EARLY DEVELOPMENT OF WATER QUALITY METHODS AND APPROACHES IN ECOLOGICAL RESERVE ASSESSMENTS

CG Palmer · P-A Scherman · WJ Muller ·
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WRC Report No. 1108/1/04

Water Research Commission

Early Development of Water Quality Methods and Approaches in Ecological Reserve Assessments

CG Palmer[1], P-A Scherman[1], WJ Muller[1], JN Rossouw[2], HL Malan[3] and S Jooste[4]

[1]Institute for Water Research
Rhodes University

[2]Ninham Shand Consulting Engineers

[3]Freshwater Research Unit
University of Cape Town

[4]Institute for Water Quality Studies, DWAF

The first of two reports to the Water Research Commission on the "Integration of Water Quality Tools for the Ecological Reserve into a Risk-based DSS"

WRC Report No.: 1108/1/04
ISBN No.: 1-77005-227-5

OCTOBER 2004

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Printed by Silowa Printers: 012 804 1164

EXECUTIVE SUMMARY

Development of water quality methods within ecological Reserve determinations

This project was initiated in 2000 to allow the ecological Reserve team for water quality to undertake additional research while working on ecological Reserve determinations funded by the Department of Water Affairs and Forestry. The results of this work form the basis of the methods reported in the Water Research Commission report Hughes DA (Ed)(in press) *SPATSIM, an integrating framework for ecological Reserve determination in implementation: incorporating water quality and quantity components for rivers;* specifically in Chapter 6: Palmer, Muller and Hughes "*Water Quality in the ecological Reserve*".

This documentation of processes of development is especially important since some of the recorded "lessons learned" have not yet been taken up.

The concept of the ecological Reserve was accepted by Parliament, and became the cornerstone of the resource protection policy (DWAF 1997), with ecological Reserve assessments being required by law in 1998 (National Water Act No. 36 of 1998). During the process of policy formulation and drafting it was argued that methods existed to quantify the ecological Reserve. This was true for water quantity (Palmer 1999) – water flow and the associated ecological responses of ecosystem integrity, geomorphology, riparian vegetation, fish and invertebrates. However it was recognized that supplying adequate flows in space and time would not necessarily lead to ecosystem health if water quality (water chemistry) was impaired. The NWA therefore specifically included water quality criteria within the ecological Reserve.

Formal methods for determining water quality aspects of the Ecological Reserve were then developed in two major processes:
- DWAF organised method-development workshops and contracts; and
- DWAF contracted ecological Reserve assessments.

The combined workshop/contract process led to the publication of the first official water quality method in 1999. As ecological Reserve assessments were undertaken, methods were extended and modified. Three major ecological Reserve assessments contributed to this process: the Crocodile (Mpumalanga), Olifants (Mpumalanga) and Breede (Western Cape) Rivers. Members of the WRC project team for this project were closely involved in the Olifants and Breede River assessments, and this project was planned on the basis of using the data and methods developed during those studies. However, it became clear that other innovative approaches were developed during the Crocodile River Ecological Reserve assessment and details of this study were included.

Main contributions from Case Studies

Olifants River

- The first methods were developed and applied, including a method relating salt toxicity to resource classes.

- The need for a range of skills in a water quality team was identified: aquatic ecology, aquatic ecotoxicology, water chemistry, and flow/concentration modelling.

- The lack of routine monitoring of key variables was identified (temperature, dissolved oxygen and total suspended solids), and a recommendation made for data collection over one or two seasons preceding the study, not just during the execution of the study.

- The principal was established that environmental flows are not recommended to solve water quality problems, rather that water quality consequences of recommended environmental flows are noted. Managers then have the option of allocating flows for dilution, or enforcing stricter source directed controls. However, it was noted that there was perception amongst water resource managers that all water quality problems will be solved by source directed controls. This leads to a lack of attention to the consequences of flows on water quality. The time frame to implement source control measures is often very long and to see results often even longer. There is therefore the need to control abstractions or to find some other way to allow for interim dilution.

- The importance of the appropriate scale of management decisions was highlighted. In the Olifants study there was not the opportunity to evaluate catchment scale impacts. An attempt by the water quality team to address this was to identify "Hot spots" and "Refugia". "Refugia" were the Blyde and Mohlapitse Rivers which were essential to maintain or reach water quality objectives in the Olifants River in the Kruger Park. "Hot spots" were areas that needed urgent attention, e.g. Selati and Klip Rivers.

Breede River

- The Breede River study allowed for the simultaneous, and comparative application of water quality methods within the Downstream Responses to Instream Flow Transformations (DRIFT) and Building Block Methodology (BBM) methods:
  - DRIFT: more critical and rigorous use of water quality data (generic and specific descriptors useful).
  - DRIFT: tentative "minimum degradation flows" is useful for the water quality team to focus thinking. Water quality consequences can be refined with the information from other specialists.
  - DRIFT: Resource Quality Objectives (RQO) should be an explicit product of DRIFT (as in BBM) (some of this is captured but hidden in the descriptors and database) and linked to Classes/flow-reduction scenarios.
  - BBM: (in the current workshop) used water quality more analytically due to recent exposure to the DRIFT method (see point 1).
  - For water quality, DRIFT is a more useful approach as water quality consequences are consequences for summer and winter base flows vs. a single month in summer and winter for BBM. The latter may lead to missing peaks (e.g. TDS water high flow peak in the Western Cape) outside of the 2 selected months.
  - BBM: confidence in prediction per site while in DRIFT give confidence, severity, data source and direction of change for each water quality variable and element of flow reduction.

- In the Breede River study the water quality procedure was better integrated with the quantity-based workshop procedure. Data and actions required for the workshop were listed:
  - A starter document describing both reference conditions (RC) and present ecological state (PES) for specified water quality variables per identified reach/Instream Flow Requirement (IFR) site.
  - Prepared scatter plots (especially flow vs. water quality variable, and trend of changes in water quality over time).
  - Box and whisker plots to examine and describe seasonal distributions.

- Summary statistics (recommend inclusion of 5% and 95%): mean, median, 5%, 25%, 75% and 95%.
- Maps of **all** the water quality monitoring points and biomonitoring points on a catchment land-use map (including point sources etc.).
- All available water quality data, biomonitoring data, toxicology data, pollution data (POLMON).
- Flow-concentration water quality modelling templates for RC and PES (NB: need hydrological information).
- Time-series profiles: if doing stressor-responses (T-Soft) for RC and PES (if data available) flow-concentration matrices, concentration-stress matrices for IFR sites and for specified water quality variables are needed.
- Prepare generic descriptors for water quality variables (required for DRIFT workshop only).

- Toxicology data was confirmed as being invaluable.

- The mismatch between **daily** hydrology data and **monthly** water quality data was noted.

Main contributions from this report

A proposed modified method was provided to DWAF and formed the basis of developments reported in Hughes (Ed)(in press).

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LIST OF ABBREVIATIONS

AEV	- Acute Effects Value
ASPT	- Average Score Per Taxon
BBM	- Building Block Method
BRB(S)	- Breede River Basin (Study)
C	- Concentration
CEV	- Chronic Effect Value
CL	- Confidence Level
DEMC	- Default Ecological Management Category
DRIFT	- Downstream responses to instream flow transformations
DWAF	- Department of Water Affairs and Forestry
EC	- Electrical Conductivity
EIS	- Ecological Importance and Sensitivity rating
EMC	- Ecological Management Class
ERC	- Ecological Reserve class
GCTMA	- Greater Cape Town Metropolitan Area
GSRR	- Generic Stressor Response Relationship
IFR	- Instream flow requirement
IWR	- Institute for Water Research
LC_{50}	- Lethal concentration for 50% of the test population
LC_x	- Lethal concentration for x% of the test population
LOEC	- Lowest Observed Effective Concentration
NOEC	- No Observed Effective Concentration
NRHP	- National River Health Programme
NWA	- National Water Act
OREWRA	- Olifants River Ecological Water Requirement Assessment
PES	- Present Ecological State
PESC	- Present Ecological State Category
PS	- Present State
Q	- Discharge
RC	- Reference Condition
RDM	- Resource Directed Measures
RQO	- Resource Quality Objectives
RS	- Reference State
SRR	- Stressor Response Relationship
SSRR	- Site specific Stressor Response Relationship
TCEL	- Threshold Critical Effect Level
TDS	- Total Dissolved solids
TIN	- Total Inorganic Nitrogen
TNEL	- Threshold No Effect Level
TP	- Total Phosphorus
TSS	- Total Suspended solids
TWQR	- Target Water Quality Range
UCEWQ	- Unilever Centre for Environmental Water Quality
WCSA	- Western Cape System Analysis
WQRU	- Water Quality Resource Units
WRC	- Water Research Commission
WRYM	- Water Resources Yield Model

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CHAPTER 1

METHOD DEVELOPMENT THROUGH APPLICATION – CASE STUDIES

This project was initiated in 2000 to allow the ecological Reserve team for water quality to undertake additional research while working on ecological Reserve determinations funded by the Department of Water Affairs and Forestry. The results of this work form the basis of the methods reported in Hughes (Ed)(in press).

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Formal methods for determining water quality aspects of the ecological Reserve were then developed in two major processes:

- DWAF organised method-development workshops and contracts; and
- DWAF contracted ecological Reserve assessments.

The combined workshop/contract process lead to the publication of the first official methodology (DWAF 1999). As ecological Reserve assessments were undertaken, the DWAF methodology was extended and modified. Three major ecological Reserve assessments contributed to this process: the Crocodile (Mpumalanga), Olifants (Mpumalanga) and Breede (Western Cape) Rivers. Members of the WRC project team for this project were closely involved in the Olifants and Breede River assessments, and this project was planned on the basis of using the data and methods developed during those studies. However, it became clear that other innovative approaches were developed during the Crocodile River Ecological Reserve assessment. Therefore the Crocodile River assessment is introduced in this chapter, and detailed in Appendix 1.

1.1 CROCODILE RIVER (MPUMALANGA)

The water quality required to maintain ecosystems associated with the Crocodile River was determined through applying the ecological Reserve concept and the DWAF (1999) method:

Step 1	Delineate geographical boundaries
Step 2a	Determine ecoregional type
Step 2b	Delineate resource units
Step 2c	Select sites
Step 3	Reference conditions
Step 4a	Present status
Step 4b	Importance + sensitivity
Step 5a	Determine Management class
Step 5b	Set management class

Step 6a	Quantify Ecological Reserve
Step 6b	Set Resource Quality Objectives (RQO)
Step 7	Monitoring Programme
Step 8	Notice of Resource Directed Measures (RDM)

The Crocodile River Ecological Reserve determination followed this process with the following modifications:

- Steps 5a and 5b: These were deemed management responsibilities, and this study therefore only provided inputs to such decisions.
- Step 6a: In quantifying the reserve, biological and water quality monitoring data are used to supplement the generic dose-response relationships (DWAF 1999).
- Step 6b: Resource quality objectives should include ecological considerations, human health considerations and resource use considerations (NWA Act 36 of 1998). This study addressed only the first of these requirements, therefore RQO's were not set.
- Step 8: An administrative requirement that was addressed.

The terms of reference for the assessment were:

i. The reserve should be determined with the highest level of confidence possible, given the time constraints (1 month). The availability of data for this river will allow the assessment to approximate an intermediate reserve determination.
ii. The resource unit for which the reserve will be determined is defined by quaternary catchment X24H.
iii. The study should determine whether the ecosystem in the Komati River, downstream of the confluence with the Crocodile River, is more sensitive than that in the Crocodile River. If this is the case, then the reserve should also provide adequate protection for this section.
iv. The following variables for which the reserve should be determined were identified: Temperature, pH, Dissolved O_2, Cl, TSS, TDS/EC, NH_3 and NO_3, PO_4, Total phosphorous, N:P ratio, Pb, NH_3.
v. The ecological Reserve for water quality should be specified as percentage exceedance values.

Details of the Crocodile River Ecological Reserve study and results are provided in Appendix 1.

1.2 OLIFANTS RIVER (MPUMALANGA)

1.2.1 Introduction

The water quality component of the Olifants River Ecological Water Requirements Assessment (OREWRA) study commenced in January 1999. At that stage the first draft of the RDM Water Quality methodology was under development and the Olifants River study was used as the first application of the draft methodology. The study has been completed and the water quality report has been submitted to the Department of Water Affairs and Forestry for final approval (DWAF 2000).

1.2.2 Application of the seven step RDM Method to the Olifants River

1.2.2.1 Step 1 – Delineate study area

The first step and second step of the water quality reserve study were closely linked. The terms of reference for the OREWRA Study specified the whole of the Olifants River catchment, but selected upper river reaches and tributaries were excluded. To develop a common way of referring to river reaches in the study team, all of the Olifants River and the selected tributaries included in the study were divided into numbered segments. These segments were referred to in the water quality report to delineate the sections of river which were classified and for which a water quality reserve was specified.

The Olifants River catchment area was sub-divided into three main study areas to facilitate the execution of the study and to facilitate interactions with stakeholders. These were:

- *Upper Olifants* which was the catchment upstream of Loskop Dam and included the Upper Olifants, Klein Olifants, and Wilge Rivers;
- *Middle Olifants* which was the catchment downstream from Loskop Dam to just downstream of the Mohlapitse River and included the Middle Olifants, and the Elands River; and
- *Lower Olifants* which was the Olifants River from downstream of the Mohlapitse River, up to the border of South Africa and Mozambique and included the Steelpoort, Blyde, and Selati Rivers.

Where data were available for tributaries not formally included in the study, and whose water quality affected the main stem of the Olifants River (such as the Klip River and the Steenkoolspruit in the upper Olifants, the Moses River in the middle Olifants, and the Ohrigstad River in the lower Olifants), these data were examined, and used to interpret conditions within the affected main stem river reaches.

In this section of the water quality report, a description of the key water quality features of the three sub-study areas was included as background information.

Some remarks on the method used – The terms of reference compiled by the Department defined the boundaries of the study area as well as which rivers and tributaries were included. Negotiations between the project team and Department about the terms of reference were largely designed to meet the needs of the quantity component. The result was that some tributaries that had a significant impact on water quality were excluded from the study. In these cases, the water quality team dealt with these as "hot spots" or "refugia" depending on whether the quality was poorer or better than the mainstem river.

1.2.2.2 Step 2 – Delineate water quality resource units

The next step was to sub-divide the three main study areas into water quality reaches (resource units), which were significantly different from each other to warrant their own specification of a water quality reserve.

In some cases, the size of the reaches for the water quality component differed from the water quantity river reaches because the quantity reaches were more closely aligned with Level 2 ecoregions, whereas water quality in rivers was more closely related to geological regions (Dallas *et al.* 1998) and Level 1 ecoregions (DWAF 1999). In most cases the minimum size of a water quality reach was more or less equivalent to a quaternary catchment size.

The delineation of the water quality reaches consisted of:

Step 1: The three study areas were mapped, and the reservoirs in each study area were identified. The dam wall formed the upstream boundary of a water quality reach and the inflow point into the reservoir formed the downstream boundary of a water quality reach. It was argued that reservoirs modify water quality to a large degree and that it justified using reservoirs as the first break between reaches.

Step 2: The main river channel was then divided into ecoregion (Level 1) boundaries that became further sub-divisions of the reaches.

Step 3: All the DWAF water quality monitoring points in each water quality river reach were then identified. The water quality monitoring point closest to the downstream boundary of each water quality reach became the site where the present ecological state (PES) was defined for that reach.

Where no downstream monitoring point could be identified, the nearest monitoring point in the next water quality reach was used to determine the PES, provided that this monitoring point fell within the upper quarter of the next downstream water quality reach. Otherwise the reaches were combined into one reach with one downstream PES site.

The location of monitoring points dictated to a large degree the length of water quality reaches because without a downstream monitoring point, it was very difficult to define a present state.

A table was compiled describing the key features of each reach as well as the present state and reference sites in the reach.

Some remarks on the method used – The first attempts to delineate water quality reaches using Level 2 ecoregions were stopped because it was found that the water quality data did not support the theory that water quality also changed at a Level 2 ecoregion. The team reverted to a Level 1 ecoregion as a guide to delineate water quality reaches.

1.2.2.3 Step 3 – Define reference conditions

Reference conditions form an integral component of the ecological Reserve determination (DWAF 1999). The *water quality* reference condition is used to describe the natural unimpacted characteristics of a particular section of a river course and to describe the seasonal variation in the data. The reference condition is used to (1) assess the present ecological status by determining the degree of modification from the unimpaired state, and (2) in assigning the ecological Reserve. The reference state therefore represents a baseline for assessing the present water quality in a particular ecoregion of a river.

Problems arise in the middle and lower reaches of rivers where water quality is currently in an altered impaired state, and historical data are not available to provide information about unimpacted conditions. In the OREWRA, data used to describe "reference" conditions for many quality river reaches were the best available "reference" data. These data did not represent the unimpacted state, but rather the best available estimate of the unimpacted state. In the OREWRA, TDS tolerance-test results were used as the primary tool for classifying the resource for water quality, recommending the water quality ecological management class; and for setting the ecological Reserve for TDS. The tolerance results were independent of the reference condition, and therefore the reference condition was used mainly for qualitative comparisons.

Reference conditions are determined for three categories of water quality variables (DWAF, 1999; **Table 1.1**):

System variables - Total dissolved salts (TDS), pH, dissolved oxygen (DO), suspended solids and water temperature;

Nutrients - Soluble phosphorus and N:P ratio; and

Toxic chemicals - Ammonia, organics, inorganics and trace metals.

Table 1.1	Summary of the determination of reference conditions for the three categories of water quality variables.	
System variables	TDS	Defined in terms of hazard to aquatic organisms
	pH	Monthly values are specified for each system variable
	DO	
	Temperature	
	TSS	
Nutrients	SP:TP ratio	Defined in relation to an A Class river status
	Unionised NH_3	
	N:P ratio	
Toxics	Toxic substances	Defined in relation to an A Class river status

Reference conditions for the Olifants River Study

System variables

For each of the system variables, a time series plot was prepared and the data record was examined for positive or negative trends. If a trend existed, the most unmodified ("best") part of the water quality record was selected as the reference condition. Where no trend existed, the most recent five years of data were used. The monthly 25th, 50th and 75th percentiles were then calculated for each of the system variables.

- *Total Dissolved Solids (TDS)*
  Reference conditions for salinity were originally defined by Bath *et al.* (1999) as the guideline Target Water Quality Range (TWQR) (DWAF 1996): or 15% above or below the monthly median at a reference site. In this study, we undertook salinity tolerance experiments, and used the results to add a hazard-based definition of the reference condition following the method of Palmer and Scherman (2000). Mayfly nymphs of the species *Tricorythus discolor*, collected from a reference site in the Upper Olifants, were used as the test taxon, and were assumed to be a sufficiently sensitive species. Class A rivers (in a natural or reference state) are defined for water quality as "Unmodified. Allow minimal risk to sensitive species. Remain within the TWQR". Following Palmer and Scherman (2000), "minimal risk" was defined as the salinity condition where less than 1% of the experimental test population would risk death. Tolerance testing was undertaken using a test population of the same species from the Lower Olifants and the results were considered adequate to accept the extrapolation of the Upper Olifants test results to the Middle and Lower Olifants.

- *pH*
  In the case of the Olifants River study, the monthly 25^{th}, 50^{th} and 75^{th} percentile statistics were calculated for pH at the reference site.

- *Temperature*
  In the absence of adequate stream water temperature records from the study area, a reference temperature time series was derived using average air temperature measurements available from the Weather Bureau for weather stations close to the middle of the study area (upper, middle and lower Olifants). An empirical relationship developed by Pilgrim *et al.* (1998) was used to relate average daily stream water temperature to average daily air temperature.

- *Dissolved oxygen*
  In the absence of adequate dissolved oxygen observations in the study area, reference saturation dissolved oxygen concentrations were derived using the derived stream water

temperatures and altitude, using the APHA method described in Chapra (1997). However, observed dissolved oxygen data sets could not be found for Olifants River study area to confirm the estimated DO concentrations. The few observed data collected during site visits were within the derived range.

- *Suspended solids (TSS)*
  There were no long term suspended solids data sets available to determine reference conditions for the Olifants River study area. The qualitative study by Moolman, Quibell and Hohls (1999) was used for information.

Nutrients : Phosphorus and nitrogen

For the nutrients, phosphorus and nitrogen, it is assumed that the reference condition is specified by the limits for a Class A river (DWAF 2000) (**Table 1.2**).

Table 1.2 Reference conditions for nutrients.

Nutrient descriptor	**Value of reference condition (equivalent to a Class A river)**
Ammonia (expressed as un-ionised NH_3 in mg-N/L)	< 0.007
SP:TP ratio	< 10%
N:P ratio where [SP] is <0.01 mg/LL	10:1
N:P ratio where [SP] is <0.05 mg/LL	20:1

Toxics

Toxic substances are assumed to be those specified in the South African Water Quality Guidelines Volume 7 (DWAF 1996). For toxic organic and inorganic compounds and trace metals, it was assumed that the reference condition is specified by Class A limits (DWAF 1999) which are the target water quality ranges (TWQR) specified in the SA Water Quality Guidelines Volume 7 (DWAF 1996).

Some remarks on the method used – in the case of rivers downstream of reservoirs the team often used the in-lake quality as a "reference condition". Reservoirs often modify water quality to a large degree and set a new starting condition for the river. In this case, the term "Reference condition" is probably not correct, even though it is used to evaluate how much water quality has changed at a site downstream of the dam by comparing it to the in-lake quality.

1.2.2.4 Step 4 – Assess present water quality status

The present ecological state is used to describe the current water quality conditions in the river. The aim is to describe the current water quality at the downstream boundary of a specific water quality river reach. As is the case with reference conditions, the present ecological state is determined for the system variables, nutrients and toxic substances. The most recent three to five years of data were used depending on the number of observations in the data set. In cases where samples were collected infrequently, longer data periods than the previous 5 years were used.

The PES assessment class for TDS was an experimentally derived hazard-based range. For the other system variables, an assessment class was assigned based on the *difference* between the reference condition and the present status. Nutrients were evaluated against tables provided in DWAF (1999). Evaluation of toxic substances proved to be difficult. The terms of reference specified the use of existing data only. The DWAF pollution database could not be made available to the OREWRA as the records were *in-transit*, and therefore difficult to access. It was judged that the available information was not worth the time it would take to access. Specialists with local knowledge were interviewed, and available data were used. A limited instream toxicity test-series was undertaken, and served to indicate the value of that approach. It should be noted that where instream toxicity was measurable, the reach was classified as being in an E/F Class. If regular instream toxicity testing were undertaken, the frequency of measurable instream toxicity might be shown to be acceptably low, but in the absence of such data, measurable instream toxicity was taken to indicate a high risk of ecosystem degradation.

System variables: TDS

When the Bath *et al.* (1999) approach to assessing salinity was applied, nearly all the Olifants River reaches were classified as E/F class status for salinity. This was not consistent with the bio-assessment results in this study. The Bath *et al.* (1999) method for salinity was therefore judged to be too stringent, as it did not discriminate between relative degrees of salinity impact. We therefore undertook a toxicity testing approach, using resources available from a Water Research Commission research project. Mayfly nymphs (*Tricorythus discolor*) were collected from a reference site in the Upper Olifants and subjected to a salinity tolerance testing experiment. The results were used to derive a hazard-based salinity classification for the Olifants River. The experiment was repeated for the same species collected from the Lower Olifants. Difficulties were experienced with transporting the experimental organisms, and the results were less statistically reliable than those for the Upper Olifants. The Lower Olifants results were judged to be adequate to decide that the Upper Olifants results could be extrapolated to the whole Olifants study area. However, at the Lower Olifants specialist meeting it was noted that the Blyde River has a natural salinity profile more similar to the Sabie River than to the main stem of the Olifants River, and other major tributaries. Similar tolerance experiments have been undertaken for the same genus of mayfly (*Tricorythus tinctus)* from the Sabie River (Palmer and Scherman 2000), and the results from those experiments were then applied to the Blyde River.

Classification procedure

In addition to the detailed status descriptions of each reach, the present ecological state (PES) is finally presented as a class. Arriving at a class for water quality is a complex procedure as water quality comprises many water chemistry variables.

The RDM Methodology (DWAF 1999) does not give guidance on calculating an integrated PES class for the water quality of a specific reach. The OREWRA water quality project team developed a procedure for summarising the monthly classes and for integrating across the variables to calculate an overall water quality class. The procedure was submitted to the RDM Office for review.

In addition, for each water quality reach, the bioassessment classes (for fish and invertebrates) were compared with the overall water quality class. Discrepancies, where bioassessment classes were lower than water quality classes, provided the cue for an assessment of instream toxicity testing. This approach needs to be amplified and extended into a formal toxicity-monitoring programme.

Procedure to establish an overall water quality class for each water quality reach:

Step A: Establish the TDS class

- select the appropriate data set (e.g. median monthly TDS values for the past 5 years)
- compute and tabulate median monthly TDS concentrations
- use either river-specific experimental data, or the tables in DWAF (2000), which use tolerance test results to link a class to a TDS range
- identify the classes for the 4 worst-case consecutive months
- identify the classes for the 3 best consecutive months
- compute the average TDS for the year in which the present state is being assessed, and assign an overall TDS class using the tolerance data classification tables
- if the average of the 3 best consecutive months falls in a class above average class for the year, then use the average class for the year as the TDS class
- if the average of the 3 best consecutive months falls in the same class as the class for the year, then use the worst monthly class as the TDS class.

Step B: Establish nutrient class

- exclude ammonia (recent advice from the RDM team is that ammonia should be treated as a toxic chemical not a nutrient)
- use the three nutrient assessments derived from the median concentrations over the previous 5 years
- take the average, or low average (e.g. if B, C and D - then C; or B, C and C - then C); or if only 2 nutrient assessments were made, and these 2 are consecutive then take the lower class (e.g. C and D - then D)
- if any of the assessments is in an E or F class, then the overall nutrient assessment is an E class.

Step C: Record the monthly median pH range

- use this in Step 5.

Step D: Record any incidence of instream toxicity

- use this in Step 5.

Step E: Establish overall water quality class

- start with the TDS class
- check with the nutrient class:
  - if the nutrient class is the same, 1 class below, or 1 class above the TDS class, then the overall water quality class is the same as the TDS class
  - if the nutrient class is 2 or more classes below the TDS class then the overall water quality class is 1 class below the TDS class
- only consider allowing the pH to influence the overall water quality assessment if the median pH is less than 5.5 or greater than 9.5 (general standard)
- if pH is outside of this range decrease the class by one level
- if instream toxicity is detected the reach should be recorded as an E/F class, with notes on the possible source of toxicity.

Step F: Procedure for using fish and invertebrate bioassessments

- *Invertebrates*

The SASS scores and associated classes for invertebrates (**Table 1.3**); and the classification for fish recorded in the OREWRA EMC report were listed, and compared with the water quality class. Where the bioassessment class was lower than the water quality class the reach was selected for an instream toxicity test, and/or the role of turbidity and habitat impairment was assessed.

Table 1.3 Palmer (OREWRA EMC report) was used to link SASS scores to classes:

Table 1.3 SASS scores and average score per taxon (ASPT) related to classes (A – F)		
Class	**SASS Score**	**ASPT**
A	>175	>7
B	140 – 175	6.2 - 7
C	85 – 139	5.6 - 6.1
D	60 – 84	4.8 - 5.5
E	41 – 59	4 - 4.7
F	<40	

Where SASS scores and ASPT score for a single site were in different classes, the class description was modified by using primarily the SASS score, but adding "high" or "low" (e.g. a SASS score of 100, with an ASPT of 5.4 would be a low C).

- *Fish*

The fish scores and categories recorded by the fish specialist were listed and used.

1.2.2.5 Step 5 – Ecological management class

The ecological management class for each reach was set at the specialist workshop. The water quality team prepared a Water Quality Starter document for the three IFR specialist workshops. After the Upper Olifants specialist workshop it was clear that the water quality team had to respond to a number of questions that could not be answered with the information obtained as part of the routine water quality reserve methodology. At the Middle and Lower Olifants specialists workshops, these questions were refined and the water quality team were able to respond appropriately after additional preparation was done for the workshops. The questions posed at the specialist workshops that were related to water quality were:

- Describe water quality reference conditions, present state (including trajectory of change and resulting class) and provide contributing factors for the resulting PES.
- Describe the attainable ecological class in terms of water quality (including an assessment how difficult it would be to attain).
- Ecological class as specific objectives.
- Water quality consequences of the recommended flows.

In order to answer these questions, the team prepared information for each reach on the following aspects in addition to Step 1 to 4 of the water quality Reserve method (**Figure 1.1**):

- A qualitative assessment of the quality of the data and information on which the water quality assessments were based (rated on a scale of good to poor).
- A description of the trajectory of change (better, poorer or unchanged). Only long-term TDS trends were examined. An examination of seasonal trends was part of the Reserve method for system variables.
- An estimate of the medium and long-term water quality state if current trends continue unchanged.
- Causes and origins of water quality problems.

- The attainable water quality class and how difficult it would be to achieve the class.
- Initial water quality objectives (only considering water quality and not the other components of the study).
- The relationships between flow and constituent concentrations to assess the water quality implications of recommended maintenance, drought and flood flows.

At the lower Olifants River specialist workshop, a flow stress relationship was developed for the lower Blyde River. This provided further opportunity to examine the flow/TDS and TDS/stress relationships.

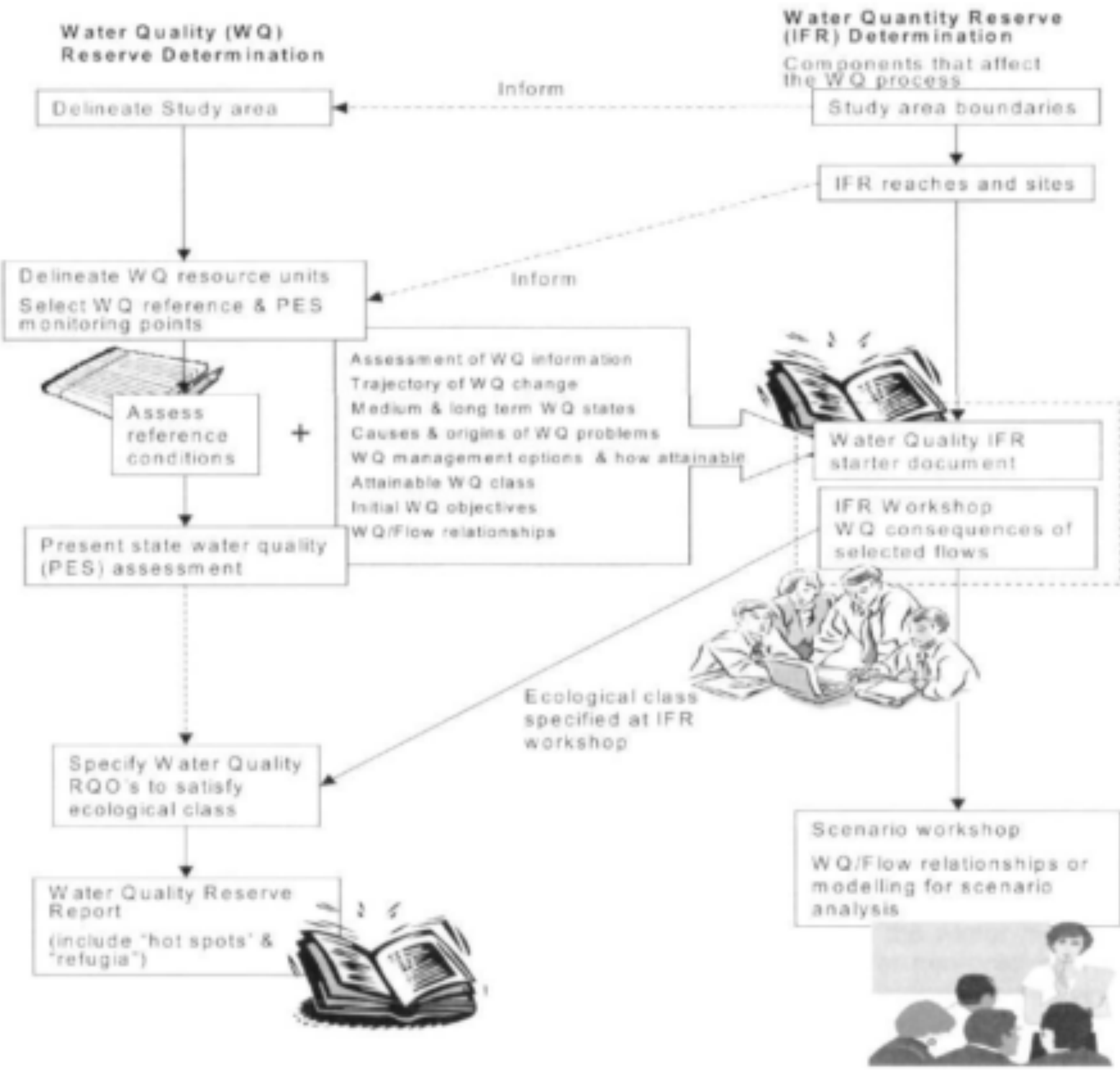


Figure 1.1 Diagram showing the interactions between the quantity and quality Reserve determinations and the additional information required for the specialist workshops.

1.2.2.6 Step 6 – Set resource water quality objectives

Once the ecological class was set for the biota, the resource water quality objectives could be set to satisfy the water quality needs of the selected class. In many cases it meant the water quality situation had to improve, in some cases remain the same and in others, utilization of the resource could be authorized which will result in a lower water quality class. Setting resource water quality objectives meant unpacking the water quality category, almost reversing the order in which the summary water quality category was determined (**Figure 1.2**).

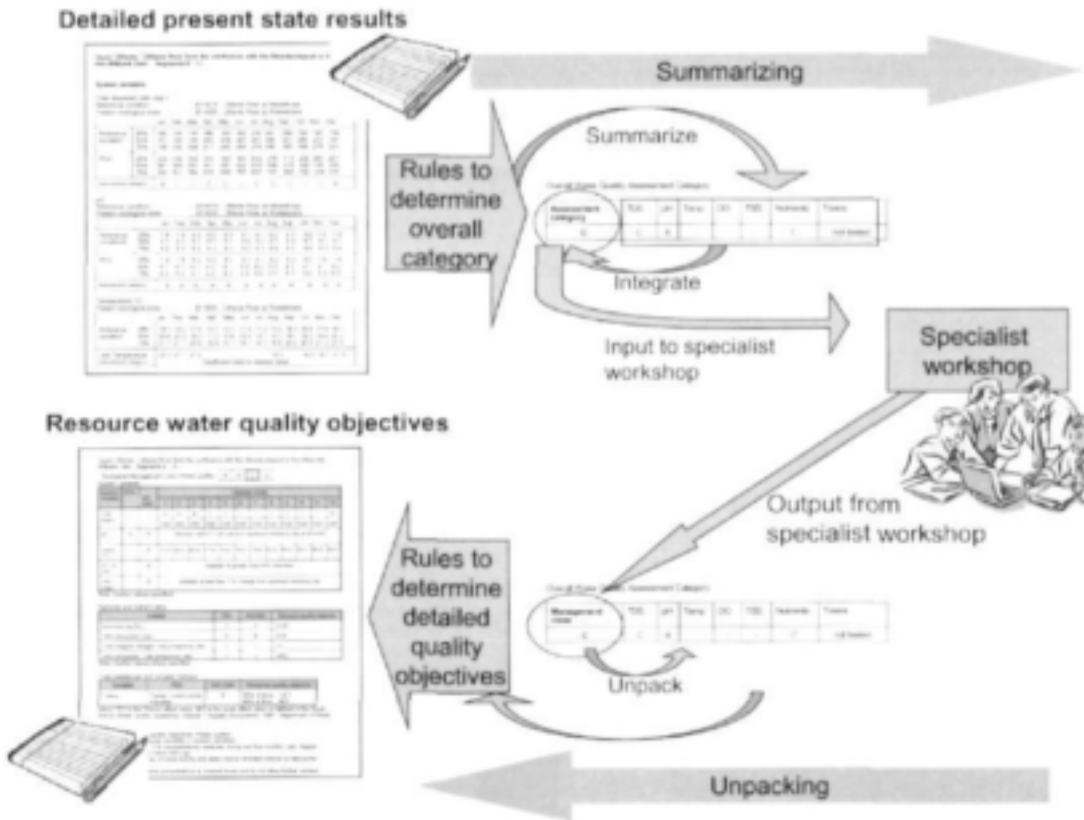


Figure 1.2 Diagram illustrating the processes of summarizing the water quality status and unpacking the resource water quality objectives once the EMC has been set.

The following aspects were also considered when setting the resource water quality objectives:

- In many cases where the EMC was better than the PES water quality class, the poor water quality was the result of high TDS concentrations during the winter months and this could be traced back to discharges from known point and/or non-point sources in the catchment. In these cases, reducing the winter TDS concentrations and leaving the summer values unchanged could achieve the desired EMC.

- If the PES category for a specific variable (for example pH) was better than the desired EMC and there was no obvious reason why this should be changed, the RQO was set equal to the PES.

Some remarks on the method used – The team followed a conservative approach for specifying RQOs for individual water quality variables. It was argued that the cumulative effect of specifying RQOs for all the water quality variables equal to the EMC, may result in a lower overall water quality category.

1.2.2.7 *Step 7 – Monitoring system design*

This step was not part of the terms of reference for the OREWRA study.

1.2.3 A hazard-based approach to applying tolerance testing in the development of Ecological Reserve evaluations for salinity in the Olifants River

1.2.3.1 *Introduction*

This section provides a summary of the results from the use of tolerance data in setting the salinity values for the ecological Reserve determination for the Olifants River (Gauteng/Mpumalanga). Details of the philosophy and the hazard-based approach are given in Palmer and Scherman (2000). In summary, the method is based on the site-specific tolerance response of selected invertebrates.

The resource classes A - D are envisaged as providing conditions of increasing risk to biota (Palmer 1999). Hazard and risk are linked concepts. Hazard is a measure of the danger something poses, and risk is a measure of the likelihood of encountering that danger. Tolerance experiments provide a quantified measure of hazard - the danger posed, in this case, by sulphate-dominated elevated salinities. If the likelihood of exposure is modelled, by linking concentration to the hydrology, then risk can be estimated.

Methods for the determination of water quality aspects of an ecological Reserve assessment were still under development at the start of the OREWRA study. In the RDM (Resource Directed Measures) method of Bath *et al.* (1999), TDS was listed as a system variable, for which the reference condition is described in terms of the median, and $25^{th}/75^{th}$ percentiles of TDS/conductivity data collected at a reference site; and the classes for the present ecological state (PES), were defined as deviations from the reference condition. Bath *et al.* (1999) successfully applied this method in the Crocodile River, but it proved to be unsatisfactory for the Sabie River, and an alternative method was developed which allowed tolerance test data, rather than a percentage deviation from the reference condition, to link TDS values to resource classes (Palmer and Scherman 2000). This alternative method was applied in the OREWRA study, and the method then formed the basis of a revised RDM method (DWAF 1999). The OREWRA thus contributed directly to the development of RDM methods. (Note: Funds for the Upper Olifants salinity tolerance testing were provided by the Water Research Commission.)

Unlike the Sabie River, the Olifants River is heavily impacted with regard to TDS, which is a conservative, multi-variate water quality parameter (many different organic and inorganic ions go make up electrical conductivity and total dissolved solids). When the Bath *et al.* (1999) method was applied to reaches in the upper Olifants River, there was a considerable difference between the TDS values in the reference site and most of the other sites, and most of the water quality reaches were therefore classified in the E/F resource class for salinity. This seemed to be misleading as the SASS results indicated lower levels of impact. We suggest that this discrepancy relates to the nature of biotic responses to elevated salinity. Research over the past five years (Palmer *et al.* 1996, Binder 1999, Palmer and Scherman 2000) has indicated that elevated salinity acts as a slow toxicant, and with both acute (96 hour) and chronic (10 day) responses recorded at concentrations well above those usually experienced in rivers. A review of biotic responses to salinity is given in Binder (1999). However, salinities in parts of the Olifants River are considerably elevated, particularly in winter, and it was important that we should be able to present a biologically sensible assessment of the PES.

Experimental toxicology results have been used in setting water quality guidelines (DWAF 1996), and the ecological Reserve for toxic substances (DWAF 1999), and it seemed probable that empirical tolerance data for salinity would be useful for setting resource quality objectives for salinity. However, worldwide there have been few studies of the impact of elevated salinity on freshwater biota (Hart *et al.* 1990,1991; Lowell *et al.* 1995). The mayfly *Tricorythus discolor* is one of the dominant filter feeding, riffle-dwelling macroinvertebrates in the Olifants River. *T. discolor* were collected from minimally salt-impacted reference sites: B1H018 – a reference site for the Upper Olifants and from the National River Health Programme biomonitoring site near Mamba weir, in the Lower Olifants. Nymphs were transported to the artificial stream laboratory at Rhodes University in Grahamstown, and experimentally exposed to elevated salinities (sodium sulphate-based), with the aim of experimentally establishing the concentration-response relationship.

1.2.3.2 Study Site And Methods

More than 2000 *T. discolor* nymphs were collected from a riffle on the farm Middelkraal, near to the DWAF water quality monitoring site B1H018 in the upper reaches of the Olifants River. Water chemistry data showed this site was minimally salt-impacted, with salinities rising from reference conditions only in the winter months.

Individual nymphs were collected off rocks, using a paintbrush to avoid damaging them, and were placed in cooled, aerated river water in a cooler box. Sponges were placed in the water to which the nymphs could cling. The cooler box, with aerated water and test organisms, was transported by air to the artificial stream laboratory at Rhodes University, where 29 artificial stream channels, each recirculating 20 litres of charcoal-filtered tap water, had been prepared. Between 40 and 50 undamaged nymphs, which were not in the final instar, were placed in each channel. After 48 hours of acclimation, dead individuals were removed. There was less than 1% mortality in each channel, and the experiment was started with the addition of sodium sulphate to fresh filtered tap water. Three replicate experiments were run: with three filtered tap-water controls, and three replicates of each of the following concentrations: 0.2, 0.5, 1, 2, 3, 4, and 6 g/L Na_2SO_4.

Channels were checked for mortalities in the morning and evening from the start of the experiment until the end of Day 10. Fresh salt solutions and controls were made up on Days 4 and 8. Organisms were fed with Tetramin (Haigh and Davies-Coleman 1997) on Day 5. Full chemical analysis of the test water was conducted by IWQS, DWAF, from water collected from each channel on Days 1, 4, 8,10 and temperature, pH and EC (electrical conductivity) were checked daily. Mortality results were analysed using the parametric Probit analysis, and the non-parametric Trimmed Spearman-Karber analysis (APHA 1992). Both methods yielded similar values. Probit results were used because a variety of LC values were provided. (LC values estimate the Lethal Concentration - this term has been used rather than EC for Effective Concentration even though a surrogate for death - total immobility - was used as the experimental end-point. EC could have been confused with EC for electrical conductivity).

All the collection and experimental methods had been previously applied (Palmer *et al.* 1996; Binder 1999; DWAF 2000; Palmer and Scherman 2000).

Essentially the same procedure and methods were followed for *T. discolor* nymphs collected from the Mamba Weir site in the Lower Olifants. However, the travel time back to Grahamstown was longer, and was attempted by road. Nymphs were in cooler boxes for a minimum of 17 hours (collecting took 4 hours). Either the time from collection, or the prolonged agitation of the water in a vehicle, proved too stressful for the mayflies, and unacceptably high travel and acclimation mortalities (up to 40%) were recorded. The travel mortalities forced a decision to run an unreplicated experiment, with a larger number of concentrations, and fewer nymphs per channel. Unfortunately there were also high mortalities in control channels. At the end of a 10 day experiment only 96 hour results could be computed. Although the statistical variability was too high for these results to be acceptable for scientific publication, the results for acute exposure (96 hours) were computed, and compared with the acute exposure results from the Upper Olifants. In the judgement of the OREWRA water quality team, the Upper and Lower Olifants data were sufficiently similar to reasonably accept the application of the Upper Olifants data to all the OREWRA river segments. (Results of the Lower Olifants experiments are given in **Table 13**).

1.2.3.3 Results

There were 3 independent sets of mortality results from the Upper Olifants experiments. These were analysed separately to check that the results were within a reasonably similar range, (i.e. as a check on the degree of variability in response within the test population) and then the results were pooled so as to take advantage of the large numbers of test organisms used (**Table 1.4**). Data were analysed as nominal exposure concentration (Na_2SO_4), measured SO_4 concentration and conductivity. The pooled data were used in the PES state evaluation. Probit analyses provided the best estimates of the concentrations at which 1% (LC_1), 5% (LC_5), and 50% (LC_{50}) of the test population died, and these values, as well as their 95% confidence limits are listed (**Tables 1.5** and **1.6**).

These LC values are used, as described in Palmer and Scherman (2000) to calculate the AEV and CEV values (Roux *et al.* 1996, **Tables 1.7** and **1.8**).

The LC, AEV and CEV values were then used to derive a ranked hazard table (**Table 1.9**) and **Table 1.10** gives hazard descriptions associated with these ranked tolerance end points.

Table 1.11 provides a risk-based site-specific salinity guideline, related to resource classification, for the upper Olifants River.

Table 1.12 shows a comparison of the salinity class ranges in the Sabie and Olifants Rivers.

Table 1.4	LC$_1$, LC$_5$ and LC$_{50}$ results of the Probit analysis for the individual experiments and the pooled data, expressed as Na_2SO_4 and SO_4 concentrations (g/L) as well as conductivity (mS/m).						
		Acute (96h) Results			Chronic (10d) Results		
		Na_2SO_4 (g/L)	SO_4 (g/L)	EC (mS/m)	Na_2SO_4 (g/L)	SO_4 (g/L)	EC (mS/m)
LC$_1$	Pooled data	1.07	0.72	155	0.46	0.31	90
	Experiment 1	0.9	0.61	145	0.65	0.44	115
	Experiment 2	0.9	0.61	145	0.75	0.51	125
	Experiment 3	1.92	1.30	280	0.34	0.23	75
LC$_5$	Pooled data	1.6	1.08	235	0.68	0.46	125
	Experiment 1	1.5	1.01	225	0.84	0.57	140
	Experiment 2	1.38	0.93	205	0.99	0.67	160
	Experiment 3	2.56	1.73	340	0.57	0.39	105
LC$_{50}$	Pooled data	4.4	2.97	515	1.82	1.23	265
	Experiment 1	4.6	3.11	535	1.56	1.05	230
	Experiment 2	3.6	2.43	445	1.72	1.16	250
	Experiment 3	5.11	3.45	5.85	2.02	1.37	285

Table 1.5	Olifants River - Upper Reaches - Sodium Sulphate - ACUTE toxicity results (96h/Day 4): Estimated LC Values and Confidence Limits (CL) expressed as nominal Na_2SO_4 concentration and conductivity.					
LC Value	Concentration (g/L) Na_2SO_4	Lower 95% CL	Upper 95% CL	EC mS/m (Na_2SO_4)	Lower 95% CL	Upper 95% CL
LC$_1$	1.08	0.29	1.71	155	65	250
LC$_5$	1.63	0.65	2.29	235	110	315
LC$_{50}$	4.45	3.69	5.59	515	445	625

Table 1.6	Olifants River - Upper Reaches - Sodium Sulphate - CHRONIC toxicity results (Day 10): Estimated LC Values and Confidence Limits (CL) expressed as nominal Na_2SO_4 concentration and conductivity.					
LC Value	Concentration (g/L) Na_2SO_4	Lower 95% CL	Upper 95% CL	EC mS/m (Na_2SO_4)	Lower 95% CL	Upper 95% CL
LC$_1$	0.49	0.002	1.06	90	30	165
LC$_5$	0.72	0.009	1.34	125	45	200
LC$_{50}$	1.82	0.46	2.65	265	90	355

Table 1.7 Calculations of possible Acute Effects Value (AEV) and Chronic Effects Value (CEV) (DWAF 1996) using salinity tolerance test results expressed as conductivity (mS/m). The hazard-based use of LC values is described more fully in Palmer and Scherman (2000). The values in bold are used in **Tables 1.8** and **1.9**.

* Although test results from only one taxon were used, the safety factors designated in DWAF (1996) have not been applied because LC_1 *(with a 1% mortality risk) and* LC_5 *(with a 5% mortality risk), rather than the mean* LC_{50} values from a range of taxa, were used.

Acute Effects Value (AEV)

A. Calculate the Final Acute Value (FAV) - use the LC_5^* or LC_1^* for acute (4 day) tests:

4 Day exposure:	Na_2SO_4 (mS/m)
LC_5^*	235
LC_1^*	155

B. Calculate the Acute Effects Value (AEV)

$\frac{FAV}{2}$ =

	Na_2SO_4 (mS/m)
LC_5^*	**118**
LC_1^*	**78**

Chronic Effects Value (CEV)

CEV can be calculated in 2 ways:

A. Use the LC_5^* or LC_1^* for chronic (10 day) tests:

10 Day exposure:	Na_2SO_4 (mS/m)
LC_5^*	**125**
LC_1^*	**90**

B. 1. Calculate the acute:chronic ratio (ACR) - $\frac{\text{acute } LC_{50}}{\text{chronic } LC_{50}} \quad \frac{515}{265} = 1.94$

2. CEV = $\frac{FAV}{ACR}$

	Na_2SO_4 (mS/m)
LC_5^*	**121**
LC_1^*	**80**

Table 1.8 Acute Effect Value (AEV) and Chronic Effects Value (CEV) calculated from LC_1 and LC_5 values (mS/m).

LC_5 - based AEV	118	LC_5 - based CEV	125
LC_1 - based AEV	78	LC_1 - based CEV	90
		LC_5 - based CEV - using the acute to chronic ratio calculation	121
		LC_1 - based CEV - using the acute to chronic ratio calculation	80

Table 1.9 List of classes and tolerance end-points with the associated conductivity; ranked by increasing hazard posed to the mayfly *T. discolor*, from exposure to sodium sulphate solutions. where: acute = 4-day exposure, chronic = 10-day exposure, H= hazard sequence, and * indicates 30 mS/m - the annual median salinity for the reference site (DWAF 2000).			
	Sodium Sulphate (Na_2SO_4)		
H	**Tolerance test result**	**Conductivity (mS/m)**	**Class**
1*	Annual median salinity for the Upper Olifants reference and test organism collection site, and below the lower 95% confidence limit of the chronic LC_1	30	A
2	The lower 95% confidence limit of the chronic LC_5	45	B
4	LC_1 - based AEV	78	C
5	LC_1 - based CEV - using the acute to chronic ratio calculation	80	
6	The lower 95% confidence limit of the chronic LC_{50} and Chronic LC_1	90	D
7	The lower 95% confidence limit of the acute LC_5	110	
8	LC_5 - based AEV	118	
9	LC_5 - based CEV - using the acute to chronic ratio calculation	121	
10	Chronic LC_5	125	E
11	Acute LC_1	155	F
12	Acute LC_5	235	

Table 1.10 Examples of selected tolerance endpoints and associated hazard descriptions. Similar descriptions can be derived for any LC value and associated confidence limits. Of those listed here, the LC_{50} is the most accurate, and the LC_5 and LC_1 indicate concentrations posing a lower hazard.	
Tolerance end-point	**Risk description**
Below the low 95% confidence limit for the LC_1	Concentrations at which each nymph has <1% chance of mortality
Below the low 95% confidence limit for the LC_5	Concentrations at which each nymph has <5% chance of mortality
LC_1	Best estimate of concentration where each nymph has 1% chance of mortality
LC_5	Best estimate of concentration where each nymph has 5% chance of mortality
LC_{50}	Best estimate of concentration where each nymph has 50% chance of mortality
AEV	A measurable acute effect - may be associated with 1% (LC_1) or 5% (LC_5) risk of mortality
CEV	A measurable chronic effect - may be associated with 1% (LC_1) or 5% (LC_5) risk of mortality

Table 1.11 A hazard-based, site-specific salinity guideline for the upper reaches of the Olifants River, for resource classes A-F. The ranges given indicate the median instream salinity that should not be exceeded in the lowest flow month. Classes E and E are indicated by salinities which would result in biotic degradation. The water quality objective for each class is given (Palmer 1999).

Class A **20 - 30 mS/m**

Water quality Unmodified. Allow minimal risk to sensitive species. Remain within the target water quality range (TWQR, sensu DWAF 1996).

Risk *Within this range each nymph would face a less than 1% risk of mortality from exposures of up to 10 days.*

Class B **30 - 45 mS/m**

Water quality Use Aquatic Ecosystems guideline values (DWAF 1996) such as chronic effects value (CEV) and TWQR to set objectives that pose slight risk to intolerant organisms.

Risk *Within this range each nymph would face a less than 5% risk of mortality from exposures of up to 10 days.*

Class C **45 - 80 mS/m**

Water quality Use aquatic ecosystems guideline values such as Acute Effects Values (AEV), CEV, and TWQR to set objectives that allow moderate risk only to intolerant biota.

Risk *Within this range there is a risk of 1-5% mortality from exposures of longer than 10 days, and 1% mortality from exposures of more than 4 days.*

Class D **80 - 120 mS/m**

Water quality Use aquatic ecosystem guidelines values (AEV, CEV<TWQR) to set objective that may result in high risk to intolerant biota.

Risk *Within this range falls the best estimate of that 1% mortality would occur, but that less than 5% mortality would occur, from exposures of longer than 4 days. In addition, 5% mortality would be expected, and up to 50% mortality could occur, from exposures of longer than 10 days.*

Class E **120 - 155 mS/m**

Water quality The water quality poses the risk of ecological degradation. The minimum management objective should be those for a D Class.

Risk *Within this range there is the best estimate the 5% mortality would occur after exposures of longer than 10 days.*

Class F **> 155 mS/m**

Risk *Above 155 mS/m mortality would be expected to occur after short term (4day) exposure.*

Table 1.12 A comparison of the salinity class ranges in the Sabie and Olifants Rivers.

Class	Sabie (mS/m)	Olifants (mS/m)
A	8 - 17	20 - 35
B	17 - 30	35 - 45
C	30 - 40	45 - 80
D	40 - 50	80 - 120
E	> 50	120 - 155
F		>155

1.2.3.4 *Discussion points*

The method used here is still to be published in the peer-reviewed literature. Until then, the method should be regarded as preliminary and exploratory. It has been presented at conferences in South Africa and Australia and was well received. At this stage there are a limited number of important points to be made:

- the same approach, using organisms of the same genus, in two rivers (Sabie and Olifants) with very disparate natural and impacted salinity profiles produced results which were consistent with each river's assessed ecological condition, and biomonitoring results;
- this points to the value of this as a site-specific approach;
- the approach gave a much better assessment of the present state for salinity than the Bath *et al.* (1999) method - in both rivers, with degraded reaches in the Olifants successfully identified and discriminated from less impacted reaches.

1.2.3.5 *Lower Olifants Tolerance Data*

Table 1.13 shows a selected analysis of the acute (96 hour) results from the Lower Olifants exposure experiment. Control mortality was 30% and the variability of response was high. The OREWRA water quality team compared this analysis, and alternative analyses, with those presented in **Table 1.5**, and agreed that they fell within the range of the better replicated and more reliable results from the Upper Olifants, which were then used as described above to derive a range of salinities to be linked with each Class.

Table 1.13 Olifants River- Lower Reaches -Sodium Sulphate - ACUTE toxicity results (96h): Estimated LC Values and Confidence Limits (CL).

LC Value	Concentration (g/L) Na_2SO_4	Lower 95% CL	Upper 95% CL	EC mS/m (Na_2SO_4)	Lower 95% CL	Upper 95% CL
LC_1	0.7	0.2	1.3	115	60	185
LC_5	1.1	0.3	1.7	155	85	215
LC_{50}	2.6	1.7	3.4	310	215	385

1.2.4 Lessons learnt in the OREWRA study

The water quality team compiled a list of lessons learned during the execution of the project. This list is by no means comprehensive but it probably reflects the ones that had the biggest impact on the project.

- Water quality team composition – The Olifants reserve determination was done at a comprehensive level. The water quality team consisted of a water quality specialist (Nico Rossouw), and aquatic ecologist (Tally Palmer) and a water quality modeller (Heather Malan). This combination worked very well because each team member had skills that complemented each other. The water quality specialist understood the macro-scale water quality processes and driving forces in the catchment, the aquatic ecologist understood the implications of water quality in terms of invertebrates and was able to liaise effectively with the invertebrate specialists. The water quality modeller added substantial value because she brought the quantity and quality together.

- Reserve methodology – When the OREWRA project was initiated, the water quality reserve methodology still in very much a draft format. The OREWRA project added substantial value to RDM WQ method development project because it acted as the first comprehensive test of the draft methods. The project had a direct impact on the water quality reserve methodology that was published in October 1999. However, there are mechanisms in the RDM Office to capture lessons learned on an ongoing basis to ensure a more robust method is eventually gazetted.

- Problems with data – The POLMON data was unavailable to the project team due to operational problems experienced by the DWAF regional offices. The team was uncertain about the degree to which this would have added value to the determination. However, the team felt that it would have helped to explain some discrepancies between the water quality data or category and the invertebrate data or category, especially in those cases where the invertebrate category was much poorer that the water quality data indicated it to be.

- Water quality variables – A major shortcoming of the study was the non-availability of observed temperature, dissolved oxygen and total suspended solids data in the study area. DWAF does not collect this data on a routine basis. The data should be collected but if the data are to be useful, it should be collected over one or two seasons preceding the study, not just during the execution of the study.

- Restriction on data collection – A restriction was placed on the collection of additional data. This meant that data on the system variables such as temperature, dissolved oxygen and total suspended solids could not be collected. The team was however allowed to collect samples for toxicity tests. These tests added substantial value to the study and DWAF should be commended for this foresight.

- Recommended flows violate the water quality objectives – One of the issues that arose at the specialist workshops was what to do in cases where the recommended flows would result in the water quality objectives being violated, specifically because the poor water quality is the result of point or non-point source pollution. In these cases, the recommended flows were NOT adjusted to effect dilution. However, in the report the consequences of the recommended flows were stated and it was recommended that management actions be taken to address the water quality problems. Examples of this was middle Olifants situation where TDS from Loskop irrigation scheme dominate water quality until it is diluted by the Steelpoort and by the Blyde rivers. The same applied to TSS that is a consequence of poor land-use practices in the former Lebowa homeland.

- Toxicity tests – The toxicity tests and risk based TDS experiments that IWR Rhodes undertook were crucial to the successful completion of the project. For the first time it provided real data for the evaluations of the effects of TDS on aquatic invertebrates.

- Water quality guidelines for Aquatic Ecosystems – the OREWRA study pointed out inadequacies in the Water Quality Guidelines for Aquatic Ecosystems. The original water quality reserve methodology was based on the guidelines but when it was applied in practice, it was found that some guidelines were obviously inappropriate. It indicates that the aquatic ecosystem guidelines need revision to address these problems.

- Research needs – some of the lessons learned require additional research that can be funded by the Water Research Commission. It is recommended that the WRC fund a project to capture successes and problems and use this information as the basis for a research strategy to improve the scientific basis of the water quantity reserve methodology.
- Interaction with hydrology team – provision should be made for the water quality team to work with the hydrology team and to explain their data requirements.

- Managing water quality impacts – There appears to be a perception amongst water resource managers that all water quality problems will be solved by source directed controls. This leads to a lack of attention to the consequences of flows on water quality. The time frame to implement source control measures is often very long and to see results often even longer. There is therefore the need to control abstractions or to find some other way to allow for interim dilution. In the Olifants study there was not the opportunity to evaluate catchment scale impacts. An attempt by the water quality team to address this was to identify "Hot spots" and "Refugia". "Refugia" were the Blyde and the Mohlapitse Rivers, essential to maintain or reach water quality objectives in the Olifants River in the Kruger Park. "Hot spots" were areas that needed urgent attention, e.g. Selati and Klip Rivers.

1.3 BREEDE RIVER (WESTERN CAPE)

1.3.1 Introduction

The development of the Greater Cape Town Metropolitan Area (GCTMA) and adjacent towns, as well as the agricultural needs of the south western region of the Western Cape, make the efficient utilisation of available water resources a necessity. Rapid population growth, industrial development, improving living standards and the needs of irrigation have severely taxed the water resources of the GCTMA and the surrounding environment. One of the potential supply options identified in the Western Cape System Analysis (WCSA) was that of transferring water from the Breede River to the Berg River catchment and ultimately to the GCTMA. A comprehensive study of the Breede River Basin was initiated to evaluate the potential use of water resources in the basin and assessing the quantity and quality of water that could be made available for external use. A component of the basin study was a water quality situation assessment.

A further component of the Breede River Basin Study (BRBS) was an investigation of the aquatic ecosystems which included the determination of the ecological Reserve (quantity and quality) for the key river reaches and principal wetlands in the basin. The Department of Water Affairs and Forestry and the Water Research Commission also gave approval to use this opportunity to apply the DRIFT and BBM methodologies and to critically compare the two methods.

Most of the runoff in the Breede River and its tributaries originates in relatively pristine high rainfall source areas underlain by Table Mountain Sandstone. Consequently, the natural quality of the stream flow is excellent. Historically, the dominant water quality issue in the Breede River catchment has been progressively rising salinities in the middle and lower reaches of the main stem, as well as in certain tributaries, such as the Kogmanskloof and the Poesjenels, due to increasing irrigation return flow impacts.

The two key water quality problems in the Breede River basin that have received most attention have been salinisation of the river and high nutrient concentrations in some reaches of the river or its tributaries. However, concerns have also been expressed about suspended sediment loads as a result of inappropriate farming practises such as not adhering to guidelines for establishing buffer strips next to watercourses or changing watercourses. Concerns have also been expressed about the bacteriological quality of the water in the Breede River and its tributaries. Lastly, concerns have also been expressed about the presence of agrochemicals in surface and ground waters in the Breede River catchment.
At the time of compiling this report, the specialist workshops have been completed and the Breede River water quantity and quality reserve documents were in preparation.

1.3.2 Application of the seven step RDM method to the Breede River

1.3.2.1 Step 1 – Delineate the study area

The Breede River catchment comprises the drainage areas of six basins. These are the Ceres basin (upper part of H100), the Upper Breede River catchment (rest of H100), the Hex River catchment (H200), the Middle Breede River catchment (H300, H400, H500), the Riviersonderend catchment (H600), and the Lower Breede River catchment (H700), which includes the Breede River estuary.

1.3.2.2 Step 2 – Delineate water quality resource units

The Breede River was divided into five resource units for the purpose of determining the water quantity reserve:

1. IFR1 – Upper Breede River from Witbrug to confluence with the Wit River
2. IFR2 – Moolenaars River from confluence with the Elands River to the confluence with the Tierkloof River
3. IFR3 – Middle Breede River from Moordkuil to Bonnievale
4. IFR4 – Lower Breede River from confluence with the Buffeljags River to the head of the estuary
5. IFR5 – Riviersonderend River from confluence with the Baviaans River to the town of Riviersonderend
6. IFR6 – Baviaans River from the dam to the town of Genadendal.

For the water quality reserve determination, the study area was divided into 10 water quality resource units (**Table 1.14**), most of which corresponded to the IFR resource units but adding resource units so that the complete Breede River was covered by water quality resource units.

Table 1.14 Water quality resource units defined for the Breede River water quality Reserve determination.

Water quality resource unit	Description	Corresponding IFR unit
1	Upper Breede River from to Wit River confluence	IFR 1
2	Upper Breede River from Wit River to Moolenaars confluence	
3	Middle Breede River from Moolenaars to Kogmanskloof confluence	IFR 3
4	Middle Breede River from Kogmanskloof to Riviersonderend confluence	
5	Lower Breede River from Riviersonderend to Buffelsjags River	
6	Lower Breede River from Buffelsjags to estuary	IFR 4
7	Riviersonderend River from Theewaterskloof Dam to Bok River	IFR 5
8	Riviersonderend River from Bok River to Breede River confluence	
9	Baviaans River	IFR 6
10	Moolenaars River	IFR 2

Water quality reserves were not determined for the other Breede River tributaries but were referred to in the descriptions of the different mainstem reaches.

1.3.2.3 *Step 3 – Define reference conditions*

For the BRBS water quality reserve, the approach followed in the Olifants River study with regard to reference conditions (and described in the RDM documentation), was used.

Reference conditions were defined for those water quality resource units (WQRUs) situated close to the headwaters of the system. These were water quality resource units 1, 7, 8, 9 and 10. For these WQRUs, monitoring points that were regarded as largely unmodified, were identified in the upstream reaches of the resource unit. In fact, the Baviaans River (WQRU 9) and the Moolenaars River (WQRU 10) were regarded as largely natural and probably equal to a reference condition (not for nutrients in the case of the Moolenaars site).

However, water quality in the middle and lower reaches of mainstem Breede River was modified to a large degree by intensive agriculture along the river and its tributaries as well as the control of water quality through releases from Brandvlei Dam. For these river reaches no data could be found to characterize unimpacted conditions. In these cases, the present water quality state was compared to a monitoring point in the upstream reach to illustrate the degree of change that took place in that particular section of the river. A new term should be found for this approach as "reference condition" as originally defined is probably inappropriate.

1.3.2.4 *Step 4 – Assess present water quality status*

The approach followed for present state assessment was similar to that described for the Olifants River. A monitoring point was identified in each water quality resource unit to characterize the present water quality status. However, many of the lessons learned and integration rules developed in the Olifants River study were applied in the BRBS. Once again there was no observed temperature, dissolved oxygen or total suspended solids data.

To assess the TDS status, the Institute for Water Research at Rhodes University undertook an investigation of the tolerance of biota from the basin to different salinity levels (Appendix 2). One of the findings of their investigation was that the invertebrate biota was more tolerant to increased salinity than invertebrates from the Olifants River.

1.3.2.5 *Step 5 – Ecological management class (EMC)*

The EMCs were set at the BBM specialist workshop.

When preparing the starter document, the brief to the water quality team was to focus on the present water quality status, largely due to limited time and resources. Although the team quantified the reference conditions as part of their calculations, it was not included in the starter document. This turned out to be an inconvenience at the specialist workshop when interacting with other specialists on water quality issues. Reference conditions are, however, part of the BRBS water quality reserve documents currently being prepared.

The DRIFT and BBM workshops took place one week apart in July 2001. The DRIFT methodology required a much greater understanding of the relationship between flow and concentration at the different IFR sites. The water quality team therefore prepared flow/concentration scatter plots for different water quality variables to examine the changes in water quality at low, medium and high flows. Greater consideration was also given to understanding the catchment processes that affect water quality at the different IFR sites in order to describe the water quality changes with given reductions in flow. The fact that the water quality team also undertook the BRBS water quality situation assessment made this task easier and less time consuming. The DRIFT methodology places much greater demands on the specialist team to understand the site-specific water quality characteristics.

1.3.2.6 *Step 6 – Set resource water quality objectives*

Once the ecological class was set at the specialist workshop, the resource water quality objectives could be set to satisfy the water quality needs of the selected class. Setting resource water quality objectives meant unpacking the water quality category, almost reversing the order in which the summary water quality category was determined (**Figure 1.2**).

The following aspects were also considered when setting the resource water quality objectives:

- In many cases where the EMC was better than the PES water quality category, the poor water quality was the result of high TDS concentrations during the winter months and this could be traced back to discharges from known point and/or non-point sources in the catchment. In these cases, reducing the winter TDS concentrations and leaving the summer values unchanged could achieve the desired EMC.
- If the PES category for a specific variable (say pH) was better that the desired EMC and there was no obvious reason why this should be changed, the RQO was set equal to the PES.

1.3.2.7 *Step 7 – Monitoring system design*

This is step was not part of the terms of reference of the BRBS reserve determination.

1.3.3 Lessons learned from the Breede River study

At the conclusion of the Breede River BBM specialist workshop, N. Rossouw, H. Malan, P.-A. Scherman, W.J. Muller, W. Kamish and T. Zokufa, reviewed and captured some of the lessons learned during the Breede River DRIFT and BBM specialist workshops. These water quality requirements and procedures are listed below:

Data preparation before the workshop, and to take to the workshop:

- Starter document: Describe both RC and PES for specified water quality variables per identified reach/IFR site.
- Prepare scatter plots (especially flow vs. water quality variable, and trend of changes in water quality over time).
- Box and whisker plots to examine and describe seasonal distributions.
- Summary statistics (recommend inclusion of 5% and 95%): mean, median, 5%, 25%, 75% and 95%.
- Map **all** the water quality monitoring points and biomonitoring points on a catchment land-use map (including point sources etc.).
- Get all water quality data, biomonitoring data, toxicology data, pollution data (POLMON).
- Set up flow-concentration water quality modelling templates for RC and PES (NB: need hydrological information).
- Set up time-series profiles: if doing stressor-responses (T-Soft), for RC and PES (if data available)(need flow-concentration matrices, and concentration-stress matrices for IFR sites and for specified water quality variables).
- Prepare generic descriptors for water quality variables (required for DRIFT workshop only).

Some key issues:

- Need information on potential reference conditions (time and budget should be allowed to include this information, particularly for the starter document).
- Toxicology data proved to be invaluable.

- Water quality team must comprise more than 1 person. Skills needed: data modeller and biotic links person (minimum) (This is however, dependent on level of determination needed, e.g. rapid vs. intermediate and comprehensive).
- Starter Document should not just address water quality issues at IFR sites, but provide a more comprehensive picture of water quality in the study area.
- Water quality, hydrology and geomorphology (i.e. abiotic template) should be available to biotic specialists **before** they prepare their input to the starter document.
- Need a chapter in the Starter Document on operating rules for the system (from DWAF) to understand the physical limitations and current water resource management practices. It would be useful if a short presentation on Day 1 and DWAF representative be available on Day 1 for questions. Starter Document can then also be used as a check for the hydrology. For water quality, the source water is very important (e.g. whether the dam release is from the top or bottom, spill, inter-basin transfer etc.).
- Create a **project team** specific web site for sharing data, information, documents and maps etc. between project team members. Inform team members by e-mail when the site is updated. An issue to resolve is web-site security.

Some gaps identified:

- Need "integrated" picture of each site (specialist information) **before** start working on each site in detail.
- There is a need for information on current constraints on the system from a DWAF representative.
- DO, temperature and TSS are not measured routinely, but are essential.
- DWAF must develop an approach to deal with toxic substances and pesticides.
- There is a mismatch between **daily** hydrology data and **monthly** water quality data.
- Major limitation is amount and quality of water quality data available.
- Assessing the implications of changed water quality for the biota: this is the result of the changed concentration and duration of the change.

Some comments on DRIFT vs. BBM:

- DRIFT: more critical and rigorous use of water quality data (generic and specific descriptors useful).
- DRIFT: tentative "minimum degradation flows" is useful for the water quality team to focus thinking. Water quality consequences can refine with the information from other specialists.
- DRIFT: RQO should be an explicit product of DRIFT (as in BBM) (some of this is captured but hidden in the descriptors and database) and linked to Classes/flow-reduction scenarios.
- BBM: (in the current workshop) used water quality more analytically due to recent exposure to the DRIFT method (see point 1).
- For water quality, DRIFT is a more useful approach, as water quality consequences are consequences for summer and winter base flows vs. a single month in summer and winter for BBM. The latter may lead to missing peaks (e.g. TDS water high flow peak in the Western Cape) outside of the 2 selected months.
- BBM: confidence in prediction per site while in DRIFT give confidence, severity, data source and direction of change for each water quality variable and element of flow reduction.

At the workshop - DRIFT procedure:

- Respond to "minimum degradation flows" for each water quality variable by making use of generic list to give site-specific consequences.
- As above for each flow reduction scenario and Classes of floods.
- To respond, use: scatter-plots, box-and-whisker plots, etc.

- For each flow reduction and consequence (i.e. how concentration changes), note severity of effects (implies considering impacts on biota, therefore biotic-links specialist important on water quality team), direction of changes, data sources and confidence in the prediction.

At the workshop - BBM procedure:
Some of the key water quality issues identified for the BBM workshop are identified in **Figure 1.3**.

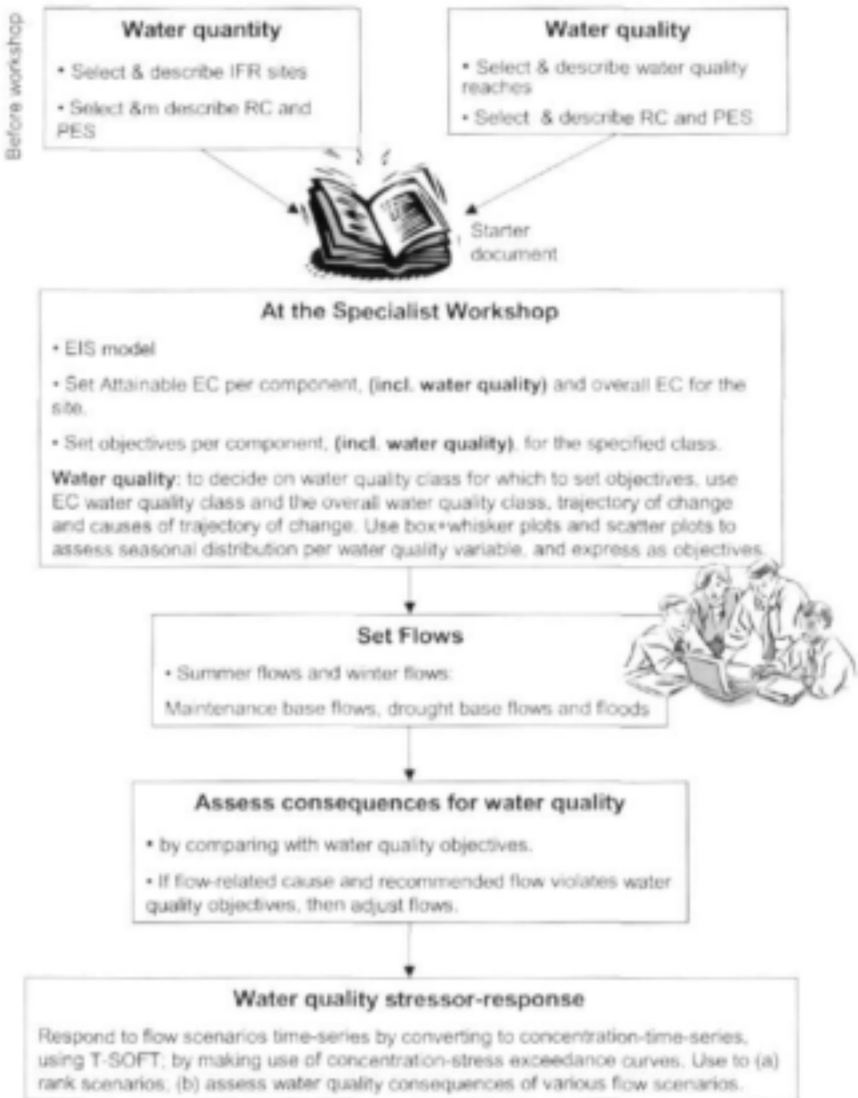


Figure 1.3 Diagram illustrating the activities of the water quality team at a BBM specialist workshop.

CHAPTER 2

INTEGRATION OF WATER QUALITY AND WATER QUANTITY

Heather Malan
Freshwater Research Unit, Department of Zoology, University of Cape Town

2.1 INTRODUCTION

It is recognised within the Reserve process that, in order to maintain a given level of ecological functioning in an aquatic resource, both water quality and water quantity must be considered. Resource quality objectives are set which represent the limits (concentrations of chemical constituents or values of physical variables) that should not be exceeded at given reaches within the water body. It is well known that the concentration of chemical constituents, as well as the values of physical variables may vary significantly with changes in discharge (Malan and Day 2002b). Therefore, in order to ensure that in meeting the Ecological Reserve with regard to *quantity* the water *quality* Reserve is also attained, predictions of the water quality that will result from the proposed flow regime need to be made. The term "*water quality modelling*" is used in this report to describe techniques employed to obtain *quantitative predictions* of what the concentration of chemical constituents in a particular river reach would be under given conditions of discharge (e.g. a proposed IFR flow regime). This represents a separate aspect of the water quality Reserve and is distinct from the assessment of water quality, which is a standard feature of the Building Block Method. In this report, the methods used to integrate water quality and quantity in the Olifants River Ecological Water Requirement Assessment (OREWRA) as well as the Breede River Basin Study is described. Until the above two projects, the question of water quality had not been adequately addressed within the IFR process, since only present, and reference condition, but not *future* water quality were described in a quantitative manner.

2.2 OUTLINE OF THE METHOD FOR INTEGRATING WATER QUALITY AND QUANTITY

Integration of water quality and quantity was undertaken using two basic tools:

i Discharge-concentration modelling
Discharge-concentration (Q-C) modelling was used to estimate the concentration of a particular chemical constituent that would be expected to occur in a river reach at a given discharge.

ii Time-series modelling
Discharge time-series were converted to time-series of concentration of a given water variable using the relationships derived using Q-C modelling. Different discharge scenarios could then be compared using this method with regard to their water quality consequences.

In addition, results from ecotoxicological testing were used to predict the effect that a given concentration of a water quality variable would have on the aquatic macroinvertebrates in the river, and thus the stress that might be exerted on the system could be inferred. Concentration time-series were converted to stress time-series using the concentration-stress relationships derived using ecotoxicological data.

The method is discussed in more detail below. In addition, some of the results from Tulani (IFR site 13), a site on the Olifants River, are presented in section 2.4 in order to make the discussion clearer. The type of information that can be obtained is presented, as well as the limitations and assumptions in the method. Finally, some of the lessons learned from application of the techniques to the Reserve determinations for the Olifants and Breede Rivers are discussed and recommendations for future research made.

2.3 METHODS AND DATA SOURCES USED

2.3.1 Discharge-concentration modelling

Discharge-concentration (Q-C) modelling was used to estimate the concentration of a particular chemical constituent that would be expected to occur in a river reach at a given discharge. This technique was developed as part of a Water Research Commission-funded study (project number K5/956) undertaken by Malan and Day at the Freshwater Research Unit, University of Cape Town. The project is titled "Development of numerical methods for assessing water quality in rivers, with particular reference to the instream flow requirements process". The method is discussed in detail in the final report of the above project (Malan and Day 2002a).

For each IFR site, present day water quality was assessed using data from the nearest DWAF monitoring site. Reference condition (RC) water quality was derived from either historical data, an un-impacted tributary or by inference and expert knowledge using the procedure described in the Resource Directed Measures manual (DWAF 1999). The discharge data (Q) used were either observed monthly discharge from the nearest DWAF gauging site, or simulated discharge used in the water quantity determinations of the IFR, as supplied by the consultant hydrologist. Where possible, the data used for Q-C modelling were the same as those used in the water quality assessment. At a few sites on the Olifants and Breede Rivers however, the purposes of modelling required that different data sets be used. This was usually where the RC water quality data in the assessment had been taken from a reservoir (i.e. non-flowing water). Or in cases where IFR sites had been combined in the water quality assessment, but it was considered to be more useful to model the sites separately. This information was recorded in a tabulated form such as in **Table 2.1**.

Mean monthly discharge values were correlated with median monthly concentration values for each variable (C). *Mean* discharge values were used, because this is the convention in the field of hydrology. *Median* water quality values were used on the other hand, since concentrations can range widely and a single extreme event can alter the mean significantly. It is therefore statistically correct to use median values. Correlation of concentration and discharge values was carried out separately for the reference condition (i.e. least impacted state) as well as for the present state. The water quality constituents examined included TDS (total dissolved solids), other conservative constituents (e.g. fluoride, SO_4^{-2} sulphate), TP (total phosphorus), PO_4^{-2} (ortho-phosphate), TIN (total inorganic nitrogen), NO_3^- and NO_2^- (combined nitrate and nitrite). The choice as to which chemical constituents were modelled depended on the availability of data, as well as the perceived water quality problems at each site. *It is important to note that all predictions of water quality using this method are made under the assumption that the present load of pollution will remain the same.*

The relationship between discharge and concentration was examined separately for both the Reference condition as well as the present ecological state (which may or may not be impacted). Graphs were drawn of concentration versus discharge (an example of TDS concentration versus discharge for Tulani is shown in **Figure 2.1**).

Table 2.1 Example of a table (taken from OREWRA) listing the sources of water quality and discharge data used for Q-C modelling * Indicates that water quality data from a different DWAF monitoring station was used than in the water quality assessment.					
IFR Site	**Water quality data**		**Discharge data**		**Comments**
	RC	**PES**	**RC**	**PES**	
1	*B1H002Q01 and B1H006Q01	*B1H002Q01 and B1H010Q01	VTI model	B1H002 and B1H010	Mass balance calculation used to derive WQ.
2	No water quality data				Not modelled
3	B1H026Q01	B1H015Q01	VTI model	B1H015	
4	PS discharge data (B2H015) unreliable				Not modelled
5	*B3H007Q01 ('92-'98)	B3H001Q01	BKS simulation	BKS simulation	
6	Discharge data suspect				Not modelled
7	*B3H007Q01 ('92-'98)	B5H002Q01	BKS simulation	BKS simulation	No PES water quality data after 1988
8	B7H013Q01 (entire data series)	B7H009Q01 (entire data series)	BKS simulation	BKS simulation	B7H009Q01 is downstream of Steelpoort
9	No PES water quality data				Not modelled
10	B4H007Q01	B4H011Q01	BKS simulation	BKS simulation	
11	Mean of B7H009Q01 ('79-'84) and B7H013Q01	B7H009Q01 (entire data series)	BKS simulation IFR 8 + IFR 10	BKS simulation IFR 8 + IFR 10	Discharge derived by adding IFR 8 + IFR 10

A regression line was then drawn through each of the data points using the trend line function in the spreadsheet package (Microsoft Excel). This provides a choice of five regression equations (linear, exponential, power, logarithmic and polynomial). The "best" fit was chosen by using the relationship (the "model") that gave the highest value of the correlation coefficient R^2. Where there was little difference between R^2 values the simplest relationship i.e. linear or logarithmic was used.

Using the described model, the concentration of each water quality variable was predicted for each month under the prescribed IFR *base flow regime*. These calculations were carried out separately for both maintenance as well as for drought flows. Predictions were made for base flows alone, as opposed to total discharge (which would include floods and any excess flow in the system). *Therefore, in the case of chemical constituents, which decrease in concentration with increased discharge (i.e. exhibit a "dilution effect"), the predictions from Q-C modelling represent the "worst case scenario".* Predictions of concentrations of chemical constituents that *increased* markedly with increasing discharge (for example, TDS in the Breede River, section 2.5.2) were not made. It was felt that in such situations reliable predictions could not be made using the Q-C method. In addition, predictions of concentrations that would be expected to occur during flood events were generally given a lower confidence rating. The reason for this is, that extrapolation to discharges for which very little observed data are available is likely to be unreliable. In addition, it is during periods of low flow rather than floods that water quality is likely to be poor.

By comparing the predicted concentration against the reference condition and the present ecological state value, the extent of current and future deviation from natural could be assessed. In addition, using the predicted concentrations, the predicted assessment category for each month, for each water quality variable under the IFR flow regime could be derived. The criteria used for defining assessment categories for each variable are described in Appendix 2. Using these results it was possible to assess if there was a likelihood that the water quality Reserve might not be attained at that site under the recommended flow regime and current pollution loads.

The degree of confidence in the accuracy of the simulations was recorded for each site. This was assessed by taking the following factors into consideration:

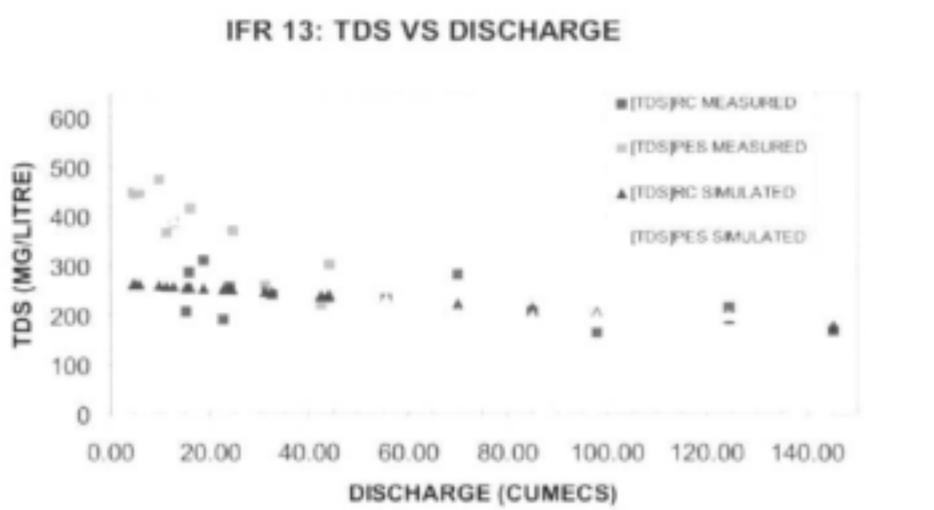
- The completeness of the data-set used to assess reference condition water quality for each water quality variable as well as an assessment of how representative the data was of the natural state.
- The completeness of the data-set used to assess present ecological state water quality for each water quality variable.
- How representative the water quality data (both reference condition and present ecological state) were of the IFR site under consideration. This depended largely on how close the monitoring station used as data source was to the IFR site, as well as if a hydrological feature (e.g. minor tributary, weir etc.) was situated between the IFR site and data source site.
- The reliability and availability of the discharge data. This was assessed by consultation with the hydrologist for the project.
- The accuracy of the water quality simulations. Simulations for which the correlation coefficient between measured and predicted values was less than 0.5 were discarded. It was concluded in such cases that factors other than discharge were influencing the relationship and accurate water quality predictions using Q-C modelling could not be made.

2.3.2 Time-series modelling

The software package T-SOFT was developed at the Institute for Water Research, Rhodes University, by Prof. Hughes and is described in Hughes, Forsyth and Watkins (2000). This software was used to transform time-series of discharge to time-series of concentration at each IFR site. This type of modelling was necessary in order to compare different flow scenarios (that is time-series of discharge) with regard to the effect on water quality. The method is discussed in more detail in Malan and Day (2002a).

The concentration-discharge relationship that had been derived at each site using Q-C modelling was used to set up a transformation matrix such as that shown for TDS at IFR 13 on the Olifants River (**Figure 2.1**). T-SOFT requires a transformation matrix as input, which must consist of exactly 20 discharge values and the corresponding concentration value (or concentration values and corresponding stress level, see later). In order that the profiles obtained be as accurate as possible, the matrix should cover the discharge range that occurs at that site. Therefore discharges were set ranging approximately from the lowest flows that occur at the site to approximately the 1:10 year flood. The Q-C model was used to calculate concentrations for the middle portion of the matrix. Since the Q-C model uses monthly mean discharge values, it does not extend to the full range shown by the time-series. A complete Q-C graph for the site, which utilises all the present state water quality data points (rather than monthly medians) with corresponding discharge was also drawn up. These observed data were used to complete the transformation matrix for very low flows and for floods.

Concentration profiles (time-series) were prepared using different flow time-series in order to compare the flow scenarios with regard to their water quality consequences. In the case of the Olifants River Reserve determination the discharge time-series were generated using the Water Resources Yield Model (WRYM). This was part of a system analysis to assess whether the Ecological Reserve could be supplied from the water available in the current system (Louw and Maré 2001). The concentration time-series could then be manipulated using facilities built into the software. The various proposed flow scenarios were examined and the consequent water quality implications assessed. In the case of the Breede River, the flow scenarios that were compared were generated by the IFR model (Hughes 1999). They included natural, present day, as well as the recommended IFR flow regime. Concentration duration curves were prepared and an examination made of the percentage time that a given chemical constituent (e.g. TDS, fluoride) would be in each assessment category.



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

Figure 2.1 TDS Q-C plot for IFR site 13 on the Olifants River. Also shown is the transformation matrix used to convert discharge to salinity in the preparation of concentration time-series for the same site.

If ecotoxicological data were available, a transformation matrix was set up to convert concentration to corresponding stress levels. The stress index was expressed on a scale of 1-10, where 1 represents very low stress (low concentration) and 10 extreme stress. Different stress matrices were drawn up for different rivers. The stress-concentration matrix was derived by using data obtained from toxicity testing. The toxicity of Na_2SO_4, or NaCl, the most important (by mass) chemical constituent of TDS in the Olifants and Breede systems respectively, was determined. The effect of different concentrations of these salts on the percentage survival of mayfly nymphs exposed in an artificial stream situation was examined. The experimental details of the toxicity testing are described in Appendix 2. These results were used to predict the stress that a given level of TDS would exert on the macroinvertebrates in the river.

In an analogous manner to discharge time-series converted to concentration time-series, a transformation matrix was used to transform the concentration time-series to a time-series of corresponding stress levels. Duration curves (stress profiles) were also derived.

The final step in the process was to consider the concentration (and to a lesser extent stress, see later) profiles arising from the various flow scenarios. The scenarios were then ranked according to their perceived impact on the aquatic biota.

2.4. APPLICATION OF THE MODELLING METHOD

The results for Q-C modelling as well as time-series modelling for total dissolved solids (TDS) at IFR site 13 (Tulani) are presented as an illustration of the type of results that were obtained. This site is on the Olifants River, downstream of the confluence with the Blyde River and upstream of the Phalaborwa barrage and the confluence with the Selati River. The discharge data for the reference condition were simulated data provided by the hydrologists involved in the OREWRA project. Present state discharge data were derived from data collected by DWAF at the gauge station B7H007 (Olifants River at Oxford). The water quality data for the reference condition for this site were derived from the mean of the Blyde River and the Olifants River at Oxford (B6H004Q01 data set 1992-1997 and B7H007Q01 data set 1992-1997). Present day water quality was taken from Oxford (B7H007Q01 data set 1992-1997).

2.4.1 Q-C modelling

Figure 2.1 shows the Q-C plot for TDS at this site, as well as, the matrix that was used to convert discharge to concentration. A fairly typical Q-C relationship is exhibited, in that the *present state concentration of TDS is considerably higher than the reference condition levels at* low flows, and that TDS shows a negative trend with increasing discharge. At discharges of approximately 70 m^3/s and higher, there was little difference between the reference condition and present state concentrations of TDS. As a result, in the transformation matrix, discharges greater than 70 m^3/s were assigned a concentration value of 200 mg/L TDS. The results from Q-C modelling for TDS, as well as other the other water quality variables that gave acceptable simulations at this site are summarized in **Table 2.2**. This table records, for each water quality variable, the correlation between concentration and *increasing* discharge. As is to be expected from the dilution effect at high discharges, TDS and SO_4 showed a slight to marked decrease with increasing discharge at all sites. Ortho-phosphate concentrations, on the other hand, was found to increase very slightly with discharge at IFR site 13 at low flows, and at higher flows was constant. A perusal of the individual sampling data however, indicated that over the range of discharges exhibited at the site, median ortho-phosphate concentration was 0.02 mg/L (S.D. = 0.019). Thus, although ortho-phosphate concentration increased with discharge at this site this trend was very slight, and Q-C modelling is not recommended in such situations. As a consequence, it could be predicted with a fair degree of confidence that this constituent would not be expected to increase to a C assessment category for ortho-phosphate, even during floods.

An estimate of the "goodness of fit" between the measured and simulated present ecological state concentration values is given by the correlation coefficient (shown in column 3 of **Table**

2.2). In general, higher correlation coefficients were obtained for conservative variables (TDS, sulphate). There was a poor correlation between discharge and nutrient levels (non-conservative variables), especially in the case of the nitrogen-containing constituents. For this reason, results for nutrients are presented only where satisfactory simulations were obtained.

The median assessment category and/or concentration for the reference condition and present ecological state as well as the ecological Reserve class (ERC; formerly termed "Ecological Management Class", Louw, D. IWR Environmental, *pers. comm.*) are also given in **Table 2.2.**

Column 14 of **Table 2.2** gives the concentrations for each variable that are predicted to occur under maintenance base flow as well as under drought base flow. The corresponding assessment category for each water quality constituent is given in parenthesis in the same column. Assessment categories have not been defined for sulphate in the Reserve methodology, and consequently only predicted concentrations are given for this chemical constituent. Predicted concentrations for maintenance, and drought base flow, but not for high discharges, are reported, since as stated previously these tended to be inaccurate.

Also given in the table is the number of months of the year that water quality will be in each assessment category, the worst water quality (i.e. highest concentration), as well as when (within which months) this is likely to occur. Finally, whether the water quality Reserve will be attained with respect to the proposed ecological Reserve category for that variable is reported. From **Table 2.2**, it can be seen that there is a danger that the concentration limits set in terms of the water quality Reserve will be exceeded for TDS during winter when the recommended drought IFR flow regime is implemented.

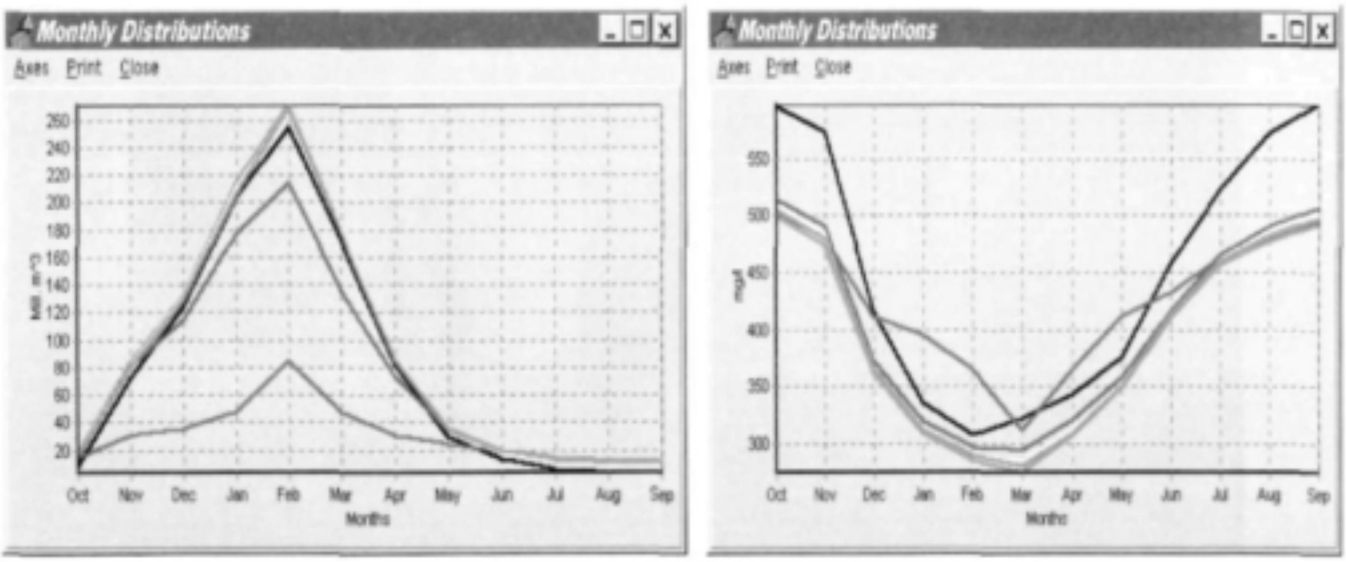
Table 2.2 Summary of the results obtained using Q-C modelling for all water quality variables modelled at Tulani (IFR 13, Olifants River). The trend in concentration with increasing discharge is shown as well as the correlation coefficient (R^2). The ecological Reserve class (ERC) for each variable is given as well as the median concentrations and assessment categories in parenthesis for the RC and PES. The predicted median concentration and assessment category under the IFR regime (maintenance and drought base flow) is given in column eight and in the adjacent columns, the number of months that each water quality variable will be in each assessment category. The predicted highest median monthly concentration is presented as well as the month in which this is likely to occur. In the final column whether the water quality Reserve (as determined by the ERC) will be attained in all months is recorded. All concentrations expressed as mg/L.
(Q=discharge, C=concentration, RC=reference condition, PES=present ecological state, ERC=ecological Reserve class)

Q-C relationship			ERC	Median RC conc	Median PES conc. and (cat.)	IFR regime	Predicted Median conc. and (cat.)	Predicted no. of months in each category					Predicted highest conc. and (cat.)	Month	Reserve ?
Variable	Trend with flow	R^2						B	C	D	E	F			
TDS	Decrease	0.89	C	241	342 (C)	**1. MAINT. BASE FLOWS** Drought base flows	404 (C) 521 (D)		12 6	 6			438 (C) 558 (D)	Sep Sep/Oct	Yes No
SO_4	Decrease	0.84	-	No data	33	**2. MAINT. BASE FLOWS** Drought base flows	37 50						40 55	Sep/Oct Sep/Oct	- -
PO_4	Very slight increase	0.89	D	0.02	0.019 (B)	**3. MAINT. BASE FLOWS** Drought base flows	0.018 (B) 0.015 (B)	12 12					0.021 (B) 0.016	Jan/Apr Jan/Apr	Yes Yes

2.4.2 Time-series modelling

Using the Q-C matrix, discharge time-series were transformed to time-series of concentration. Salinity (TDS) was considered to be a key water quality variable and therefore time-series modelling was undertaken for all key sites on the Olifants River, including IFR 13. As mentioned previously, in the case of the Olifants River Reserve determination the discharge time-series were generated using the Water Resources Yield Model (WRYM). The period over which the discharge time-series (and consequently that of concentration and salinity-induced stress, see later) extended, was from 1920-1987. The same Q-C matrix was used for the entire time period. **Figure 2.2** shows the mean monthly discharge in million cubic meters (a) and mean monthly salinity in mg/L (b) for IFR 13 as predicted under the different flow scenarios using TSOFT. The various flow scenarios that were compared are given in the legend at the foot of the figure. In brief, the scenarios ranged from present day discharge ("No-IFR") to scenarios in which drought was allowed to occur in the system 10% or 20% of the time (named "10% drought" and "20% drought" respectively). The flow scenarios considered in the scenario modelling phase of the OREWRA are outlined in Malan (2001), and a comprehensive description is given in Louw and Maré (2001).

a b



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

Figure 2.2 Mean monthly discharge in million cubic meters (a) and mean monthly salinity in mg/L (b) for IFR 13 as predicted under the different flow scenarios using TSOFT. Values calculated using the entire flow time-series (1920-1987). Flow scenarios given below the figures.

Figure 2.3 shows the concentration duration curves (or more correctly exceedence curves, Hughes, D., IWR, Rhodes University *pers. comm.*), for all the flow scenarios examined at site 13. The "No-IFR" scenario (present day scenario) appears to represent the worst water quality consequences since TDS reaches the highest concentrations under this scenario. Indeed under this scenario TDS would be in a D category (i.e. higher than 520 mg/L) 42% of the time, whereas for all the other scenarios, TDS will be above 520mg/L for only 5% of the time. **Figure 2.3** also shows that approximately 22% of the time, all scenarios except "IFR-rule" (this represents just the IFR as determined by the specialists, with no additional discharge) would be in a B category, or better for TDS (less than 300 mg/L). Under the "IFR-rule" flow scenario, TDS would be less than this threshold only 4% of the time. Thus with regard to the water quality scenarios, for site IFR 13, the "No-IFR" flow scenario was ranked worst, followed by the "IFR-rule", and the other four flow scenarios equally in third place.

a b

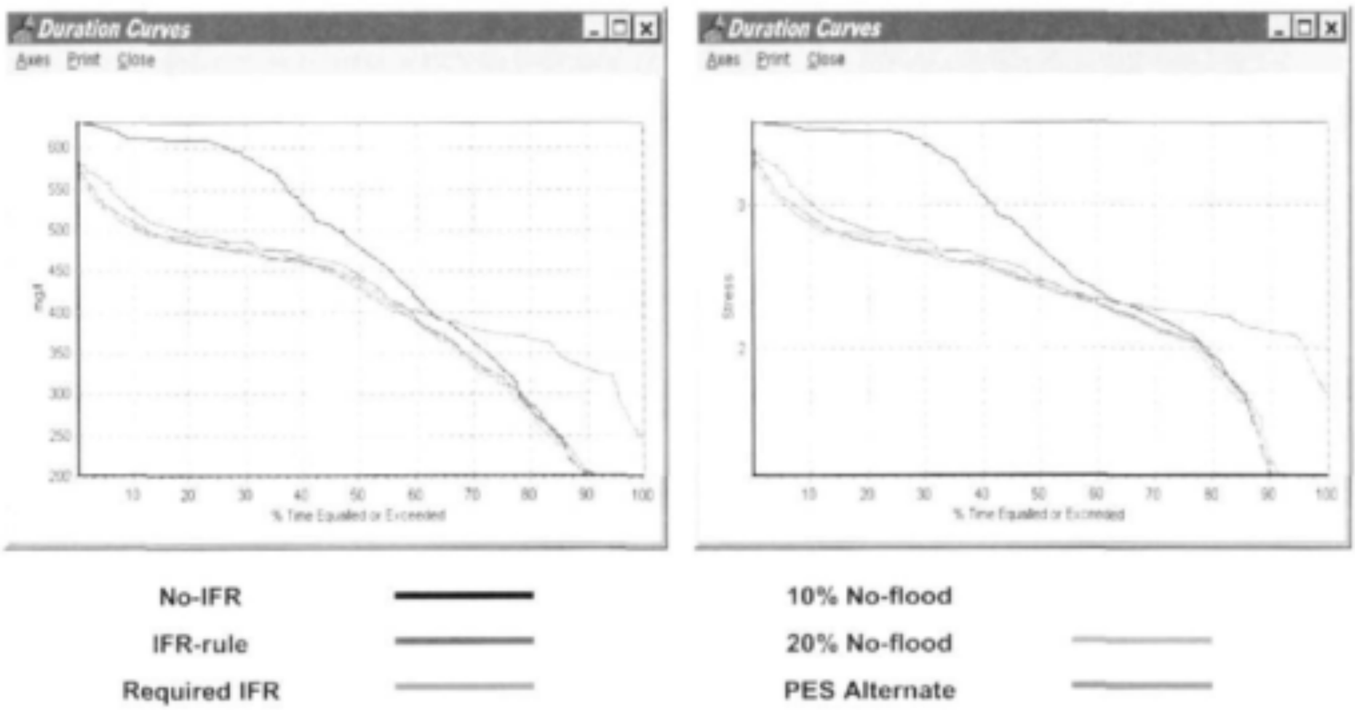


Figure 2.3 Salinity in mg/L (a) and stress (b) duration curves for IFR 13 as predicted under the different flow scenarios. Stress derived using a scale of 0-10 as described in the text.

2.4.3 Stress profiles

Stress time-series were prepared using salinity-stress relationships derived from ecotoxicological parameters. The stress index was expressed on a scale of 1-10, where 1 represents very low stress (low salinity) and 10 extreme salinity-induced stress. **Table 2.3** shows the ERC, TDS and sulphate concentration, and the corresponding value of the stress index. Also indicated are the toxicity descriptor (given in terms of confidence level for the various Lethal Concentration values) as well as the likely biotic response. The derivation of this table is described fully elsewhere (Appendix 2).

The stress matrix that was used for all sites on the Olifants River is shown in **Table 2.4**. Stress profiles were analysed for frequency and duration of the highest stress levels as in the case of salinity profiles. The salinity-induced stress duration curves for IFR 13 are shown in **Figure 2.3**. Similar results were obtained to the salinity profiles. Consequently, the ranking of the stress profiles yielded identical results to those obtained from considering salinity.

Table 2.3 A comparison of the ecological Reserve class (ERC), concentration of TDS* and sulphate (SO_4), stress index value, toxicity descriptor and biotic response descriptor for the Olifants River.
* Note that TDS was calculated by multiplying the EC of the Na_2SO_4 solution by 6.5 (CL=confidence level, LC=lethal concentration)

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

Table 2.4 Transformation matrix used to convert TDS concentration to associated stress levels for aquatic invertebrates in the Olifants River.

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

2.5 POINTS TO NOTE AND LESSONS LEARNED

2.5.1 Olifants River (OREWRA)

- The Q-C modelling method can also be used to estimate the discharge that would be required to attain a given assessment category with regard to a particular water quality variable under the current pollution loading. For example the discharge required to attain a "D" category, which is the lowest Ecological Reserve category that is considered to be sustainable (DWAF 1999).

- The Q-C method can be used for TDS, sulphate and other conservative constituents. Nutrients can also be modelled in this manner, although they often exhibit considerable scatter when concentration is plotted against discharge. This is likely to be a consequence of the various processes that influence instream concentrations (e.g. microbial conversion between chemical forms, adsorption/desorption from sediment particles, uptake by the biota).

- Because of the inherent inaccuracies in extrapolating to discharges for which no observed TDS values are available, the Q-C transformation matrix is set up to cover only up to the 1:10 year flood. Floods equal to, or larger than this, are all set at the minimum concentration value. The fact that the concentration of most variables usually decreases as discharge is increased is considered to be justification for this approximation. Thus, it is during periods of low flow, rather than during floods that water quality problems are most likely to occur.

- The Q-C plot for the present ecological state is used to prepare the transformation matrix and hence concentration profiles of different flow scenarios. If a concentration profile that is representative of the natural state is required however, the Q-C relationship derived from reference condition water quality and discharge data is used to set up the transformation matrix.

- TDS time-series were produced for the key sites on the two rivers. The reasons for focusing on this chemical constituent were as follows. Firstly, TDS was considered by the project team to be one of the most important water quality variables and to be important in determining the overall present water quality status of a water resource. This is supported by the fact that TDS is classed as a system variable in South African Water Quality Guidelines (DWAF 1996). Secondly, the UCEWQ-IWR (Rhodes University) team had already carried out preliminary salinity toxicity testing in other catchments and an experimental protocol for this variable had been developed. Other water quality variables can also be modelled in the same manner however. For example, fluoride profiles were prepared for the IFR site 15 (at Mamba) on the Olifants River and combined nitrate and nitrite profiles for the Moolenaars River (part of the Breede River system). These results are presented in Malan and Day (2002a). Taking into account the limitations and approximations in the modelling methods (section 2.6), any water quality variable that is suitable for Q-C modelling and gives good simulations can be converted to concentration time-series.

- Use is made of monthly median values of concentration and monthly average discharge through which a trend-line is fitted. An alternative approach is to use all concentration data points. Comparisons have been made of the trend patterns, as well as the correlation coefficients obtained between measured and simulated data using either all data points or monthly data. It was found that using monthly data Q-C trends could be more clearly discerned, than if all data points were used. Plots of all data points were useful however to confirm the trend found using monthly values, to examine the variability in response to discharge, as well as, to view the Q-C relationship over the full range of discharges found at a given site. The additional effort required in preparing Q-C plots of all data points is necessary if time-series modelling is to be carried out.

2.5.2 Breede River (BRBS)

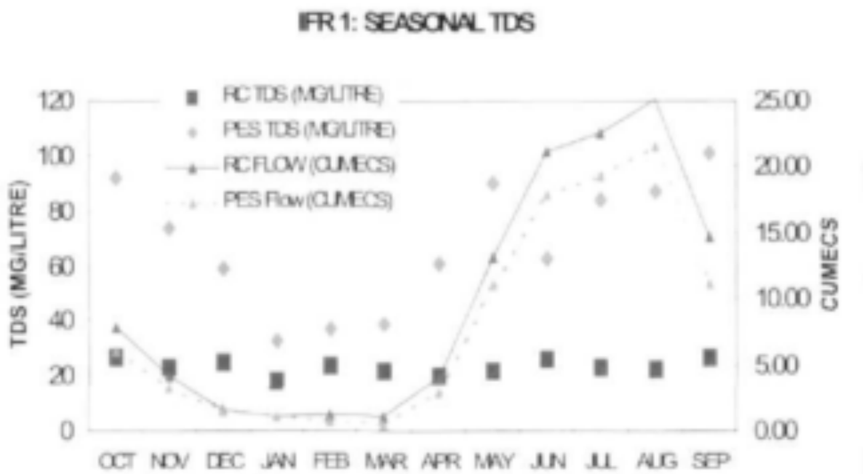
Discharge-concentration modelling was undertaken for all key water variables at the six designated IFR sites in this system, with varying success (see below). Of these sites only two were considered to be suitable for time-series modelling. Despite the limited time-series modelling, useful insight was gained in considering the suitability of various sites for the modelling techniques, and this is discussed below.

- Two of the sites, namely IFR 1 and 3 showed increased concentrations of TDS during the onset of winter rains. Extensive irrigation of agricultural lands takes place in the catchment during the long, hot, summer. This results in a build-up of salts in the surrounding soils, which are flushed into the river during the winter rains. The seasonal trend of TDS at IFR site 1 (Mooiplaas) as well as the Q-C plot is shown in **Figure 2.4**. The elevated instream salinity levels during the winter months in the present state compared to the natural condition can be clearly seen. If concentration is plotted against discharge, it can be seen that at discharges less than approximately 15 m^3s^{-1}, TDS increases with discharge (wash-off effect), but above this value, TDS decreases with increasing discharge (dilution effect). As noted in section 2.3, Q-C modelling cannot be used to predict water quality in situations where concentration markedly increases with discharge. The pollution load in such cases is largely due to diffuse pollution, rather than instream sources (for example from the discharge of effluents). Predicted concentrations under a given flow regime will depend heavily on not only discharge, but also the manner in which discharge is manipulated. Thus Q-C modelling is simplistic in such situations and confident predictions cannot be made. This aspect is discussed more comprehensively in the final report in which the Q-C modelling method is detailed (Malan and Day 2002a).

- Another two sites, namely IFR 2 on the Moolenaars River and IFR 6 on the Baviaans River (part of the Riviersonderend system) were relatively unimpacted with regard to TDS. The present state levels of this variable were largely the same as the reference condition levels. In addition, the Q-C plots show very little variation in TDS concentration with discharge. As a result, all flow scenarios resulted in very similar salinity duration curves, and were all within the limits required to attain the water quality reserve. Time-series modelling in such cases does not yield very useful results.

- Q-C plots for IFR site 5 (Riviersonderend) could not be modelled accurately because no water quality monitoring station was close by. Furthermore, attempts were made to model the water quality using data either from the closest upstream and or downstream station, or using a mean concentration value derived from the two sites. These attempts were not successful since a clear trend between discharge and the concentration of the chemical constituents examined (TDS, sodium, nitrate and nitrite, ortho-phosphate) was not obtained. It is likely that factors, other than discharge are influencing water quality at this site.

a



b

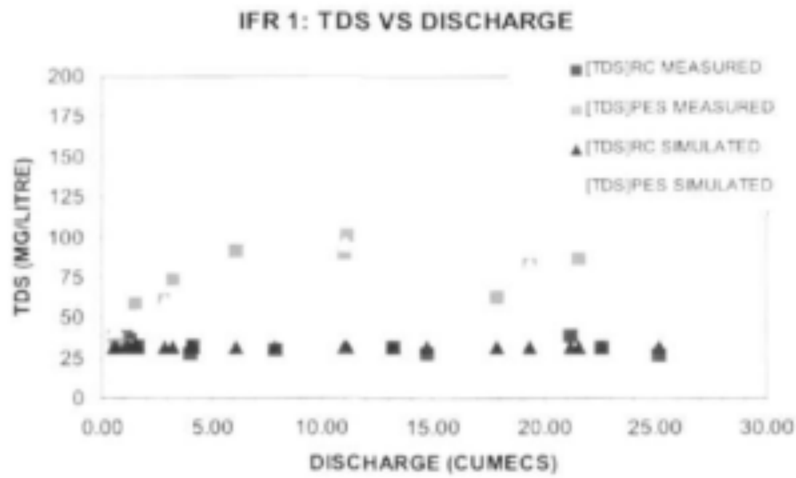


Figure 2.4 TDS at IFR 1 (Mooiplaas) on the Breede River. (**a**) Seasonal plot of discharge and TDS concentration for the reference condition (RC) and present day (PES). (**b**) Q-C plot of TDS versus discharge. Note the wash-off effect in the PES during the winter months and that TDS concentration increases with increased discharge at low flows and then shows a dilution effect at higher discharges.

- IFR site 4 was suitable for Q-C and time-series modelling of TDS, since this variable showed a marked decrease in concentration with increased discharge. TDS duration curves were generated for the RC and PES, however the discharge time-series for the flow regime recommended at the specialist workshop was not available at the time of writing this report (Louw, D., IWR-Environmental, *pers. comm.*).

2.6 ASSUMPTIONS AND LIMITATIONS IN THE METHOD

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

- Unless there are measured water quality data for very low discharges and very high discharges, extrapolation to these flows (required when converting discharge- to salinity time-series) is likely to be inaccurate.

- The modelling method is severely constrained by the availability of water quality and discharge data. It should perhaps be noted that this is not unique to Q-C modelling and that the absence of appropriate data makes the use of any water quality modelling method difficult, if not impossible. The degree of confidence in the predictions is influenced by the accuracy and completeness of the data used, as well as, the proximity of the water quality monitoring stations to the IFR site in question. The greater the distance between the site and the monitoring station, the less it can be assumed that the water quality predictions are representative of that IFR site. In addition, if a major tributary or other hydrological feature (e.g. a weir) is between the monitoring station and the IFR site, confidence in the results is also diminished.

- From the above, it can be seen that expert judgment is important in the use of this method. In addition, derivation of the monthly median water quality concentrations, particularly for the reference condition is not always straight forward. Frequently there were no pre-impact data available. Although guidelines are given in the RDM manual (DWAF 1999) for such situations as this, experience of the water quality to be expected in different ecoregions, is essential.

- It is assumed in the method that, if discharge is altered, apart from the concentration or value of the water quality variable under concern, all other parameters (e.g. the pollution load) will remain constant. In practice, if discharge is altered drastically it is likely that the source of the water will be altered (for example by means of impoundment of tributaries). In addition, changes in operation of upstream impoundments may also be involved. In other words, changes in management scenarios have not been incorporated into the method. Thus the predictions made using this modelling method are valid only if the system is operated in the same manner as used to derive the Q-C relationships.

- Discharge-concentration modelling is not suitable for simulation of DO or temperature, and may sometimes be inaccurate for simulation of non-conservative constituents.

- The method makes use of the standard "trendline" regression functions available in commercial spreadsheet packages. Considering the inaccuracy inherent in the modelling method, it is considered that this simplification will not detract from the overall accuracy of the results.

As explained above, due to the many factors that can influence Q-C relationships, the predictions made using Q-C modelling are not particularly accurate. Using these relationships to obtain time-series of concentrations therefore represents a further approximation. It is felt that the specific results of this type of modelling (e.g. the percentage time that a chemical constituent will be within a given assessment category) will therefore not be particularly precise. Nonetheless, the method is useful for comparing and ranking different flow scenarios with regard to their water quality consequences (and resultant stress on the biota). Since the results from Q-C modelling are used to prepare the concentration profiles, any limitations that apply to Q-C modelling are also likely to apply to time-series modelling. In addition to those mentioned above, various other simplifications of the real situation are made when applying time-series modelling. These are discussed below. A more complete description of the assumptions made and the limitations in using ecotoxicological parameters to infer water quality-induced stress for the aquatic biota is given elsewhere (Appendix 2).

- If discharge time-series spanning a considerable period are used, as was the case in OREWRA and Breede River Basin study, it is assumed that the same Q-C relationship was in place the entire time. This is unlikely to be true, as changes in the pollution load carried in the river have undoubtedly occurred in both catchments over the last few decades. The same Q -C relationship was used for all flow scenarios however, and it was therefore felt that the method was satisfactory for the purpose of comparing and ranking them in terms of potential impacts on water quality.

- In the case of the BRB study, daily discharge data was transformed to daily concentration using a Q-C relationship derived using monthly data. This is likely to introduce inaccuracies.

- Because of limited time and financial resources, ecotoxicological testing was carried out using only Na_2SO_4 (in the case of the Olifants River) and NaCl (in the case of the Breede River) and these results were extrapolated to make predictions of the effect of TDS. The water quality constituent TDS is a combination of many chemical compounds and not just Na_2SO_4 or NaCl. Due to the occurrence of extensive mining in the Olifants catchment, sulphate is the most important contributor to TDS (salinity). In a similar manner NaCl is the most appropriate surrogate for TDS in the Breede River. It was therefore considered that this is a valid approximation.

- Because of limited time and financial resources, ecotoxicological testing was carried out using only one genus (three species) of macroinvertebrates, and these results were used to derive the implications of changed salinity conditions for the entire macroinvertebrate assemblage. It is very likely that some species may well be more sensitive to the surrogate TDS salts that were selected and similarly that others are likely to be less sensitive.

- Ecotoxicological testing was carried out under artificial stream conditions in which environmental parameters were controlled. It is difficult to extrapolate the levels of stress experienced under artificial conditions to those in the field. In addition, In the case of the Olifants River, the experiments were carried out using filtered Grahamstown tap water (rather than river water as in the case of the BRB study – Appendix 2) to which Na_2SO_4 had been added. Due to differences in chemical speciation of Na_2SO_4 in filtered tap water compared to river water, the toxicity of these chemical compounds in the field may not be the same as in the laboratory.

- Lethal concentration parameters (i.e. LC_1, LC_5), were used to infer chronic effects on invertebrate populations.

- The stress experienced by an aquatic organism is the cumulative effect of salinity as well as possible effects caused by other water quality variables, hydraulic habitat, biotic effects such as predation etc. Ignoring the other stressors and considering just salinity is therefore a gross simplification of the actual situation. Nevertheless, if employed as a tool for comparative purposes, it is a useful approach.

2.7 INFORMATION THAT CAN BE GAINED FROM WATER QUALITY MODELLING

Using Q-C and time-series modelling, depending on the availability and reliability of data at each IFR site, the following information could be obtained:

- Discharge-concentration relationships derived for the key water quality variables.
- Estimates could be made as to how many months of the year, under the proposed IFR *base flow*, the water quality Reserve would be attained with regard to the various water quality constituents (TDS, nutrients) as well as the likely assessment category (A, B, C etc.).
- In what month the worst water quality would be likely to occur and what concentrations could be expected.
- What discharges, in the absence of pollution control, would be required to dilute pollutants in order to attain the water quality Reserve.
- In the case of "natural water quality problems" (see section 2.9), what discharges would be needed to dilute the chemical constituents in order to attain the water quality Reserve. Note though, that in neither of the Reserve assessments, were cases of "natural water quality problems" considered to occur.
- Approximate estimates of the duration of periods of poor water quality under different flow scenarios.
- Ranking of flow scenarios with regard to the water quality consequences.
- Ranking of flow scenarios with regard to the likely impact on aquatic macroinvertebrates.

2.8 WHEN SIMPLE Q-C AND TIME-SERIES MODELLING SHOULD NOT BE USED

Predictions of water quality using Q-C and time-series modelling were not considered to be valid in the following situation:

1. If the available present ecological state water quality data did not satisfy the requirements as laid out in the Resource Directed Measures manual (DWAF 1999). In other words, a minimum of 60 water quality data points was required covering the entire hydrological year etc. In the absence of reference condition data, predictions of future water quality could still be made, but obviously the extent of deviation from natural could not be assessed.

2. If the nearest water quality monitoring station to the IFR site was in a different water quality reach from the site. This may be either because of the distance between the two was too great, or because of a significant hydrological feature, e.g. large tributary, point-source of pollution or impoundment is situated between the monitoring station and the IFR site. In other words if no data that were considered to be representative of the WQ at the IFR site were available, modelling could not be carried out.

3. If accurate present day discharge data for the water quality reach under consideration were not available.

4. If the correlation coefficient between measured and predicted values was less than 0.5, the simulations were discarded. It was concluded in such cases that factors other than discharge were influencing the relationship and water quality predictions were not made.

5. If the concentration of the water quality variable that showed a marked *increase* in concentration with increasing discharge. Such a situation was sometimes encountered for nitrates in the Olifants and Breede River systems, as well as TDS in the latter.

2.9 INTEGRATION OF WATER QUALITY MODELLING INTO THE RESERVE DETERMINATION PROCESS

Whilst it is generally acknowledged that inclusion of water quality modelling within the IFR process is important, exactly how and where it should fit in, is not clear. An important principle within the Building Block Methodology is that the environmental flows recommended should be those that satisfy the requirements of the aquatic biota with regard to hydraulic habitat. Flows should not be recommended because they are required to dilute pollutants to a level acceptable to the biota. If they are, it should be stated clearly that this is a management decision and that the "extra" water required for dilution is not part of the ecological Reserve. Yet it is a fact of life that many South African rivers, including the Olifants and Breede are impacted due to high levels of pollutants. Thus, if point and diffuse sources of pollution are not addressed and implementation of IFR regimes leads to a reduction in base flow, deterioration of water quality is a likely result. It was therefore decided that although IFR flows would not be adjusted to compensate for poor water quality, the predicted *consequences* of the recommended flow regime (in the absence of pollution control) should be quantified as far as is possible using water quality modelling.

In the context of IFR determinations it is important to distinguish between "natural" and "anthropogenic" water quality problems. Natural water quality problems would include instances where due to the geology of the surrounding catchment, water draining that region is naturally saline resulting in elevated concentrations in the river. The aquatic biota in such reaches are adapted to high salinity levels. Implementation of an IFR flow regime however, for which the maintenance and drought flows represent a reduced discharge volume may well result in unacceptably high levels of TDS. In such a case, incorporation of dilution flows into the flow requirement is necessary. It should be emphasised however that no cases of poor *natural* water quality were found to occur in the OREWRA or BRB study.

2.10 RECOMMENDATIONS FOR FUTURE RESEARCH

The following aspects of integration of water quantity and quality (amongst others) require further investigation:

- Scenario modelling appears to be frequently undertaken, in which no consideration is given to the manner in which the system will be operated. In other words no information is available as to how the relative sources of water would change between flow scenarios. Thus the actual effects on water quality cannot be determined, and all predictions are made on the premise that the source of water for all flow scenarios will be the same. Although this is adequate if only the water quantity Reserve is to be determined, for assessing the water quality Reserve, it is simplistic. Qualitative statements such as "the discharge from tributary X, which carries good quality water should be maintained in order to ensure that salinity at site Z downstream of the confluence is not compromised" were included in the scenario report for the OREWRA. This was intended to encompass likely effects on pollutant loads resulting from changes in the source (tributaries, impoundments etc.) of water. Although it is likely to increase the complexity of the scenario phase, there is an urgent need to include more realistic scenarios that consider the water quality of the various water sources. It should be possible using a combination of mass-balance modelling and time-series modelling to obtain more accurate predictions of water quality and thus ranking of flow scenarios.

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

- More attention should also be paid to the relationship between water quality and biotic response. Although stress profiles were developed they were not particularly useful in these studies because the response between concentration and stress was roughly linear in concentration range under consideration. Therefore Q-TDS and TDS-Stress duration curves were very similar and the same ranking of flow scenarios was obtained with both. It is conceivable however, that other water variables would have very different concentration-stress relationships. For example a toxic constituent may have little effect on an aquatic organism up to a critical threshold concentration, where after, even slight increases in concentration may have a drastic effect on percentage mortality. Transformation of concentration profiles to stress profiles in such cases may yield interesting results. It may well be that ranking of flow scenarios according to the perceived stress on aquatic organisms yields different results to ranking according to consideration of concentration maxima alone.

- Attention still needs to be given to integration of the different water quality variables to obtain the actual stress that would be experienced by the biota. Due to the additive, synergistic and antagonistic effects between individual chemical constituents and physical attributes of water bodies, this is not a straightforward process. It is probably best undertaken by using data from biomonitoring (such as SASS scores for example) and linking these results to water quality data. Using such data, coarse predictions of the likely effect of water quality on aquatic biota can be made. As part of the WRC project titled "Development of numerical methods for assessing water quality in rivers, with particular reference to the instream flow requirements process", a 'Biotic Protocol', has been developed. This outlines a method for assessing the implications of predicted water quality for the biota. The protocol has not been refined fully and in particular needs to be applied during an Instream Flow Requirement (IFR) project. The method is discussed in detail in the final report of the above project (Malan and Day 2002a).

2.11 CONCLUSION

Several tools have been developed that can be used to integrate water quality and quantity within the Reserve determination process. They are complementary and given the assumptions and limitations in the methods, can yield useful information. Discharge-concentration modelling for instance can be used to predict the concentration of chemical constituent that could be expected at a given discharge. Because of the inherent extrapolations and simplifications in the time-series modelling method it is not particularly accurate for predicting the exact proportion of time that a water quality variable is likely to be in each assessment class. It is however a very useful tool for comparing the water quality that will result from different flow time-series and for ranking such scenarios. Similarly, stress time-series can also be potentially useful in ranking the implications of different scenarios for the aquatic biota.

In conclusion, prescribing environmental flows for a river, especially systems that are large and complex such as the Olifants and Breede, is not an easy task. Methods for determining the quantity and timing of flows required have evolved in South Africa over several years. The field of water quality lags considerably behind that of quantity and this is the first major attempt to integrate the two fields within the Reserve determination process. Some steps have been taken in making quantitative predictions of expected water quality under different flow regimes. It is likely however that much development of the method is still required. In particular, considerable feedback is required from water resource managers in order to make the method as useful and relevant as possible. Despite the approximations and assumptions inherent in the approach, these methods represent an attempt to predict the water quality that will be experienced under a proposed IFR regime, as well as to assess the implications for the aquatic biota. As such it represents considerable progress on work that has gone before.

CHAPTER 3

CONCEPT OF RISK

Sebastian Jooste
Resource Quality Services, Department of Water Affairs and Forestry, Pretoria

Notes on an adaptive approach to the hazard-based derivation of water quality EcoSpecs for the Ecological Reserve in Fresh Water Resources

> *"I often say that when you can measure what you are speaking about, and express it in numbers, you know something about it; but when you cannot measure it, when you cannot express it in numbers, your knowledge is of a meagre and unsatisfactory kind; it may be the beginning of knowledge, but you have scarcely in your thoughts advanced to the state of Science, whatever the matter may be."*
>
> **- Lord Kelvin (William Thompson) 1824-1907**

EcoSpecs:	The set of quantitative expressions of stressor levels that characterises the ecological component of the Reserve.
Hazard:	Having the potential to cause a specific type and level of harm.

3.1 PHILOSOPHY AND RATIONALE

The need exists to derive descriptors of water quality that serves as an expression of the ecological component of the Reserve (referred to here in a shorthand form as water quality EcoSpecs). These EcoSpecs serve as an input to the resource quality objectives, which will eventually drive water resource management. It is therefore important to have a scientifically tenable procedure to derive these EcoSpecs.

The core of an ecological water quality Reserve is a knowledge base. The knowledge base needs to contain some information on how the system is expected to respond to an applied stressor. This knowledge is captured in a stressor response relationship. EcoSpecs are derived from the knowledge base by selecting benchmark levels of response and assessing the stressor levels that correspond to the selected benchmarks.

It is accepted that EcoSpecs derive from different disciplines that are in various stages of scientific maturity. Therefore, the method rests on two legs: 1) generic hazard assessment formulated as a stressor response relationship, and 2) site-specific adaptation or bias applied to the generic hazard assessment to produce a site-specific hazard assessment.

1) A generic hazard assessment captures the generic knowledge base that primarily leads to identifying ecological stressors. The generic knowledge base is used to derive a generic stressor response relationship (GSRR). The GSRR is a hypothesis about the relationship between stressor levels and a chosen type and level of effect. This hypothesis summarises the most objective knowledge on how hazard conditions arise in the aquatic environment. The rationale is that, unless any other data exist to adapt the GSRR, it will serve as the basis for EcoSpecs. The process by which the

GSRR is constructed from the knowledge base could introduce a degree of precaution.

2) Where sufficient evidence exists that the generic knowledge base in not suitable to a specific site or situation, there needs to be a procedure to adapt the GSRR to a site specific stressor response relationship (SSRR) that has greater scientific defensibility. The bias introduced in this way is not scientific bias but rather situational bias.

3.2 GENERIC HAZARD ASSESSMENT

It is assumed that:

- In a system that has to be utilised, it is not feasible to protect all aquatic species at all times, but the aim is to protect most of the species most of the time. A hazard would be considered to exist if this criterion is not met. It is suggested that "most of the time" should be interpreted as 95% of the time and "most of the species" should be interpreted as "all but the projected most sensitive species" (or as alternative if sufficient data exists, "95% of the species").

- The term "protect" is interpreted to mean that a **population** of the theoretical "most sensitive species" (or "5th percentile organism") has at least an even chance for survival. This species as well as those more sensitive (or less tolerant) to the stressor may not actually exist, but they are postulated on a statistical projection basis.

- The hazard resulting from the stressor is characterised in terms of the likelihood of observing a hazard condition. It is postulated that at least two benchmarks should be used: one representing the knowledge of stressor level where no hazard is expected and the other the hazard is practically certain to exist.

- The most scientifically tenable assessment of hazard is based on the most objective information possible within the definition of the stressor. The "most objective" is taken to mean then the "least subjective" information in the sense of containing the least site-specific bias.

A **stressor-response relationship** is a hypothesised relationship between the occurrence of any given level of stressor and the likelihood of "non-survival" of a small fraction of the aquatic species derived from the most objective knowledge base available.

3.2.1 Site-specific adaptation or bias

Several conditions may arise that require site-specific adaptation of the stressor response relationship such as:

- Special value of the site/water body (e.g. a fragile pristine site).
- Sufficient evidence exists that the biota at a site are either more or less tolerant to the stressor than expected (e.g. a highly modified site where only a few hardy species occur).
- The stressor presents itself in a manner which could not / was not accounted for in the compilation of the data base (e.g. the water chemistry is uncommon).

A site-specific stressor response relationship (SSRR) is obtained by adapting a generic stressor response relationship (GSRR) based on site-specific response observation.

What constitutes "sufficient evidence", may differ from discipline to discipline. It is suggested that evidence should be weighted with data derived from scientific observation and be given the greatest weight while anecdotal evidence be given the least weight.

Procedures for updating the GSRR would be stressor specific, depending on the type of site-specific information available.

3.2.2 Definition of stressor

A stressor is here defined in terms of any situation to which the ecosystem may be exposed to which it is not expected to adapt to within the time frame of management action. This situation may be brought about by changing the concentrations of substances, levels of kinetic energy, flow or habitat integrity in a way that disrupts that which is naturally found in the system.

3.3 CONTENT OF A RESPONSE KNOWLEDGE BASE

The knowledge base may be expressed as a relationship between the level of the stressor and the expectation of a specific unacceptable response. The format of such expression may range from a mathematical function to a fuzzy rule-base. Generally, this relationship could be interpreted in the form of an **IF**... **THEN** ... rule or set of rules.

For example:

> Rule 1: **IF** pH is greater than x1 **OR** pH is less than y1 **THEN** likelihood of critical ecological effect is VERY LOW
>
> Rule 2: **IF** pH is less than x2 **OR** pH is greater than y2 **THEN** likelihood of critical ecological effect is VERY HIGH
> Etc.
>
> Where "VERY LOW likelihood" is < 10% and "VERY HIGH" likelihood is > 80%

OR

Likelihood of critical effect = $0.0843 * e^{[(\text{Optimal pH} - \text{Observed pH}) / 3.46]}$

OR

Likelihood of critical effect $= \begin{cases} 0 & \text{if } x \leq x_0 \\ \frac{x - x_0}{x' - x_0} & \text{if } x_0 < x < x' \\ 1 & \text{if } x \geq x' \end{cases}$ where x is the value of a variable and x' and x_0 are benchmark values for that variable.

3.4 TYPES OF RESPONSE KNOWLEDGE BASES

One may distinguish two types of knowledge bases:

- a generic knowledge base and
- a situation specific knowledge base.

A **generic knowledge base** would contain the background knowledge of the type: "It is generally believed that...." OR "From laboratory experiments it is known that......" OR "Most experts agree that......"

A **situation-specific knowledge base** would contain the knowledge of the type: "For this specific river it is expected that......", OR " Due to the observations in this river it is likely that......", etc.

A situation-specific knowledge base results from updating the generic knowledge base by applicable site-specific observation.

3.5 DISCRETISING THE GSRR OR SSRR

In the form derived above, the GSRR/ SSRR is an expression defining three domains:

1. The **no-hazard domain**. This stressor domain representing the no expected hazard condition that will most likely be associated with pristine sites.
2. The **high-hazard domain**. This stressor domain represents a definite expected hazard. These stressor levels are associated with an unacceptable hazard level and needs no further discretisation for management purpose since they are assumed to be unacceptable as management objectives.
3. The **transitional-hazard domain**. This domain represents a variable expectation of hazard. The implication is that there is also a "grey scale" of acceptability in stressor levels based on corresponding hazard level. This domain may be discretised into any arbitrary number of hazard categories as required by policy.

> The response of the stressor that defines the hazard is divided into a no-hazard domain, a transitional hazard domain and a high hazard domain. The no-hazard domain corresponds to the pristine state while the high hazard domain corresponds to the actively degrading state. The transitional domain can be discretised in any arbitrary number of domains as required by policy. At present it is divided into three domains with corresponding stressor levels as boundaries to ecological water quality classes.

In the current management practice requiring 5 categories (A, B, C, D and E/F), the first domain corresponds to category A, the second domain to category E/F, which means that the third domain can be divided into three categories (B, C and D). Without any further information to the contrary, it is suggested that the transitional-hazard domain is divided into three equally spaced hazard domains (see **Figure 3.1**).

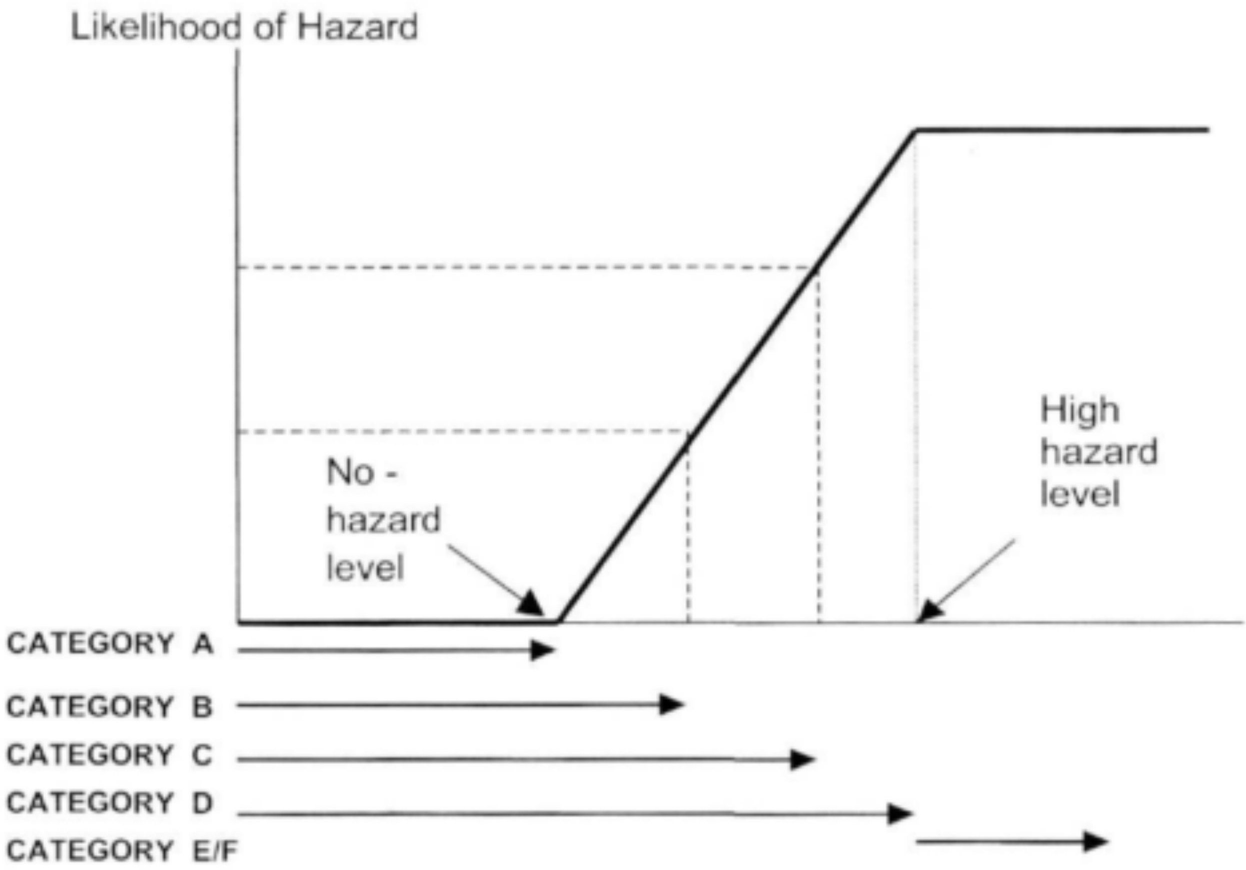


Figure 3.1 A typical stressor response relationship.

This procedure is largely involved in steps 4a (determine the present status of resource units – resource quality) and 6a (quantify reserve for each resource unit: water quality) of the current reserve methodology (see Chapter 1). It supplies a tool for deriving categorisation criteria and Reserve stressor boundaries.

However, an alternative procedure for applying an adaptive approach to deriving a hazard-based ecological water quality reserve is shown in **Figure 3.2**. An important prerequisite to the application of this procedure is the identification of reference condition (if at all possible). The product of the procedure supplies the criteria to categorise the present state. The intention is that this procedure will at the same time produce the stressor criterion values that will be used in setting/assessing the ecological aspect of the reserve. A decision on the desired ecological management class is not a prerequisite for this procedure.

It is further assumed that the integration between water quantity and quality will broadly proceed along the lines of a waste load allocation as described in the USEPA documentation (USEPA 1984) and that this is a management function.

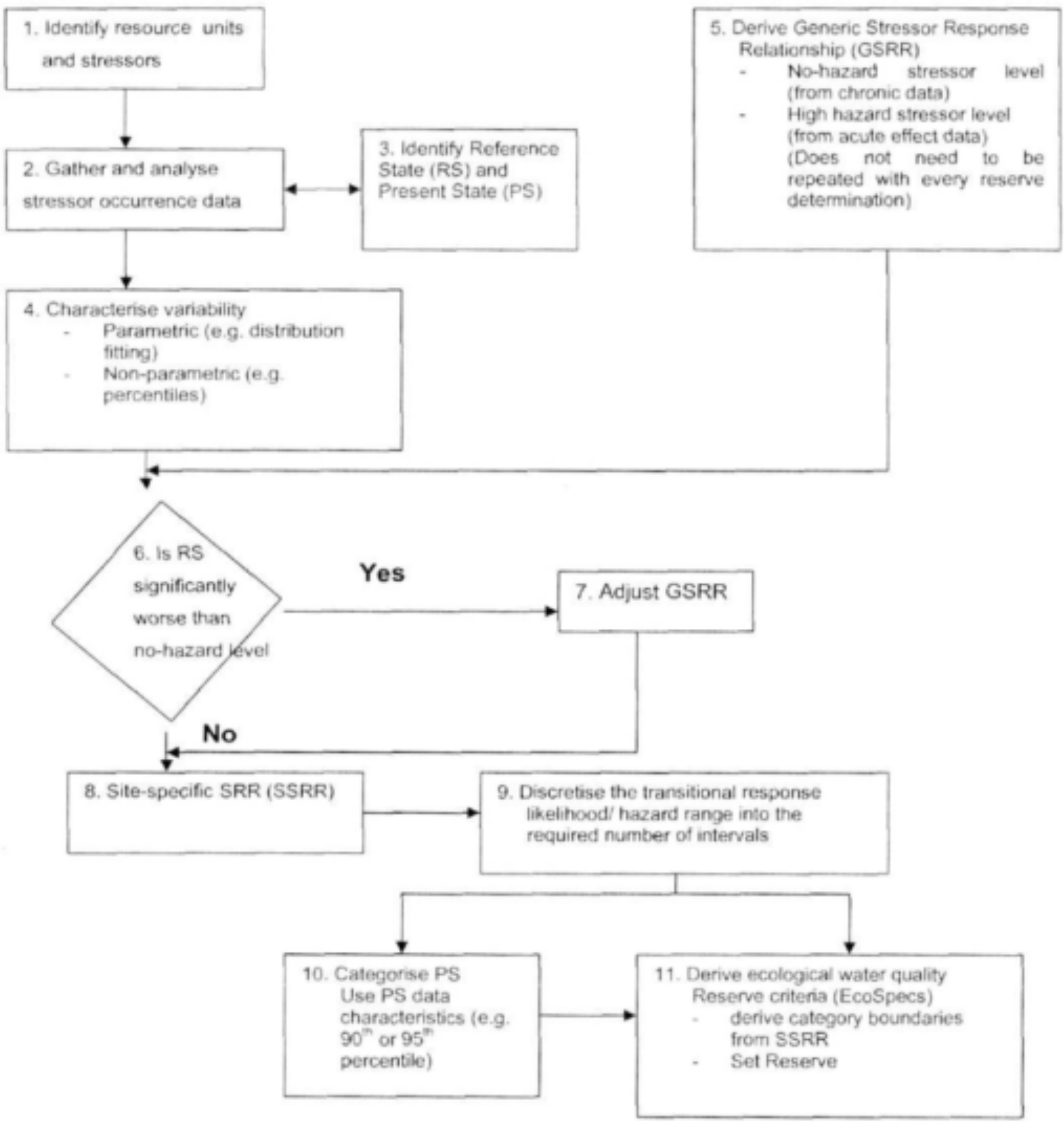


Figure 3.2 A procedure for applying site-specific stressor response relationships in setting the reserve.

The steps in **Figure 3.2** are briefly outlined below.

Step 1 - Identify resource units and stressors

Identification of relevant resource units is a procedure described in the current RDM documentation. The identification of stressors should be performed carefully. It should not be assumed that the commonly measured variables represent stressors. A useful guide in this regard is USEPA (2000). Particular attention should be paid to scientific evidence regarding seasonality in measured variables and its importance in defining stressors.

Step 2 - Gather and analyse stressor data

The guidelines provided before should be followed. It is important the there be at least 60 records in the data set

Step 3 - Identify the reference state (RS) and present state (PS)

There are two-steps: a) identify a water quality reference site and b) charcaterise the PS and RS.

The reference state by definition is one that is as close to pristine as possible. Although there are unlikely to be any pristine site left, factors such as a clear lack of impacts upstream and sustained good biotic integrity would contribute to the identification of such as site. Alternatively, it may not be possible to select a geographic location that will comply to this description, but a time period in the data record for a site may well comply. The RS is important in assessing the applicability of the SRR. If no RS is identifiable then it should be recorded, as "Not Available" and the default SRR would be valid until evidence to the contrary exists.

Step 4 - Characterise stressor occurrence variability

Both random and seasonal variability may exist. It should be determined if seasonal variability is important in a specific situation. If seasonality is important then the random variability in seasonally grouped data should be established.

Variability may be described in either parametric or non-parametric statistical terms. The simplest nonparametric description of variability is the use of percentiles. However, parametric estimates may be obtained by function fitting techniques, where a "best fit" parametric distribution is obtained. Percentiles may then be calculated from the parametric equation.

Box plots may serve as a useful graphical representation of the data.

Step 5 - Derive generic stressor response relationships (GSRR) for each stressor

The procedure is described in Section 3.6 below. This function need not be repeated every time a reserve is being assessed.

Step 6 - Is the RS significantly worse than the no-hazard level?

This test is necessary to determine whether the RS represents a worse quality than the no-hazard level of the GSRR. This has to be repeated for every water quality variable. If the RS water quality is worse than the no-hazard level, and it still qualifies as a RS, then the implication is that the biota under those conditions can tolerate that water quality.

In principle, both the water quality and the no-hazard level are non-deterministic quantities and so hypothesis testing forms the basis of this decision. The no-hazard level in fact derives from a chronic effect distribution and the water quality also present itself as a distribution of values. For this decision the null hypothesis is set as:

"the RS water quality is better or the same as the GSRR chronic effect distribution". A suitable test needs to be used to accept or reject the null hypothesis (e.g. a one-tailed t-test). It is suggested that the Type I error criterion, α, is set at 5%. A simplification can be brought about by using the projected no-hazard value as a benchmark of the chronic effect distribution while using the 95^{th} percentile as a benchmark of the RS water quality variable distribution. The test could then involve the question: "Is the water quality benchmark larger than the no-hazard level?"

Step 7 - Adjust the GSRR

It is suggested that either the Bayesian or the Dempster-Schafer approach is used in the updating the GSRR. It is not foreseen that this step will be needed very often and the detail of the approaches above is quite complex. It is suggested that expert opinion is obtained in updating the GSRR. The possibility of updating the GSRR should be noted in the documentation.

Step 8 - Produce site-specific stressor response relationship (SSRR)

The site-specific stressor response relationship is applicable only to a specific site and should not be extrapolated.

Step 9 - Discretise the transitional response likelihood/hazard range

There are two clearly defined categories when using this model: the category with values better than the no-hazard level and the category worse than the high-hazard level. The transitional hazard domain can be divided as required by water management policy. Currently, with the 5-category classification, the transitional domain is divided into 3 equal categories.

Step 10 - Categorise the PS data

The PS data are characterised in a manner similar to step 4. The category bound values derived in this way are also the EcoSpecs criteria.

Step 11 - Derive ecological water quality reserve criteria

As a precautionary approach, the EcoSpecs are set according to the PS category. The ecological water quality criteria for each variable will be the upper (worse quality) bound of the current class. This procedure may change with current policy.

3.6 GENERATING GENERIC STRESSOR RESPONSE RELATIONSHIPS

There are two fundamental assumptions used here:

I. The endpoint is described in terms of phenomenon or an event that happens independent of how or why it happens, because we are dealing with a domain where mechanistic knowledge is often insufficient or totally lacking. This means that we consider an endpoint effect, such as loss of integrity, as a phenomenon without necessarily having detail on how the different stressors may interact to bring about an observed endpoint, or, perhaps even how any specific stressor interacts with the system to bring about that endpoint. All that may be known is that:
   - the stressor does interact with the system (by definition of the term stressor) and
   - the intensity of interaction may vary with level of stressor.

The scientific contribution to the process is to determine the relationship between the level of stressor and the endpoint response (i.e. a stressor response relationship).

II. We are dealing with an endpoint that is epistemologically vague, i.e. the quality of our knowledge about the endpoint and what determines it is such that the endpoint cannot be described precisely. This means that although we might use integrity as a practical and reasonably precise measure to describe deviation from pristine, sustainable conditions, the critical level of loss of integrity that corresponds to the regulatory endpoint of un-sustainability remains a matter of conjecture. It is therefore not scientifically defensible to set a deterministic endpoint in the sense of a unique critical level of integrity. At best one might describe an increasing likelihood that a critical level of loss of integrity is reached.

The implication of this is that the stressor response relationship is expressed in terms of the likelihood of observing the endpoint.

In general at least two critical points are needed to generate a SRR:

1. **No expected hazard level**. Up to what level(s) of stressor is there no critical response expected (i.e. what is the no-expected-response level of the stressor)?

2. **Hazard level**. Beyond which level(s) of stressor is critical response expected (i.e. what is the critical effect level of the stressor)?

For non-nutrient chemicals or chemicals present at hyper-nutritional levels, curves such as those in **Figure 3.1** above can be generated. In order to do that, it is necessary to decide:

What is a suitable end-point for the hazard assessment?
How do the available lab end-points translate to that end-point?
How should stressor levels be expressed?

In the case of substances for which toxicity data exist (such as common salts, heavy metals and a number of anions and organics), the example below might be valid.

1. Suitable end-point

If populations of species survive, then system integrity is expected to remain intact and, by implication, sustainability is expected to be unimpaired. Population survival is sometimes used as an ecotoxicological end-point.

For chemical substances the Reserve end-point might be population survival.

2. Translating lab end-points to Reserve end-point

Generally a range of end-points are reported varying from no observable effect of any kind to mortality. If the data for a range of species are available, then a threshold of effect may be determined at which very few species would display that effect. For example the threshold for acute effect may be derived from LC_{50} values for a range of species. This threshold may then be interpreted as that level of stressor at which all but the most sensitive specie would exhibit more than 50% survival (Threshold Critical Effect Level, TCEL). A similar rationale would hold for chronic effects (Threshold No Effect Level, TNEL).

Furthermore, if less than half of individuals of the most sensitive species is expected to survive, then it is quite possible that populations of all but the most sensitive species will survive. Consequently, the TNEL and TCEL define benchmarks of population survival.

A threshold acute effect level and a threshold chronic effect level could be used to define a SRR for population survival.

3. Expressing stressor levels

Although body burden is toxicologically more important than concentration, in ecotoxicological testing concentration is usually used to express the level of stressor. For substances that do not occur in surface water naturally or that are not physiologically necessary, this would be a suitable measure of stressor level.

For substances that occur naturally (e.g. major salts) the stressor should be expressed in terms of deviation from natural (or necessary) levels. However, generally the interest is not in managing the lower physiologically detrimental levels.

For most substances concentration may be a suitable measure of stressor level.

3.7 SELECTING, INTERROGATING AND REFINING A SUITABLE DATABASE

For Salts and Toxics

For water quality (salts and toxics) the ECOTOX database maintained by the USEPA (http://www.epa.gov/ecotox) was considered the most extensive and most accessible. All toxicity data were extracted subject to:

- Fresh water as medium.
- Laboratory results.
- Single chemical species.
- LC_{50}, EC_{50}, LOEC, NOEC.
- All end-points (mortality, immobilisation, behaviour, physiological, population, etc.).

Refinement of data base:

- Data were rejected if experiments were flagged due to insufficient control data (blank and/or positive control).
- End-point values reported as "larger than" were replaced by the value.
- Concentrations were converted to mg/L units.
- The end-point concentrations from otherwise identical records were aggregated by geometric mean.
- Results were aggregated to genus mean values also by geometric mean.
- Care was exercised to ensure representation of toxicity data for: a) fish, b) other aquatic vertebrates, c) invertebrates, d) plants (no distinction was made between vascular plants and algae).

3.8 PROJECTING THE MOST SENSITIVE SPECIES

The procedure below is followed for both acute and chronic data. "Acute" data are used in the context of mortality end-point data (usually expressed as LC_{50}) but without reference to the duration of exposure. "Chronic" data are used in the sense of data on sub-lethal end-points with exposure durations that are relatively long in comparison to the life expectancy of the organism or estimates thereof at the sub-organismal level (usually expressed as LOEC/NOEC):

1. The most sensitive projected species is found by linear regression on the cumulative probability data. The data for most salts were approximately lognormally distributed, but with significant tails.
2. Percentiles (50^{th}, 10^{th}, 5^{th} and 1^{st}) of the toxicity data were generated. The natural logarithm of these were used as ordinate data points in a linear regression against the corresponding fractile (the percentile expressed as a proper fraction) as abscissa. The projected most sensitive species' benchmark concentration was found by solving the linear fit equation for the concentration corresponding to a zero ordinate (y-value).
3. The no hazard level is derived from the chronic data benchmark and the high hazard level is set at 0.5*acute data benchmark. This is analogous to the CEV and AEC respectively of the South African Water Quality Guidelines (DWAF 1996).

3.8.1 Deriving the GSRR for major salts

The no hazard and high hazard levels for some major salts are shown in **Table 3.1**. The GSRR calculates the response possibility, *y*, as defined by Eq. 1. The category boundaries are derived from y using **Table 3.2**.

$$y = \begin{cases} 0 & if\ x \le x_0 \\ \dfrac{x - x_0}{x' - x_0} & if\ x_0 < x < x' \\ 1 & if\ x \ge x' \end{cases} \qquad \text{Eq. 1}$$

Table 3.1 Stressor response benchmarks for major salts and ions in aquatic environments.

Variable	No hazard (x_0)	High hazard (x')
NH_3 mgN/L	0.007	0.1
PO_4^{3-} mgP/L	0.01	0.1
Na_2SO_4 mg/L	6	85
$MgCl_2$ mg/L	10	65
$CaCl_2$ mg/L	35	150
KCl mg/L	46	65
$MgSO_4$ mg/L	55	270
$CaSO_4$ mg/L	20	960
NaCl mg/L	3	1 200
Na mg/L	5	500
K mg/L	46	65
Mg mg/L	17	88
Ca mg/L	13	340
Cl mg/L	62	932
SO_4 mg/L	70	956

Table 3.2 Categories corresponding to the possibility *y*.	
Value of y	**Category**
0	A
$0 < y \leq 0.33$	B
$0.33 < y \leq 0.67$	C
$0.67 < y < 1$	D
$y \geq 1$	E/F

3.8.2 Deriving the GSRR for other water quality variables

While the basic principle remains the same, the difference with the salts and toxics above relates primarily to the use of laboratory-derived data. With these variables no experimental data were at hand, so the data contained in DWAF (1999) were used.

For pH:

$$y = \begin{cases} 1 & if\ x \leq a\ or\ x \geq d \\ \dfrac{b-x}{b-a} & if\ a < x < b \\ \dfrac{x-c}{d-c} & if\ c \leq x < d \\ 0 & if\ b < x < c \end{cases}$$

where x is the median pH and a = 5, b = 6.5, c = 7.5 and d = 9.

For DO:

$$y = \begin{cases} 1\ if\ x \leq x' \\ \dfrac{x_0 - x}{x_0 - x'}\ if\ x' < x < x_0 \\ 0\ if\ x \geq x_0 \end{cases}$$

where x is the median % Dissolved Oxygen and x' = 40% and x_0 = 120%

For nutrients:

The TIN:SP ratio is dependent on the PO_4^{3-} concentration. Using the data in Table D7 (DWAF 1999) the categories were replaced by values from **Table 3.2**. The PO_4^{3-} concentrations were replaced by the upper boundaries and the ratios were replaced as follows: 10:1 replaced by 5, >10:1and<20:1 replaced by 15, >20:1and<30:1 replaced by 25 and > 30:1 replaced by 35. A plot of these values generates a 3-dimensional curve (**Figure 3.3**).

It is possible to find a correlation between PO_4^{3-} concentration and TIN:SP ratio for an A-category as well as for a D category. These would yield a high hazard response R' and a no-hazard response R_0. At any given phosphate concentration P, the No-effect ratio R_0 and the Critical effect ratio R' is determined by:

R' = 1.344613 * exp(0.027361*P), and
R_0 = 12.5277 * exp(0.018592 * P) (**Figure 3.4**)

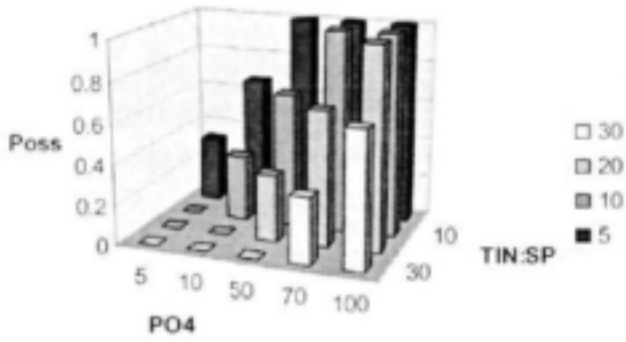


Figure 3.3 *Possibility of response as a function of nutrient parameters.*

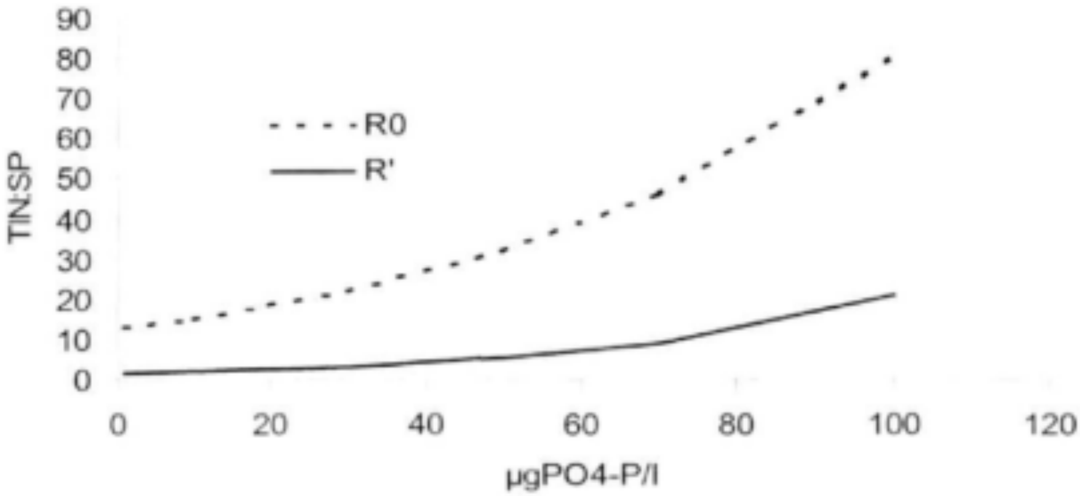


Figure 3.4 The No-hazard (R_0) and High hazard (R') curves for nutrients.

The possibility *y* is then calculated from the TIN:TP ratio *R* by:

$$y = \begin{cases} 0 & if \;\; R > R_0 \\ \dfrac{R_0 - R}{R_0 - R'} & if \;\; R' \leq R \leq R_0 \\ 1 & if \;\; R < R' \end{cases}$$

3.9 GENERAL NOTES

- No account has yet been taken of seasonal and ephemeral systems and water quality conditions in pools.
- How will dams be treated in ecological Reserve assessments?
- How will inter-basin transfers be treated in ecological Reserve assessments?
- How to integrate biomonitoring?
- How to use toxic units?
- How/whether to use algal growth potential as a water quality assessment criterion?
- Make sure boundaries are consistent with emerging classification policy.
- Confirmation that at intermediate and comprehensive level Reserve determinations, field visits and biomonitoring are essential to assess PES. Resource unit boundaries including dams, tributaries and joint sources are important in defining the present state.
- Seasonality issues are still to be discussed further, and refined in next draft. Current ideas include seasonality being taken into account in a total account of risk, and resource quality objectives being specified in terms of duration, frequency and magnitude.
- Discussions indicated that time-series data and trajectories of change were no longer needed, however, further discussion is needed.
- Clarify the concept of continua and the limitations of "boundary values"
- Care must be taken not to create the impression that it will be operationally possible to manage for water quality conditions in terms of magnitude, frequency and duration in good and fair management classes given the highly complex nature of the combined impacts of waste discharges (both point and non-point) on rivers whether they are regulated or unregulated.
- TSS is currently not part of the Reserve water quality assessment procedure. Boundary criteria, which define riparian buffer-zones, must be developed.
- A method and place in the process for making decisions about the level of confidence needed must be developed.
- Decide on a method for a selection process to determine which ecosystems are to undergo a Reserve assessment process.
- Clarify the role of management classes and ecological classes within an ecological Reserve assessment.

Useful note from Olifants Reserve study:
Some remarks on "reference conditions" – in the case of rivers downstream of reservoir can the in-lake quality be used as a "reference condition"?

Reservoirs often modify water quality to a large degree and set a new starting condition for the river. In this case, the term "Reference condition" is probably not correct even though it is used to evaluate how much water quality has changed at a site downstream of the dam by comparing it to the in-lake quality.

CHAPTER 4

RECOMMENDATIONS, COLLABORATION AND CAPACITY BUILDING

The set of recommendations from this study resulted in a proposal to the WRC for completing input into the water quality aspects of the Ecological Reserve Decision Support System, which was completed in 2004 (Hughes (Ed.) in press). These two projects have provided the momentum, resources and direction for the successful inclusion of water quality methods into Ecological Reserve determinations.

Critical areas that still need research attention are:

- Development of risk-based boundary values for nutrients and sediments.
- Further development of integration of quality and quantity.

However the most important needs are management-related:

- Finalization of a classification system.
- Implementation of an integrated quality-quantity ecological Reserve.

This whole project was undertaken in a collaborative manner by a team from different institutions, notably: Rhodes University, Ninham Shand, University of Cape Town and the Department of Water Affairs and Forestry.

Three workshops were organized as part of the project which involved the wider stakeholder group with interests in the ecological Reserve.

Mrs. T. Zokufa and Mr. M. Papo of DWAF participated as part of the collaborative capacity building initiative of the IWR.

Mr. W. Kamish was part of the Ninham Shand team and was trained in water quality methods during the project.

Ms. M. Stewart, a disabled student, worked as a research assistant at the UECWQ-IWR laboratory.

CHAPTER 5
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APPENDIX 1

WATER QUALITY SPECIFICATION FOR THE ECOLOGICAL RESERVE: CROCODILE RIVER (MPUMALANGA)

Marius Claassen
CSIR, Environmentek, Pretoria

1.1 INTRODUCTION

1.1.1 Concept of risk and water quality Reserve determinations

The protection philosophy was entrenched in the fundamental principles and objectives for a new water law in South Africa (DWAF 1996b), calling for the protection of water resources (quantity, quality and reliability) to maintain the ecological functions to ensure long-term sustainability of aquatic and associated ecosystems. The white paper on water policy (DWAF 1997) supported this philosophy, stating, "only that water required to meet basic human needs and maintain environmental sustainability will be guaranteed as a "right" (known as the Reserve)". This approach initiated in the principles has subsequently been enacted (Statutes of the Republic of South Africa 1998), with Part 3 of the National Water Act (NWA) stating, that the ecological Reserve relates to the water required to protect the aquatic ecosystems, referring to both the quantity and quality of the water in the resource, and will vary depending on the class of the resource. Being in accordance with a class, the Reserve allows a balance to be sought between the need to protect and sustain water resources on the one hand and the need to develop and use them on the other. The ecological Reserve can be described as ecological specifications that provide specific levels of protection to water resources. The level of protection is related to the goods and services that society derives from the resource, and the potential for the resource to provide such goods and services for future generations.

Jooste and Claassen (2000) investigated the application of risk concepts to water resource quality management and concluded that risk may reasonably be used to aid water resource quality management decisions and activities related to, but not necessarily limited to, the following areas:

- Basis for water quality criteria: The current South African Water Quality Guidelines (aquatic ecosystems) was derived from toxicological data and some qualitative assumptions regarding exposure. These criteria are limited because expected effects differ from substance to substance, co-occurrence of different stressors are not considered, and the criteria do not necessarily relate to the same ecological effect. Setting criteria on a risk basis would induce a measure of transparency into the interpretation of such criteria.

- Site-specific criteria: Applying risk criteria to the site-specific adaptation of criteria supplies a rational basis for incorporating new or locally significant data in a manner that is open to peer review.

- Resource management classification: The provision in the National Water Act for the classification of water resources can be linked to risk concepts. Management objectives may be expressed in terms of allowable risk to the Reserve. This provides an explicit communality between the receiving water quality/risk objectives and the Reserve as well as effluent criteria and/or standards.
- Hazard ranking: In some situations, it is neither necessary nor feasible to calculate absolute risks. In the case where different hazards within the same scenario or hazards in different scenarios need to be compared, risk is often a suitable basis for comparison.

These approaches were considered in determining the Reserve for a section of the Crocodile River (Mpumalanga). The relationship between risk and water resource classification is fundamental to this assessment. The classes (A-F) are based on the protection of ecological integrity, with increasing risk of irreversible damage to basic ecosystem processes (structure and function; **Figure A1**). Management goals are set along this continuum, although classes E and F have been defined as unacceptable. A decision as to the management goals and acceptability of specific risks are a function of societal values, with science providing an input to the decision. For this study, acceptable condition (class D) are defined as habitat conditions and biological integrity deviating significantly from that associated with ecotype under normal conditions, basic ecosystem processes (structure and function) being intact and indicator species representing the ecotype still occurring. Class A represents natural conditions. The provision of goods and services are linked to the classification system, with an A-class river providing non-consumptive services and having the maximum potential to provide goods. D-class rivers provide the maximum goods and services that can be delivered on a sustainable basis, albeit at higher risk. Classes E and F provide goods at a rate that poses high (unacceptable) risk to basic ecosystem processes. For this study, the risks associated to the different classes are based on the probability that basic ecosystem processes (structure and function) will be irreversibly damaged in a given year (**Table A1**). In an A-class, this risk is one in a million, which would typically be due to long term natural change such as interglacial cycles and relate to geological time. In a D-class, the risk is one in a thousand with the system being sensitive to small disturbances at the level of ecological succession processes.

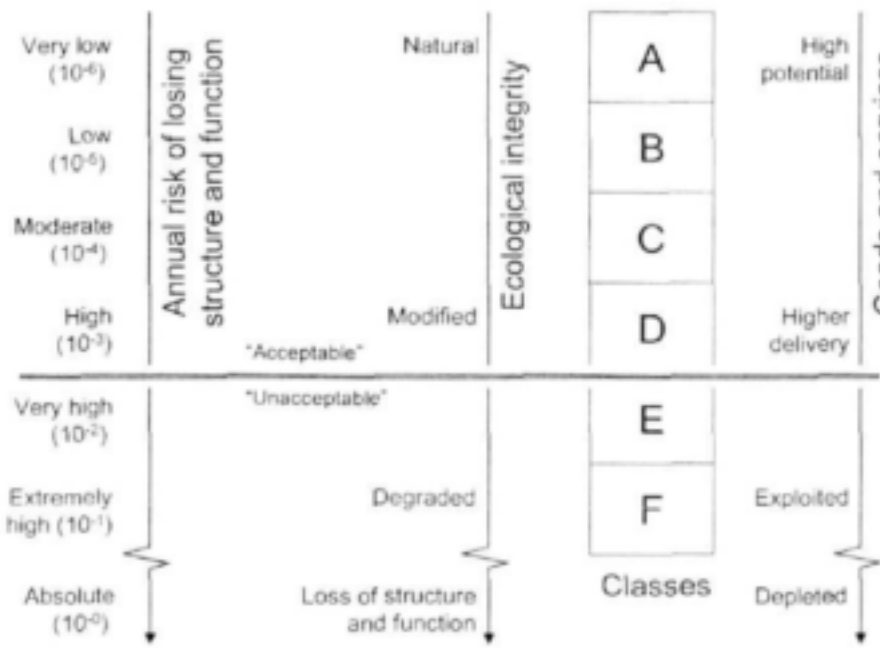

Figure A1 Relation between risk and water resource classification (Claassen, in prep).

Table A1	Risk associated with different classes (Claassen, in prep.). * Drivers that could cause change in structure and function (Forman and Gordon 1986)				
Class	**Risk**	**Temporal scale**	**Spatial scale**	**Typical drivers ***	**Ecosystem response**
A	$1x10^{-6}$	1000 000 years	$1x10^{12}$ m^2	Interglacial cycles	Evolution
B	$1x10^{-5}$	100 000 years	$1x10^{9}$ m^2	Climate change	Speciation, extinction
C	$1x10^{-4}$	10 000 years	$1x10^{7}$ m^2	Catchment impacts	Migration
D	$1x10^{-3}$	1 000 years	$1x10^{5}$ m^2	Floods	Succession
E	$1x10^{-2}$	100 years	$1x10^{3}$ m^2	Pathogens	Competition
F	$1x10^{-1}$	10 years	$1x10^{2}$ m^2	Disturbance	Productivity

1.1.2 Crocodile River Project – Terms of reference

The water quality required to maintain ecosystems associated with the Crocodile River is determined through the applying the ecological Reserve concept. The present method for Reserve determination (DWAF 1999), identifies the following steps:

1 Delineate geographical boundaries
2a Determine ecoregional type
2b Delineate resource units
2c Select sites
3 Reference conditions
4a Present status
4b Importance + sensitivity
5a Determine Management class
5b Set management class
6a Quantify Reserve
6b Set RQOS
7 Monitoring programme
8 Notice of RDM

The Crocodile River Reserve determination follows this process with the following modifications:

- Steps 5a and 5b: These are deemed management responsibilities, and this study only provides inputs to such decisions.
- Step 6a: In quantifying the Reserve, biological and water quality monitoring data is use to supplement the generic dose-response relationships (DWAF 1999; DWAF 1998).
- Step 6b: Resource quality objectives should include ecological considerations, human health considerations and resource use considerations (NWA Act 36 of 1998). This study addresses only the first of these requirements, therefore RQOs will not be set.
- Step 8: An administrative requirement that will not be dealt with in this section.

Scoping meetings and discussions were held on during which the terms of reference for the assessment was formulated as follows:

vi. The Reserve should be determined with the highest level of confidence possible, given the time constraints (1 month). The availability of data for this river will allow the assessment to approximate an intermediate Reserve determination.

vii. The resource unit for which the Reserve will be determined is defined by quaternary catchment X24H.

viii. The study should determine whether the ecosystem in the Komati River, downstream of the confluence with the Crocodile River is more sensitive than that in the Crocodile River. If it is the case, then the Reserve should also provide adequate protection for this section.

ix. The following variables for which the Reserve should be determined were identified: Temperature, pH, Diss. O_2, Cl, TSS, TDS/EC, NH_3 and NO_3, PO_4, Total phosphorous, N:P ratio, Pb, NH_3.

x. The Reserve should be specified as percentage exceedance values.

The study area (**Figure A2**) extends from Maroela Weir to the confluence with the Komati River. The Lowveld and Lebombo Uplands ecoregions (Level I classification; DWAF 1999) transect the study area. The Lowveld ecoregion in the study area is comprised of Level II regions 5.06 and 5.07, with the Lebombo Uplands representing the 6.01 Level II ecoregion. These characteristics of the region (Balance et al. 2001; DWAF 1995) as summarised in **Table A2** supported the decision to use one resource unit for the study. This resource unit integrates water quality influences from the biggest part of the Crocodile River catchment.

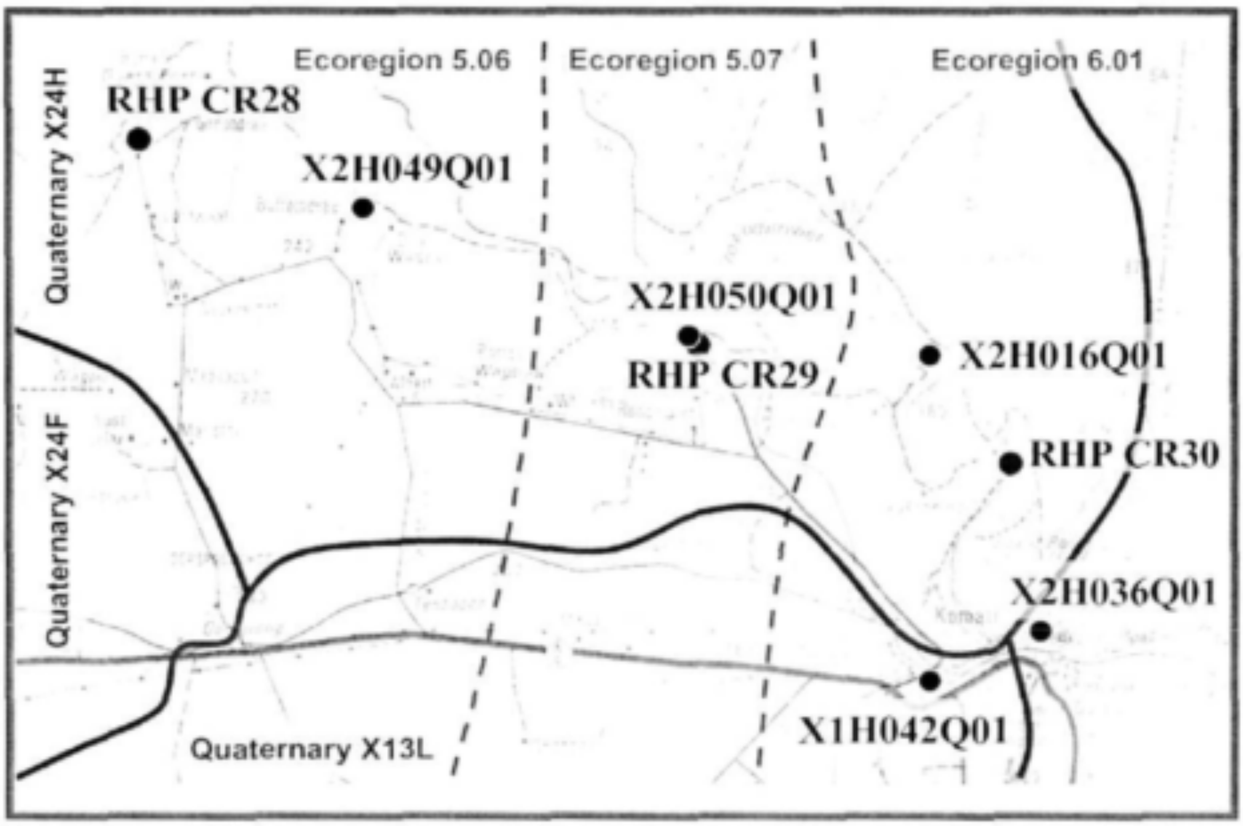

Figure A2 Crocodile River from Maroela Weir to Komatipoort.

Table A2 Physical and climatologic characteristics of the study area.			
	Ecoregion 5.06	**Ecoregion 5.07**	**Ecoregion 6.01**
Topography:	Flat moderate relief, typical of Lowveld proper.	Moderately undulating plains with a moderate relief	Moderate relief with Lebombo arid maintain bushveld
Precipitation: (mean)	400-800 mm/annum	400-600 mm/annum	400-800 mm/annum
Temperature: (mean annual)	20-22 °C	≥ 22 °C	≥ 22 °C
Altitude: (m.a.m.s.l)	300-600	200-400	100-300
Soils:	Shallow black, brown or red clayey soils	Shallow calcareous red sandy clay to clay overlay basalt of the Karoo system	Shallow acidic, sandy soils overlay basalt, tuff, breccia and rhyolite.
Veld type:	Sweet lowveld bushveld	Sweet lowveld bushveld	Lebombo arid mountain bushveld
River:	The river is 40-50 m wide and slow flowing. With mostly large sandy pools.	The river is 40-50 m wide and slow flowing. With mostly large sandy pools.	The river is 40-50 m wide and slow flowing. Rocky pools and large sandy pools dominate, with occasional rapids.

The selection of water quality variables was based on their importance to ecological processes and their prevalence in the Crocodile River catchment. Current and potential sources of pollution were also considered in the selection. Based on the inputs from stakeholders the variables listed in **Table A3** were identified for the Reserve determination.

Table A3 Variables selected for the Reserve determination * Conductivity (mS/m) = TDS (mg/L at 25°C) / 6.5 ** Nitrate/Nitrite is represented in the ratio of inorganic (TIN) and phosphorus				
System variables		**Nutrients**	**Toxics**	**Micro-biological**
Temperature PH Dissolved O_2	Cl TSS TDS/EC *	NH_3 and NO_3 PO_4 Total phosphorous N:P ratio (NO_2/NO_3**)	Pb NH_3 Active chlorine and chloramines	Basic human needs

Water quality and biological assessments have been done in the lower Crocodile River catchment (DWAF 1995; Heath and Claassen 1999; NAEBP 1998). Survey sites and available data are shown in **Figure A2** and listed in **Table A4**.

Table A4	The study sites for the Reserve determination. [1] DWAF (2001) water quality monitoring points [2] Quaternary catchments [3] Monitoring stations used by the River Health Programme (Hill et al. 2000)			
Section	**Water quality** [1]	**Ecological**		**Hydrology Modelled** [2]
		Desktop [2]	**Actual** [3]	
Upstream section	-	X24F	-	X24F
Lower Crocodile	X2H049Q01 X2H050Q01 X2H016Q01	X24H	CR28 - 1 CR29 - 6 CR30 - 1	X24H
Komati River	X1H042Q01 X2H036Q01	X13L	-	-

The ecological parameters established for the determination of the desktop Reserve was used as a baseline for the study (DWAF 2000). The present ecological state category (PESC), ecological importance and sensitivity rating (EIS) and the default ecological management category (DEMC) as derived in the desktop assessment are listed in **Table A5**. The view from Kruger National Park (Deacon 2001) is that the Crocodile River is within the park and that all rivers in the park have a long-term management goal of "A".

Table A5	PES, EIS, DEMC and derived EMC used for the assessment.					
Quaternary	**PES category**	**EIS**	**Quaternary DEMC**	**Best AEMC**	**Resource unit EMC**	**KNP goal**
X24F	B/C	High	B	B	B	A
X24H	C/D	High	B	C	C	A
X13L	C/D	High	B	C	C	-

The geology of the lower Crocodile River (from Kaapmuiden to the confluence with the Komati River) contributes to the water quality in the resource unit. A summary of the geology is provided in **Table A6**.

Table A6	Combined lithological units and water quality impact (DWAF 1995) * Geological Survey, Barberton 2530, 1:250 000 Lithological Units	
Lithological Units *	**Area (%)**	**Ionic contribution**
Q	1	Al, Fe, Si, Minor K, Ca Mg
Ztk, Ztt, Zu, Zt	9	Mg, Ca, Fe, K, Al Minor: Au, As, Mn, Cr, Ni
Zg, Zk	6	Si, Al Na, Ca, K, Fe Minor: Au, As, Cu, Ni
Zn, Znm, Zh, Rm, Md	69	K, Si, Fe, Al, Ca, CO_3, Na
Zm, Zf, Zmb, Zj, Zmc, Zfs, Zb, Zfh	2	Mg, Si, Al, Fe, Ca, K Minor: Au, Ni, Mn
Rs	2	Si, Al, K
Mt	3	Si, Al, Fe, K
P-*h*, Jd, *h*-t, *h*-r	3	Si, Al, Fe
Jl	5	Si, Al, Fe, K
Jj	1	Si, Al, K

2.1 WATER QUALITY RESERVE ASSESSMENT METHOD

2.1.1 Reference conditions

The reference conditions and the present water quality status are obtained from the analyses of water quality data. According to the desktop determination of EIS, the sensitivity of the ecosystem to water quality changes is low (DWAF 2000).

2.1.1.1 Temperature

Temperature affects the rate of chemical reactions, solubility (including that of oxygen) and physical characteristics of water and temporal patterns are behavioural triggers. The known generic exposure-response is as follows:

- Natural (A): Temperature should not vary from background average by more than 10% or 2°C whichever is the more conservative. (DWAF 1996a, TWQR)
- Deviation from reference median (DWAF 1999)
  - 12% or 3°C is largely natural (B)
  - 15% or 4°C is moderately modified (C)
  - 20% or 5°C is largely modified (D)

The historical water temperature at X2H016Q01 in **Figure A3** indicates no clear trend in the data, although the last few years (associated with low flows) have elevated summer and suppressed winter temperatures. The diel variability of temperature (**Figure A4**) indicates a 6°C differential between daily minimum and maximum temperatures (04h00 - 20h00), with the differential between summer and winter months being 8 – 10°C. Biological monitoring data from 1992/1993 listed in **Table A7** shows that invertebrate communities were generally in a moderately modified state, with near natural and largely modified conditions being approached at times. Knowledge specific to the ecosystem is that the red breasted tilapia (*Tilapia rendalli*) can tolerate temperatures from 11 – 37°C (Skelton 1993), while breeding is arrested when summer temperatures drop below 21°C (Pienaar 1978). The characteristics of ambient temperature at Komatipoort (average of daily minimum and maximum temperatures) are shown in **Table A8** (DEAT 2000). Although the median is within 1°C of the median water temperature, the range of ambient air temperature is much narrower. The reference conditions (**Table A9**) for water temperature closely resemble the trend of ambient air temperature.

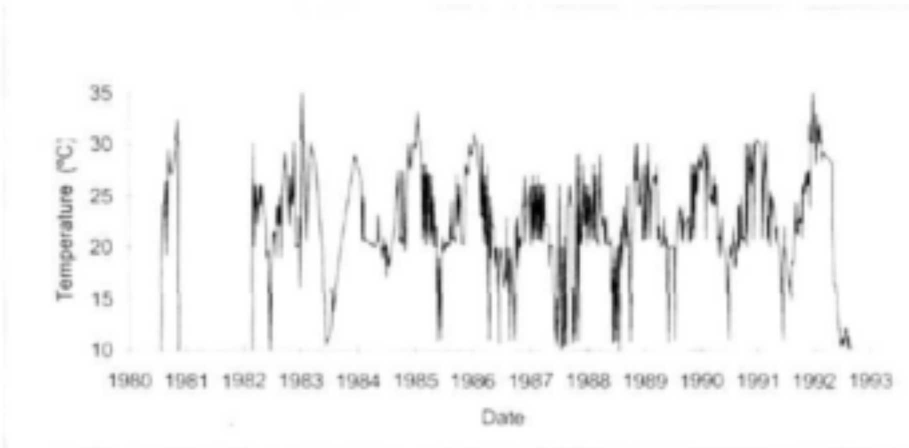

Figure A3 Historical water temperature at X2H016Q01.

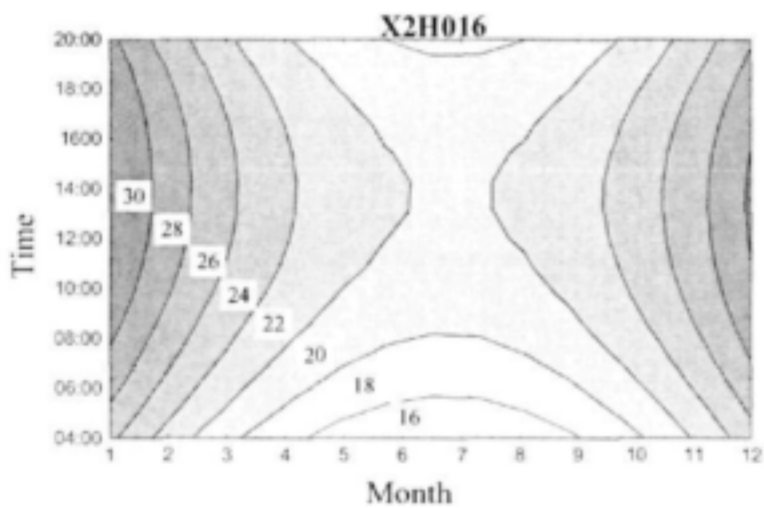

Figure A4 Diel and seasonal trends of water temperature at X2H016Q01 (°C).

Table A7	Historical SASS scores and temperature. * : MacMillan 2001and ←: Thirion 2001				
Site	Date	SASS	ASPT	State	°C
CR29*	5/1992	100	4.8	Moderately to largely modified	16.5
CR29*	6/1992	104	4.7	Moderately to largely modified	14.0
CR29*	7/1992	118	5.1	Moderately modified	10.3
CR29*	9/1992	139	5.1	Largely natural to moderately modified	10.2
CR29←	7/1993	111	5.3	Moderately modified	21.6
CR29←	8/1993	118	5.1	Moderately modified	16.8
CR29←	10/1993	96	4.4	Largely modified	22.2
CR29←	6/1994	122	4.9	Moderately modified	17.1
CR29←	9/1994	66	4.4	Largely modified	18.1
CR29←	6/1995	77	4.8	Largely modified	15.8
CR29←	9/1996	78	7.8	Moderately modified	24.0
CR28←	9/1996	133	7.0	Largely natural	28.1
CR30←	9/1996	146	8.1	Natural	23.9

Table A8	Historical ambient temperature at Komatipoort from 1993 to 2001. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)			
	Temperature (5th %tile – median – 95th %tile)			
	Summer	**Autumn**	**Winter**	**Spring**
Historical ambient temperature	23-26-31	22-25-27	17-19-21	21-23-26

Table A9	Reference conditions for water temperature. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)			
	Temperature (5th %tile – median – 95th %tile)			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions (A)	22-26-32	21-25-28	16-19-22	20-23-27

2.1.1.2 pH

The equilibrium of chemical reactions, bioavailability of metals and other compounds as well as ionic and osmotic processes are all affected by pH. According to generic knowledge about the dose-response relationship, pH should not vary from background average by more than 5% or 0.5 of a pH unit whichever is the more conservative (DWAF 1996a, TWQR). Deviations from this reference median are specified as (DWAF 1999):

- 7% or 0.7 of a pH unit is largely natural
- 10% or 1.0 of a pH unit is moderately modified
- 12% or 1.2 of a pH unit is largely modified

The historical pH of water at X2H016Q01 and an analysis of earliest and most recent pH values are provided in **Table A10** and **Figure A5**. The earliest available data shows clear seasonal trends (**Table A11**). Biological monitoring data and associated pH values from 1992/1993 (**Table A12**) shows that invertebrate communities were generally in a moderately modified state, with near natural and largely modified conditions being approached at times. This assessment is confirmed by Balance et al. 2001, stating that invertebrate fauna in this section give a varied response, but fish communities show fair (moderately modified) health. **Figure A5** clearly shows an increase in pH from 1977 to 2000, a trend that is confirmed in **Table A10** (noting that there was a change in the methodology between 1990 and 1992). The seasonal variability of pH (**Table A11**) is less apparent in more recent data. The potential impacts from geology, soil and vegetation do not explain this trend. The expected pH under natural conditions (**Table A13**) is based on the 1977 – 1979 conditions, although industrial and agricultural impacts would have had an impact on these values. The 1977 - 1979 conditions are thus adjusted according to generic knowledge to ensure that the deviation is not more than 0.5 of a pH unit.

Table A10 Earliest and most recent pH values at X2H016Q01.

	pH unit percentiles				
	5th	**25th**	**50th**	**75th**	**95th**
1970 – single value for February	7.4				
1977 - 1979 (n = 61)	6.9	7.1	7.4	7.6	8.1
1996 – 2000 (n = 285)	7.5	7.8	8.2	8.4	8.5

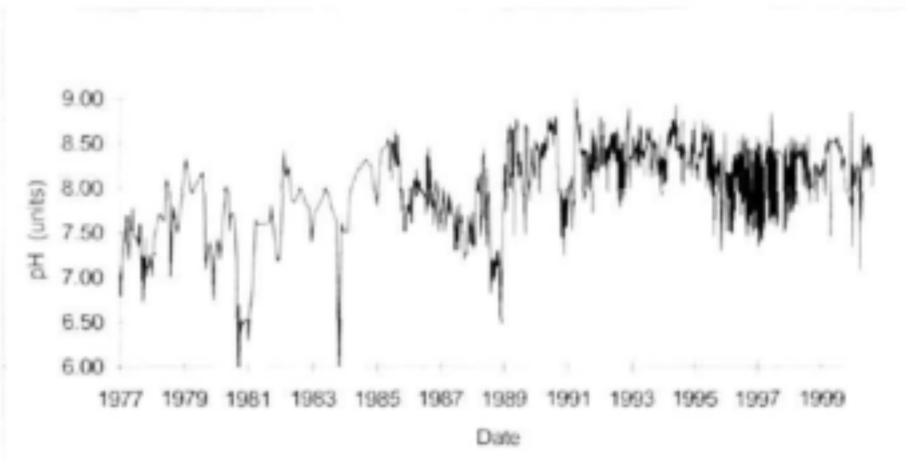

Figure A5 pH at X2H016Q01.

Table A11 Seasonal trends for pH (1977-1979 and 1996-2000). (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)

	pH units (5th %tile – median - 95th %tile)			
	Summer	**Autumn**	**Winter**	**Spring**
Earliest data (1977-1979)	6.9 - 7.2 – 7.8	6.8 – 7.1 – 7.3	7.2 – 7.6 – 8.1	7.3 - 7.6 – 7.9
Most recent (1996-2000)	7.5 - 8.1 – 8.5	7.4 – 8.1 – 8.6	7.5 – 8.2 – 8.5	7.6 – 8.3 – 8.6

Table A12	Historical SASS scores and pH. * : MacMillan 2001 and –: Thirion 2001				
Site	**Date**	**SASS**	**ASPT**	**State**	**pH**
CR29*	6/1992	104	4.7	Moderately to largely modified	7.8
CR29*	7/1992	118	5.1	Moderately modified	8.6
CR29*	9/1992	139	5.1	Largely natural to moderately modified	8.1
CR29*	10/1992	81	4.5	Moderately to largely modified	8.7
CR29–	7/1993	111	5.3	Moderately modified	8.4
CR29–	8/1993	118	5.1	Moderately modified	8.1
CR29–	10/1993	96	4.4	Largely modified	8.0
CR29–	6/1994	122	4.9	Moderately modified	8.3
CR29–	9/1994	66	4.4	Largely modified	8.3
CR29–	6/1995	77	4.8	Largely modified	8.2
CR29–	9/1996	78	7.8	Moderately modified	8.5
CR28–	9/1996	133	7.0	Largely natural	8.6
CR30–	9/1996	146	8.1	Natural	8.4

Table A13	Reference conditions for pH. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)			
	pH units (5th %tile – median - 95th %tile)			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions (A)	6.9 - 7.2 – 7.7	6.8 – 7.1 – 7.4	7.2 – 7.6 – 8.1	7.3 - 7.6 – 7.9

2.1.1.3 Oxygen

Oxygen is required for respiration in all aerobic organisms and also affects speciation of metals, sulphur and other compounds. DWAF (1996a) suggests that 80-120% oxygen saturation, with a 7-day mean minimum of 60% and 1-day minimum of 40% represents natural conditions. Deviations from this level are specified as follows:

- 80-100% of saturation is largely natural (DWAF 1999)
- 60-80% of saturation is moderately modified (DWAF 1999)
- 40-60% of saturation is largely modified (DWAF 1999)

Super-saturation can cause gas bubble disease in fish and inhibit photosynthesis in green algae and prolonged exposure to oxygen concentrations less than 50% of saturation can cause significant changes in community composition.

Dissolved oxygen (DO) was not measured at X2H016Q01, with limited measurements available for X2H050Q01 (DWAF 2001; Heath and Claassen 1999; RHP 2001) provided in **Table A14**. Biological monitoring data from 1992/1993 listed in **Table A15** shows that invertebrate communities were generally in a moderately modified state (ranging from natural and largely modified). The seasonal variability of DO (**Table A14**) illustrates the effect of low flows. The shallower water has a bigger surface to volume ratio and also higher turbulence across rapid and riffle sections. The solubility of oxygen is also higher during winter due to the colder water. The expected DO under natural conditions (**Table A16**) is based on generic knowledge and adjusted according to measured values.

Table A14	Dissolved oxygen data at X2H050Q01. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)							
	Dissolved oxygen (median)							
	Summer		**Autumn**		**Winter**		**Spring**	
	mg/L (%)	**n**	**mg/L (%)**	**n**	**mg/L (%)**	**n**	**mg/L (%)**	**n**
1989-1993	6.7 (86)	2	6.2 (78)	1	10.5 (115)	5	7.7 (93)	5

Table A15	Historical SASS scores and dissolved oxygen. * : MacMillan, 2001 and –: River Health Programme				
Site	**Date**	**SASS**	**ASPT**	**State**	**DO (mg/L)**
CR29*	6/1992	104	4.7	Moderately to largely modified	7.0
CR29*	9/1992	139	5.1	Largely natural to moderately modified	7.7
CR29*	10/1992	81	4.5	Moderately to largely modified	10.0
CR29–	7/1993	111	5.3	Moderately modified	11.2
CR29–	8/1993	118	5.1	Moderately modified	8.9
CR29–	10/1993	96	4.4	Largely modified	7.9
CR29–	6/1994	122	4.9	Moderately modified	7.7
CR29–	6/1995	77	4.8	Largely modified	9.5
CR29–	9/1996	78	7.8	Moderately modified	7.9
CR28–	9/1996	133	7.0	Largely natural	8.2
CR30–	9/1996	146	8.1	Natural	7.3

Table A16	Reference conditions for DO. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)			
	Dissolved oxygen : 5^{th} %tile – (median range) - 95^{th} %tile (% saturation)			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions	90-100-110	80-90-100	100-110-120	90-100-110

2.1.1.4 Nutrients

Absolute and relative inorganic nitrogen and phosphorus concentrations are key drivers for primary production. The generic dose-response relationship proposed by DWAF (1996a) suggests a that total inorganic nitrogen (TIN) and inorganic phosphorus (iP) should not deviate by more than 15% from natural, the trophic status of the system should not deteriorate and natural TIN and iP cycles should be maintained. The trophic status is defined as (DWAF 1996a):

	TIN	**iP**
Oligotrophic	< 0.5 mg/L	< 5 mg/L
Mesotrophic	0.5 – 2.5 mg/L	5 – 25 mg/L
Eutrophic	2.5 – 10 mg/L	25 – 250 mg/L
Hypertrophic	> 10 mg/L	> 250 mg/L

The Resource Directed Measures (RDM; DWAF 1999) provides the following specifications for nutrients.

Category	NH_3	PO_4	TP	TIN:TP (with PO_4 at 0.01 and 0.5)
A	0 - 0.007	0 – 0.01	0.1	>20:1
B	0.007 – 0.015	0.01 – 0.05	0.1 – 0.175	>10:1 and <20:1
C	0.015 – 0.03	0.05 – 0.07	0.175 – 0.25	>5:1 and <10:1
D	0.03 – 0.07	0.07 – 0.1	0.167 – 0.175	<5:1

The historical values for nutrients at X2H016Q01 are provided in **Figures A6 – A9**, with a comparison of the earliest (1977-1979) and most recent data (1996-2000) provided in **Table A17**. The range for nitrate/nitrite values is not significantly different between these periods, although the median was 63% higher in the latter term. The values for ammonium decreased from 1977 to 2000 (60%). The range for ortho-phosphate was narrower during 1996-2000, while the median increased from 0.01 in 1977-1979 to 0.02 in 1996-2000. **Table A18** shows that invertebrate communities at CR30 were generally in a natural to largely natural state from 1996 to 2000. A conclusion that can be drawn is that the nutrients listed in **Table A17** for 1996-2000 are sufficient to maintain the invertebrate community in a near natural state. The values listed in **Table A18** shows a TIN:PO_4 ratio of 35:1 for 1977-1979 and a TIN:TP ration of 10.2:1 for 1996-2000. Compared to generic knowledge the 1977-1997 ratio represents a natural state and the 1996-2000 ratio a largely natural state. The recent biological monitoring (**Table A19**) supports this view, with invertebrate communities being in a natural to largely natural state with TIN:PO_4 ratios between 25:1 and 50:1. Recent TP and PO_4 values exceed those required to maintain a natural system given generic knowledge, but still maintains the system in a near natural state. The reference conditions for TP and PO_4 are thus adjusted from the generic knowledge (**Table A19**). The TIN:TP ration showed similar deviations from generic rules and are adjusted accordingly.

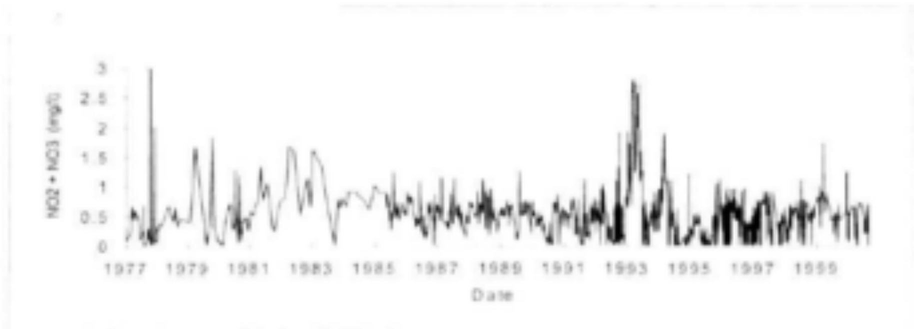

Figure A6 NO_2 and NO_3 at X2H016Q01.

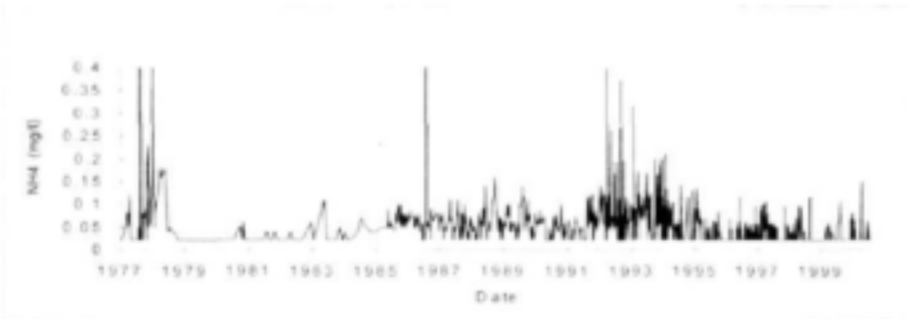

Figure A7 NH_4 at X2H016Q01.

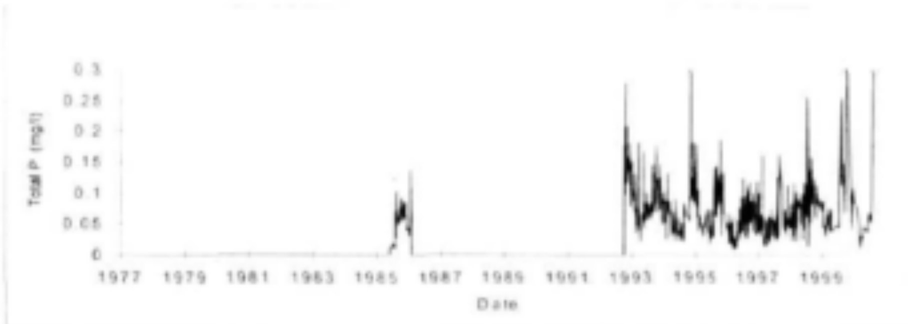

Figure A8 Total phosphorus at X2H016Q01.

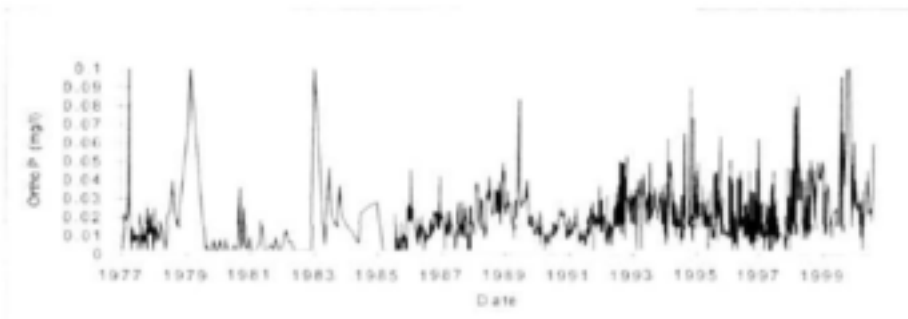

Figure A9 PO_4 at X2H016Q01.

Table A17 Nutrients at X2H016Q01 for the periods 1977 – 1979 and 1996 – 2000.

	Nutrients (5th %tile – median - 95th %tile) mg/L			
	$NO_2 + NO_3$	**NH_4**	**TP**	**PO_4**
1977 - 1979:	0.02 - 0.30 - 0.96	0.02 - 0.05 - 0.21	N/A	0.003 – 0.01 – 0.07
1996 - 2000:	0.02 - 0.49 - 0.93	0.02 - 0.02 - 0.08	0.02 – 0.05 – 0.12	0.006 – 0.02 – 0.05

Table A18	Historical SASS scores and nutrients (mg/L). * : Monthly median at X2H016 ** : MacMillan 2001 –: Thirion 2001								
Site	**Date**	**SASS**	**ASPT**	**State**	**NO_2+NO_3**	**NH_4**	**TP**	**PO_4**	**TIN:PO_4**
CR29**	6/1992	104	4.7	Moderately to largely modified	0.06*	0.07*	-	0.011*	61*
CR29**	7/1992	118	5.1	Moderately modified	0.60*	0.07*	-	0.015*	45*
CR29**	9/1992	139	5.1	Largely natural to moderately modified	0.20*	0.07*	-	0.022*	12*
CR29**	10/1992	81	4.5	Moderately to largely modified	0.10*	0.06*	-	0.025*	6*
CR29–	7/1993	111	5.3	Moderately modified	0.07	0.09	-	0.019	8
CR29–	8/1993	118	5.1	Moderately modified	0.04	0.05	-	0.007	13
CR29–	10/1993	96	4.4	Largely modified	0.08	0.08	-	0.022	7
CR29–	6/1994	122	4.9	Moderately modified	0.04	0.08	-	0.007	17
CR29–	9/1994	66	4.4	Largely modified	0.11	0.04	-	0.017	9
CR29–	6/1995	77	4.8	Largely modified	0.07	0.04	-	0.020	6
CR29–	9/1996	78	7.8	Moderately modified	0.38	0.04	0.05	0.015	28
CR28–	9/1996	133	7.0	Largely natural	0.49	0.04	0.07	0.019	28
CR30–	9/1996	146	8.1	Natural	0.69	0.04	0.05	0.016	46
CR30–	5/1999	148	7.4	Natural	0.59*	0.02*	0.04*	0.016*	38*
CR30–	7/1999	159	6.6	Largely natural	0.81*	0.02*	0.04*	0.017*	49*
CR30–	9/1999	182	5.9	Natural	0.50*	0.02*	0.05*	0.020*	26*

Table A19	Reference conditions for nutrients.			
	Nutrients (5th %tile – median - 95th %tile) mg/L			
	NH_3	**TP**	**PO_4**	**TIN/TP**
Reference conditions (A)	0.008 - 0.008 - 0.03	0.01 – 0.04 – 0.1	0.005 – 0.015 – 0.04	20:1 - 15:1 - 10:1

2.1.1.5 *Total dissolved salts*

Total dissolved salts (TDS) leads to chronic and acute physiological effects and affects buffering capacity and metabolism. The generic dose-response relationship suggested by DWAF (1996a) is a maximum deviation of 15% from natural and the maintenance of natural cycles. The specification provided for resource directed measures (DWAF 1999) rates 0 - 163 mg/L as natural (A), 163 - 228 mg/L as largely natural (B), 228 - 325 mg/L as moderately modified (C) and 325 - 520 mg/L as largely modified (D).

The historical values for TDS at X2H016Q01 are provided in **Figure A10**. **Table A20** shows that invertebrate communities were generally in a moderately - to largely modified state between 1992 and 1995 and in a natural to largely natural state from 1996 to 2000. It can be concluded that the changes in biotic state may have been affected by TDS, since TDS was significantly different during these periods (**Table A21**). The 1996-2000 TDS values represent a largely natural-to-natural state (**Table A20**). The reference conditions for TDS are thus adjusted from the generic knowledge by specifying the highest TDS values during natural conditions as the 95th percentile and the range according to the earliest data (**Table A22**).

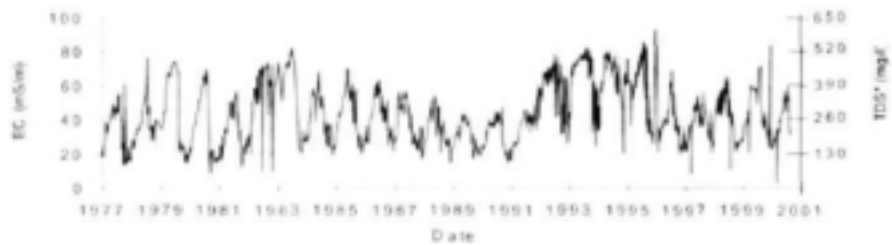

Figure A10 EC (TDS) at X2H016Q01. (*TDS = EC x 6.5).

Table A20 Historical SASS scores, EC (mS/m) and TDS (mg/L).
*Monthly median at X2H016 and TDS = EC x 6.5
** : MacMillan, 2001
–: Thirion, 2001

Site	Date	SASS	ASPT	State	EC	TDS$^{\#}$
CR29**	6/1992	104	4.7	Moderately to largely modified	63.7*	414*
CR29**	7/1992	118	5.1	Moderately modified	69.0*	449*
CR29**	9/1992	139	5.1	Largely natural to moderately modified	71.5*	465*
CR29**	10/1992	81	4.5	Moderately to largely modified	71.3*	464*
CR29–	7/1993	111	5.3	Moderately modified	-	636
CR29–	8/1993	118	5.1	Moderately modified	-	572
CR29–	10/1993	96	4.4	Largely modified	-	480
CR29–	6/1994	122	4.9	Moderately modified	-	491
CR29–	9/1994	66	4.4	Largely modified	-	437
CR29–	6/1995	77	4.8	Largely modified	-	360
CR29–	9/1996	78	7.8	Moderately modified	-	369
CR28–	9/1996	133	7.0	Largely natural	-	366
CR30–	9/1996	146	8.1	Natural	-	404
CR30–	5/1999	148	7.4	Natural	38.8*	252*
CR30–	7/1999	159	6.6	Largely natural	59.0*	384*
CR30–	9/1999	182	5.9	Natural	58.3*	379*

Table A21 Historical values for EC at X2H016Q01.	
	EC (5th %tile – median - 95th %tile) mS/m
1977-1979	15.8 – 37.5 – 72.1
1992-1995	34.2 – 65.6 – 79.7
1996-2000	22.1 – 39.0 – 60.7

Table A22 Reference conditions for TDS.	
	TDS (5th %tile – median - 95th %tile) mg/L
Reference conditions (A)	100 - 250 – 400

2.1.1.6 Total Suspended Solids

Suspended solids affect water temperature, reduce visibility (consequently affecting predation) and lead to physiological effects such as gill efficiency. The target water quality range (DWAF, 1996a) suggests that change is less than 10% from natural and natural cycles are maintained (where background is <100 mg/L). Modifications for natural are specified as follows (DWAF, 1999):

Largely natural (B)	< 15% deviation from natural
Moderately modified (C)	< 20% deviation from natural
Largely modified (D)	< 25% deviation from natural

The historical values for TSS at X2H016Q01 provided in **Figure A11**. **Table A23** shows that invertebrate communities were generally in a moderately - to largely modified state between 1992 and 1995. The TSS measured between 1992 and 1995 (**Table A24**) was thus sufficient to maintain the system at a moderately to largely modified state (but probably not necessary). The monthly analysis of TSS at X2H006 before 1966 (**Figure A12**) indicates unimpacted conditions and seasonal variability. Downstream conditions would not deviate significantly from these values and they are therefore set as the reference values (**Table A25**).

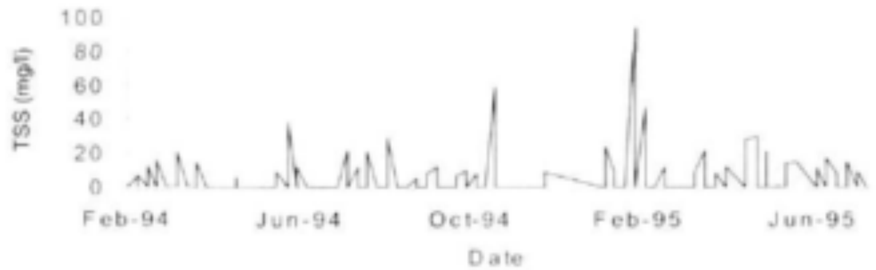

Figure A11 TSS at X2H016Q01.

Table A23	Historical SASS scores and TSS (mg/L). *Monthly median at X2H016 ⤚: River Health Programme				
Site	**Date**	**SASS**	**ASPT**	**State**	**TSS**
CR29⤚	6/1994	122	4.9	Moderately modified	25*
CR29⤚	9/1994	66	4.4	Largely modified	8*
CR29⤚	6/1995	77	4.8	Largely modified	8*

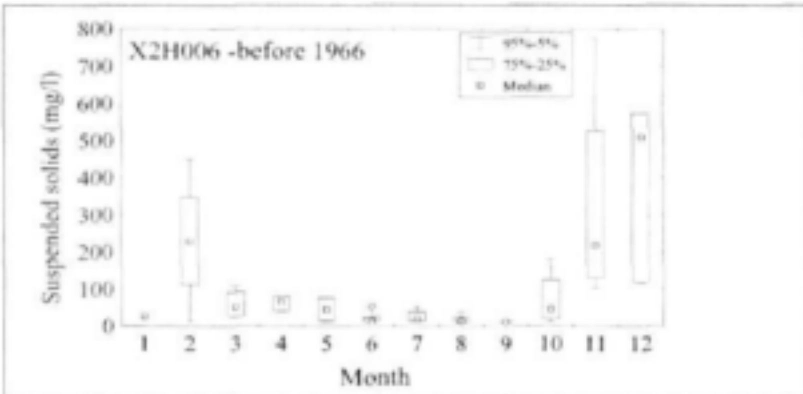

Figure A12 TSS at X2H016Q01.

Table A24	Historical values for TSS at X2H016Q01. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)			
	TSS (mg/L): 5th %tile – median – 95th %tile			
	Summer	**Autumn**	**Winter**	**Spring**
1992-1995	6 – 12 – 80	8 – 15 – 26	1 – 12 – 30	6 – 8 – 11

Table A25	Reference conditions for TSS. Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)			
	TSS (mg/L): 5th %tile – median – 95th %tile			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions (A)	13 – 229 – 775	26 – 65 – 107	13 – 14 – 77	13 – 20 – 182

2.1.1.7 Toxics

Toxics lead to acute and chronic effects through a wide range of mechanisms. The limits (DWAF, 1996a; TWQR) proposed for lead and residual chlorine levels are 0.5µg/L and 0.2µg/L respectively, with the chronic effect values (CEV) being 1.0µg/L and 0.35µg/L and the acute effect values (AEV) being 7.0µg/L and 5.0µg/L respectively (hardness: 60-119 mg/L). The specifications for RDM (DWAF, 1999) are as follows:

Natural (A)	90% < TWQR, 99% < CEV, 100% < AEV
Largely natural (B)	95% < CEV, 99% < AEV
Moderately modified (C)	75% < CEV, 90% < AEV
Largely modified (D)	50% < CEV, 85% < AEV

The historical values for lead at CR28, CR29 and CR30 are provided in **Table A26**. Chlorine concentrations have not been recorded in the study area. Geological sources are not expected to contribute significantly to lead (**Table A6**). The levels reported in **Table A26** are expected to be at or above the natural background. This was also the detection limit for the specific analyses, therefore the natural level can be stated as < 0.02 µg/L. Chlorine does not occur naturally in the environment and the reference condition is thus set at 0 µg/L.

Table A26 Historical values for lead.

	Lead (diss. µg/L)
CR28: Sept 1996	0.02
CR29: Sept 1996	0.02
CR30: Sept 1996	0.02

Table A27 Reference conditions for lead and chlorine.

	Lead	**Chlorine**
Reference conditions (A)	<0.02µg/L	0 µg/L

2.1.2 Ecological Reserve

The DEMC for quaternary X24H is C (indicated with light shading), although specifications for all categories of the Reserve are provided for comparative purposes. The present state for each variable is indicated with "Ⓟ".

2.1.2.1 Temperature

The temperatures specified in **Table A8** were sufficient to maintain the ecosystem in the states listed. Given this relationship and what is known generically about the effect of temperature on aquatic ecosystems, the temperatures that would maintain different ecological states are provided in **Table A28**. The 95^{th} percentile for temperature is limited to 35°C, due to requirements of specific species, physiological requirements (oxygen depletion) and evidence from historical data (**Figure A3 and A4**). The deviation for minimum temperatures during winter and spring is greater than the generic 20%, but is realistic in the context of historical data.

Table A28	Ecological Reserve for water temperature*. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct) * : Temperatures fluctuations during the diel cycle are incorporated into the ranges. Minimum early morning temperatures are guided by the 5th percentile and late afternoon temperature by the 95th percentile. ** : Where a median range is specified, the 5th percentile is relative to the lower median and the 95th percentile relative to the higher median.			
	Temperature (°C) : 5th %tile – (median range)- 95th %tile **			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions (A)	22-26-32	21-25-28	16-19-22	20-23-27
Largely natural (B)	21-(25-27)-33	20-(24-26)-30	14-(18-20)-23	17-(22-24)-28
Moderately modified (C) Ⓟ	19-(23-29)-34	18-(22-28)-32	12-(17-21)-24	14-(20-26)-30
Largely modified (D)	17-(21-31)-35	16-(20-30)-35	10-(15-23)-26	11-(18-28)-32

2.1.2.2 *pH*

Applying the generic dose-response relationship to the reference conditions (**Table A13**) and considering actual monitoring data (**Table A12**) specifications are derived that would maintain the system in the different ecological states (**Table A29**).

Table A29	Ecological Reserve for pH*. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct) * pH fluctuations during the diel cycle are incorporated into the ranges. Minimum early morning pH is guided by the 5th percentile and late afternoon pH by the 95th percentile.			
	pH units : 5th %tile – (median range) - 95th %tile			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions (A)	6.9 - 7.2 – 7.7	6.8 – 7.1 – 7.4	7.2 – 7.6 – 8.1	7.3 - 7.6 – 7.9
Largely natural (B)	6.5-(6.8-7.6)-8.1	6.4-(6.7-7.5)-8.0	6.8-(7.2-8.0)-8.5	6.9-(7.2-8.0)-8.3
Moderately modified (C) Ⓟ	6.1-(6.4-8.0)-8.5	6.0-(6.3-7.9)-8.6	6.4-(6.8-8.4)-8.9	6.5-(6.8-8.4)-8.6
Largely modified (D)	5.7-(6.0-8.4)-8.9	5.6-(5.9-8.3)-9.2	6.0-(6.4-8.8)-9.3	6.1-(6.4-8.8)-9.0

2.1.2.3 *Dissolved oxygen*

Generic knowledge suggests that a minimum DO of 40% for a largely modified state. This value is set as the 5th percentile for autumn, and other values adjusted according to the natural seasonal cycle in **Table A30**. The Reserve values (**Table A14**) for other classes are interpolated with generic knowledge and actual measurements (**Table A15**).

Table A30	Ecological Reserve for dissolved oxygen*. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct) * DO fluctuations during the diel cycle are incorporated into the ranges. Minimum early morning temperatures are guided by the 5th percentile and late afternoon temperature by the 95th percentile.			
	Dissolved oxygen (% saturation) : 5th %tile – (median range) - 95th %tile			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions (A)	90-100-110	80-90-100	100-110-120	90-100-110
Largely natural (B) ℗	80-(90-100)-110	70-(80-90)-100	90-(100-110)-120	80-(90-100)-110
Moderately modified (C)	65-(75-85)-95	55-(65-75)-85	75-(85-95)-105	65-(75-85)-95
Largely modified (D)	50-(60-70)-80	40-(50-60)-70	60-(70-80)-90	50-(60-70)-80

2.1.2.4 *Nutrients*

The available data for nutrients in **Table A18** represents conditions from natural to largely modified. The predictions of conditions that would result in a largely modified system based on the generic relationships are supported by this data. The specifications to maintain the ecosystem in these states are presented in **Table A31**.

Table A31	Ecological Reserve for nutrients.			
	Nutrients (mg/L) : 5th %tile – median - 95th %tile			
	NH_3	**TP**	**PO_4**	**TIN/TP**
Reference conditions (A)	0.008 - 0.008 – 0.03	0.01 – 0.04 – 0.1	0.005 – 0.015 – 0.04	20:1 - 15:1 - 10:1
Largely natural (B) ℗	0.016 - 0.016 – 0.05	0.015 – 0.06 – 0.15	0.01 – 0.05 – 0.015	13:1 – 10:1 - 7:1
Moderately modified (C)	0.035 - 0.035 – 0.08	0.02 – 0.08 – 0.2	0.03 – 0.10 – 0.25	9:1 – 7:1 - 5:1
Largely modified (D)	0.08 – 0.08 - 0.16	0.025 – 0.1 – 0.25	0.05 – 0.15 – 0.4	7:1 – 5:1 - 3:1

2.1.2.5 *Total dissolved salts*

The available data for TDS in **Table A20** represents conditions from natural to largely modified. The predictions of conditions based on the generic relationships are not supported by the data, since a TDS value of 636 mg/L was measured in conjunction with a moderately modified system. The conditions for largely modified conditions were adjusted according to the expectations in **Table A20** and conditions for other ecological states specified according to the non-linear relationship suggested from generic knowledge (**Table A32**).

Table A32 Ecological Reserve for TDS.	
	TDS (mg/L) : 5th %tile – median - 95th %tile
Reference conditions (A) ℗	100 – 250 – 400
Largely natural (B)	120 – 300 – 480
Moderately modified (C)	150 – 375 – 600
Largely modified (D)	200 – 500 – 800

2.1.2.6 Suspended solids

The reference conditions for TSS (**Table A25**) are used as a basis to derive specifications for other ecological states (**Table A33**), by using the deviations specified in the generic dose-response relationship.

Table A33 Reference conditions for TSS. (Summer: Nov-Feb; Autumn: Mar-Apr; Winter: May-Aug; Spring Sep-Oct)				
	TSS (mg/L): 5th %tile – median – 95th %tile			
	Summer	**Autumn**	**Winter**	**Spring**
Reference conditions (A)	13 – 229 – 775	26 – 65 – 107	13 – 14 – 77	13 – 20 – 182
Largely natural (B)	15 – 263 - 890	30 – 75 - 123	15 – 16 - 89	15 – 23 – 209
Moderately modified (C) ℗	16 – 275 - 930	31 – 78 - 128	16 – 17 - 92	16 – 24 –218
Largely modified (D)	16 – 286 - 969	33 – 81 - 134	16 – 18 - 96	16 – 25 - 228

2.1.2.7 Toxics

The available data do not suggest that the tolerance of the system is different to that expected based on generic knowledge. The specifications (**Table A34**) are thus based on these values (DWAF 1996a; DWAF 1999).

Table A34 Ecological Reserve for toxics*. * Specified as the % time that the given values should not be exceeded		
	Lead (µg/L)	**Residual chlorine (µg/L)**
Reference conditions (A) ℗	90% < 0.5, 99% < 1.0, 100% < 7.0	90% < 0.2, 99% < 0.35, 100% < 5.0
Largely natural (B)	95% < 1.0, 99% < 7.0	95% < 0.35, 99% < 5.0
Moderately modified (C)	75% < 1.0, 90% < 7.0	75% < 0.35, 90% < 5.0
Largely modified (D)	50% < 1.0, 85% < 7.0	50% < 0.35, 85% < 5.0

2.2.1 Integration

The lower Crocodile River receives water from the adjacent aquifer (gaining river), with the groundwater being of good quality. There is no evidence of the flow being reversed through abstractions. The relationships between flow and water quality variables were investigated, with seasonal analysis being done where such a relationship is evident. The upstream resource unit (X24F) is of better quality than X24H, with the EMC being higher. A water quality Reserve in this unit should also be protective of the current unit (X24H). The downstream unit (X13L) is of similar quality to the current unit, and would therefore not be compromised by the Reserve. The proximity of an international border would have implications for relevant water quantity obligations and quality requirements for basic human needs. The confidence in the assessment, based on the availability and characteristics of data and the methods followed, is as follows:

Temperature	High
pH	High
Dissolved oxygen	Low
Nutrients	High for natural, Moderate for largely modified
TDS	Moderate
TSS	Moderate
Toxics	High

3.1 EVALUATION

The method employed for the Reserve determination is summarised in **Figure A13**. The geographical boundaries of the study were set by regulatory requirements, with a technical evaluation confirming its relevance. The resource unit was based on Level II ecoregion information and cathment characteristics. This approach was proven sufficient to derive a resource unit with high confidence that it is sufficiently homogenous to apply the same Reserve specifications. The reference conditions were based on historical water quality data, with biological monitoring data and generic dose-response knowledge being employed where unimpacted data were insufficient. A limitation in this regard was the limited dose-response data for species relevant to South African aquatic ecosystems. The water quality database from DWAF (DWAF 2001) could be used to derive reference conditions for most variables with high confidence. The desktop Reserve assessment was used as a basis for ecological importance and sensitivity ratings, although it was verified during field assessments. The present ecological state of the resource units was based on biological monitoring results. The management class was based on the best DEMC as specified in the desktop assessment and also verified in the field. The specifications for the Reserve were derived through comparing actual biological monitoring data with water quality measured at the same time. This approach proves sufficiency, but not necessity. Generic knowledge was employed where evidence from actual monitoring was lacking. The approach provided specifications that are more relevant to the specific ecosystem than generic knowledge would suggest, with values often being more lenient. The implementation of the River Health Programme would support the determination of the water quality Reserve in future. The present state for each water quality variable was based on the derived specifications for each ecological state.

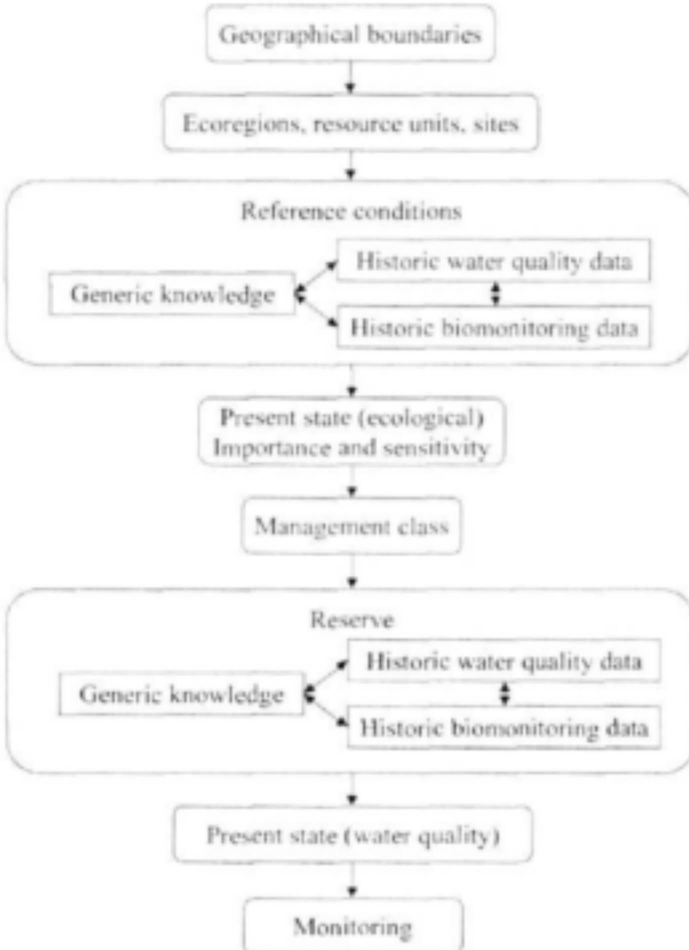

Figure A13 Framework used for developing specifications for the ecological Reserve.

4.1 ACKNOWLEDGEMENTS

The following people are thanked for their inputs to the study: T. Bokako, A. Deacon, D. Grobler, S. Jooste, P. MacMillan, E. van Niekerk, M. von Mollendorf, S. Oelofse, C. Thirion and J. Wentzel.

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APPENDIX 2

THE USE OF SALINITY TOLERANCE DATA IN THE CLASSIFICATION OF WATER QUALITY REACHES

Scherman P-A, Muller WJ and Palmer CG
UCEWQ, IWR, Rhodes University, Grahamstown

The approach of integrating ecotoxicological data in ecological Reserve determinations has been published:

Scherman P-A, Muller WJ, and Palmer CG (2003) Links between ecotoxicology, biomonitoring and water chemistry in the integration of water quality into environmental flow assessments. *River Research and Applications* **19:** 483-493.

In summary:
Organisms are collected from representative rivers/river reaches, and exposed to concentrations of selected salts following the protocol of DWAF (2000). If land-use and analysis of water chemistry data indicates that salinisation is a problem, the likely cause is evaluated. In agriculturally impacted areas sodium chloride is used as the salt toxicant; where mining is the impact, and sulphate is present, sodium sulphate is used as the toxicant.

The results of tolerance experiments are used to set salinity boundaries for each class:

Specific toxicity endpoints related to ecological health classes		
Ecological Health Class		**Ecotoxicity endpoint**
A	Natural	Lowest measurable 10 day toxicity endpoint: < lower 95% confidence limit of the LC1
B	Good	Measured 10 day (chronic) toxicity = 10 day LC1
C	--------------------	
D	Fair	Measured 4-day (acute) toxicity = 96 hour LC1
E/F	Poor	> 96 hour LC1

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FM Chutter

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The English system has been modified to suit South African conditions, widely tested and renamed SASS (South African Scoring System). The SASS index offers a cost-effective method for assessing the health of the macro-invertebrate community in a river. During the development of SASS, the methodology was widely discussed, and it is now routinely being used as part of the biological monitoring of river health. It is also one of the indices which form the basis of the National River Health Programme.

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