As part of the KNPRRP (Kruger National Park Rivers Research Programme), a suite of models have been set up for the Sabie River. These consist of a hydrology model (ACRU) which is used to simulate daily flow and sediment yield, and three qualitative rule-based models which describe fish, geomorphic and riparian vegetation response to changes in flow and sediment (Jewitt, Heritage, Weeks *et al.* 1998). Although water quality was not included in the models, it appears that stream chemistry-biotic response may be amenable to such an approach.

6.5 Conclusion

Aquatic ecosytems are complex. There are many interactions between biotic and abiotic components, some of which are known and can possibly be quantified, others of which are totally unknown. Part of the reason for a lack of understanding of these interactions is the paucity of research on indigenous organisms. In addition, 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). Thus any predictions relating the implications of changed water quality to the aquatic biota are also likely to suffer from the same disadvantage. Herricks, Sale and Smith (1981) concluded that "progress towards predictive ecosystem analysis has been slow. One cause is that biologists are reluctant to make ecological predictions because of a lack of detailed, quantitative data. Typically, the inherent variability of ecosystems and the limitations to the collection of statistically valid data, constrain both interpretation and prediction". This statement was made in 1981 but is equally valid now. 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 in making predictions of the implication of changes in water quality for aquatic biota. A start on this task has been made by the compilation of the BIOBASE, which 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). The results from the River Health 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 SASS scores. Such collections of data represent an invaluable resource to river ecologists.

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

APPENDIX A

A list of models encountered in the literature

<u>Riverine water quality models</u>

CE-QUAL-RIV1 = dynamic, river quality model (Wurbs 1995).

DOMOD3 = steady state model, simulates DO, NOD, CBOD (Gowda 1983).

HEC-Q5 = dynamic, river quality model (Wurbs 1995).

IQQM = integrated quantity and quality model, Australian model (Podger *et al.* 1994).

MAGIC = catchment process simulation model that simulates effect of acid deposition on river water chemistry, developed by Institute of Hydrology, UK (Crockett 1994).

MIKE 11, MIKE 21 & MIKE 3 = a complex of dynamic water quality models developed by the Danish Hydraulic Institute, 1,2 and 3 D. (Crockett 1994; Fawthrop 1994; http://www.dhi.dk).

QUAL-2E and QUAL-2E-UNCAS = 1D steady semi-dynamic water quality model (http://www.epa.gov)

QUASAR = consent-setting model for continuous point-source pollution, developed by Institute of Hydrology, UK (Crockett 1994).

Riverware = University of Colorado and US Bureau of Reclamation (http://CWRRI.colostate.edu)

RIVMIX = 2D, steady state riverine water quality model (Wurbs 1995).

SARAH = 2D, steady state riverine water quality model (Wurbs 1995).

Simple Model = Urban catchments (Chandler 1994).

SIMCAT = consent-setting model for continuous point-source pollution, developed by NRA Anglian Region (Crabtree *et al.* 1986; Crockett 1994).

SQUAL = specifically for rivers with hydroelectric projects (Tharme 1996).

STEADY = 1D, steady state riverine water quality model (Wurbs 1995). Only models temperature, DO and BOD interactions (Dortch and Martin 1989).

TOMCAT = consent-setting model for continuous point-source pollution, developed by Thames Water Authority (Brown, 1986; Crabtree *et al.* 1986; Crockett 1994).

TOXIWASP = dynamic model for simulating the transport and fate of toxic chemicals in water bodies (Crockett 1994).

TVARMS = dynamic model produced by Tenessee Valley Authority

USGS Streeter-Phelps = DO (NOD, CBOD, IP, coliforms, 3 conservative substances, NOT temperature). Steady-state, 1-D.

WASP = dynamic, river quality model (Wurbs 1995)

WQRRS = dynamic, river quality model (Wurbs 1995)

CATCHMENT PROCESS SIMULATION MODELS

HSPF = for well mixed reservoirs and streams, developed by Hydrocomp/US EPA (Van Rensberg *et al.* 1997).

IMPAQ = South African wash-off and river water quality model (Ninham Shand)

IQQM = Australian model. Water quality component = QUAL2E.

SHE = Systeme Hydrologique Europeen, a physically based distributed hydrological model – but has been extended to predict nitrate levels (Abbott *et al.* 1986; Crockett 1994; Fawthrop 1994).

SWRRB-WQ = developed by US ARS for ungauged agricultural catchments (Wurbs 1995).

WITQUAL = an urban water quality model for South African conditions (Coleman and Simpson 1996)

WITSIM = developed for salinity modelling in the Upper Olifants catchment, SA.

RESERVOIR WATER QUALITY MODELS

Note: some models can be used to simulate rivers as well as reservoirs.

BATHTUB = steady state eutrophication model, of use when data is limited (EPA webpage 1998; http://www.epa.gov)

BETTER = TVA model (Brown 1986) Also used for Mohale reservoir LHDA by CSIR 1997 (LHDA report 648-02). 2D flow, temperature, and water quality model for reservoirs.

SELECT = 1-D, reservoir release model (Dortch and Martin 1989)

CATCHMENT INFORMATION SYSTEMS/DECISION SUPPORT SYSTEMS

Basins = EPA catchment information system (contains HSPF called NPSM, as well as QUAL2E)

CMSS = catchment management support system developed by CSIRO (Horn 1995).

ICIS = South African catchment information system which can incorporate HSPF (http://www.ccwr.ac.za/icis/).

MIKE BASINS = an integrated catchment management tool.

MISCELLANEOUS HYDROLOGY MODELS AND SUB-ROUTINES

ANNIE = a general program for interactive hydrologic analysis and data management that can be used as an editor for HSPF (Hogarth *et al.* 1995).

ACRU =South African hydrology model with limited water quality simulation capability.

AQUALM-XP= Australian Stormwater Quality Analysis model. Gives predictions of rainfall runoff and non-point source pollutant export from different land uses. (Horn, A.M. 1995; Hogarth *et al.* 1995).

DISA = "Daily Irrigation and Salinity Analyses", developed to assist decision-makers with salinity management in the western Cape.

HYRROM = 'lumped' hydrology model (Fawthrop 1994)

HYSIM = 'lumped' hydrology model (Fawthrop 1994)

IHDM = Institute of Hydrology Distributed Model, - hydrology only. (Fawthrop 1994).

IHACRES = hydrological model based on the unit hydrograph theory (Fawthrop 1994).

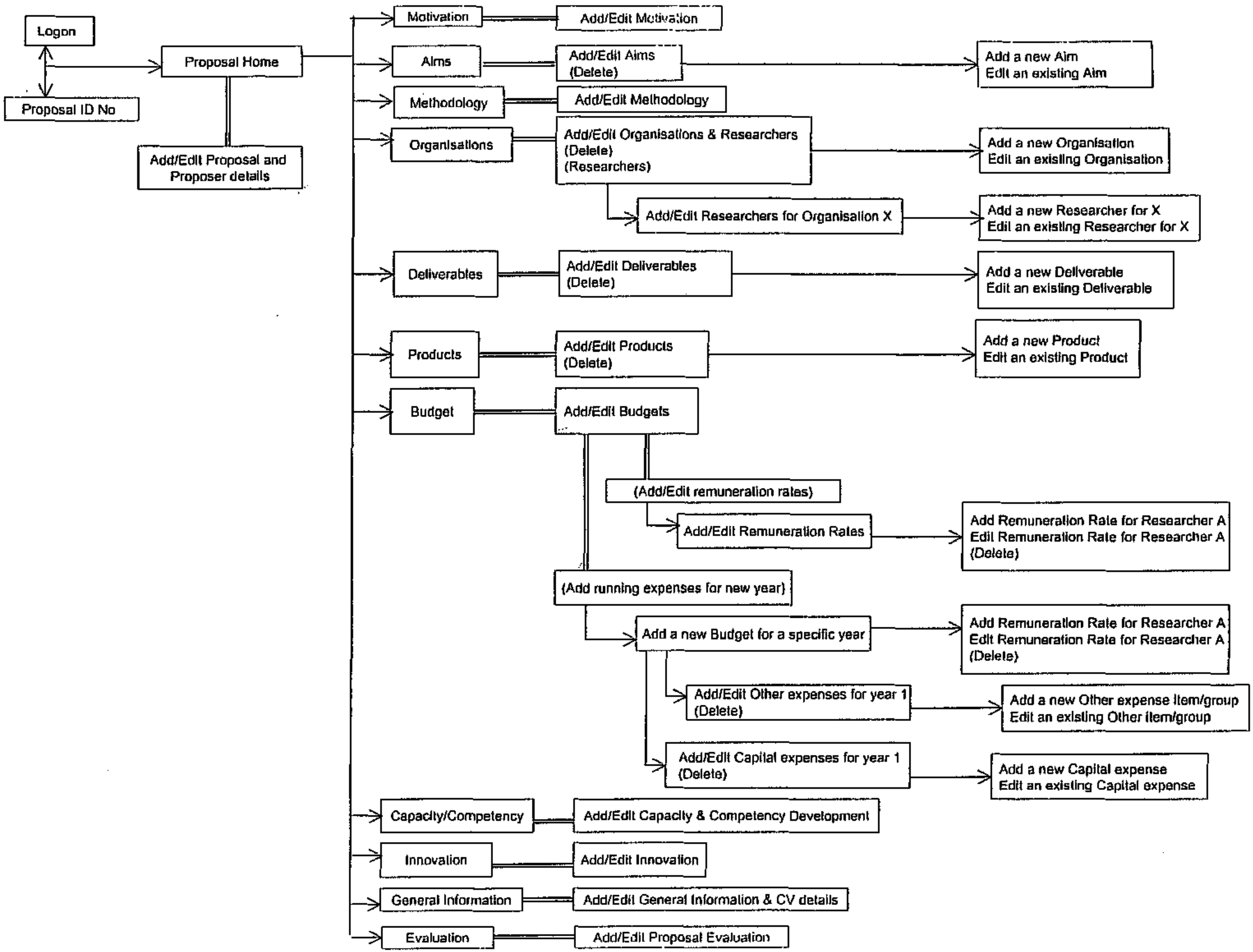
ISIS = 1D hydrodynamic model, developed in the UK. Used on Orange River (Whitlow *et al.* 1998). Not clear if the model can also simulate water quality.

Medli = developed by CRC waste management and pollution control, Australia. Expert system framework to assess effects of land disposal options and fate of contaminants (Horn 1995).

SWMM III RECEIVE = Storm Water Management Model. Urban run-off model that models some quality constituents but not sediments. Use when a drainage network is involved (Crockett 1994). Used by CMC, Australia.

TOPMODEL = semi-distributed hydrological model –ie. No water quality modelling? (Fawthrop 1994).

WITSKM and **WITQUAL** = urban run-off model, able to simulate TSS, phosphorus, developed at Wits University, SA (Simpson and Coleman, 1993).


Logon
Proposal ID No
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Motivation
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Methodology
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Deliverables
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Add a new Deliverable Edit an existing Deliverable
Products
Add/Edit Products (Delete)
Add a new Product Edit an existing Product
Budget
Add/Edit Budgets
(Add/Edit remuneration rates)
Add/Edit Remuneration Rates
Add Remuneration Rate for Researcher A Edit Remuneration Rate for Researcher A (Delete)
(Add running expenses for new year)
Add a new Budget for a specific year
Add Remuneration Rate for Researcher A Edit Remuneration Rate for Researcher A (Delete)
Add/Edit Other expenses for year 1 (Delete)
Add a new Other expense item/group Edit an existing Other item/group
Add/Edit Capital expenses for year 1 (Delete)
Add a new Capital expense Edit an existing Capital expense
Capacity/Competency
Add/Edit Capacity & Competency Development
Innovation
Add/Edit Innovation
General Information
Add/Edit General Information & CV details
Evaluation
Add/Edit Proposal Evaluation