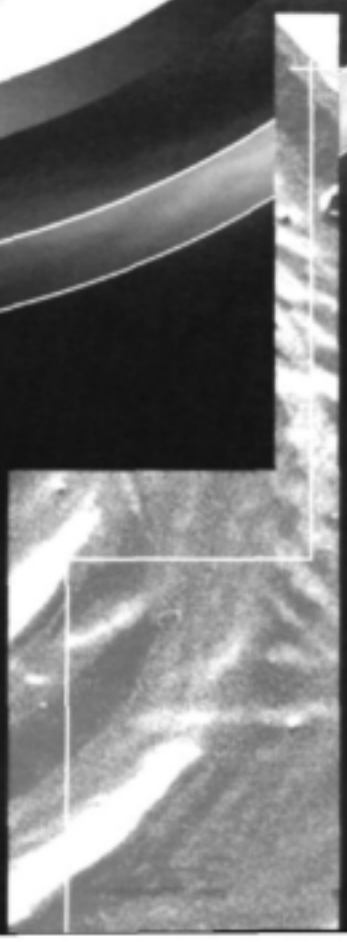
WETLAND WATER QUALITY AND THE ECOLOGICAL RESERVE

HL Malan · JA Day

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Water Research Commission

RESERVE WATER QUALITY
VOLUME 1

WETLAND WATER QUALITY AND THE ECOLOGICAL RESERVE

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Water Research Commission
by

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This report emanates from the WRC research project K5/1310 entitled:

Refinement of aspects of the reserve determination methodology
for water quality, including the assessment of potential impacts
to aquatic biota

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EXECUTIVE SUMMARY

A Water Research Commission-funded project titled "*Refinement of aspects of the Reserve determination methodology for water quality including the assessment of potential impacts to aquatic biota*" was undertaken from July 2002 to December 2004. Two reports emanate from this work:

- Volume 1: "*Wetland water quality and the ecological Reserve*" (Malan and Day 2005), including WetlandsJet, an Access database on CD; and
- Volume 2: "*Assessment of trophic status in aquatic resources with particular reference to the water quality Reserve*" (Malan and Day 2005).

Because the subjects covered in the two research thrusts of the project are somewhat disparate, they have been recorded in separate documents (each addressing separate objectives – see below). But at the same time, there is also some overlap in the topics, and for this reason the same executive summary precedes each.

OBJECTIVES

The objectives of this project were to:

A) *To assess the methodology and the availability of tools (and data) for carrying out water quality Reserve determinations in wetlands and where possible, develop tools/methodology still outstanding;*

B) *To refine existing Reserve protocols for nutrients by;*

i) recommending the nutrient criteria (e.g. total phosphate, dissolved phosphate, total nitrogen or a ratio of two or more nutrients) that should be used to assess the nutrient status of a river or wetlands. A critique of the method now being used will be included;

ii) validating (or otherwise) the assumption in the Reserve methodology that the Reference Condition (i.e. the un-impacted state) levels of nutrients in all ecoregions are equal to or less than the Target Water Quality Ranges as given in the South African Water Quality Guidelines.

Objective A is addressed in **Volume 1**, and the objectives grouped under **Objective B** are considered in **Volume 2**.

RATIONALE

The aim of this project is to fill in some gaps in the methodology for the determination of the water quality part of the ecological Reserve. Although the basic ideas and concepts of determining the Ecological Reserve are documented in DWAF (1999a,b,c,d) there are some areas that require more attention. This may be either because a proper methodology for that step has not yet been developed (in the case of the water quality method for wetlands) or because it has, but it needs further evaluation (in the case of the nutrient assessment method). By far the greatest effort was spent on development of methods for setting the water quality component of the ecological Reserve for wetlands. Less effort was directed towards the assessment of trophic status in aquatic resources (rivers, lakes and wetlands) because considerable progress has already been made nationally in this field.

Wetlands water quality

Wetlands are an important, but neglected, component of the limited water resources in South Africa. The National Water Act (1998) recognises that both the water quantity and water quality aspects of the ecological Reserve need to be determined for this type of aquatic ecosystem. The method for water quality, however, is rudimentary and procedures for determining this aspect of the Reserve need to be developed and refined. At the same time, there is considerable pressure on these systems with regard to development in the surrounding catchments, as well as with regard to the discharge of effluent into them.

Nonetheless, setting water quality specifications for wetlands is not a straight–forward process. Wetlands are naturally variable with regard to water quality – both spatially and temporally. This is a consequence of the many different types of wetlands, differing in (amongst other attributes) water chemistry and geology, and also because levels of inundation (and hence the dilution capacity) vary with time. In addition, water quality data are limited, and in general we have only a poor understanding of the ecological functioning of these systems. A further complicating factor to this project is, that any method that is developed to determine the water quality Reserve for wetlands, and consequently any water quality specifications that are derived, need to be linked to different levels of utilization of the resource. Thus, whatever system for setting water quality standards is developed, it needs to address the problem of allowing maximum benefits from the aquatic resource, but at the same time ensuring sustainable use and minimised impacts.

Assessment of trophic status

There are many different water quality variables that can be measured to describe the nutrient status of an aquatic ecosystem. These range, in the case of phosphorus from total phosphorus (TP) to dissolved forms containing this element (soluble reactive phosphorus, ortho-phosphate). In the case of nitrogen there are also many different chemical species (examples are, nitrate, nitrite, ammonia/ammonium, total inorganic nitrogen (TIN)), all of which can be measured. Furthermore, biotic response variables such as the measurement of chlorophyll *a*, the biomass of phytoplankton, or dissolved oxygen concentrations are also potentially useful assessment indicators. In the original method as given in the RDM manual (DWAF 1999a) ratios of total inorganic nitrogen to total phosphate, or total phosphate to ortho-phosphate were used to describe the nutrient status of rivers. This was often problematic, since data on some of these variables (especially total phosphate) were often lacking at a site. Although this method has been superseded by a new approach, there is still a need to evaluate what criteria should be used to assess nutrient status and the validity of the new method.

STRUCTURE OF THE REPORTS

A brief outline of each of the two reports is given below:

Volume 1: "Wetland water quality and the ecological Reserve"

- **Chapter 1, Introduction:** explains the original objectives for the project, how and why these were amended, and presents the final objectives. The rationale for undertaking this study is discussed and an explanation of the overall approach used to achieve the objectives is given;
- **Chapter 2, Literature review:** presents a wide scope of issues pertinent to water quality and wetlands. Topics covered include; the types of wetlands found in South Africa and their water chemistry, benefits to humans, ecological functioning, and causes of degradation. The project is contextualised by providing a discussion of DWAF's Resource Directed Measures (RDM) methodology as it applies to wetlands. The topic of water quality standards that have been developed internationally is also explored;
- **Chapter 3, Wetland water quality ranges in South Africa:** discusses the need for, the development of and the attributes of the database developed during the project. Results from interrogation of the database are presented, including the ranges of water quality variables (the concentration of chemical constituents and values of physical variables) exhibited by different wetland types, and the effect of geographic region on these ranges;

- **Chapter 4, Wetland water quality objectives:** explains theoretical approaches to setting water quality objectives, and some of the inherent problems and limitations. A Decision Support System (DSS) is formulated which gives a proposed method of how to set the water quality specifications for a given wetland. A description of the "best-guess" default values for the key water quality variables, how these were derived, when and how they should be used, and the limitations to this approach, is provided;
- **Chapter 5, Application of the DSS:** describes how the DSS developed in this project can be applied to real wetlands in order to determine the water quality component of the Reserve. The systems investigated are; Nylsvlei, Rocher Pan and Verlorenvlei;
- **Chapter 6, Major results, conclusions and recommendations:** summarises the major results and products of the wetland part of the project. Recommendations for further work are discussed.

Volume 1 is distributed with a CD containing "WetlandsJet", a Wetland Water Quality Database which was compiled as one of the products of the WRC project K5/1311 by Malan and Day (2005).

Volume 2: "Assessment of trophic status in aquatic resources with particular reference to the water quality Reserve."

This report considers the refinement of nutrient assessment methods within the ecological Reserve process. It presents a review of the indicators used internationally to describe the nutrient status of an aquatic resource, and discusses the latest method prescribed for use in South African rivers. A critical evaluation is given of the Target Water Quality Range (TWQR) values for nutrients that are given in the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF 1996).

APPROACH USED IN THIS STUDY

Wetlands water quality

The following approach was used to achieve Objective A (setting the water quality Reserve for wetlands):

1) a literature review was carried out to identify the important issues relevant to wetland water quality and to establish how other countries manage this aspect;

2) an attempt was made to establish the extent and availability of water quality data for wetlands in South Africa and to collate these data;
3) a database was developed to house the above water quality data;
4) the database was interrogated and the range and median values of key water quality variables examined. How these varied with wetland type, climatic region and season was also investigated;
5) from theoretical approaches and examination of water quality data-sets for several wetlands a Decision Support System for setting the water quality Reserve was designed;
6) using the information obtained from step 4), in conjunction with standards from the literature, "best-guess" default values for key water quality variables were drawn-up. These values were verified by the scrutiny of wetland experts in the country;
7) the DSS was applied to several wetlands, and the method amended as required.

Assessment of trophic status

The approach used to achieve Objective B (assessment of nutrients in aquatic ecosystems) is detailed below:

1) a literature review was carried out to assess how the nutrient status of aquatic ecosystems is described internationally;
2) this was followed by critical examination of the latest method for assessing nutrients (DWAF 2002);
3) reference condition values of nutrients were examined at different sites on rivers in different ecoregions. The sites were first screened for possible impacts by examining the land use in the catchment and using biomonitoring (SASS) data;
4) a comparison was made between the reference condition values of nutrients obtained above and the South African Water Quality Guidelines for aquatic ecosystems;
5) conclusions regarding the validity of the guidelines were made.

THE MAJOR PRODUCTS

The major products emanating from this project are:

- a literature review of water quality and wetlands;
- a database (Access 2000) of water quality and other information for South African wetlands (on a CD at the back of this document);
- an analysis of some of the major trends and patterns in water quality in South African wetlands;

- a Decision Support System to aid in setting the water quality component of the wetland ecological Reserve;
- a review of some issues concerned with nutrients in aquatic ecosystems.

THE MAJOR RESULTS

The major results obtained were those arising from interrogation of the Wetland Water Quality database.

- The lowest median values for electrical conductivity (EC), phosphate and ammonia in all the types of wetland examined, were found in springs ("eyes"). This is perhaps to be expected because the data are all from DWAF and many are used as sources of potable water in arid areas.
- Lakes were significantly different from all non-impacted wetland types with regard to EC, but were not different to impacted wetlands. Lakes were thus higher in conductivity than springs, but lower in conductivity than endorheic and palustrine systems.
- Endorheic wetlands (pans) showed the highest median conductivity of all the wetland types.
- Preliminary results indicate that for some wetlands, anthropogenic impacts have taken the form of dilution, rather than concentration of some water quality constituents (although these results were not statistically significant).
- The range in median pH values for the different wetland types in the non-impacted state was small and pH tended to be circumneutral. There were no significant differences between the non-impacted wetland types. Impacted lakes, on the other hand, exhibited a significantly higher median pH value than all other groups (a likely consequence of eutrophication).
- The highest phosphate levels exhibited by non-impacted wetlands were found in palustrine systems and these were significantly different with regard to all other natural wetland systems. The difference between palustrine and *impacted* wetlands, however, was not significant.
- The median nitrate concentrations exhibited by the non-impacted wetlands (lacustrine, palustrine and endorheic systems) did not vary significantly between the different types. Nitrate levels in the water column of impacted wetlands on the other hand, were significantly higher than in natural systems.
- The lowest median EC values were exhibited by the wetlands that fell into the "All year, summer maximum" (the eastern coastal) region of the country.

This area was statistically different from all the other rainfall regions. This region is one of the most mesic in the country and thus the "dilution capacity" is likely to be high.

- The highest median EC values (1 985 mS/m) were exhibited by the wetlands of the summer rainfall region and this value was significantly higher than all other wetland groups.
- It was concluded that whilst there seems to be ranges of water quality variables characteristic of different rainfall regions, it is difficult to pick out clear patterns. Water quality also varies with wetland type and this is likely to confuse any regional patterns.

CONCLUSIONS ARISING FROM THIS PROJECT

Wetland water quality

The following major conclusions were reached with regard to Objective A (setting the water quality Reserve for wetlands) in this project.

- There is a lack of water quality monitoring data for wetlands, in conjunction with limited research that has been carried out into the fundamental understanding of ecological functioning of these systems. Until the knowledge base concerning wetland water quality and ecological response is increased, the ecological status of a wetland under a particular water quality regime cannot be predicted with any certainty. As a consequence the DSS developed in this project is only a first step towards development of an effective water quality approach for wetlands.
- For non-marine-influenced wetlands there are four basic wetland water quality types, based on drainage pattern and the water source. These are riverine, runoff-fed, and groundwater-fed systems and freshwater lakes.
- The 'best-guess" default values should be used only where no site-specific data are available since objectives defined in this manner represent a low level of confidence. Whatever the approach used, a monitoring programme would need to be implemented to ensure that water quality objectives were appropriate and to ensure compliance.
- Useful approaches to setting water quality guidelines for wetlands were obtained from examination of international practices, especially those of the USA and Australia/New Zealand. No approach, however, was transferable in its entirety to the South African situation, but rather a combination of methods was considered to be the most useful.

- Narrative statements may be useful as a management tool, until the links between concentration and ecosystem response are better understood.

Assessment of trophic status

The following major conclusions were reached with regard to Objective B (assessment of nutrients in aquatic ecosystems) in this project.

- From analysis of the data it would appear that the current Target Water Quality Range (TWQR) of 0.005 mg/L for inorganic phosphate as specified in DWAF (1996a) is too conservative. From consideration of the 75%ile for the DWAF chemical monitoring data and from the data collected during biomonitoring, a more reasonable value would be around 0.02 mg/L (rather than 0.005mg/L). With regard to Total Inorganic Nitrogen, it would appear that the value of 0.5 mg/L is possibly too high and that a value of around 0.25 – 0.3 mg/L should be used. Because of the critical role that phosphate plays in limiting eutrophication, however, it is important that although realistic values be set, these should err on the side of conservatism (see below).
- The detection limits for DWAF during the time period from which the data were extracted was 0.011 mg/L for soluble reactive phosphorus (SRP) and 0.04 mg/L for combined nitrate and nitrite. Thus records with measurements lower than these values (roughly 42% of the SRP and 75% of TIN data points) may be unreliable in terms of the exact concentrations determined. Given this uncertainty, **the present TWQR values should be retained, especially with regard to SRP,** until the detection limits of routine sample analyses have been improved.
- It should be noted that in the revised method for assessing the present state for nutrients (Approach 2, DWAF 2002), the default value of TIN used to differentiate between the Natural and Good categories is 0.25 mg/L (and 0.005 mg/L for SRP). A mechanism is included in the assessment method to take into account *naturally* high nutrient levels. Thus, if the Reference Condition level of a nutrient is higher than the TWQR, this can be accommodated and the entire set of boundary values used to delineate management categories is adjusted accordingly. Nevertheless the results from the current project are useful in that it indicates that most values for SRP would need to be adjusted.

RECOMMENDATIONS AND FUTURE RESEARCH AREAS

Wetlands water quality

The recommendations and topics for further research in wetland water quality are presented below.

- Development of assessment techniques, and if possible a rapid biomonitoring system for water quality in wetlands should be a high research priority. This should be underpinned by research into abiotic stressor-biotic response relationships, where water quality is one of the abiotic stressors included.
- The DSS developed in this project to facilitate setting water quality objectives for wetlands needs to be refined through use by wetland specialists. The particular aspects, *inter alia*, that requires further investigation are:
    - the recommended allowable % deviation from the median,
    - the use of the 80%ile from a comparable wetland,
    - the default "best-guess" values.
    - the classification system of wetland water quality types.
- It would be useful to continue adding data to the Wetlands Water Quality Database, and in particular, wetlands from regions other than the winter rainfall region should be included. This would aid in making the statistical analysis of wetland water quality more robust. In order to ensure compatibility, however, and to maximize the value, cognizance needs to be taken of the final form of the database used to house the National Wetlands Inventory, so that the two can be merged.
- During interrogation of the Wetland Water Quality database, the correlation between rainfall areas and water quality was investigated. It would be useful to extend this work to ascertain if a link between for example, bioregions or water management areas and water quality could be found. This would further our understanding of water quality in the wetlands of this country.
- It was found that very few of the literature sources that were used to compile the Wetlands Water Quality Database recorded linked (i.e. measured at the same place and time) hydrological and water quality data. The question of linking water quantity and water quality requirements in wetlands is another important topic of research that requires attention.
- It would be useful if during the compilation of the National Wetland Inventory or during extension of the Wetlands Water Quality Database a system of Reference Condition sites could be identified.

These would be wetlands of different types and regions that were un-impacted with regard to water quality (and other aspects e.g. habitat).

- It may also be useful to extend the database to the inclusion of biotic data e.g. the plant, fish or macroinvertebrate species present at a wetland site. These could then be correlated with water quality so that eventually an understanding of the tolerances and relationships between water chemistry and various species of wetland plants and animals could be established.
- In order to limit the levels of pollutants that enter into wetlands, it is suggested that there should be more stringent enforcement of existing legislation (e.g. Conservation of Agricultural Resources Act – act 43 of 1983) protecting natural vegetation adjacent to wetlands.
- The topic of setting the lower limits of water quality objectives for wetlands, specifically for salinity requires further research. Minimum concentrations of inorganic salts need to be incorporated into the DSS.
- Ultimately sediment quality guidelines should be drawn up for aquatic ecosystems, and particularly those such as wetlands in which these materials accumulate. At the moment, however, not enough is known about water quality in general and that of sediments in particular to draw up such guidelines but future work should be directed towards this goal.

Assessment of trophic status

Recommendations arising from Objective B (assessment of nutrients in aquatic ecosystems) are as follows:

- Total N and TP should be used as indicators of rivers, streams, lakes and wetlands in addition to measures of dissolved nutrients. Even seasonal measurements of TN and TP (in addition to more frequent measurement of dissolved nutrients) would be useful.
- Research needs to be carried out to draw up default boundary values between Reserve categories for TP and TN in order to demarcate the assessment categories.
- It is important that there is integration between the water quality and River Health Programme monitoring systems. Collecting measurements of water quality as well as biological data (presence/absence of macroinvertebrate taxa and SASS scores) at the same time and same place maximises the usefulness of both.

CONCLUSION

For effective protection of water quality in wetlands it is essential that reasonably stringent pollution limits be set. At the same time, to enable optimum utilization of those systems for economic development, the water quality objectives should not be too conservative. A method to help determine suitable water quality objectives was developed in this project. The DSS is the result of preliminary deliberations and needs verification by application to many wetlands. It is recognized that "expert knowledge" currently plays a large role in applying this protocol and that the process is not definitive. There is, however, intense pressure on wetlands and many licence applications have, and are being received by DWAF for the discharge of effluents, or for activities that might impact on the water quality of wetlands (R. Stassen, Directorate of Resource Directed Measures, *pers. com.* October 2003). Although not perfect, it is considered by the authors that DSS should at least help to set some limits to protect these assets. Wetlands have long been neglected in South Africa in terms of research and monitoring of all aspects, not just water quality. In a country that is limited in terms of freshwater it is important that attention now be directed towards these vitally important resources. Once more is known about these systems the methods of setting water quality objectives can be improved so that a greater confidence can be placed in the values obtained. Otherwise there is a very real danger of some wetlands being irretrievably damaged if steps are not taken to protect them.

In a similar vein eutrophication of all types of aquatic resources (not just wetlands, but also rivers and lakes), is an increasing problem both in South Africa and world-wide. Furthermore, because enhanced nutrient levels result in shifts in ecosystem functioning many aspects could potentially be measured to assess the trophic status of a water body. It is important that in order to protect aquatic resources the optimal variables are measured. This is further supported by the fact that monitoring of aquatic resources invariables involves expending considerable resources. The current method used to assess trophic status in this country appears to be on a sound footing. It is however important that measurements (even on a seasonal basis) of total nutrient concentration also be recorded.

The two major thrusts of this project are very different, the one directed at setting the water quality Reserve for wetlands, and the other for assessing nutrient levels (primarily in rivers). Nevertheless they are both concerned with managing water quality in aquatic resources. As water becomes even scarcer in this country, and pollution loads increase, managing water quality will become an ever-challenging concern.

Sophisticated tools and new ways of tackling the problem will need to be engaged. The project team hopes that the products developed in the course of this project will go some way in helping to protect South Africa's vulnerable aquatic resources.

WETLAND WATER QUALITY AND THE ECOLOGICAL RESERVE

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ABBREVIATIONS

Chl *a* – chlorophyll *a*
DO – dissolved oxygen
DSS – decision support system
EC – electrical conductivity
EFA – Environmental Flow Assessment
EIS – ecological importance and sensitivity
EWR – Ecological/environmental water requirement
FRU – Freshwater Research Unit
RC – Reference condition
RDM – resource directed methods
RQO – resource quality objective
SASS – South African scoring system
Secc – secchi disk depth
STWs – sewerage treatment works
TDS – total dissolved solids
TIN – total inorganic nitrogen
TN – total nitrogen
TP – total phosphorus
TSS – total suspended solids
TWQR – target water quality range

CSIR – Council for Scientific and Industrial Research
DEAT – Department of Environmental Affairs and Tourism
DWAF – Department of Water Affairs and Forestry
UCEWQ-IWR – Unilever Centre for Environmental water quality – Institute for Water Research, Rhodes University
US EPA – United States Environmental Protection Agency
WRC – Water Research Commission

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CHAPTER 1
INTRODUCTION

1.1 OBJECTIVES OF THE PROJECT

This document forms part of the final report for Water Research Commission (WRC) project K5/1311 "Refinement of aspects of the Reserve determination methodology for water quality including the assessment of potential impacts to aquatic biota". The project lasted for three years, starting in April 2002 and was completed in Dec 2004. Two reports emanate from this work:

- Volume 1: "*Wetland water quality and the ecological Reserve*" (Malan and Day 2005), and
- Volume 2: "*Assessment of trophic status in aquatic resources with particular reference to the water quality Reserve*" (Malan and Day 2005).

The revised objectives of the project were as follows:

A. *assess the methodology and the availability of tools (and data) for carrying out water quality Reserve determinations in* ***wetlands*** *and where possible, to develop tools/methodology still outstanding;*

B. *refine existing Reserve protocols for* ***nutrients*** *by:*

i) *recommending the nutrient criteria (e.g. total phosphate, dissolved phosphate, total nitrogen or a ratio of two or more nutrients) that should be used to assess the nutrient status of a river or wetlands. A critique of the method now being used will be included;*

ii) *validating (or otherwise) the assumption in the Reserve methodology that the Reference Condition (i.e. the un-impacted state) levels of nutrients in all ecoregions are equal to or less than the Target Water Quality Ranges as given in the South African Water Quality Guidelines.*

The findings for **Objective A** are given in the current document **(Volume 1)**. The findings for **Objective B** are presented in **Volume 2** ("*Assessment of trophic status in aquatic resources with particular reference to the ecological Resrve*" by Malan and Day, 2005).

The above objectives are based on those originally laid out in the project proposal which are given below.

THE OVERALL AIM OF THE PROJECTwas to develop tools that are still outstanding and that are needed for carrying out water quality Reserve determinations in rivers and wetlands.

1. *To refine existing Reserve protocols for* ***nutrients*** *by:*

a) *identifying which nutrient criteria (e.g. total phosphate, dissolved phosphate, total nitrogen or a ratio of two or more nutrients) should be used to assess the nutrient status of a river or wetland;*

b) *validating (or otherwise) the assumption in the Reserve methodology that the reference condition (i.e. the un-impacted state) levels of nutrients in all ecoregions are equal to or less than the Target Water Quality Ranges as given in the South African Water Quality Guidelines for freshwater ecosystems (DWAF 1996);*

c) *providing, if necessary, a more appropriate and practicable protocol for evaluating the nutrient status of rivers for the purpose of Reserve determinations.*

2. *To enhance the protection of aquatic biota by:*

a) *refinement of the* ***'Biotic protocol'*** *developed in WRC project K5/956 for assessing potential impacts of water quality on the aquatic biota, and to expand its use in the Reserve determination process;*

b) *integration of data (especially SASS scores) concerning reference sites obtained from the River Health Programme, within the Reserve determination methodology and thus ensure that there is optimal use of all relevant information.*

3. *To assess the entire methodology and the availability of tools (and data) for carrying out water quality Reserve determinations in* ***wetlands*** *and, where possible, to develop tools/methodology that are still outstanding.*

At the inaugural Steering Committee Meeting (Sept 2002) it was pointed out by the project team that subsequent to the submission of the project proposal, various changes had occurred in the water management field in South Africa and specifically within the environmental water quality sector. Furthermore, the need for tools to aid setting water

quality objectives in wetlands was becoming urgent. Because of these developments, it was proposed (and accepted by the committee and the WRC) that changes to the original objectives be made. Thus in this project, Objective 1 from the original objectives was amended, Objective 2 was not addressed at all, and the emphasis of the project was placed on Objective 3. The motivation for these changes is given below.

The Department of Water Affairs and Forestry (DWAF) had already initiated a project to refine the water quality assessment method used in determining the ecological Reserve for rivers – including the protocols for dealing with nutrients (DWAF 2002). Consequently, it was decided that rather than developing a new approach (Objective 1), the latest DWAF nutrient assessment method should be evaluated. Another consequence of the refinement of the Reserve water quality method for rivers was that consideration of biomonitoring data (in addition to other "biotic response variables" such as chlorophyll *a* concentration) is now formally included in water quality assessments. With regard to further development of the "Biotic protocol", although it is considered that this is a useful tool, further development at this stage may well be premature. The regional extent of SASS samples recorded is still limited and although the ecological Reserve has been determined for many rivers in the country, virtually none are being implemented. The "Biotic protocol" can best be refined by examination of biomonitoring data gathered before and after implementation of ecological/environmental flows. Thus Objective 2 was not addressed.

By far the greatest effort in this project was spent on the development of methods for setting the water quality component of the ecological Reserve for wetlands. This emphasis was supported by the fact that, nationally, there is a growing recognition of the urgent need for the protection of wetlands. As a result, the scope of Objective 3 was greatly expanded (see section 1.4) and the major emphasis of the project was shifted towards this aspect. The project team feels that the final objectives and the products emanating from these will be useful in water resource management and represents a more sensible use of the funding than would have been achieved were the original objectives retained.

1.2 RATIONALE FOR THE PROJECT

The primary objective of this project is to fill in some gaps in the methodology for the determination of the water quality part of the Ecological Reserve. Although the basic ideas and concepts of determining the Ecological Reserve are documented in DWAF (1999a,b,c,d) "Resource directed measures for protection of aquatic resources" (referred to hereafter as the RDM manuals), there are some areas that require more attention. This may be either because a proper methodology for that step has not yet been developed (in the case of the water quality method for wetlands) or because it has, but it needs further evaluation (in the case of the nutrient assessment method).

The rest of this document reports the findings and conclusions for **Objective A** (development of methods for setting the water quality component of the ecological Reserve for wetlands). Aspects relevant to **Objective B** (assessment of trophic status) are given in Volume 2 ("*Assessment of trophic status in aquatic resources with particular reference to the ecological Reserve*" Malan and Day 2005).

Wetlands are an important, but neglected, component of the limited water resources in this country. The National Water Act (1998) recognises that both the water quantity and water quality aspects of the ecological Reserve need to be determined for this type of aquatic ecosystem. The method for water quality however, is rudimentary and procedures for determining this aspect of the Reserve need to be developed and refined. At the same time, there is considerable pressure on these systems with regard to development in the surrounding catchment, as well as with regard to the discharge of effluent into them (R. Stassen, *pers. com.*, DWAF RDM office, July 2003). Thus, addressing this deficiency, by setting realistic and sensible water quality specifications (objectives), is one of the major aims of the project.

Nonetheless, setting water quality specifications for wetlands is not a straight–forward process. Wetlands are naturally variable with regard to water quality – both spatially and temporally. This is a consequence of the many different types of wetlands, differing in (amongst other attributes) water chemistry and hydrology, and also because levels of inundation (and hence the dilution capacity) vary with time. In addition, water quality data for these systems are limited, and in general we have only a poor understanding of their ecological functioning.

A further complicating factor to this project is, that any method that is developed to determine the water quality Reserve for wetlands, and consequently any water quality specifications that are derived, need to be linked to different levels of utilization of the resource. Thus, as outlined in the National Water Act (1998), different levels of protection (ecological categories) need to be assigned to aquatic resources, including wetlands, which are linked to different intensities of utilisation of the resource (Chapter 4). Thus, whatever system for setting water quality standards is developed, it needs to address the problem of allowing maximum benefits from the aquatic resource, but at the same time ensuring sustainable use and minimised impacts.

1.3 STRUCTURE OF THE REPORT

A brief outline of each chapter in this document is given below:

- **Chapter 1, Introduction:** (this chapter) presents the objectives for the project and explains how and why they were amended from the original project proposal. The rationale for undertaking this study is discussed, followed by an outline of the major chapters of the document, and an explanation of the overall approach used to achieve the objectives;
- **Chapter 2, Literature review:** presents a wide scope of issues pertinent to water quality and wetlands. Topics covered with regard to wetlands include; the types found in South Africa and their water chemistry, benefits to humans, ecological functioning, and causes of degradation. The project is contextualised by providing a discussion of the RDM methodology as it applies to wetlands. The topic of wetland water quality standards that have been developed internationally is also explored;
- **Chapter 3, Wetland water quality ranges in South Africa:** discusses the need for, the development of and the attributes of the database developed during the project. Results from interrogation of the database are presented, including the ranges of water quality variables (the concentration of chemical constituents and values of physical variables) exhibited by different wetland types, and the effect of geographic region on these ranges;
- **Chapter 4, Wetland water quality objectives:** explains theoretical approaches to setting water quality objectives, and some of the inherent problems and limitations. A Decision Support System (DSS) is formulated which gives a proposed method of how to set the water quality specifications for a given

wetland. A description of the "best-guess" default values for the key water quality variables, how these were derived, when and how they should be used, and the limitations to this approach, is provided;

- **Chapter 5, Application of the DSS:** describes how the DSS developed in this project is applied to real wetlands in order to determine the water quality component of the Reserve. The systems investigated are; Nylsvlei, Rocher Pan and Verlorenvlei;
- **Chapter 6, Major results, conclusions and recommendations:** summarises the major results and products of the wetland part of the project.

1.4 APPROACH USED IN THIS STUDY

The broad objective for this part of the project given in section 1.1 was expanded as follows:

A i) develop water quality objectives (standards) for wetlands in South Africa. In other words to decide upon the;
- maximum (and possibly minimum) allowable concentrations of chemical constituents (e.g. salts, nutrients, toxic substances);
- and the allowable range of physical variables (e.g. temperature, dissolved oxygen, suspended sediments);

A ii) link the above with the DWAF classification system of levels of protection (and wetland use);

A iii) develop a protocol (or a Decision Support System) to apply these objectives (standards) within the Reserve determination procedure.

The above sub-objectives were addressed in Chapters 3, 4 and 5.

The following approach was taken to achieve these objectives:

1) a literature review was carried out to identify the important issues relevant to wetland water quality and establish how other countries manage this aspect;

2) an attempt was made to establish the extent and availability of existing water quality data for wetlands in South Africa and to collate these data;

3) a database was developed to house the above water quality data;

4) the database was interrogated and the range and median values of key water quality variables examined. How these varied with wetland type, climatic region and season was also investigated;
5) from theoretical approaches and examination of water quality data-sets for several wetlands a Decision Support System for setting the water quality Reserve was designed;
6) using the information obtained from step 4), in conjunction with standards from the literature, "best-guess" default values for key water quality variables were drawn-up. These values were verified by the scrutiny of wetland experts in the country;
7) the DSS was applied to several wetlands and the method amended as required.

1.5 TERMINOLOGY USED IN THIS REPORT

The Resource Directed Measures (RDM) approach in South Africa (which includes determining the ecological Reserve and associated activities) makes use of specific terminology. This terminology is in a process of flux, and it was found, for example that whereas for rivers, ecological classes were termed "Natural", "Good", "Fair" etc. (section 2.5.5), for wetlands the terms "Class A", "Class B" etc. were still being used (A. Singh, RDM office, DWAF, *pers. com.*, April 2004). The term water quality "ecospecifications" is a new term now being used during Reserve determinations for rivers (but not apparently, at the moment for wetlands) to specify the maximum concentrations of chemical constituents and ranges of physical variables that should not be exceeded at a given site. In this document the following terms were used:

- water quality objectives – i.e. the maximum (and possibly minimum) concentration of chemical constituents, and the range of physical variables that should not be exceeded in a given wetland;
- Ecological Category (category A, B etc.) – the level of ecosystem integrity for which a system should be managed.

CHAPTER 2
WATER QUALITY AND WETLANDS:
A REVIEW OF THE LITERATURE

2.1 INTRODUCTION

The objective of this literature review is to provide a summary of the relevant information that is currently available concerning water quality in wetlands. A brief discussion is given of the types of wetlands that occur in South Africa, their functions in the landscape, and the threats that they face (with particular reference to pollution). The current status and details of the procedure for setting the ecological Reserve for wetlands is discussed. The review concentrates on water quality in different wetland types and the methods used internationally for setting water quality objectives for these aquatic ecosystems. The topics covered in this review are listed below:

- **section 2.1: Introduction** (this section)
- **section 2.2: Types of wetlands**
- **section 2.3: The importance of wetlands in the environment**
- **section 2.4: Causes of degradation to wetlands.**
- **section 2.5: Resource Directed Measures and wetlands**
- **section 2.6: Physico-chemical conditions in South African wetlands**
- **section 2.7: International water quality regulations for wetlands**

2.2 TYPES OF WETLANDS

2.2.1 Definition

Wetlands represent a very diverse range of ecological systems. This is illustrated by the range of wetland habitat-types recognized by the Ramsar Convention (also known as the "Convention on wetlands of international importance, especially as waterfowl habitat"), which range from estuaries, open coastline, floodplains, freshwater marshes, lakes, peatlands and swamp forests (Dugan 1990). The Ramsar Convention is an international movement that seeks the protection and wise use of wetlands. Because of the diverse range in types, there are many definitions that are commonly in use (over

fifty according to the above author). These ecosystems are defined in the South African National Water Act (DWAF 1998) as follows;

"Wetlands are land which is transitional between terrestrial and aquatic systems, where the water table is usually at, or near the surface, or the land is periodically covered with shallow water and which land in normal circumstances supports, or would support, vegetation adapted to life in saturated soil."

This definition is suited to the types of wetland systems found in South Africa and will be used in this document. Note that this definition includes seasonal or ephemeral pans, a wetland type that is characteristic of arid, or semi-arid regions that cover much of Southern Africa (Allan, Seaman and Kaletja 1995). Whilst distinctive plant and animal communities are characteristic of seasonal and permanent wetlands, ephemeral and saline wetlands are often entirely without vegetation (Davies and Day 1998). From the definition given in the National Water Act, it can be seen that the distinguishing feature of all wetlands is the interplay between the land and water, and the consequent characteristics which reflect both. Standing water or water-logged soils in the landscape can arise from several different causes. These include the periodic flooding of riverine floodplains, tidal fluctuations, impeded surface flow due to geological or geomorphological processes (such as tilting, uplift or landslip, land subsidence, deposition of sediments in estuaries or deltas), the impediment of sub-surface flow by aquicludes, or the rising of the water table to above surface level (Cowan and van Riet 1998). The above refers to naturally-occurring wetlands. Obstruction of water flow by man-made means (e.g. by the construction of berms or roads) is also frequently used to construct artificial wetlands.

Wetlands may be either be permanent (always wet), seasonal (they may not always be wet but become inundated at about the same time each year), or ephemeral (varying from year to year in the degree of inundation and when this occurs). According to Cowan (1995), because the inundation of many wetlands in South Africa tends to be erratic, wetlands of no apparent importance may become significant at different times. They are also biologically very diverse and because water is a scarce resource in South Africa represent a vitally important habitat. A similar sentiment is expounded by Whigham (1999) with regard to the temporary wetlands in the arid areas of the United States

(called "dry-end" wetlands). He makes the argument that such wetlands may be more valuable than other types because of the important landscape and biodiversity functions that they perform, which includes, provision of habitat for migrating birds, improvement of water quality and maintenance of high biodiversity.

South Africa has been divided into 26 wetland regions (based on geomorphology, topography, hydrology, climate and nutrient regime) by Cowan (1995). These are shown in Figure 2.1, and an explanation of the abbreviation for each region is given in Table 2.1. The 26 regions fall into four broad groups based on the general morphology of the country, namely; plateau, mountains, coastal slopes and rimland, and coastal plain. The "nutrient regime" was used as one of the determinants to divide the four major areas into smaller sub-areas. The nutrient regime was determined from a consideration of climate (temperature and humidity zones), and the underlying rock strata. It is, however, not clear exactly how this was done. According to Cowan (1995) it is necessary to validate the above-delineated wetland regions by mapping the distributions of wetland biota.

A major focus of this project is to look at how water quality in South African wetlands varies from region to region. It will be necessary to decide if this shows a pattern and to determine if this pattern is in any way comparable with the regions shown in Figure 2.1, or alternatively with other proposed delineations (e.g. that of Allanson, Hart, O'Keeffe *et al.* 1990). A confounding factor is that wetland types typical of a region are not necessarily exclusive to that region (Cowan and van Riet 1998) and may well occur in other areas.

2.2.2 Classification of wetlands

The diversity of wetland habitats has lead to much discussion concerning cataloging the different wetland types. This topic is covered only briefly in this review, since a detailed discussion of the classification of wetlands is given in Jones (2002). A national wetland classification system has been proposed (Dini, Cowan and Goodman 1999). This method of classification is based on the well-known system of Cowardin *et al.* (1979). This method provides wetland categories that can be compared directly with existing classification systems, such as the Ramsar system. Six types of water body are recognised (rivers, estuaries, marine-, palustrine-, endorheic- and lacustrine-systems), in addition to lagoonal (included under coastal wetlands) and artificial wetlands (including

reservoirs, farm dams, wastewater treatment areas). These main types of wetland can be sub-divided further (Figure 2.2). A detailed discussion of the sub-types is given in Dini *et al.* (1999). The terms used in the methodology for determining the ecological Reserve for wetlands (DWAF 1999b), are also compatible with the above classification system.

The classification of wetlands is made even more confusing by the fact that there are many different names in use, and that the same name may be used for different types of wetlands. For example, Davies and Day (1998) refer to swamps as "wetlands with trees" whilst the Ramsar system (Cowan 1995) refers to shrub swamps and swamps with various types of herbaceous vegetation, but makes no mention of trees. The Resource Directed Measures methodology (DWAF 1999b) recognises the first six types of wetlands mentioned above, i.e.

- **Marine**
- **Estuarine**
- **Endorheic**
- **Palustrine**
- **Riverine**
- **Lacustrine**

In order to avoid complications due to detailed classification systems and because at this stage, there is only limited knowledge of water quality in South African wetlands, only the broad classes of wetlands will be considered in this project. Thus in this project the DWAF system of classification (Figure 2.2) was used only two the system level. Of the six wetland systems shown in the figure (and listed above), only endorheic, palustrine and riverine systems, and to some extent freshwater lakes, were considered. Thus, marine-influenced systems are not addressed in this project.



Fig 2.1 *Wetland regions of South Africa as proposed by Cowan (1995).*

Table 2.1 ***Wetland regions of South Africa (from Cowan 1995)***

REGION	MORPHOLOGY	CLIMATE
P	**Plateau wetland group**	
PW.w	Western plateau	Desert region
PW.sn	Western plateau	Steppe region
PS.w	Southern plateau	Desert region
PS.ss	Southern plateau	Steppe region
PE.h	Eastern plateau	Highveld region
PNB.nt	Bankenveld	Northern Transvaal region
PNW.nt	Waterberg	Northern Transvaal region
PNBb.nt	Bushveld basin	Northern Transvaal region
PNP.nt	Pietersburg plateau	Northern Transvaal region
M	**Mountain wetland regions**	
MD.d	Drakensberg/Maluti	Drakensburg region
MCF.k	Cape Fold mountains	Karoo region
MCF.m	Cape Fold mountains	Mediterranean region
S	**Coastal slope & rimland wetland**	
SW.w	**regions**	Desert region
SW.m	Western coastal slope	Mediterranean region
SSE.w	Western coastal slope	Desert region
SK.k	Southern escarpment	Karoo region
SSE.ss	Karoo	Southern steppe region
SS.a	Southern escarpment	Temperate region
SE.d	Southern coast	Drakensberg region
SE.se	Eastern coastal slope	South-east coastal region
SE.e	East coast	Sub-tropical region
SNE.l	East coast	Lowveld region
SL.l	Northern escarpment	Lowveld region
SLV.nt	Lowveld	Northern Transvaal region
SOR.w	Limpopo valley	Desert region
	Orange River canyon	
C	**Coastal plain**	
C.e	Coastal plain	Sub-tropical region

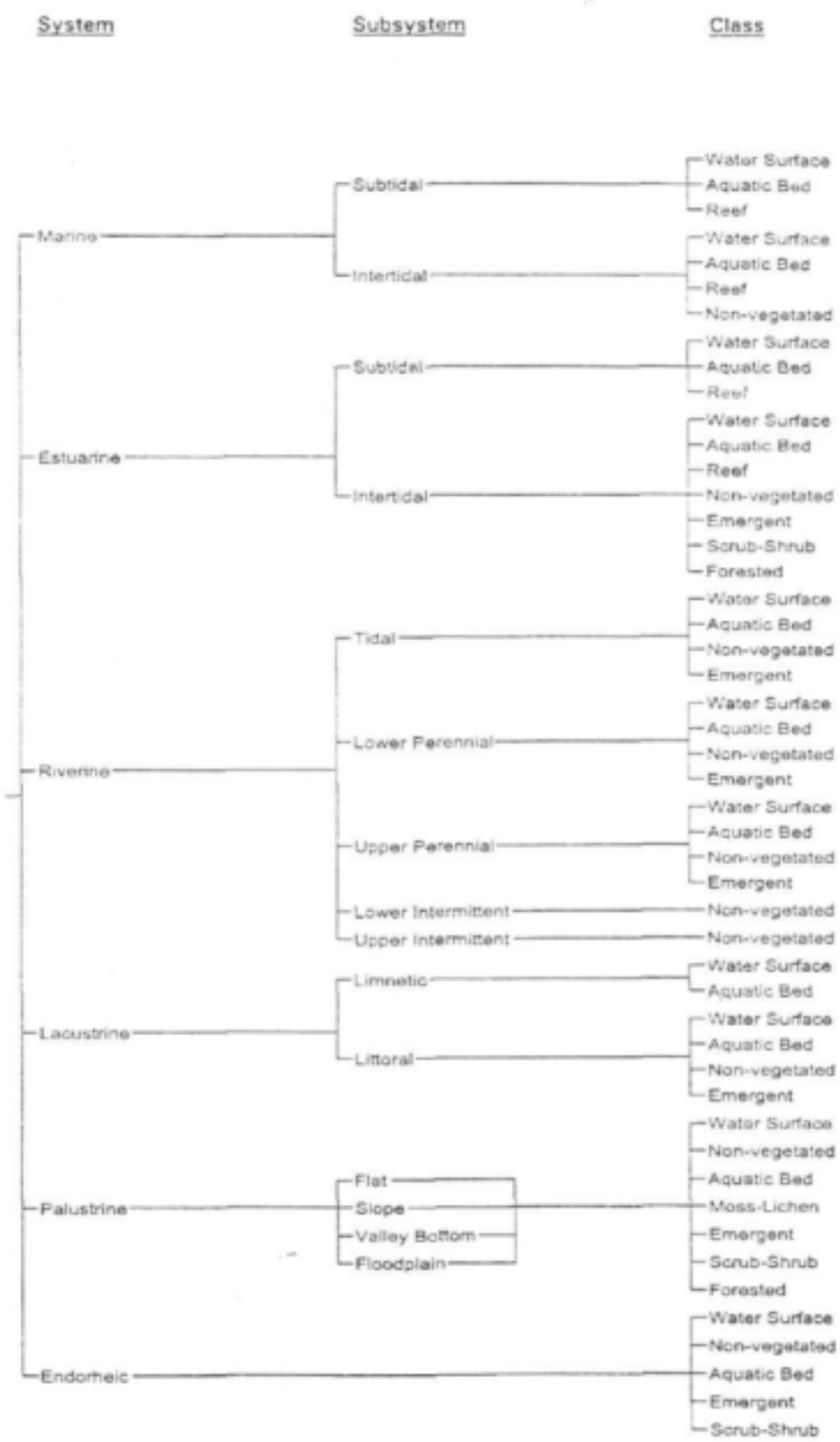


Fig 2.2 *The proposed classification system for South African wetlands (from Dini et al. 1999).*

2.2.3 A description of some wetland types

The description of individual wetland types given in the following sections is taken largely from Cowan (1995) and Cowan and van Riet (1998), in addition to Allan, Seaman and Kaletja (1995) for endorheic pans, and (Hart 1995) for lacustrine systems.

2.2.3.1 *Endorheic systems*

Endorheic pans are frequently situated in the Western, Eastern and Southern Plateau wetland regions of South Africa and are most commonly found in areas with annual rainfall *less* than 500mm and where there are high evaporative losses (Shaw 1988, cited by Cowan and van Riet 1998). Endorheic pans are defined as being typically circular to oval in shape and where two or more pans have spread and combined they form characteristic lobed or kidney-shaped wetlands. They are shallow (less than 3m deep) even when fully inundated and exhibit a closed drainage system (i.e. there are no major outlets). Most of the water is lost from pans due to evaporation, which is one of the factors leading to increased salinity. Inundation is characteristically ephemeral, some pans in the dry western areas stand dry for years between flooding, but in the wetter eastern regions many pans are seasonal and some are sufficiently large and permanent to be classified as lakes. The depth of inundation for a given pan is a function of the amount of precipitation, rate of evaporation, catchment size and basin morphometry (O'Brien and Motts 1980, cited by Rogers, Ellery, Winternitz *et al.* 1989), this in turn has a profound influence on the distribution of vegetation and hence habitat. Because pans can vary from a few metres to many hectares in diameter, Allan *et al.* (1995) recommended disregarding pans smaller than 1 hectare.

There are several interactive factors involved in the formation of pans. Water may collect in areas of poor drainage, for example, and this may lead to areas of no vegetation. When the water evaporates in arid areas, the soil in the depression is not bound by plant roots, which allows wind-action to scour out the basin further (the reason why many pans are round). According to Allan *et al.* (1995) many such pans have characteristic humps of soil on the down-wind side of the pan formed from the deposition of eroded soil. Loss of vegetation and subsequent wind-erosion is accelerated by grazing. The erosion effect of wildlife that tends to congregate at pans has been postulated as one of the factors that enhance the development of pans, although it is probably not a causative agent (de Bruiyn 1972; Seaman, Ashton and Williams 1991). Many pans of the Western

Orange Free State, however, are not primarily caused by wind, but by climate and local topography which is in turn accelerated by enhanced weathering brought about by crystalisation of salts.

According to Cowan and van Riet (1998), pans have been differentiated into the following groups:

- Salt pans, which are dry for most of the time but may contain perennial pools filled by springs.
- Temporary pans which are also dry for long periods and are moderately saline, but flood during the rainy season.
- Grass pans which are seasonal and dry up in the dry season. They are covered by a thick growth of hygrophilous grasses and other low terrestrial vegetation. Their waters are fresh to slightly saline.
- Reed pans and Sedge pans which are temporary or semi-permanent, with dense stands of *Phragmites* and members of the Cyperaceae.

From a biological point of view, endorheic pans are interesting because the biota that are found in them often need to be specially adapted in order to withstand periods of desiccation. Furthermore, some pans do not have any vegetation, whilst others may have extensive plant coverage.

Rogers *et al.* (1989) highlight some important points for the management of endorheic pans:

- Shallow pans and lakes appear to resist eutrophication (in terms of algal growth) until a critical point is reached after which they respond quickly (this is discussed more fully in section 2.4.2.1).
- Pans should be managed to preserve their temporal and spatial variability in order to maintain biodiversity of vegetation and avifauna (and presumably of other faunal groups). Thus fluctuation in water depth is important to maintain the diversity of vegetation. Stabilization of water levels due to anthropogenic activities can lead to favouring of one species (e.g. *Typha capensis* or *Phragmites australis*). This point has implications for setting the water quality component of the Reserve.
- Conservation of pans should be on a regional basis rather than preserving individual wetlands as this will ensure maintenance of diversity in space and time.

2.2.3.2 *Palustrine systems*

Palustrine systems are non-tidal wetlands dominated by emergent plants (e.g. reeds), shrubs or trees and include a variety of systems commonly described as marsh, floodplain, vlei or seep (Kotze *et al.* 2004). In contrast to endorheic pans, palustrine wetlands in this country are generally found in areas where the mean annual rainfall is *greater* than 500mm (Cowan and van Riet 1998). Exceptions do occur and palustrine wetlands are found in drier regions but this is usually along the main water-courses or around dolomitic eyes (Skelton *et al.* 1994 cited in Cowan and van Riet 1998). In a preliminary directory of South African wetlands four different types of freshwater palustrine systems (**marshes**) were recognised.

- Sedge marshes dominated by sedges and hygrophilous grasses up to 1 m high.
- Restio-marshes typical of the south-western Cape which are acid and dominated by members of Restionaceae.
- Reedbed marshes dominated by *Phragmites* species, but where water levels remain close to the soil surface during the dry season, stands of *Typha* spp., *Scirpus* spp. and *Cyperus* spp. may occur. Reedbed marshes intergrade with sedge marshes at higher altitudes.
- Cape seasonal wetlands which include plant communities of various types (e.g. coastal Renosterveld, Strandveld), each with characteristic species.

Two types of freshwater **swamps** were recognized:

- Reed swamps which are found in perennial standing water on floodplains and fringe many coastal and estuarine lakes. They are dominated by *Phragmites australis* or *P. mauritianus*, while *Typha capensis* is often present.
- Papyrus swamps which typically require a stable hydrological regime and are dominated by *Cyperus papyrus*. They are confined to the sub-tropical coastal plain wetland region.

This class of wetlands also includes peatlands. Peat in this country is found in wetlands ranging from the sub-tropical coastal region, the highveld and high-altitude areas of the Drakensberg mountains (Cowan and van Riet 1998) and localised areas in the Western Cape e.g. above the Wadrift Pan (Southern Waters 2002). The extensive deposits that are common in the northern hemisphere, however, are not found in South Africa.

Whereas all peat is associated with wetlands, not all wetlands contain peat (Grundling, Mazus and Bartman 1998). It is composed of thick layers of humified organic material that is deposited over a long period of time and can be used as a fuel. Peat accumulation (rather than complete decomposition of wetland vegetation) occurs when decaying organic matter accumulates in moist, reducing and low energy environments (Van Vuuren, 1976 cited in Grundling, Grundling and Blackmore 1999). Conditions that promote peat formation also include low temperature and low pH. Exploitation of peatlands for horticultural purposes is a serious threat to these sensitive ecosystems and recently initiatives have been taken to control the amount of peat that is harvested in South Africa (Grundling *et al.* 1999). Other threats include the uncontrolled burning of veld for grazing and afforestation (which lowers the water table).

2.2.3.3 *Riverine systems*

Cowan and van Riet (1998) support the classification of riverine wetlands as proposed by Cowardin *et al.* (1979) to include systems contained within a channel which contains moving water either continuously or from time to time. They have expanded this type of wetland however to also include inland deltas, seasonal rivers and streams and riverine floodplains. From the point of view of water quality this seems to be sensible, since all will at some time or other have water moving in a linear direction through them. This classification, however, is not supported by the new system for classifying wetlands (Figure 2.2) since under that system floodplains are grouped under palustrine rather than riverine wetlands.

According to Walmsley (1988) only a few South African rivers have well-developed floodplains. Those that are present are short and are in the middle or lower reaches. The above author cites the classification system of Noble and Hemens (1978) who categorize flood plains in this country as Karoo Salt flats (e.g. Van Wyksvlei), floodplain vleis (e.g. Nylsvlei) and storage floodplains (e.g. Pongola, Mkuze). Because relatively few rivers in this country have floodplains and their associated wetlands, it is especially important to conserve these ecosystems. According to Kingsford (2000), more than 50% of floodplain wetlands in Australia have been lost due to water resource development, a situation that is unlikely to be better in South Africa. A brief review of the ecological importance of floodplains, taken largely from the above reference, is given below.

It is the flow regime of a river and its connections to low-lying terrestrial areas that is the driving factor that governs channel formation, sediment transfer and biotic responses in floodplain wetlands (Lockaby, Stanturf and Messina 1997). Many changes, physical, chemical and biological occur in floodplains during inundation (Kingsford 2000). When a river arrives in flood at a floodplain (extensive areas of low relief), not only does the river carry water but also sediment and organic matter. In addition, organic matter from within the wetland (for example from terrestrial vegetation when the wetland was dry) or macrophytes from the previous inundation may also be present. As a result, inundated floodplains are nutrient rich and sites of high productivity and of dynamic ecological processes. Complex food webs involving microbes, zooplankton and higher trophic levels are formed. Zooplankton emerge from newly flooded eggs in the sediments, as do the seedlings of macrophytes. Flood-resistant trees and shrubs make use of the period of inundation to grow. Fish larvae from the river arrive and use the relatively calm backwaters as a nursery area. Burrowing frogs and other organisms that withstand the *times of desiccation in the sediments emerge to feed and reproduce (although initially,* due to high tannin levels and low oxygen, habitat suitability may be limited). Insects hatch in great abundance, and together with the rapid growth of aquatic macrophytes, as well as frogs and fish, provide food for water birds, which colonise the wetland from more permanent wetlands nearby. Kingsford (2000) goes on to emphasise that these systems exhibit high levels of biodiversity and that it is the (often unpredictable) river flow and the lateral connectivity between wetland areas and the river that are the principle ecological drivers.

The building of upstream dams, abstraction, weirs, levees affect these ecosystems detrimentally, either by altering the amount (or timing) of flood-waters that reach the floodplain wetlands or by reducing the amount of sediment and nutrients. Not only is the amount of water that reaches a floodplain (and thus the area that will be inundated) important, but also the residence time of the water. Increased flows can also be detrimental in that floodplain wetlands no longer dry out. As a result of the reduced heterogeneity of habitat both in time and space, biodiversity is reduced.

2.2.3.4 *Lacustrine systems*

The Ramsar convention distinguishes between permanent freshwater lakes (being greater than 8 ha) and ponds and seasonal lakes. In this work we will consider lakes to

be *permanent* relatively freshwater systems that are greater than 8 ha in surface area, whilst ponds are defined as permanent relatively freshwater systems smaller than 8ha in surface area. This is in line with (Hart 1995) who considers lakes to be water-bodies with extensive, relatively deep open waters. This author also points out that if such a classification system is used, South Africa has very few natural lakes. There is only one natural inland lake (Lake Fundudzi), and several coastal lakes in three distinct geographical areas; those of the south-western cape (mediterranean climate), those of the southern cape (southern coastal temperate region) and northern KwaZulu/Natal (sub-tropical). There is a graduation between ponds and pans (Cowan and van Riet 1998) and consequently any distinction between the two is often artificial. The national classification system Dini *et al.* (1999) proposes that freshwater systems deeper than 3 metres be designated as lakes, and those less than this depth, are ponds (permanently inundated) or pans (temporarily inundated).

Because of the relatively great depth of lakes compared to pans and ponds, physico-chemical characteristics of the water body are affected by stratification (Davies and Day 1998), a phenomenon that is not significant in shallow water bodies.

2.3 THE IMPORTANCE OF WETLANDS IN THE ENVIRONMENT

Society has grown to appreciate the wide range of goods and services, ranging from flood control to fish nurseries, provided by wetlands, and a great deal has been written about these values (Maltby, Hogan, Immirzi *et al.* 1994; Cowan 1995; Davies and Day 1998). They form some of the most productive and biodiverse ecosystems on earth, and millions of people throughout the world are dependent on these systems for fish, grazing and other materials (Dugan 1990). A list of functions that have been attributed to wetlands is given in Table 2.2., but only those that pertain directly to water quality are discussed. It is important to note that not all wetlands perform all the listed functions and not all functions are carried out by individual wetlands to the same extent. We still have an inadequate understanding of how different environmental factors and processes interact and thus how different wetlands work (Maltby *et al.* 1994).

Table 2.2 *"Goods and services" provided by wetlands. Functions of wetlands that have a direct impact on water quality are indicated in bold. (Adapted from Dugan 1990).*

Services	Products
• Groundwater recharge • Groundwater discharge • Flood control • Erosion control • Storm protection/windbreak • Micro-climate stabilisation • Water transport • Recreation/tourism • **Nutrient retention** • **Nutrient/biomass export** • **Sediment/ toxicant retention**	• Forest resources • Wildlife resources • Fisheries • Forage • Agricultural resources (crops) • Water supply
	Attributes
	• Biological diversity • Uniqueness to culture/heritage

2.3.1 Functions of wetlands that affect water quality

As water flows through a wetland area, the biomass of roots leaves and stems that are both above, and below the soil surface, tend to slow that water down. This helps to attenuate floods during periods of high flow, reducing the volume of water that reaches downstream systems. During dry periods, however, this water is slowly released, providing vital water to systems that might otherwise dry up (Reddy and Gale 1994). Reduction of the velocity of the water flowing through a wetland tends to lead to sediments being deposited on the soil surface. At the same time that sediments are trapped within the wetland, other chemical constituents that are often associated with sediments (e.g. nutrients, toxicants) also become trapped. Due to the action of bacteria, fungi and protozoa that are present amongst plant roots and in the sediments, these chemical constituents are degraded to simpler molecules. Plants take up nutrients and convert them to biomass. In addition, dissolved constituents are also removed from the water column by the biota (Davies and Day 1998). As a result of the above processes, wetlands have been found to be very efficient at removing chemicals from the water column.

For example, it has been calculated that Florida's cypress swamps remove 98% of all nitrogen and 97% of all phosphorus entering the wetlands from wastewater, and thus prevent contamination of the groundwater (Ramsar 2002). Wetlands often form around the margins of lakes and along the banks of rivers. Because of their location these boundary wetlands can intercept overland runoff and stream flow and thus influence water quality by moderating the amount of nutrients and sediment that enters the system (Dugan 1990).

Of particular importance for the regulation of water chemistry in these ecosystems, is the process of denitrification, whereby nitrate is converted to gaseous oxides or N_2. According to Maltby *et al.* (1994), this process is enhanced in wetlands where alternating aerobic and anaerobic conditions are found. Aerobic and anaerobic conditions in a given wetland may be separated spatially, or temporally. For example in wetlands that are alternately flooded and then dry out, the oxygen content of the soil fluctuates widely, being largely anoxic when inundated and aerobic when dry. Under aerobic conditions NH_4^+ formed from the break-down of plant material is converted by micro-organisms to nitrates and can be readily taken up by the biota. Under anaerobic conditions, nitrate is converted to gaseous N_2 or nitrous oxide and is lost to the atmosphere. Although both nitrogen fixation and denitrification occur in wetlands, the rates of these opposing processes vary, and in general it appears that the rates of the denitrification are the highest. As a result, wetlands can be considered to be sites where organic N arrives in run-off and detritus and is then returned to the atmosphere (Keddy 2000). Wetlands are therefore important for removing nitrogen from the water column, and this is one of the main reasons that artificial wetlands are becoming popular (Rogers, Breen and Chick 1991). Eventually, release of large proportions of stored N into the water column may occur as plants die, but this can be prevented by harvesting and removal of excess vegetation.

Keddy (2000) emphasises that in the case of phosphorus, uptake and retention within wetlands is much more variable. He goes on to cite work (Richardson 1985) indicating that the often high initial rates of phosphorus removal in wetlands are not sustained and may be followed by export of this nutrient, especially if the loading is high. Currently there is a lack of criteria that can be used as the basis for determining whether a wetland will behave as a source, sink or transformer of nutrients and pollutants (Reddy and Gale

1994; Mitchell, Chick and Raisin 1995; Raisin, Bartley and Croome 1999). Interestingly, experimental evidence suggests that, for northern temperate systems at least, natural freshwater marshes may be *phosphorus* limited while salt marshes may be *nitrogen* limited (Keddy 2000).

Environments downstream of wetlands and in coastal waters often benefit from plant material (and hence nutrients) that are carried away from the wetland by surface flow. For example, in temperate systems, plants die in winter and the material (plant material, sediments, associated nutrients) that is washed down-stream is often a crucial input for fisheries that are further down the system (Dugan 1990). According to Dugan (1990) there is commonly a cycle of wetlands tending to act as a sink during the growing season when water flows slowly, these nutrients support the growth of fish, macroinvertebrates, plant growth etc. When water flows fast, wetlands act as a source. According to Dugan (1990), this cycle has important implications in that nutrient uptake is greatest when the dilution capacity is the lowest (low flows) thus helping to decrease the impacts of eutrophication. It is not clear how relevant the above is to South Africa, for example in the summer-rainfall region where fastest flow is during the growing season.

As mentioned above, because water travelling through wetlands tends to be impeded, there is a tendency for sediments (and hence associated chemical constituents such as nutrients, pesticides, pathogens, organic contaminants) to collect in these areas. Thus wetlands tend to protect down-stream aquatic systems from excess sediments. The abundant bacteria in wetlands are often active in breaking down complex molecules, converting them to simpler molecules that can be taken up by plants. Excess toxins (for example mental pollutants) and nutrients bound to sediments may accumulate within a water body. While still bound to sediments these chemicals may be relatively harmless, especially to animals that do not come into close contact with the benthic layers. If there is a change in the physico-chemical properties of the water column and sediments (e.g. in pH, or redox condition), however, toxicants and nutrients bound to sediments can be released again into the water. During periods of inundation sediments tend to become anoxic. The low oxygen levels and reducing conditions lead to a release of phosphate, metals (especially iron, manganese) and hydrogen sulphide into the water column. Under oxidising conditions however (when the wetland is dry, or the water column is shallow) phosphates interact with many cations (e.g. Al, Fe, Ca) to form relatively

insoluble compounds that precipitate out of water. Phosphates also bind to inorganic colloids, humics and particulate material such as clays, carbonates, hydroxides. Manganese and iron, which exhibit similar chemistry to each other, also form insoluble complexes under oxidising conditions (DWAF 1996; Batchelor, Matthews and Coleman 2002). There are movements in the USA to develop Sediment Quality Guidelines as part of a national strategy to deal with contaminated sediments (McCauley, DeGraeve and Linton 2000). This is discussed further in section 2.7.4.

2.3.2 Wetlands in South Africa

A study carried out by Batchelor *et al.* (2002) was aimed at quantifying the ecosystem services of the wetlands of the Upper Olifants River catchment (Mpumalanga). The ultimate aim was to place a monetary value on the wetlands of this area and the "goods and services" they provide. An interesting review is given of the processes likely to occur in the wetlands that facilitate the removal and/or transformation of pollutants. Environmental factors (temperature, pH, availability of substrate and appropriate bacteria etc.) that influence the rate of these processes are also discussed. The pollutants considered were those likely to arise from agriculture (nutrients) and mining (sulphate and metals) which are the major land-use activities in the upper Olifants River catchment. A consideration of the general processes by which the above contaminants are transformed is followed by a discussion of how these might be modified in the specific wetland types that occur in the area, namely; endorheic pans, floodplain systems and seepage wetlands. Because of a lack of data, no quantitative estimates could be made of the contribution of the wetlands to assimilation and break-down of pollutants. The authors make some interesting conclusions with regard to water quality functioning, some of which differ from those given in the previous section – the authors however do stress that the results are largely from a desk-top study and therefore tentative. Because hydrology and water quality are inextricably linked, some of the major conclusions with regard to this aspect are also included in the summary below.

- More water is lost through evaporation and evapo-transpiration in a wetland than from an equivalent area where there is no wetland.
- Wetlands have an augmenting and regulatory effect on baseflow during dry-periods, and the Mean Annual Runoff (MAR) from wetlands is higher than from non-wetland areas.

- The wetland systems in the Upper Olifants catchment are complex, varying in land-use, position in the landscape, water quality, hydraulics and hydrology. Because there are no data on nutrient inputs or outputs it was difficult to establish the effect the wetlands were having on nutrients, let alone quantify this. A tentative conclusion was drawn that the purification value of the wetlands is likely overall to be low.
- Riparian systems and floodplains are not likely to be very efficient in nutrient removal since the contact period between wetland biota and water/substrate is short (i.e. only during the flood period).
- Seepage wetlands are the only types of wetlands occurring in the upper Olifants catchment that are likely to remove significant quantities of excess nutrients – and this is not expected to be high.
- Artificial wetlands and pans do not remove nutrients. They are nutrient rich systems in which these substances are cycled. There is unlikely to be a net change with addition of surplus nutrients.

The low level of nutrient removal concluded by Batchelor *et al.* (2002) in the Olifants River wetlands is surprising but is consistent with the variable results that have been found in the literature. These results serve to emphasise the complexity of these systems and the need to deal with them on a site-by-site basis.

Batchelor *et al.* (2002) also noted that nitrogen present in groundwater is likely to be in the form of nitrates due to nitrification in the surrounding unsaturated soil. Even if ammonium is added to surrounding land as fertiliser it is likely to be either bound to the soil or oxidised. Thus ammonium is unlikely to be present in high concentrations in groundwater. In the case of phosphates, groundwater is unlikely to be high in concentration in soluble reactive phosphorus (SRP) because this form of phosphorus binds tightly to soil particles and is likely to remain in the soil matrix. The above points are important because often groundwater is the major water source for a wetland and thus will have a major bearing on the overall water quality. The authors list the (many) routes of entry of nutrients into a wetland and say that without direct measurement it is not possible to estimate the mass of nutrients entering a system.

2.4 CAUSES OF DEGRADATION TO WETLANDS

2.4.1 Physical impacts

On a world-wide basis, wetlands have been irretrievably degraded or altered because of disruption to the hydrological system, or to ecological functioning, or to both. It is estimated that in certain areas of South Africa, such as on coastal plains, more than 50% of wetland habitat has already been lost (Shearer 1997) and a large proportion of the remaining wetlands are under threat (Walmsley 1988; Kotze, Breen and Quinn 1995). This loss may be due either to deliberate actions (e.g. draining for agriculture, urban development) or may arise inadvertently from activities such as the building of roads (which lead to alteration in drainage patterns) or a combination of the two. There is a wide range of activities that can lead, or contribute to the degradation of wetlands. This section concentrates mainly on impacts to water quality and the resultant effects on ecological functioning. A summary of possible causes of wetland loss and degradation is given in Table 2.3. Impacts that concern altered water quality are indicated.

Many of the impacts listed in Table 2.3 currently threaten, or have already led to, the loss of South African wetlands. For example, mining, agriculture, afforestation (which leads to reduced surface run-off to down-stream systems) have been reported to be the main threats to the wetlands of the Blyde River catchment, Mpumalanga (Linstrom 1999). Peat wetlands are a very scarce and valuable resource in South Africa. They have been degraded due to peat mining, but also due to pollution such as metals, leaching from mine dumps and to sewerage (Grundling *et al.* 1999). According to Cowan and van Riet (1998), the greatest threats to endorheic pans are urbanization (accompanied by draining and filling), afforestation and agricultural runoff.

Table 2.3 *Causes of wetland loss and degradation. Impacts to wetlands that are primarily due to impaired water quality are indicated in bold. (Adapted from Dugan 1990 and Shearer 1997).*

Direct impacts
• Drainage for agriculture, forestry, mosquito control.
• Poorly managed grazing, mowing and burning.
• Flooding due to construction of dams.
• Dredging for navigation, flood control.
• Filling for solid waste disposal, road construction, residential, commercial and industrial development.
• Erosion.
• Groundwater abstraction.
• **Discharges of pollutants from point-sources, diffuse sources (agriculture, informal settlements, urban areas), air pollution.**
• **Mining of wetland soils for peat, coal, gravel, phosphate, salts and other materials.**
Indirect impacts
• Sediment diversion by dams, channels and other structures.
• Hydrological alterations by canals, roads and other structures.
• Subsidence due to extraction of groundwater, oil, minerals.

Endorheic systems are especially vulnerable to water quality impacts because there are no (significant) outflows. Thus nutrients and toxicants tend to accumulate within these water bodies. Because wetlands are dependent on water and the source may be far away, impacts can originate at considerable distances from the wetland areas and in many cases beyond national borders (Cowan 1995). The loss of wetland can lead to; lower agricultural productivity, reduced water quality, less reliable water supplies, increased incidence of and severity of downstream flooding, reduced biodiversity and threatened wildlife resources (Shearer 1997).

2.4.2 Pollution

Wetlands are increasingly becoming the recipients of rising pollutant loads emanating from human settlements. Of the wide range of water-borne pollutants, nutrients, sediments, acidic compounds, salts, metals and biocides are considered to be the most significant in terms of their impact on wetlands and their biota (Coetzee 1995). When considering pollution in wetlands, the fact that these systems tend to accumulate sediments and that this substrate, in turn, is often an important sink for pollutants needs to be kept in mind. Sediments are discussed further in sections 2.3.1 and 2.7.4.

Coetzee (1995) highlights some very important points concerning pollution in wetlands, noting that wetlands in semi-arid areas such as South Africa are especially vulnerable to pollution because of the following reasons.

- Low volumes of incoming, uncontaminated water reduce the dilution capacity for resident, polluted water.
- Reduced through-flow results in longer residence time within a wetland system. This in turn increases the likelihood of sediments, nutrients and toxins dropping out of suspension and being retained in the wetland.
- Less frequent flushing flows occur to remove pollutants from a given wetland.
- The hot climate leads to high rates of evaporation, which in turn tends to concentrate pollutants.

A comprehensive review of the potential threats of pollution to South African wetlands is given in Coetzee (1995). Only the major points are mentioned here.

2.4.2.1 *Eutrophication*

One of the major threats to wetlands is the increasing occurrence of eutrophication. Addition of nitrogen or phosphorus to a wetland systems results in increased productivity. It is the direct and indirect effects of this increased productivity that results in deleterious effects as outlined below (Coetzee 1995; Keddy 2000).

In open water systems such as lakes or pans, a moderate level of eutrophication, due to increased productivity, is often linked to an increase in the biomass of macrophytes. In general, this enhanced biomass is limited to a very few species with a concomitant reduction in the overall biodiversity. In palustrine systems this is typically *Phragmites* or

Typha spp. both of which thrive in nutrient-rich environments. If the loading of nutrients into the system increases further, however, there is frequently a marked decline in the abundance of macrophytes accompanied by an increase in phytoplankton (Bailey, Boon and Morris 2002). It is hypothesised that increased phytoplankton densities result in increased turbidity. This in turn leads to reduced light penetration, resulting in shading of the macrophyte stands, which reduces plant growth. One of the reasons for increased phytoplankton densities is not just an increase in productivity, but because increasing levels of phosphate lead to a decrease in the N:P ratio. This in turn causes nitrogen-fixing cyanobacteria to replace other species of planktonic algae with a concomitant reduction in water transparency. It has been recognized that, in addition to altered hydroperiod, increased phosphorus loading is the main cause of excessive growth of algae and aquatic weeds in the Everglades wetlands in Florida, USA. Since most of the phosphorus originates from agricultural non-point sources, Best Management Practices are being developed to limit runoff from farms (Anderson and Flaig 1995).

Organisms such as zooplankton that normally feed on phytoplankton and keep the populations in check are unable to consume the increased biomass. Eventually large numbers of phytoplankton die and are broken down by decomposer organisms (fungi, protozoans, bacteria). Increases in the populations of decomposer organisms and their activities in assimilating the organic material from senesced phytoplankton can lead to depletion of oxygen in both sediments and in the water column. In estuaries, eutrophication tends to cause accelerated growth of macro-algae, this in turn can result in death of macrophytes (e.g. *Zostera*) due to shading. Anoxia leading to fish-kills is often an impact induced by the decomposition of excessive plant material.

From the above it can be seen that eutrophication can have a profound effect on the ecological functioning of a wetland. The extent of the impact will depend on severity of the eutrophication problem. Blooms of blue-green algae are notorious in that they often develop rapidly, wreaking havoc on water quality and ecosystem integrity (Harding and Paxton 2001). Many lakes and impoundments have been studied where there has been a dramatic change from a macrophyte-dominated system to a phytoplankton-dominated system. There appears to be threshold values which once exceeded result in a sudden change in trophic structure of the ecosystem (Bailey, Boon and Morris, 2002). Eutrophic conditions resulting from moderately elevated nutrient discharges can also have

beneficial effects on the other hand. These conditions often lead to vast beds of reeds, which give shelter to many species of birds (see below).

2.4.2.2 *Mine drainage and salinisation*

Another major impact to wetlands in South Africa is the introduction of effluent from mining operations which may originate either from point- or diffuse-sources. Such effluent is often acidic, high in salts (frequently sulphate), sediments and metal pollutants (Dallas and Day 1993).

A pertinent, South African case-study of the impacts of mining-derived pollution is represented by the Blesbokspruit wetland, situated 3 km east of the town of Springs. This wetland was designated a Ramsar site in 1986. The area consists of a naturally marshy area at the southern end of the spruit, dominated by *Phragmites* and *Typha*. It is unusual in that although situated in a naturally marshy area, water levels in some parts are artificially maintained by mining, as well as industrial and municipal effluents. Reed beds which provide an ideal habitat for birds are extensive in this area due to nutrient-enrichment (Cowan 1995). Changes in the operation of the mines in the area and economic necessity resulted in alterations in the way the mining complex was drained, with water being pumped from a different area. The water quality of the resulting effluent was worse than expected, and although treatment facilities were subsequently built to remove the manganese and iron loads and to neutralize acidity, the effluent is still saline (Green 1999). Damage resulting from salinisation of the wetland led to the placement of this Ramsar site on the Montreux Record – a list of wetlands of international importance under threat due to anthropogenic interference. This indicates that ecological functioning is severely impaired and this site has been the subject of intensive investigation. Because of further proposed development in the catchment and because curtailment of mining activities would lead to significant job losses, the future of this ecologically important wetland is uncertain (Green 1999).

According to a scoping report on the effect of salinisation (resulting not necessarily from mining effluent) on Australian rivers and wetlands, significant reductions in the species diversity of macroinvertebrates in wetlands can be expected. Salinities as low as 800 mg/L were reported to reduce species diversity, and there are indications that juvenile life stages may be especially sensitive. A reduced number of species emerged from

flooded sediments in salinised wetlands compared to non-saline systems. Sensitive taxa included cladocerans, gastropod snails, some water beetles, aquatic weevils, dragonflies, damsel flies and dipterans flies (LWRRDC 2000). In the case of temporary wetlands, the concentration of salts increases as water evaporates. Although flooding river or groundwater may initially be of low salinity, final salinities may be many times higher because of evaporation. A combination of high water temperature and high salinity during summer result in low concentrations of dissolved oxygen. Despite these stressful circumstances the biota *naturally* found in such systems are adapted to these conditions. It is in systems polluted by anthropogenic saline waters that the biota are likely to be impacted.

2.5 RESOURCE DIRECTED MEASURES FOR WETLAND ECOSYSTEMS

2.5.1 Background

Under the new South African National Water Act (Act No. 36, August 1998) wetlands are considered to be significant water resources requiring protection. Protection is to be ensured by means of determining and enforcing the ecological Reserve, defined as:

"The water *quantity* and *quality* required for maintenance of the ecological functioning of the aquatic ecosystem."

It is stated that "as soon as reasonable after the act comes into force, the Minister must determine the Reserve for all, or part of each significant water resource". The current project is concerned with developing tools that will aid in determining the water quality component of the Reserve. It will be seen from this section that the current procedure for water quality is rudimentary. In addition, since the hydrological regime is a critical factor in determining the water quality that will prevail in a wetland and the water quantity component is the most fully developed, this aspect forms the major focus of this section.

In the context of setting environmental/ecological water requirements (i.e. determining the amount, timing and duration of water required to maintain a given level of ecosystem functioning), King and Louw (1998) stated that provision of a suitable flow regime will not result in optimum ecosystem health if the question of water quality is not also considered. Although the above remark was made in relation to rivers, the preceding

review of the literature supports this statement with regard to wetlands. Thus, efficient ecological functioning in wetlands requires not only that an appropriate hydrological regime be in place, but that the quality of that water (i.e. the values of the physical attributes and concentrations of the chemical constituents) should be suitable.

The information in this section is taken almost exclusively from Volume 4 of the RDM manual (DWAF 1999b) which pertains to wetlands. Emphasis is given to the topics that are linked to water quality and are therefore relevant to this project. O'Keeffe (2000) gives a useful overview of the process, especially for setting the water quantity component of the Reserve for rivers. In the RDM manual (DWAF 1999b), estuaries and rivers have been omitted from the list of wetland types. This is because these types of aquatic ecosystem have their own methods to determine the ecological Reserve (see volumes 3 and 5; DWAF 1999a and c), tailored towards the unique requirements and attributes characteristic of these specific ecosystems. Some parts of the Resource Directed Measures (RDM) suite of methods have not been completed and this is especially true of the section on wetlands. Despite the variations for each aquatic ecosystem, there is a generic overarching protocol that is common to all. It is important that any tools developed in this project are also compatible with the generic protocol. The prescribed method for determining the Reserve in wetlands is discussed further on in this section.

The South African wetland types that are included in the RDM for wetlands are:

- Endorheic systems: permanent and seasonal pans.
- Palustrine systems: freshwater marshes, peatlands, springs, swamp forest and floodplains.
- Lacustrine systems: natural freshwater lakes.

The types of wetland that are being considered in this project (section 2.2.2) were chosen to be consistent with DWAF's Resources Directed Measures approach. The classification system used for wetland Reserve determinations (RDM manual, Appendix W1, DWAF 1999b), also gives details as to which type of wetland is considered to fall into what class. For example, oxbow lakes are considered to be lacustrine or palustrine systems, unless they are connected, either permanently or intermittently at both ends to a river, in which case, they are considered to be riverine.

2.5.2 Level of RDM determination and wetland size

For wetlands, as for other aquatic ecosystems, different levels of Reserve determination are prescribed: a Desktop estimate, Rapid determination, Intermediate determination and Comprehensive determination. The field-work for an Intermediate determination for a wetland should take approximately 2 months to complete and a Comprehensive-level (high-confidence) determination 8 –12 months, or more. According to the RDM manual the difference between the two higher levels of determination is the amount of information and data that are required. Guidelines are given as to when a high-confidence (comprehensive) study should be carried out. This includes situations where the wetland in question:

- is protected (e.g. a Ramsar site, or part of a national or provincial nature reserve);
- is a large freshwater lake;
- supports endangered species;
- is fed by a water-source for which major development is planned (and which may impact on the wetland).

The level of Reserve carried out is dependent on the level of water usage and the importance of the system. No Reserve determination usually needs to be carried out for endorheic pans unless they occur in areas in which streamflow reduction activities will be carried out in the catchment, or unless they are subjected to water abstraction or waste discharges. At present, only the Intermediate Reserve determination method for wetland water quantity is developed. There is none for the low-confidence (Desktop or Rapid) methods, and that for Comprehensive (high-confidence) is rather sketchy. No fully-documented method for setting the water quality component of the Reserve in these systems at any level of determination was found during the course of this project.

There is also a lack of clarity as to what exactly constitutes a "significant water resource" in relation to wetlands. In the case of lacustrine systems, these are considered to be aquatic systems that have a total area greater than 8 ha and surface area coverage by mosses, lichens, trees, shrubs or persistent emergent vegetation of less than 30%. In the case of palustrine systems, no specifications are given. In the case of endorheic pans, as mentioned above, the ecological Reserve will be determined only if catchment activities that will impact on the quantity or quality of water in the pan are likely.

2.5.3 Integration with other Reserve components

The manual clearly states that, depending on the hydrology of the system under question, integration with other Reserve components may be necessary. Guidelines are given with regard to integrating the Reserve for wetlands with river and groundwater components.

- No integration is necessary for endorheic pans.
- Integration with the groundwater component is required for wetlands that are fed by groundwater only, or by groundwater and surface water that does not originate from a river.
- Integration with the river component is required for wetlands (floodplains) whose hydrological regime is determined by river flow.
- Integration with both river and groundwater components is necessary in complex systems that depend on both water-sources.

The above is of relevance to the present project, since the water-source of a wetland is an important consideration for water quality (Chapter 4).

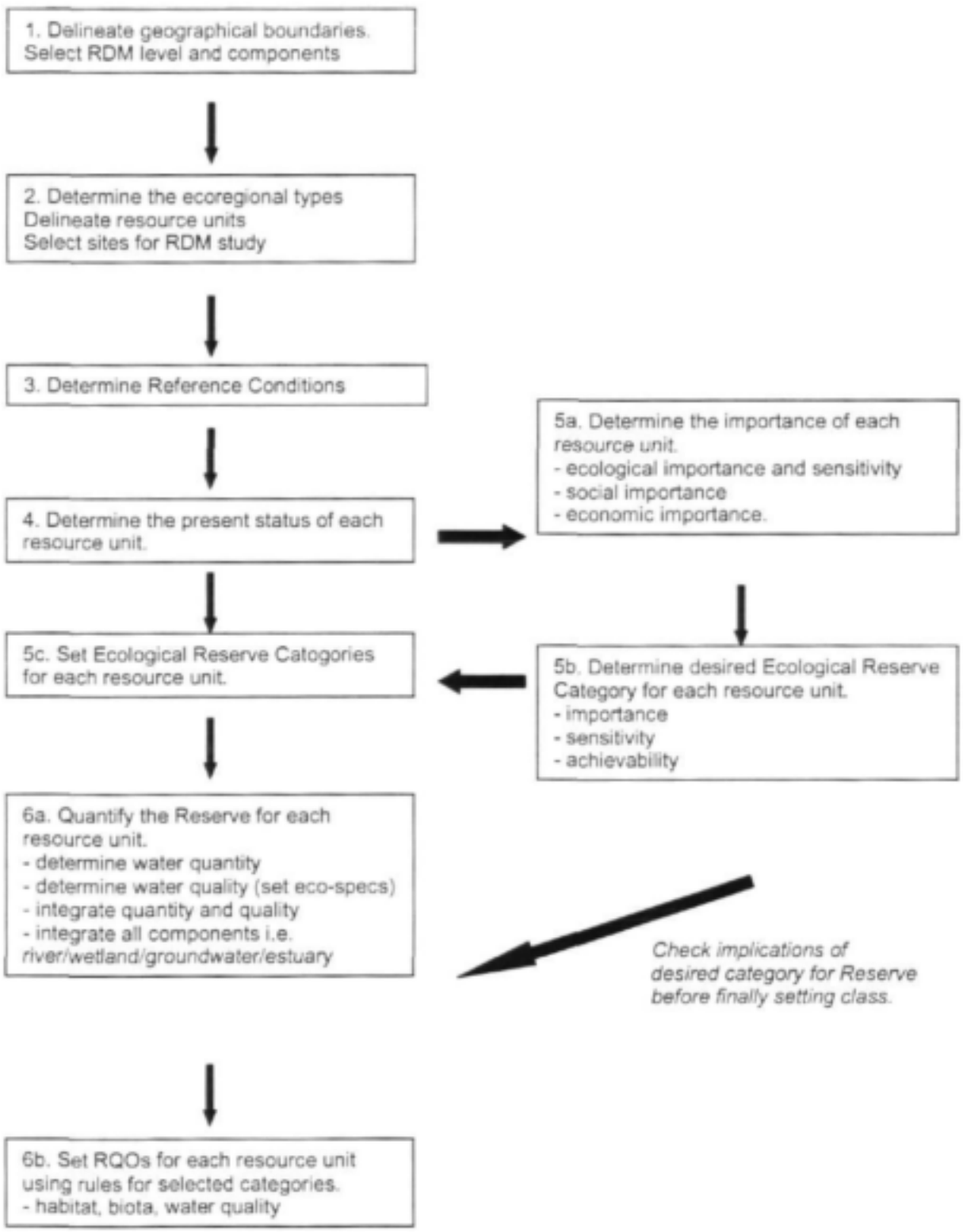
2.5.4 The Generic Reserve Determination method as applied to wetlands

Figure 2.3 summarises the six steps which form the main process for determining the ecological Reserve in wetlands. Each step is discussed separately.

Step 1: Definition of the study area

The boundary of the study area and the system components that will be included (e.g. tributaries, groundwater, estuary) are decided upon by DWAF. The level of the Reserve determination is also defined at this stage and a team of specialists is appointed to carry out the work.

Figure 2.3 *The generic procedure for determination of the ecological Reserve for aquatic ecosystems. Adapted from DWAF (1999) and O'Keeffe (2000).*



The study team leader then carries out the following tasks:

- The study area is broken down into water resource units that are each sufficiently different (e.g. in terms of hydrology) from the other to warrant their own specification of the Reserve.
- The wetland team leader identifies wetlands within the study area using maps and aerial photographs and collates all available information on each wetland.
- A site visit is conducted to determine the water-source for each wetland as well as to obtain further information (listed below).
- The wetlands are classified according to the national classification system.
- The wetlands are delineated. A delineation methodology has been developed that can be used to delineate the geographic boundaries of the wetlands (Marnewecke and Kotze 1999; DWAF 2003a) as well as to identify the various vegetation and saturation zones. These zones should be mapped in conjunction with a topographical survey of the wetlands for hydraulic modelling purposes.

Step 2: Determine ecoregional type

O'Keeffe (2000) discusses the Reserve methodology for rivers, and translates each step of the generic protocol into one or more questions that need to be answered. Extrapolating this to the aquatic ecosystems considered here, Step 2 asks the question "which wetland types are similar"? Wetlands in the study area are grouped together according to similarity in physical, chemical and biological attributes. This simplifies the Reserve determination process because the same specifications (Resource Quality Objectives) can be set for each group of wetland. A system of ecoregions has been developed for the rivers of South Africa (Kleynhans and Hill 1999). The RDM method for wetlands suggests that at a broad-level of ecotyping (ecoregion levels I and II), the delineations for rivers can be used. At a lower level (ecoregion III) the proposed national wetland classification system of Dini *et al.* (1999) should be used (section 2.2.2). The level III ecotyping determines which tools should be used to set the water quantity component of the Reserve:

- if the wetland is a **lacustrine** or **endorheic** wetland (i.e. a lake or pan), the methods developed for lakes should be used (or adapted according to site conditions);
- if the wetland is a **palustrine** wetland then the method developed for floodplains should be adapted according to site conditions. In the national classification system

(Dini *et al.* 1999), the boundary of floodplain wetlands is defined as the 1-in-10 year floodline;

- for riparian palustrine wetlands and floodplains, the methodology developed for rivers should be used.

Step 3: Determine the Reference Condition

In this step, the expected condition of the wetland in the natural, un-impacted condition is described. This represents the baseline against which any changes, brought about by anthropogenic influences can be assessed. Historical data can be used to derive this state, or otherwise by extrapolating from a nearby (un-impacted) wetland that has similar attributes. The RDM manual specifies that the Reference Condition (RC) for each wetland resource unit should be described for the most important abiotic and biotic aspects. These include:

- Geomorphology, including sediments.
- Hydrological regime (water source, hydrodynamics, wet/dry periods, saturation zones, surface water levels, groundwater levels, extent, period and depth of inundation, flow volumes, return periods of flow).
- Biological attributes.
- Water chemistry (Temperature, dissolved oxygen, electrical conductivity, TDS, pH, turbidity, TSS, nutrients, toxic constituents).

Step 4. Determine the Present Ecological State

This step answers the question "What is the wetland like now"? The status of the wetland is assessed in terms of its deviation from the Reference Condition. A method has been developed for assessing the Present Ecological State (PES) of lakes, palustrine wetlands and floodplains. The method has been adapted from the habitat integrity method designed for rivers by Kleynhans (1999) and is presented as an appendix in the RDM manual for wetlands (DWAF 1999b). A qualitative judgement is made of the condition of different attributes of the system, such as the hydrology, water quality, hydraulics/geomorphology and the biota. Each attribute is scored from 5 (largely natural) to 0 (critically modified). The sum of the scores indicates the current overall class of each particular wetland. The RDM manual notes that water quality can be assessed by direct laboratory analysis or indirectly from an assessment of the upstream landuse.

Step 5. Determine the Ecological Category

The role of this step is to answer the question "how ecologically important is each wetland sub-system and for what condition should they be managed?" In order to answer this question, the Ecological Importance and Sensitivity (EIS) of the wetland is assessed. A score sheet has been devised and is presented in the manual for floodplains (it can presumably also be used for other wetland types, but this is not made explicit). The score sheet is similar to the one used for rivers, but has some unique features. For example, the importance of a wetland system in "flood storage, energy dissipation and particulate/element removal" is included. From an examination of factors such as the number of rare and endangered species in the system, the perceived sensitivity to changes in water quality, and the protection status (e.g. a Ramsar site), an indication of the EIS can be gained. Expert judgement is used to assess these attributes. As an aid to determining the sensitivity of floodplains to changes in water quality, it is noted in the manual, that assimilation capacity for pollutants will be greater in large floodplains. Furthermore, large floods will tend to dilute pollutants. Floodplain wetlands that are the most sensitive to changes in water quality are considered to be those that are small, with a hydrological regime consisting of small multiple floods. The least sensitive systems are larger floodplains with large, infrequent (greater than annual) floods.

The EIS score provides a guideline for assigning the Ecological Category (the level of protection for which the resource will be managed), since a given EIS score range translates to a given management category (the higher the score, the higher the category and the higher the required level of protection). For example, an overall median EIS score of greater than 3 corresponds to a recommended Ecological Category of "A". This category represents a system that should be given a high level of protection and should be maintained in a largely natural state. In addition, the social and economic importance of each resource unit is defined and taken into account when deciding on the condition for which the wetland will be managed.

Step 6. Set the Reserve

This step answers the questions "what should the future inundation regime and water quality of the wetland be, in order to result in the required protection level and what practically can be done to achieve this"? In other words the ecological specifications

need to be set which will maintain the system in the recommended Ecological Category. The water quantity component of the wetland Reserve is set by examining the habitat and requirements of key biota. In the case of Intermediate ecological Reserve determinations (which are of short duration with only limited collection of field data), water requirements are determined primarily on the basis of the saturation, inundation and vegetation zones. The water requirements are described in terms of period, depth, extent, timing and frequency of inundation. A worked example for determining the water requirements of Lake Cubhu, part of the Mhlatuze system is given in the RDM manual (DWAF 1999b). The minimum depth under drought conditions is specified (as well as the length of time this can occur). In addition, the minimum dry season depth (for normal, rather than drought conditions), and the maximum depth (termed the "Full supply level"), which is applicable to the entire hydrological year is also defined. Motivations explaining how the values were derived are given, in addition to the maximum draw-down rate.

The following pointers are given in the RDM manual for determining the **water quality** component of the Reserve for wetlands:

- "For palustrine wetlands fed by river-flow and flood-flow, use the same rules as developed for rivers to determine the water quality Reserve, until such time as wetland-specific water quality rules are developed. Special attention should be given to the sediment requirements of floodplains."
- "For lakes and pans, additional attention must be given to the accumulation of toxic elements in these systems. The elevation of TDS and nutrient levels due to evaporation from these closed or semi-closed systems and limitations on mixing and dilution should also be considered."

In the Reserve determination for Lake Cubhu, mentioned above, the water quality component of the Reserve was not set, although water quality issues were taken into account when deciding on the minimum drought depth and the maintenance minimum dry-season water level. At water depths lower than those specified, it was considered that the amount of particulate organic matter and sediments that would be suspended in the water column due to wave-action would have a significant impact on the biota.

The specified maximum concentrations of water quality variables (e.g. nutrients, salts) are the "water quality resource quality objectives" (or "ecospecifications"). The results

obtained during the process above are reviewed and input from stakeholders and DWAF is obtained. The final Resource Quality Objectives for a water resource include the objectives (not only for water quality, but for geomorphology, the biota etc.) but also include economic and social considerations (N. Rossouw, Ninham Shand, Cape Town, *pers. com.* 2003).

The final step of the process is to design and implement a monitoring programme to assess whether the Reserve is being met and whether the desired Resource Quality Objectives are being achieved.

At the time of writing this review, the ecological Reserve determinations that have been carried out for wetlands are mostly for water quantity (R. Stassen, RDM office, DWAF, *pers. com.* 2003). Some consideration of water quality was taken into consideration during the Reserves carried out for the Papenkuils wetland in the Western Cape (W. Harding, Southern Waters, *pers. comm.* 2002), the Mhlatuze lakes (N. Rossouw, Ninham Shand, Cape Town, *pers. com.* 2003), and in the preliminary Reserve for the Blesbokspruit wetland (R. Stassen, RDM office, DWAF, *pers. com.* 2003). In the case of latter, it is not completely clear how water quality objectives (ecospecifications) were derived. They appear to be the values derived for riverine systems, since the water quality Reserve is for both the river and the wetland, and separate values for each are not specified.

2.5.5 The water quality component of the Reserve for other aquatic ecosystems

As noted above, the source of water and the value of the physical variables and concentration of chemical constituents in that water are likely to exert a crucial effect on the water quality of a wetland. For this reason, a brief summary is included of how the water quality component of the Reserve is determined for rivers and for groundwater.

2.5.5.1 *Determining the Water Quality Reserve for rivers*

The Reserve determination methodology is the most developed for rivers (RDM manual Volume 3). In addition, unlike wetlands, there is a reasonably comprehensive system of chemical water quality monitoring for many South African rivers and data have been collected from some sites since the 1970's. Collection of biomonitoring data (using macroinvertebrates) through the River Health Programme and other initiatives to test for

bacteriological contamination are also underway. Thus, compared to the situation in wetlands, it is usually relatively easy to assess the present water quality status of most river reaches, and it is possible to at least make an informed guess as to the historical status.

The six-step generic procedure (Fig. 2.3) that is central to the Reserve determination process is also used for rivers. The river is divided into reaches that are expected to exhibit more or less homogeneous water quality. For each reach the Reference Condition, the Present Ecological State and the Ecological Category are determined. The water quality categories are given below:

- Natural
- Good
- Fair
- Poor

Note that these four classes replace the A, B, C, D, E, F system used previously (Volume 3, Resource Directed Measures – River Ecosystems, DWAF 1999a).

Tables are provided that give the maximum allowed concentration (in the case of nutrients, salts and toxics) for each Ecological Category. These "boundary values", and how to derive them have recently been revised for rivers (DWAF 2002). A method is given for adjusting the values in cases where the back-ground levels of a constituent are naturally high. The boundary values represent a system, based on reasonably extensive data and experience, that can be used to first establish in what category a given reach currently is, both in terms of individual water quality constituents (e.g. TDS) as well as the overall status for water quality. Furthermore, they can be used to set the water quality Reserve. For example, it may be decided that a river reach is to remain in an "A" category overall for water quality. Using the new assessment (DWAF 2002) method this would mean that the maximum permitted values of selected water quality constituents are:

Total inorganic nitrogen (TIN) < 0.25 mg/L
Soluble reactive phosphorus (SRP) < 0.005 mg/L
$MgSO_4$ < 16 mg/L

The derivation of quantitative Resource Quality Objectives (called "ecological specifications" or "ecospecs" in the revised method for assessing water quality – DWAF, 2002), aids in the management of rivers and effective implementation of the ecological Reserve.

2.5.5.2 *Determining the Water Quality Reserve for groundwater*

The evolution of a methodology for determining the Reserve in groundwater resources involved considerable conceptual development, not least because the characteristics of groundwater differ significantly from those of surface water. Groundwater moves at a slow rate (usually measured in metres/year) and because it can't be seen, requires indirect methods of measurement and quantification (DWAF 1999d). Perhaps it is as a result of this that the methodology, in the extent to which it has been developed, lags behind that of rivers and even of wetlands. Due to lack of regional data it is extremely difficult to establish what the Reference Conditions should be. As a result, and because of the difficulty in linking groundwater quantity and quality to the response of surface aquatic ecosystems, a conservative approach has been taken (DWAF 1999d). In the case of the water quantity Reserve, the height of the water table "should not exhibit a long-term decline and abstraction may not impact on river and spring flow, riparian or terrestrial vegetation or land stability". It is recognised that in some cases lowering of the water table may not necessarily constitute a negative impact. In order for an exception to be made, site specific data would need to be gathered to substantiate this assessment, and permission would be required from the Minister.

In the case of the water quality component of the Reserve, the groundwater water quality of the aquifer "should not deteriorate and should remain in its ambient state". The Resource Quality Objectives would therefore be set to prevent deterioration of groundwater quality. Groundwater resources should be protected from both point-sources of pollution and from diffuse sources originating from inappropriate land-use. In addition, groundwater also needs to be protected from saline intrusion and induction of poorer quality water from adjoining aquifers, which may occur if excessive abstraction occurs in a given area. Given the vulnerability of groundwater resources to pollution and the intractability of such impacts once contamination has occurred (Bucks 1992), the conservative approach to establishing the water quality Reserve for this component of the hydrological cycle is necessary. From the above, it can be seen that monitoring of

groundwater attributes is essential in order to ensure that the Reserve is being implemented.

A preliminary classification framework for assessing the present status of a groundwater resource with regard to contamination has been drawn up. A present status category of "A" represents natural water quality. A category of "D" represents "widespread contamination, but levels relatively low, may impact the purpose for which groundwater is used." The present status categories range from "A" to "F", where in the case of the latter, contamination is so severe that groundwater use is restricted to protect users. Little further development of the water quality component of the Reserve for groundwater resources is presented in the RDM manual. Some work is currently being undertaken however as part of a Water Research Commission-funded project titled "Guidelines to set Resource Quality Objectives for groundwater" (project k5/1235; C. Colvine, CSIR, Stellenbosch, *pers. com.*, 2004).

2.6 PHYSICO-CHEMICAL CONDITIONS IN SOUTH AFRICAN WETLANDS

The values of physical variables and concentrations of chemical constituents in the water column of wetlands vary considerably. The waters of some wetlands are pure and sweet, others black and acid, and some are hypersaline (Davies and Day 1998). Even within a given region, natural water chemistry differs (King 1987; Silberbauer and King 1991a). In more temperate regions (e.g. the Florida Everglades), virtually all non-impacted wetlands will have very similar water chemistry. In South Africa, however, they vary tremendously, even within a single region. Such variability is likely to result from the fact that there are relatively few wetlands, and these are sometimes geographically wide-spaced. This is superimposed on a marked hydrological gradient across the country (which influences evaporation and salinity) and fairly variable geology (J. Day, FRU, UCT, *pers. com.* 2003). In addition to spatial variation, it will be seen from the discussion in the rest of this section that water quality also varies temporally, especially in systems that exhibit a wide range of hydrological conditions. Characterization of water chemistry in wetlands is difficult because of problems in measurement and fluctuation of the parameters according to season, time of day, or other factors. Despite these problems the description of water chemistry is important as it determines the species of organisms that will be present and has implications for utilization and management of wetlands (DWAF 1999b). Wetlands that receive most of their water from groundwater

discharge usually support more stable biological communities because water temperature and water levels do not fluctuate as much as in wetlands dependent on surface flow (Dugan 1990).

It is necessary at this point to explain the difference between water chemistry, physico-chemical conditions and water quality. Water chemistry is a term that is used to describe the chemical constituents of aquatic systems. "Physico-chemical" is used when physical variables (e.g. water temperature, water density, oxygen content) are also considered. Water quality, on the other hand is a term that is used when considering the "fitness" of a water source for a given use (e.g. irrigation, human consumption, maintenance of aquatic ecosystems). Because many of the issues pertinent to water quality apply to all types of wetland a general discussion is given. This is followed by specific aspects of water quality that relate to a particular wetland type.

2.6.1 General considerations

Silberbauer and King (1991a and b) initiated an inventory of wetlands in the Western Cape region. This work, due to lack of funding, was sadly not completed (J. King, FRU UCT, *pers. com.* 2003). The limited results that were obtained, however, represent an invaluable resource. Water samples were collected from 42 wetlands in the region and were analysed for pH, electrical conductivity (EC), total dissolved salts (TDS), and the concentration of major ions. The study covered a range of wetland types from palustrine systems, riverine-dominated wetlands to closed estuaries and endorheic pans. Electrical conductivity was found to range from nearly 1 000 mS/m to 1 mS/m; TDS from approximately 275 000 to 16 mg/L and pH from nearly 11 to 4.1. As a result, there was a wide range in the concentration of individual ions, although the authors noted that

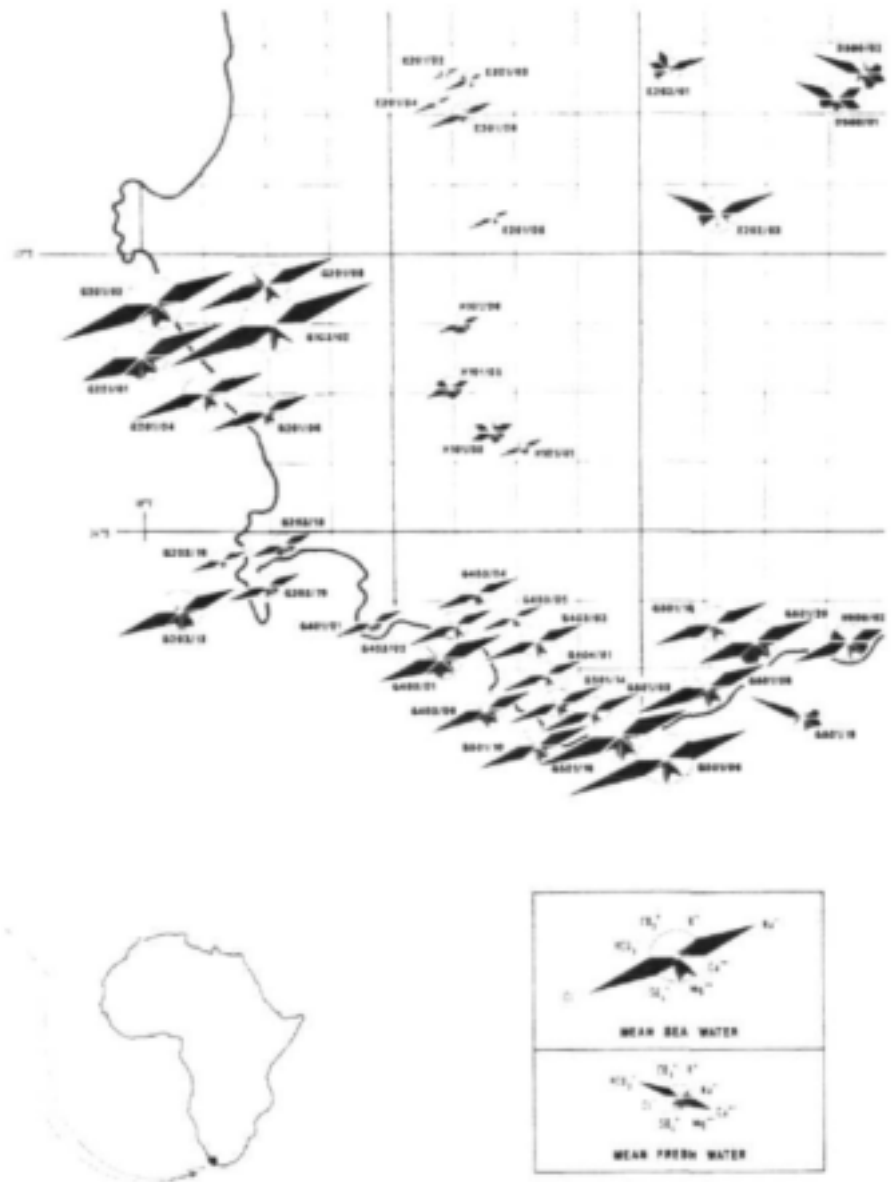


Figure 2.4 *Maucha ionic diagrams showing the relative concentrations of the major anions and cations in the waters of wetlands in the south-western Cape Province. The size of each diagram is proportional to the logarithm of the total ionic concentration. The key to the anions and cations is in the ionic diagrams for Mean Sea Water and Mean Fresh Water. The label for each wetland refers to a numbering system devised in the work refererred to and is based on the drainage region. (From Silberbauer and King 1991a).*

despite these variations there were marked similarities in the ratios of the major anions and cations (Figure 2.4). The wetlands near the coast (with one exception) were dominated by NaCl, whereas those far inland showed a predominance of HCO_3^-. It was postulated that the wetlands near the coast showed high NaCl concentrations due to a marine influence (from precipitation). Those further inland, due to the mountains were in a rain-shadow, and were influenced by the nature of the underlying geology rather than the chemical composition of the precipitation. Results from this study have been entered into a database of wetland water quality compiled as part of this project (Chapter 3).

2.6.2 Endorheic systems

Endorheic pans are usually found in arid or semi-arid areas where rainfall is low and evaporation high. As a result, water can range from freshwater to saline and the salinity can change considerably over the short period of time that the wetland is inundated (Seaman and Kok 1987). In addition, most inland endorheic systems are shallow, which has implications for water chemistry (see the case study below). In general, as for most types of wetlands, there are only limited data available on the physical and chemical properties of endorheic pans (Allan *et al.* 1995). Many pans (although not all of them) are brackish (i.e. the salinity is > 3 g/L). According to a study carried out by Seaman *et al.* (1991), the majority of the 55 pans examined were brackish/saline, but not particularly so (i.e. the salinities were greater than 3 g/L but less than 50 g/L). It was found that a great many smaller pans were relatively fresh when inundated and were vegetated. This was in contrast to the larger pans that were very saline and virtually bare. Amongst the 55 pans that were studied, sodium was the dominant cation, although a variety of patterns occurred in the dominance of the other cations, Mg, Ca and K. A common pattern was Ca>Mg>K, as well as Mg>Ca>K. The most common anion by far was Cl, followed by SO_4^{-2} (in only 7% of the 55 sites studied) followed by carbonate or bicarbonate. In the case of both cations and anions there were no obvious regional differences amongst the patterns of ion dominance, although SO_4^{-2} was usually found in the Northern Cape/Southern Namibia (in combination with NaCl-dominated pans). According to Seaman *et al.* (1991), the higher the salinity of a system, the greater the likelihood it would be dominated by NaCl. It must be noted that most of the above information was based on a single sample from each site. The authors concluded that as pans dry, salinities do not necessarily increase to high and then saturated values. There is a lack of long-term studies for South African wetlands of all types. These authors also

commented on the limited pH data available for southern African saline lakes and concluded that the pH appeared to be high (higher than 7.0) - although pans from the Western Cape Province were not examined.

The above study was followed by the work of Day (1993) who reviewed the data concerning the major ion chemistry of some southern African saline pans and lakes (including those included in the studies by Seaman *et al.* (1991), and Silberbauer and King (1991a)). The system in South Africa with the highest TDS was found to be Koekiespan in the South-western Cape (276 g/L). It was confirmed that in South Africa saline waters are dominated by NaCl. Freshwaters on the other hand (TDS < 3 g/L) were Na, or Na and Ca, dominated with CO_3^{-2}/HCO_3^- or SO_4^{-2} as the dominant anions. Cole (1968) states that in spite of marked volume changes in saline lakes, the relative ionic composition of a given lake remains stable. Thus although samples from different seasons may vary in concentration, the composition is likely to remain the same.

The distribution and origin of endorheic pans in the Western Orange Free State was examined by de Bruiyn (1972). This author classified the pans into 5 major groups each of which could be either vegetated or non-vegetated. Although the author does not specifically mention water quality, the pans were classified according to the chemical nature of the pan floor – a reflection of the major chemical ions in the water column. The Five groups were salt pans, lime pans, clay pans, soil pans and gypsum ($CaSO_4$) pans. Interestingly, the author postulates that the pans were found mainly in the western part of the country because one of the major causative agents was accelerated weathering of the substrate due to salt formation. In the east, although suitable geological formations were present, increased rainfall meant that salt formation did not occur. Pans of a certain type were often found in a given area, but sometimes, for example, the distribution of lime and salt pans was found to be so similar, with the two types occurring so close to each other, that it was difficult to draw meaningful boundaries between the various types. This work emphasises that it may not always be easy to deduce from the area or from surrounding wetlands what the natural water quality should be.

In the case of endorheic pans, the degree of inundation would be expected to exert a major effect on water quality. Batchelor *et al.* (2002) note that because pans are endorheic systems, subject to evaporation over time they are likely to become more

saline. A process that may remove material from pans is the blowing away of accumulated salts, organic nitrogen and phosphate and sulphate salts by the wind when the pan is dry (Day 1993). Outward seepage to groundwater has been reported in some systems by other authors and may help to prevent the build-up of dissolved substances such as sodium sulphate. Phosphorus may be released into the water column from the sediments under anaerobic and reducing conditions (during inundation), or under conditions of low pH. During dry periods phosphorus would be bound to the substrate.

2.6.2.1 *Case-study: Rolfes Pan*

Rogers *et al.* (1989) studied an endorheic pan on the outskirts of Johannesburg. They highlighted three limnological characteristics of pans that have a profound effect on ecological functioning and on water quality. These are:

- A high surface area to volume ratio. As a result, these systems are subject to marked fluctuations in turbidity and temperature, and evaporation is high. The action of wind is also pronounced, causing re-suspension of sediments with concomitant changes in dissolved oxygen, pH and nutrient concentrations in the water column.
- A high ratio of littoral (shore) zone to pelagic (deep-water) zone. As a result, the production of organic matter is high.
- The endorheic nature of these systems means that they are extremely vulnerable to pollution in general and eutrophication in particular. (This last point has been emphasised by others including Lemly, Finger and Nelson 1993).

The above researchers measured (amongst other attributes) the hydrological and physico-chemical characteristics of the pan during two years (1983 and 1986). The first year of monitoring (1983) coincided with a period of drought, whilst 1986 was a relatively wet year. During 1983, water depth and surface area decreased gradually and then rapidly during the dry season. In 1986 however, water depth and area were more or less constant over the entire hydrological year. Attempts were made to correlate physico-chemical attributes with inundation depth. There was a noticeable difference in the water quality regime between the two years. It was found that dissolved oxygen (DO) levels, water temperature, pH and conductivity were much more variable during 1983 compared to 1986. This was attributed to the pronounced alteration in the ratio of surface area to volume under drought conditions. In general, DO concentration and pH appeared to *decrease* with receding water levels, whilst temperature and conductivity *increased*.

Nutrient concentrations did not change in relation to evaporation losses and were extremely variable. There was an apparent marked increase in nutrient concentrations between 1983 and 1986 leading to a 5-fold increase in nitrogen and a 2-fold increase in phosphorus. Three possible reasons were given by the authors for this increase:

- *eutrophication due to changes in nutrient inflows arising from anthropogenic activities* in the catchment;
- leaching of nutrients from soil and plants that became inundated during the high water levels in 1986 (a similar situation was described by Coetzer (1981) in a West Coast pan);
- changes in the species of birds visiting the pan (namely gulls which feed outside of the water resource, but defecate in the water).

Whatever the cause, the increase in nutrient concentrations was enough to change the pan from meso- or eutrophic in 1983 to hyper-eutrophic in 1986, and to change it from a system largely dominated by submerged macrophytes to one dominated by phytoplankton (see section 2.4.2).

Primary production also appeared to influence the dissolved oxygen concentrations and pH. During periods when the dominant plant producers were submerged macrophytes, oxygen concentrations were high and the pH was above 9.0. As water levels started to recede both DO and pH began to fall. Even in spring, when the rainy season began again, DO and pH still decreased and this was attributed to the inundation of mud and dead plants (DO would be consumed during the decomposition of organic matter accumulated during the dry season). In 1986, when plankton formed the main submerged source of primary production, DO and pH levels were more stable. These authors go on to highlight the importance of natural fluctuation in water levels of pans compared to stabilisation of inundation depth due to anthropogenic influences, in order to maintain biodiversity. From the above, it also then follows that water quality should vary, rather than remaining constant, which is an important consideration when setting water quality objectives.

2.6.2.2 *Case-study: Rocher Pan*

Coetzer (1981 and 1987) studied physico-chemical variability in a seasonal, endorheic. West Coast wetland (Rocher Pan) including the effect of these factors on biotic response. Measurements were made in 1976/77 and again in 1981/82. In general the

pan starts to fill up in May with the onset of winter rains. It remains inundated until February/March of the next year. This pattern depends on rainfall and after good rains the pan may stay inundated for the entire year.

It was found that the concentration of chlorides, total dissolved solids, as well as electrical conductivity was very similar between 1976/77 and 1981/82. Not unexpectedly these variables were lowest in around September (when depth was maximal) and highest in January (when evaporation results in high concentrations of salts). Alkalinity (measured as calcium carbonate) was found to be correlated with depth and peaked a month after the water level started to recede.

Total nitrogen was more variable between years than conductivity, but also appeared to exhibit the lowest values during winter and highest values in January/February. Ortho-phosphate concentrations, however, varied greatly between the two sampling programmes, being higher in 1981 (maximum 2.0 mg/L) than in the earlier programme (maximum 0.5 mg/L). There was some similarity in the seasonal trends of ortho-phosphate, however. This nutrient frequently increased in concentration until October and decreased again. According to Coetzer (1987) this decrease is due to the rapid growth in spring of benthic macrohydrophytes and filamentous algae, which resulted in a sharp decline in the ortho-phosphate concentration. In summer, concentration again increases again towards the end of inundation when the hydrophytes die off. Concentrations of ortho-phosphate were high during winter during the time when inflow into the vlei was maximal. The author considers that this was due to the high turbidity which resulted in low algal standing crop and thus under-utilization of available dissolved nutrients. Water quality and Rocher Pan are discussed further in Chapter 5.

2.6.3 Palustrine systems

Batchelor *et al.* (2002) describe some of the processes affecting water quality in seepage wetlands which are areas in which groundwater emerge. Seeps may be either seasonal or perennial. They may be unconfined (not surrounded by hard substrates) or confined (as a spring). One of the important points of this type of wetland is that where groundwater flows are permanent and relatively constant, organic deposits may accumulate and may result in organically rich sediments (peat). This is due to the growth and senescence of plants which under the water-saturated, anaerobic conditions, do not

readily decompose. The reduction of nitrate, sulphate, iron and manganese is also likely to occur under these anoxic conditions. Because seepage wetlands are so diverse (ranging from peatlands to mossy bogs), and it is difficult to measure groundwater flow, it is very difficult to quantify nutrient removal in these systems.

2.6.4 Riverine wetlands and floodplain systems

Batchelor *et al.* (2002) note that floodplains receive water during periods of high rainfall, when the volume of water exceeds the capacity of the channel and spills into marginal areas. During the initial wetting phase, salts and nutrients may be leached from the soil into the water column. Conversely, depending on the cation exchange capacity of the soil, minerals may be taken up by the flooded soil from the flood waters. Under conditions of high flow (and thus high dilution) the concentration of dissolved substances will generally be low. Water travelling down the river will be at a high velocity and is likely to carry high sediment loads. Once it has reached the floodplain, however, the velocity slows and sediments and can be expected to be deposited. Phosphorus bound to sediments on the floodplain is likely to be recycled through plant and animal uptake. As in the case of pans, if sediments are inundated for long they are likely to become anaerobic, and phosphate in a soluble form will be released into the water column. Nutrients and organic carbon may also leach from terrestrial plants that die after being submerged for long periods under water (Kingsford 2000).

2.6.5 Lacustrine systems

As stated in the introduction to this document, lakes will not be discussed in any detail in this review. The water quality of this type of wetland is discussed in Hart (1995). Mention must be made, however, of the phenomenon of thermal stratification which has a profound influence on the water quality of deeper water bodies such as lakes and reservoirs.

The following description of thermal stratification is taken from Davies and Day (1998). The density of water increases with decreasing temperatures. In a hot country such as South Africa, during the summer, water near the surface of a water body is heated by the sun. This reinforces the effect of differential densities and the warm water remains in the upper portion of the water column. At the same time colder, denser water is trapped under the warmer, lighter upper layers, causing a sheer plane to be formed between the

two masses of water. This division of water masses is usually maintained until the ambient air temperature begins to cool in autumn or winter, or until winds blow that are powerful enough to mix the water and break the thermocline (referred to as "lake overturn"). In shallow water bodies thermal stratification does not occur, because heat from the sun penetrates a large proportion of the water column and even relatively light winds can cause enough mixing to break the water strata.

The division of the water body into two separate masses, the upper epilimnion and lower hypolimnion during the growth period of spring and summer has profound effects on aspects of water chemistry, other than temperature. Living organisms tend to thrive in the warmer epilimnion where photosynthesis can take place. This rapid growth, often leads to the depletion of available nutrients in the water column, an effect that is exacerbated by the loss of dead organisms (phytoplankton, fish, etc.) which die due to natural life cycle progression. Dead organisms sink to the deeper hypolimnion, further removing nutrients. In the hypolimnion, the accumulation of dead organic matter that settles from above causes a rapid increase in decomposing bacteria and a concomitant decrease in oxygen content. Nutrients may be released but remain trapped in the hypolimnion by the sheer plane and cannot be utilised by organisms in the epilimnion. The anoxic conditions in the bottom layer, in turn frequently lead to the reduction of sulphate and the production of large amounts of toxic hydrogen sulphide gas.

The basic method of stratification and overturn varies in different lakes depending on the local conditions, and a system for classifying lakes has been devised based on this. In some situations, dense, saline water enter lakes from the sea or from springs. Such saline water tends to remain in the bottom of lakes (for example in Swartvlei in the Southern Cape) forming a stagnant layer which mixes to a very limited extent with upper freshwater layers, even after overturn. The various types of stratification are discussed further in Davies and Day (1998). Hart (1995) gives a thorough description of the variation in water quality regimes in South African lakes, their ecology and the threats that they face.

2.6.5.1 *Case study: Zeekoevlei*

Although it is not the place of this review to discuss the rapidly- expanding field of lake restoration, it is unfortunately a fact that many lakes globally and in South Africa are

polluted to the point that rehabilitation measures need to be implemented. As noted in section 2.4.2.1, one of the major threats facing wetlands, including lakes, is the threat of eutrophication. This is especially true of lakes with urbanised catchments where loading of nutrients from defuse sources is excessive. An example of one such lake in South Africa is described by Harding and Quick (1992) who discuss the causes and consequences of hyper-eutrophy in Zeekoevlei, an urban lake in Cape Town. The lake is currently plagued by dense blooms of cyanobacteria (*Microcystis spp.*) and other algae, encroachment of reeds along the shore-line and an accumulation of sediment from algal origin. Anoxic sediments often lead to unpleasant odours and fish kills, all of which impact on the recreational use of the area. It was concluded that both catchment- and lake-based management strategies would be required in order to rehabilitate the lake. Firstly nutrient loading (particularly of phosphates) into the lake would need to be reduced. It was suggested that this could be accomplished by constructing artificial wetlands further up in the catchment to polish horticultural return-flows. This would need to be accompanied by an educational programme in the area, explaining the need to reduce nutrient loading. It was concluded that the internal loading of phosphate within sediments of the lake, however, was already so high that in order to bring about a reduction in phytoplankton blooms, this would need to be significantly reduced. The method of choice for the Zeekoevlei would be dredging, which should ultimately result in reduced phosphate levels leading to lower phytoplankton densities, increased water clarity and growth of submerged macrophytes.

2.7 INTERNATIONAL WATER QUALITY REGULATIONS FOR WETLANDS

Water quality *guidelines* establish benchmarks for assessing the acceptability of water for a particular use (Lawrence *et al.* 1994 cited by Liston and Maher 1997). Such guidelines represent the desirable condition of water bodies, but don't necessarily need to be adhered to. Water quality *standards*, on the other hand, are legally enforceable measures. Water quality guidelines and standards are based on technical and scientific information and are developed for management purposes. During the development of these guidelines, judgements need to be made as to the acceptable risk to humans or aquatic ecosystems (Liston and Maher 1997). In South Africa, water quality guidelines for the protection of aquatic ecosystems were developed by DWAF (1996). This work gives recommended values of physical variables and chemical concentrations that

should not be exceeded in aquatic resources, but does not specify if they are valid for rivers or for wetlands. Development of the guidelines was part of a gradual shift from end-of-pipe regulations to regulate point sources (e.g. the 1 mg/L phosphate standard for waste water treatment works (Heath, Steynberg, Guglielmi and Maritz 1994)) to a receiving water-based strategy (van Vliet, Swart and Roux 1994). Since implementation of the National Water Act (1998) specifications for water quality (in other words, water quality standards for the receiving waters), linked to different levels of ecosystem protection, are specified as part of the ecological Reserve (section 2.5). Currently in South Africa a combination of source-directed controls and resource-directed measures is employed to control pollution of aquatic resources, an approach that is used in many countries of the world (Lijklema 1995).

A survey of the literature was undertaken to examine the approaches used in other parts of the world with regard to the protection of water quality in wetlands. In general, apart from the USA and Australia, little mention of wetlands was found in the water quality guidelines for the respective countries, and quantitative criteria for these ecosystems were rare. The results that were obtained are discussed below.

2.7.1 United States of America

Perhaps because of the large number of lakes in the USA, water quality criteria for wetlands, in one form or another, have been enforced for a relatively long time. As will be seen, a combination of narrative statements and numeric standards are used. Narrative water quality criteria define conditions that must be protected and maintained to support a designated use. They may be as general as a statement such as "ambient water quality shall be sufficient to support life stages of all native aquatic species" or they may be more specific as in section 2.7.1.2, below. These types of standards, although vague, are an important tool for protecting wetlands from impacts and need to be considered (Robb 1992). Narrative criteria are especially important for wetlands, since many wetland impacts cannot be fully addressed by numeric criteria in (EPA 1990). For example, impacts may result from the discharge of chemicals for which numerical standards have not been designated. On the other hand, impacts may result from activities that affect the physical and/or biological, rather than the chemical aspects of water quality (e.g. turbidity), and narrative water quality criteria are required to ensure that these aspects are protected. Narrative guidelines for wetlands have slowly been

supplemented in the USA by the use of numeric guidelines, as field and ecotoxicological data relating the level of a water quality variable to the effect on biota have become available.

In much of the policy development for protection of wetlands in the USA, the term "water quality standards" is used in a rather general way, and a wider range of wetland attributes (i.e. sedimentation, hydrology and living organisms) than water quality parameters are considered. This approach serves to prevent, or limit activities in a wetland catchment that might cause (amongst others) impacts to water quality, rather than putting limits on the amount of pollutants that are allowed in water body.

Ample information on water quality in wetlands is given on the website for the Environmental Protection Agency (http://www.epa.gov)/owow/wetlands). The situation is complicated, however, because although the US-EPA gives direction to the overall policy for wetland protection, there is latitude for individual states and counties to develop their own regulatory approaches. As a consequence, there is no uniform protocol used on the ground. The development of the policy for water quality in wetlands is described below. The major emphasis is on the types of approaches that have, or are, being used rather than an accurate situation assessment or developmental history.

2.7.1.1 *The major water source*

One of the first general approaches used was to take note of the major water source for a given wetland. If the wetland was considered to be "largely riverine", numeric standards that applied to rivers were used. If wetlands were considered to be "largely estuarine" then again standards developed for estuaries were used. This is the approach used in the Decision Support System developed in this project (Chapter 4) and suggested in the RDM method for wetlands (section 2.5.4).

2.7.1.2 *Control of development in the catchment*

An example of policy using this type of approach is that of the Wisconsin Department of Natural Resources (WDNR) 1992). It is based on narrative standards. Under this approach, new activities planned for an area (or those requiring re-authorisation) first need to be assessed by the local environmental agency to see if an impact on any wetlands might occur as a result of the activity. A project planning process is instituted

that requires the project proponents to consider alternatives to avoid wetland impacts. If it is essential that the proposed development take place within the catchment of the wetland, it must be shown that there will be no significant adverse impacts to wetland functional values. The burden of proof is placed on the applicant to show compliance with the standards. If significant impacts to the wetland are likely to ensue from the activity, permission for the development would not be granted.

The standards are designed to protect not only water quality, but all the functions and values of wetlands including; biological diversity and wildlife habitat, sediment and pollution attenuation, storm and flood water retention, hydrologic cycle maintenance, shoreline protection and human uses such as recreation and education. Effects that might impact on wetland functioning, and that are taken into account during the process may either be:

- direct effects which are caused by the action under question and occur at the same time and place, or
- indirect effects which are caused by the action in question and are later in time or further removed in distance, but are still reasonably foreseeable.

The water quality standards for the discharge of materials and protection of hydrologic conditions and habitat state that the following substances (listed below) "*may not be present in amounts which may cause significant adverse impacts to wetlands.*"

- Liquids, fills or other solids or gas.
- Floating or submerged debris, oil or other material.
- Materials producing colour, odour, taste or unsightliness.
- Concentrations or combinations of substances which are toxic or harmful to human, animal or plant life when considered individually or cumulatively.

In addition:

- Significant adverse impacts must be prevented for the following parameters: water currents, erosion or sedimentation patterns, water temperature, the chemical, nutrient and dissolved oxygen regimes, movement of aquatic fauna, pH, and water levels or elevations.

2.7.1.3 *More recent approaches*

The Clean Water Act was updated in 1998 with the objective of "restoring and maintaining the chemical, physical and biological integrity of water bodies" in the country. An integral part of the Act is the specification of water quality standards for all waters, including wetlands (EPA 1998a). The water quality standards consist of several components:

- Designated beneficial uses of the wetland.
- Narrative water quality criteria for supporting each use.
- An anti-degradation statement.
- Numeric water quality criteria for supporting each use.

Different levels of protection can be assigned. Designation of a wetland (or other type of water body) as being an "outstanding national resource water" offers special protection, and more protective water quality criteria can be applied in such cases. A task force has recently been set up to examine the water quality criteria for the protection of aquatic organisms. This task force is charged by the US EPA with reviewing and revising the guidelines for deriving numerical water quality criteria for all types of water body over the next three years (EPA 2002). This revision is considered to be necessary to take into account recent developments in (amongst others) risk assessment methodology.

Designated uses

The designated uses mentioned above, include functions such as aquatic life support, groundwater recharge, propagation of fish and wildlife etc. They may be current uses or attainable uses that can be achieved using best management practices and represent the environmental goals for the water body. Under no condition may the designated use be waste transport or waste assimilation, although certain other activities involving the discharge of pollutants into wetlands may be permitted. This must be accompanied by monitoring to ensure that wetland functioning and values are preserved.

Narrative water quality standards

Narrative water quality criteria have already been discussed. These must support the "anti-degradation policy" - a statement to ensure that even if water quality in a resource is better than the minimum level established by the state water quality standards, deterioration will not occur.

Numeric water quality standards

The US-EPA has developed quantitative water quality standards for wetlands (EPA 1998b). The criteria are stated in terms of chemical, physical and biological parameters. The values of these criteria (as in many countries) depend on the designated uses that have been set for each resource (or part of a resource). Individual states and tribes may use these values as they stand to enforce maximum acceptable concentration levels of pollutants in ambient water. States may also adjust these values to suit local environmental conditions (e.g. climate, geology), local human exposure patterns, or use other, scientifically proved methods. A major thrust in water quality management for all water bodies, not only wetlands, is towards the control of excessive nutrient levels. New nutrient criteria have recently been drawn-up that vary (and are tailored towards) different ecoregions (see below). An ecoregional approach has been found to be necessary because different parts of the country vary significantly in terms of background levels of phosphates and nitrogen (EPA 1998b).

The values of the recommended EPA criteria for nutrients (for *lakes and reservoirs*) in each region of the USA are given in Table 2.4. Individual values for the different ecoregions value widely, but represent water quality conditions that are minimally impacted by anthropogenically-induced nutrient enrichment. Thus a low level of enrichment is allowed, but this would not be enough to cause adverse effects. The standards provide for protection and propagation of aquatic life and have been derived by taking into account:

- Characterisation of the reference condition.
- Consideration of historical water quality data.
- Use of predictive models.
- Expert judgement.
- Consideration of down-stream effects.

Table 2.4 *Nutrient standards for lakes and reservoirs in the USA. Ecoregions have been aggregated into areas. (From http://EPA.gov/nutrients).*

Para-meter	Aggregate Ecoregion											
	II	III	IV	V	VI	VII	VIII	IX	XI	XII	XIII	XIV
TP (mg/L)	0.009	0.017	0.020	0.033	0.038	0.015	0.008	0.02	0.008	0.01	0.018	0.008
TN (mg/L)	0.10	0.40	0.44	0.56	0.78	0.66	0.24	0.36	0.46	0.52	1.27	0.32
Chl a (µg/L)	1.90	3.40	2.00	2.30	8.59	2.63	2.43	4.93	2.79	2.60	12.35	2.90
Secc. (m)	4.50	2.70	2.00	1.30	1.36	3.33	4.93	1.53	2.86	2.10	0.79	4.50

TP =Total phosphorus, TN = Total Nitrogen, Chl a = chlorophyll a, Secc. = Secchi disk depth.

For lakes and reservoirs two ways were used to derive standards:

1. Selecting the 75th percentile of a distribution of nutrient measurements from *reference condition sites* as the recommended target. This is considered to give a sufficiently protective target and excludes the effects of outliers.

2. If information on minimally impacted sites on a local scale is not available, the 25th percentile of a distribution of samples from the *entire population of water bodies* (within, for example, a given ecoregion) can be used. The EPA have found that for lakes and reservoirs, the 25th percentile of a sample distribution from the entire population serves as a surrogate for the 75th percentile of a sample distribution from reference sites.

Currently sufficient data are not available to develop nutrient criteria for *wetlands* (other than lakes), although the gathering of monitoring data on nutrients and response variables (chlorophyll *a* and turbidity) is being undertaken.

Numeric values for different toxic substances are given in the EPA's "National Recommended Water Quality Criteria: (2002)". These have been derived from ecotoxicological databases and from field observations. Values are given for saltwater and freshwater systems to protect aquatic life. No indication is given as to how these

should be applied to different freshwater ecosystems e.g. wetlands as opposed to rivers. Recommended water quality standards for over 120 toxicants are listed, although there are no values for system variables (e.g. dissolved oxygen, sediments, temperature) or naturally occurring chemical components.

2.7.2 Australia and New Zealand

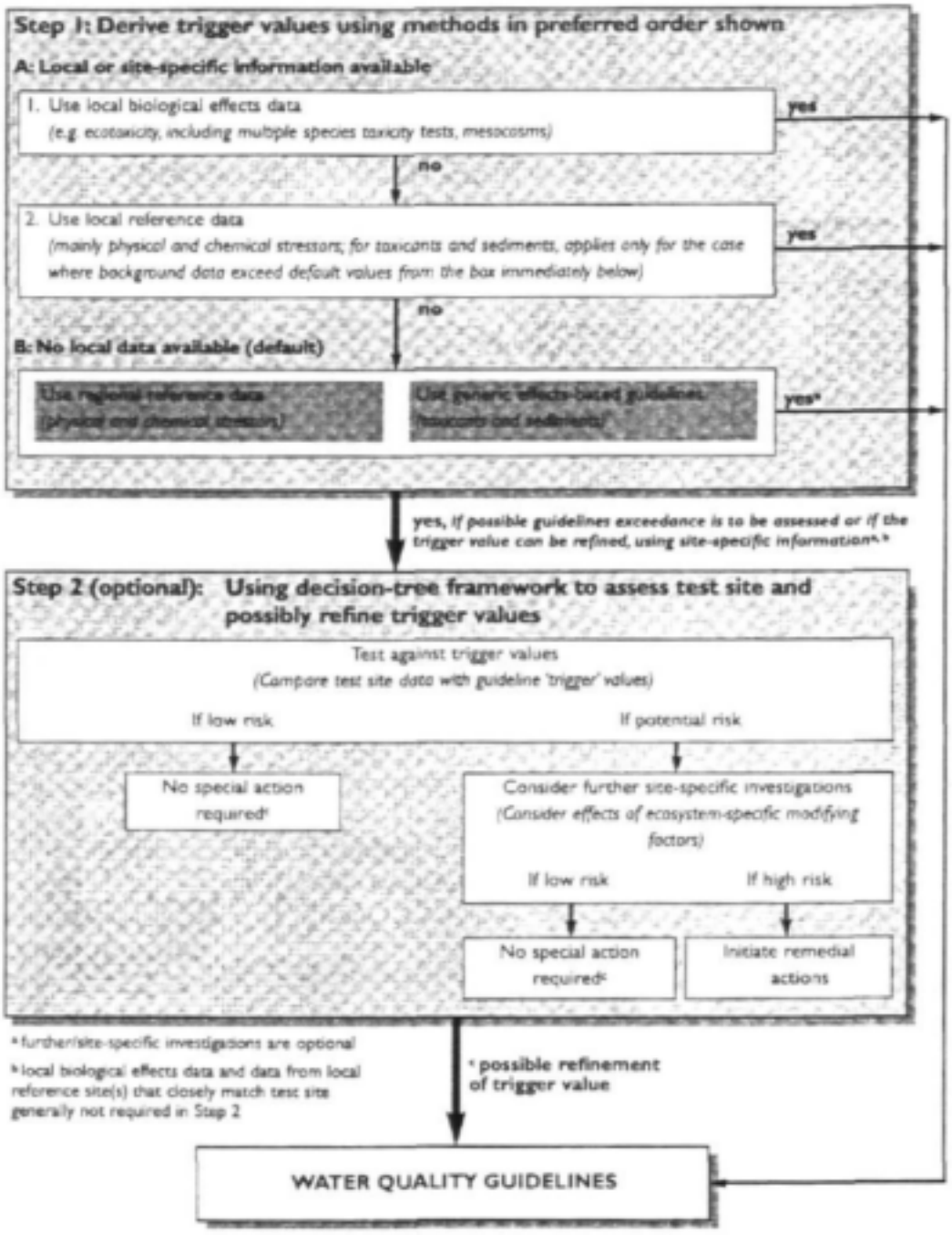
The revised version of the Australian and New Zealand guidelines for fresh and marine water quality (ANZECC/ARMCANZ 2000) forms part of a new national water quality management strategy for those countries. The guidelines are considered to be largely risk-, rather than hazard-based, since generic criteria can be modified to give site-specific values. The guidelines present an interesting philosophy with regard to aquatic pollution and because of this, the overall approach, rather than just the derivation of water quality standards is discussed.

Figure 2.5 shows an outline of the procedure for deriving and refining the water quality guidelines. Two different approaches (A and B) are shown in the upper box for deriving trigger values (termed "trigger values" because if these, or higher, concentrations are detected in the system in question this must trigger a management response). It is recommended by the authors that the use of local data (i.e. approach A) to derive trigger values is the most suitable method. At the same time, however, it is recognised that frequently such data will not be available and that generic standards (approach B) will need to be used.

In the case of approach A, the preferred method is to use site-specific biological effects data; this may include the results of ecotoxicity tests that have been carried out on local species, or the results of mesocosm experiments. Such data takes into account all site-specific environmental factors, and trigger values derived from this approach will be tailored to protecting the biota at a given site.

These trigger values are used (along with stakeholder input) to derive the water quality objectives for the site and do not need to be further refined. This approach emphasises the importance given to biological data that are able to give a direct assessment of the ecological integrity at a site (be it on a river, wetland, lake etc.).

Figure 2.5 *Procedure for deriving water quality guidelines as given in the Australian and New Zealand guidelines for fresh and marine water quality ANZECC/ARMCANZ (2000).*



The guidelines stress that such an approach is necessary because there is still, in general, insufficient knowledge to predict the likely effect of a given level of a pollutant on the aquatic biota.

If no local biological effects data are available, physical or chemical data that represent the "target condition" can be used to set the trigger values (Approach A2, Figure 2.5). Such data can be obtained from the following sources:

- from a nearby, unimpacted system (but which is apart, from the impact, considered to have similar water quality to the test site);
- historical (pre-impact data) – from the test site (or presumably from a comparable site);
- other sources – e.g. literature, modelling, expert opinion.

If such chemical and physical information is available, it can be used to extrapolate to the test site and set water quality guidelines. It should be noted that in ANZECC/ARMCANZ (2000), such "target condition sites" are termed "Reference Condition sites". The usage is not quite the same as in the South African sense, since the guidelines state "the Reference Condition should not be defined in terms of ecological targets that are impossible to attain. Conversely the Reference Condition should represent a substantial achievement in environmental protection that is agreeable to the majority of stakeholders". Thus the RC may represent the pristine condition, but usually will represent some slightly impacted state. To avoid confusion, in this report the term "Reference Condition" in the sense used in the ANZECC/ARMCANZ guidelines is replaced by the term "target condition".

As an example of this approach, the following case study can be considered. The major concern at a wetland that is under investigation is excessive growth of nuisance algae. The monitoring data for a nearby wetland (which is considered to have similar water quality to the test wetland – but normal nutrient levels) are used to set the trigger values. Data for the past two years are collated for the "target condition site". The 80th percentiles of the total phosphorus, total nitrogen and chlorophyll *a* concentrations are calculated from data representing the season or growth period when algal growth is likely to be the greatest. These values represent the trigger values for nutrients and chlorophyll *a* at the test site. In some cases, concentrations are considered, in others,

algal biomass is considered to be proportional to the available mass of nutrients, and loads (mass/unit time) rather than concentration are used to set trigger values. The concept of the "target condition" is also used for chemical constituents whose background level is naturally high in a given system (higher than the values specified in the generic default tables).

In the case of approach B (Figure 2.5), use is made of default water quality standards (quantitative values) that have been prescribed for a variety of fresh-water (e.g. upland rivers/streams, lowland rivers) and marine systems, including wetlands. These values are conservative and communities, industry, or other water users are encouraged to modify the default guideline values for individual sites in a formalised manner (see below). The values given for wetlands in the guidelines are not comprehensive and not all climatic regions are represented. It is also recognised that there is insufficient information to characterize the water quality requirements of ephemeral rivers or salt-water lakes. Updates to the default water quality standards are now available on the website for the Australian Government Department of Environment and Heritage (http://www.deh.gov.au/water/quality/targets). The values that have been specified are for different climatic regions and are summarised in Table 2.5. These represent the values for slightly, to moderately, disturbed systems and were derived from measured data from wetlands in each climatic region. It is stated that where a system is highly disturbed, the same values can be used as default targets. Where the system is of high conservation value and/or essentially undisturbed, the set target should ensure that there will be no detectable change from natural variation.

If, on examination of the chemical monitoring data for a site, the ambient concentration is less than the trigger (target) value, no further action (other than on-going monitoring of chemical, physical and biological attributes) needs to be carried out, since the risk of an impact is low. If the trigger value is exceeded, however, more complex investigation can be undertaken by the water user to measure factors that may modify the toxicity/bioavailability of the stressor.

Table 2.5 *Default target values for Australian wetlands, lakes and reservoirs (adapted from "Water quality targets online" http://www.deh.gov.au/water/qualilty/targets).*

Climatic regions	Ecosystem type	Nitrogen		Phosphorus		Turbidity (NTU)	Salinity (surface and groundwater) (µS/cm)
		Total Nitrogen (µg/L)	Oxides of Nitrogen (µg N/L)	Total Phosphorus (µg/L)	FRP[1] (µg P/L)		
South-West: Western Australia	Lakes/Reservoirs Wetlands	350 1500	10 100	10 60	5 30	10-100[2] 10-100[3]	300-1500[4] 300-1500[4]
South-central: South Australia	Lakes/Reservoirs Wetlands	1000 No data	100 No data	25 No data	10 No data	1-100[2] 1-100[2]	300-1000[5] 300-1000[5]
South-east:	Lakes/Reservoirs Wetlands	350 No data	10 No data	10 No data	5 No data	1-20[2] No data	20-30[6] No data
Tropical: Northern Territories	Lakes/Reservoirs Wetlands	350[7] 350	5 10	10 10	5 5	2-200[2] 2-200[2]	90-900[8] 90-900
Tropical: Queensland	Lakes/Reservoirs Wetlands	350[7] 350	10 10	10 10	5 5	2-200[2] 2-200[2]	90-900 90-900
Tropical: Western Australia	Lakes/Reservoirs Wetlands River pools	350[7] 350 1200	10 10	10 10 50	5 5 25	2-200[2] 2-200[2]	90-900 90-900

[1] *FRP = Filterable Reactive Phosphorus*
[2] *Low in deep lakes; High in shallow lakes, often due to wind; varies with geology*
[3] *Varies depending on catchment condition and water level*
[4] *Low values during seasonal rainfall events; wetlands often higher due to summer evaporation.*
[5] *Highly variable due to saline intrusions and evaporation.*
[6] *Values typical of Tasmanian lakes and Resrvoirs.*
[7] *For turbid lakes only; Clear lakes have lower values.*
[8] *Low in Northern Territory permanent billabongs; High in Western Australia wetlands and in summer due to evaporation.*

This step requires permission from the relevant authority, however. The default trigger values (given in Table 2.5) can be altered according to a decision framework (Step 2 in Figure 2.5). This takes into account site-specific environmental factors such as water hardness, pH, turbidity etc. that may modify toxicity or the impact of a water quality variable. Because the trigger values are conservative, examination of site-specific environmental factors will frequently lead to an

increase in the allowable concentration. For example, the default values represent the concentration of the "bioavailable fraction", whereas typically, monitoring data may be of total concentration (and thus overestimate the possible impact). If after consideration of site-specific investigations it is still considered that there is a high risk to the biota, management steps must be taken to reduce the pollution. At any stage in the process, it may be recognised that the cost of investigating possible ameliorating factors may be higher than implementing more stringent water quality standards and the choice can be made to rather accept the default criteria.

2.7.2.1 Biological indicators

There is a strong emphasis in the guidelines on monitoring ecosystem health through the use of appropriate biological indicators as opposed to just the concentration of specific instream chemicals. For this reason, Approach A in Figure 2.5 is the preferred approach, rather than the automatic use of default trigger values. The guidelines for aquatic ecosystems are directed at protecting the environment by identifying any concerns at a site (e.g. toxicity, algal blooms or loss of biodiversity). The environmental processes that result in the problem are identified and monitoring using appropriate indicators needs to be undertaken (e.g. phytoplankton counts if algal blooms are the major concern, or multiple-species ecotoxicological testing in the case of toxins). A wide range of biological assessment methods is specified in the guidelines for the different aquatic ecosystems, and for different situations. These range from Rapid Biological Assessment methods using macroinvertebrates, to methods that are designed to give an "early warning" of possible impacts to ecosystem processes.

Three different levels of aquatic ecosystem condition are recognized, with an appropriate level of protection ascribed to each:

- high conservation value (largely pristine ecosystems);
- slightly or moderately disturbed;
- highly disturbed systems.

The guidelines recommended that several complementary biological assessment methods are used for monitoring, and the higher the level of protection required, the more rigorous the biomonitoring methods. Advice is given as to which combinations of methods should be used to suit the level of protection, the perceived threat to the ecosystem and the type of aquatic ecosystem (river, wetland, marine system etc).

2.7.3 United Kingdom and other countries

During the 1990's a new system of statutory water quality objectives was introduced in the UK. This system included rivers, canals, groundwater, coastal waters and lakes (NRA 1991). The aim was to maintain the water quality of the country's water bodies and improve it where necessary. The situation is made complicated by the fact that sometimes European Union Directives (e.g. for the discharge of toxic contaminants into aquatic ecosystems) are also relevant. The importance of wetlands as sites of high biodiversity and the potential threat of effluent discharges to these systems has been recognised. An extensive national programme is underway (to be completed by 2005) to remedy these impacts (Environment Agency 2003).

Table 2.6 shows the water quality standards for rivers and lakes that are relevant to the protection of ecosystem functioning (NRA 1991). Standards are expressed as the 95th percentile, except for dissolved oxygen (5th percentile). Thus in the case of dissolved oxygen for lakes used as salmonid fisheries, 95% of the monitoring values over a given period must be above 5mg/L. Water quality standards for wetlands specifically were not reported, but may be available in more recent documents.

Although water quality guidelines for the protection of aquatic life have been developed in Canada, no mention could be found of any developed specifically for wetlands (Environment Canada 2003).

Table 2.6 *Proposed water quality standards for rivers and lakes that are relevant to the protection of ecosystem functioning (Adapted from National Rivers Authority (NRA) 1991).*

Use of resource	Applies to: Riv	Lakes	WQ parameter	Value	Comment
General ecosystem	*	*	Biological index criteria set.		Still to be developed.
Special ecosystem	*	*			Still to be developed.
Salmonid fishery	*	*	DO BOD Un-ionised ammonia Total ammonia pH Temp difference	5 mg/L 5 mg/L 0.02 mg N/L 0.78 mg N/L 6-9 1.5˚C	 Note 1
Cyprinid fishery	*	*	DO BOD Un-ionised ammonia Total ammonia pH Temp difference	3 8 mg/L 0.02 mg N/L 1.5 mg N/L 6-9 3.0˚C	 Note 2 Note 2 Note 1
Migratory fishery	*	*	DO BOD Un-ionised ammonia Total ammonia pH Temp difference	3 mg/L 6 mg/L 0.02 mg N/L 1.5 mg N/L 6-9 1.5˚C	 Note 1

Note 1: *pH range may exceed this range if due to natural causes.*

Note 2: *The ammonia value for cyprinid fisheries may be relaxed if it can be proved that fish populations will not be affected.*

Note 3: *Toxic substances (including metals) are listed separately.*

2.7.4 Guidelines for sediments

Because wetlands are frequently systems in which sediments tend to accumulate, a brief discussion of water quality guidelines for sediments is included here. Sediments can act as a sink for some contaminants entering aquatic ecosystems, where they can accumulate over long periods of time. Importantly, contaminated sediments may also serve as a continued source of contamination to other components of the environment and as a route of entry into the food web. Not only do contaminated sediments directly affect benthic organisms, such as insects, worms, molluscs and bottom-dwelling fish that

spend a large portion of their life cycle in or on sediments, they can also indirectly affect other life forms that feed on such organisms (Environment Canada 2003).

World-wide there seems to be a growing appreciation of the threat posed by contaminated sediments to the biota of aquatic systems. The US-EPA is committed to developing guidelines for sediments, although these are not yet fully developed (EPA 1998b). At present the United Kingdom does not have any sediment quality guidelines (SQGs) for chemicals, although the increasing need for this has been recognised. A recent review was carried out of world-wide methodologies (Rowlatt, Matthiessen *et al.* 2002), with a view to drawing-up a usable approach to setting sediment guidelines. It was found that the Netherlands, USA and Canada were the most advanced in this respect. Two general approaches have been used world-wide – one based on theoretical equilibrium partitioning calculations which extrapolate from existing quality standards for water. The second approach, recommended by the UK Environment Agency, was an empirical approach using data gleaned from a North American database of sediment contaminant concentrations and benthic community and toxicity data. It was advised that derived values should be subject to further validation with UK sediments. A further recommendation was that because of the limited knowledge of sediment-toxicity-biota interactions, failure to comply with a SQG should be used as a trigger for further investigatory work.

A beginning has been made in the ANZECC/ARMCANZ (2000) guidelines towards the use of criteria that specify the permissible concentrations of various chemical constituents in sediments. The values have in general been obtained from other countries and subsequently adjusted to suit Australian conditions. At the present time these values are tentative (and are referred to as "interim sediment water quality guidelines"). Different values have not yet been drawn up for the different aquatic ecosystems – thus the values represent both marine and freshwater systems. For some chemicals no data are available. In such cases it is recommended that the sediment concentration should not exceed twice (or in exceptional cases, 3 times) the background concentration. As an alternative, the water quality guideline default values can be applied to the pore water.

An important point made in the ANZECC/ARMCANZ guidelines is that, in order to ensure the protection of wetlands, it is necessary to consider the cumulative effects of pollutants on downstream ecosystems. This is of particular importance where there is likely to be accumulation of sediments such as in wetlands and estuaries. This need to consider aquatic systems on an integrated basis has also been noted by Lemly (1999). It has been found in some parts of the USA that standards for selenium have been set for rivers with no regard to downstream aquatic ecosystems. Whilst the permissible riverine concentrations for selenium may be non-detrimental to the biota in lotic environments, in some situations these are set to high for downstream wetlands. Selenium accumulates in the sediments of wetlands and has in some places led to demonstrable cases of severe toxicity. The South African Resource Directed Measures (RDM) approach to protection of aquatic resources does recognise the need for an integrated approach. Because, however, the development of methods for wetlands lags considerably behind that of other aquatic ecosystems, and because implementation of effective protection of wetlands is frequently lacking, there is a danger that issues such as accumulation of toxicants may be neglected. This is an important point that these authors would like to emphasise.

CHAPTER 3

WATER QUALITY RANGES OF SOUTH AFRICAN WETLANDS

3.1 INTRODUCTION

It was considered that a logical approach to developing water quality objectives for wetlands was to first gather all available water quality data for this type of ecosystem and then critically examine these data. It was hoped that from an inspection of the ranges of concentrations of chemical constituents and physical variables, useful insight could be gained into this aspect of wetland functioning. Furthermore, it was felt that this insight would aid in setting sensible and meaningful water quality objectives. These objectives need to give adequate protection to wetlands, but at the same time they need to be in line with the underlying rationale of the New National Water Act (1998), namely; that aquatic ecosystems need to be managed for the maximum good of people and that a balance needs to be maintained between resource protection and resource use (Palmer, Berold and Muller 2003).

The following aspects are discussed in this chapter:

- the status of the National Wetland Inventory;
- the need for a database to house data collated during this project and examination of potential, existing databases;
- the Wetlands Water Quality Database developed in the course of this project;
- the type, source and quality of the wetland information and water quality data in the database;
- the results from interrogation of the database, including, median values and ranges of selected water quality variables for different wetland types and climatic regions.

3.2 THE NATIONAL INVENTORY OF SOUTH AFRICAN WETLANDS

A complete inventory of South African wetlands has not yet been completed (J. Dini, DEAT, Working for Wetlands, *pers. com.* February 2004) and the local level of knowledge of these ecosystems varies across the country. For example, a project to classify the wetlands of the Natal Drakensberg Park was carried out by Dely, Kotze, Quinn and Mander (1999). Inventories of other parts of KwaZulu-Natal have also been

compiled (Begg 1988) and further lists of wetlands in various areas of the country have been initiated during the past few decades. Although it is recognized that an inventory of the wetlands of the Western Cape would be a valuable management tool, only now are there movements to compile such a list (K. Snaddon, FRU-UCT, *pers. com.* 2005). A review by the Department of Environmental Affairs and Tourism (DEAT) of available information on South African wetlands carried out in 1998, revealed 69 wetland inventory data-sets (Ramsar 2002), unfortunately most of the data-sets were incompatible and could not simply be combined to produce a national database. It is perhaps pertinent to point out here that as well as listing all the wetlands in an area, drawing up an inventory usually (and most usefully) involves collecting other data at the same time (e.g. an indication of the ecological condition, size and geographical position, lists of plant species). Water quality data may sometimes be one of the attributes that is included, although in the course of this project no particularly extensive collections of this type of information were found for wetlands. A closely allied activity is classifying the wetlands into the type (lacustrine, palustrine etc.) since this is also important information that needs to accompany each wetland. This subject is considered again when discussing the information entered into the Wetlands Water Quality Database produced during this study (section 3.3.1).

3.2.1 Status of the National Wetland Inventory

As a result of the fragmented nature of our knowledge of wetlands in this country, the pilot phase of a national project was undertaken for DEAT to develop the tools and the methods required for establishing a cost-effective, accurate and comprehensive national wetland inventory (Wetlands Inventory Consortium 2002). Determination of the distribution and current status of South African wetlands is vital baseline information needed for monitoring and conservation of these ecosystems. A national inventory is also required of countries (such as South Africa) that are signatories to the Ramsar convention (Ramsar 2002). As part of the pilot inventory project, sites in four geographically distinct areas of the country were investigated. The wetland types included were; rivers (and associated floodplains), pans, permanent, seasonal and temporary marshes in different topographical areas, and artificial wetlands.

Under the National Land Cover 2000 project, sponsored by DEAT, and carried out by CSIR, satellite imagery combined with topographical modelling is being used to identify

wetland areas. Coverage at a scale of 1:50 000 will be produced. The eastern and northern parts of the country have already been completed, and the more southern and westerly regions will be surveyed and the data analysed in the next year (J. Dini, Working for Wetlands, *pers. com.*, Feb 2004). Although the data-set generated through this project will miss some small or transformed wetlands (T. Landmann, CSIR, South African Wetland Action Group meeting, Barbers Pan, *pers. com.* October 2003) this information will be invaluable in compiling a preliminary national wetland inventory. It will also be used in the prioritization of catchments and wetlands for remedial measures through the public works programme "Working for Wetlands".

3.2.2 Consideration of databases to house wetland data

Data collected during the compilation of a national inventory, in addition to on-going monitoring data needs to be stored in a suitable database. Furthermore, water quality data and site information on specific wetlands collated during the present study also needed to be stored in a database. It was considered to be highly preferable that any data generated in the present project be in a form that can be integrated eventually into the national database. Thus, the database used in this project should preferably be the same as used for the national inventory, or if not, at least compatible so that data transfer can be easily undertaken. At the time of initiating this part of the project, a decision had not been taken as to the final database that would be used for the National Wetland Inventory (J. Dini, DEAT, Biodiversity Planning, *pers. com.*, March 2003) although a candidate system (MedWet) had been identified. Thus, as part of the current project, the latest version of MedWet was examined (see below). In addition, a locally-developed database (BIOBASE) was also assessed for use in the project. A professional database designer was employed to assess the two potential databases with regard to the requirements of the present project. Any database used for the current project needed to satisfy the following criteria:

- be able to house text information and time-series of data (e.g. dates and corresponding concentrations of chemical constituents or values of physical variables);
- be relatively user-friendly, and inexpensive;
- be amenable to interrogation so that ranges and median/mean values of water quality variables could easily be calculated for various subsets of data.

Useful optional criteria were that the database:

- be a useful tool for future management and conservation initiatives concerning wetlands in this country;
- be compatible with the database used for the National Wetland Inventory;
- be able to house spatial data such as GIS overlays, maps;
- be able to house images such as photographs.

3.2.2.1 *Examination of MedWet*

MedWet is a professionally-designed database that was developed in order to aid in the classification and description of wetlands in the Mediterranean regions of Europe and North Africa (Costa, Farinha, Hecker *et al.* 1996), and has evolved through several versions. Its fundamental purpose is to provide a source of information from which management and conservation decisions can be made concerning the wetlands included within the database. Part of the Terms of Reference for the project team that undertook the pilot National Wetlands Inventory in South Africa was to examine one version of the database, "MedWet 2000". The project team considered that MedWet 2000 was not suitable as the national database. The principle reasons given for the decision were as follows (Wetlands Inventory Consortium 2002):

- significant modifications would need to be made in order for time-series data and spatial data (GIS) to be stored,
- MedWet is based on Microsoft Access which is not considered to be suitable for very large collections of data, such as would be accumulated when compiling a national database.

The recommendation was that a new database be developed specifically designed for the South African national wetland inventory, but that this should be done only after the newest version of MedWet (3.0) had been examined (which was unavailable at that time). In the current project, MedWet (Version 3.0) was assessed with regard to its suitability as a vehicle for a Wetlands Water Quality Database. The criteria listed above (section 3.2.2) were tested and the following points for and against MedWet (Version 3.0) were found:

- it has a user-friendly interface that is ideal for entering new data or for extracting data to inform decisions;
- the database initially gave trouble installing, and the underlying data itself could not be accessed, just the "front-end" or user-interface;

- the database is heavily orientated towards classification systems which summarize information rather than provide raw data. While such a setup is preferable for users making management decisions, it does not help users who want to analyze the data to find trends and groupings that are not already provided;
- there are very few interfaces for entering wetland water quality data, which was the main reason for using this database platform;
- there is no apparent way of linking multiple site visits to any given wetland site. This may simply be an artifact of the way the access data form had been designed, though this could not be confirmed as the database's root tables could not be accessed.

Many of the problems that were encountered may well have been a result of faulty installation of the database program and errors in the version supplied to the team. It is possible that simple reprogramming of MedWet's interface system to incorporate the kind of data necessary to meet our needs would have solved the difficulty. Nevertheless, because of the above problems, as well as, potential issues regarding licensing agreements, it was decided not to follow this route.

3.2.2.2 *Examination of BIOBASE*

BIOBASE is a database created as part of the Water Research Commission-funded project K5/626 "Water quality requirements for riverine biotas" and is described in Dallas and Janssens (1998). It is a database compiled from literature reports where macroinvertebrate and water quality data have been collected (more or less) simultaneously. Because the BIOBASE consists of a similar format to that required during this project it was deemed to be important to see if, by means of simple modification, it could be tailored to the present project. An advantage of this option is that BIOBASE is a tried and tested system that has powerful querying functionality. In addition, since it was developed at the Freshwater Research Unit local expertise was available. However, BIOBASE is in Access 97 format and it is unclear what the impacts and costs of converting to a more recent version would be. Similarly, the effect that changes in the database necessary for the wetlands project requirements might have on the data query facility was unknown (J. Schmidt, *pers. com.* Feb 2003). Because the cost of constructing a simple database from scratch was cheaper than altering

BIOBASE, and because the ultimate database that will be used for the national inventory had not yet been finalized (and is not likely to be BIOBASE unless the latter is substantially modified to hold spatial data), it was decided to construct a custom-made database for this project.

3.3 DEVELOPMENT OF THE WETLANDS WATER QUALITY DATABASE

3.3.1 General considerations

The database developed to house the data collated during the present project is based on Microsoft Access 2000. Figure 3.1 shows the different information fields that were included in the database and the relationships between them. With regard to compliance with the criteria listed in section 3.2.2, the database can house both text information and time-series of data. The database is very user-friendly with regard to addition of data and interrogation and the results are easily down-loaded to a spreadsheet for further statistical manipulation. Furthermore, the database should be a useful tool for future wetland management and conservation initiatives, and may possibly be used to contain data for an inventory of Western Cape wetlands (H. Dallas, FRU-UCT, *pers. com.* Feb. 2004). In its present form, however, spatial data and images cannot be accommodated. This is not considered to be a serious omission for the requirements of the present project, and inclusion of this facility would have greatly increased the complexity of the database. With regard to the compatibility of the Wetlands Water Quality Database with the future database used to house the National Wetlands Inventory, it is difficult to comment except to say that Microsoft Access is one of the most powerful and commonly-used database systems.

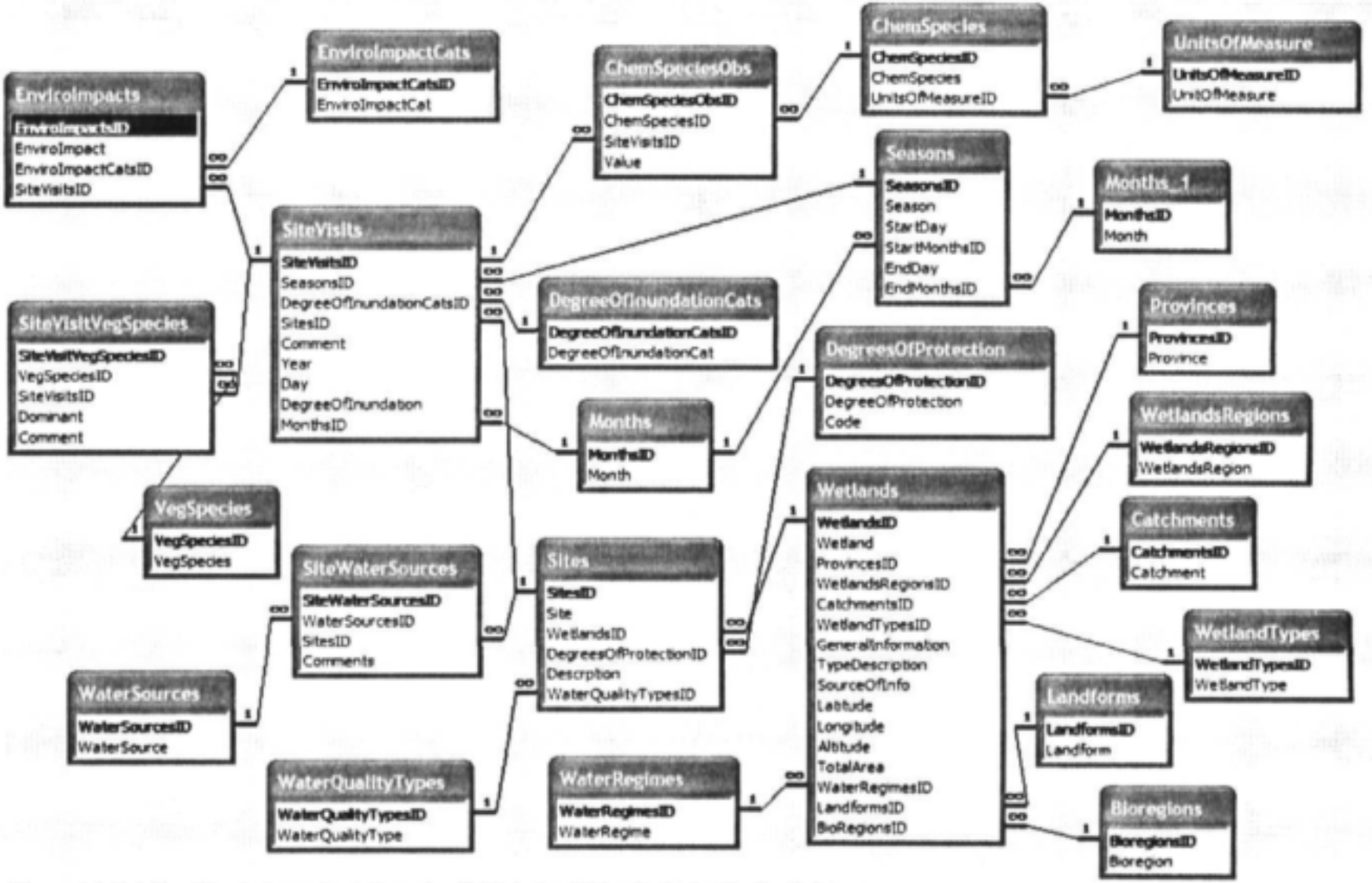


Figure 3.1 *Relationships between the information fields in the Wetlands Water Quality Database.*

Access also forms the basis for the RIVERS database used to house the biomonitoring data gathered as part of the National River Health programme (H. Dallas, FRU-UCT, *pers. com.* Feb. 2003).

3.3.2 Sources of water quality data

No water quality sampling was carried out during the course of this project. Instead, a concerted effort was made to gather together historical water quality data that had previously been collected for wetlands in this country. These data were gathered from various sources including published papers, theses, consultancy reports, and directly from institutions such as DWAF, or municipalities. Due to time constraints and the financial limitations of the project, and also because of the difficulty of getting hold of the data, this collection is by no means comprehensive. The best represented area of the country in the database is the Western Cape, although data for most other regions are included. One of the recommendations emanating from this project is that this collating of data be continued (Chapter 6).

One of the major aims of compiling the database was so that the data could be interrogated, and the ranges in values of water quality variables amongst the different wetland types be examined. Furthermore, regional patterns in water quality were investigated, so as to answer questions such as:

"What is the mean pH of wetlands in the fynbos/winter rainfall region?" Or,

"What is the range of conductivity values exhibited by impacted wetlands?"

The ultimate aim of compiling the database was to help compile preliminary, "best-guess" default values of water quality variables (section 4.6).

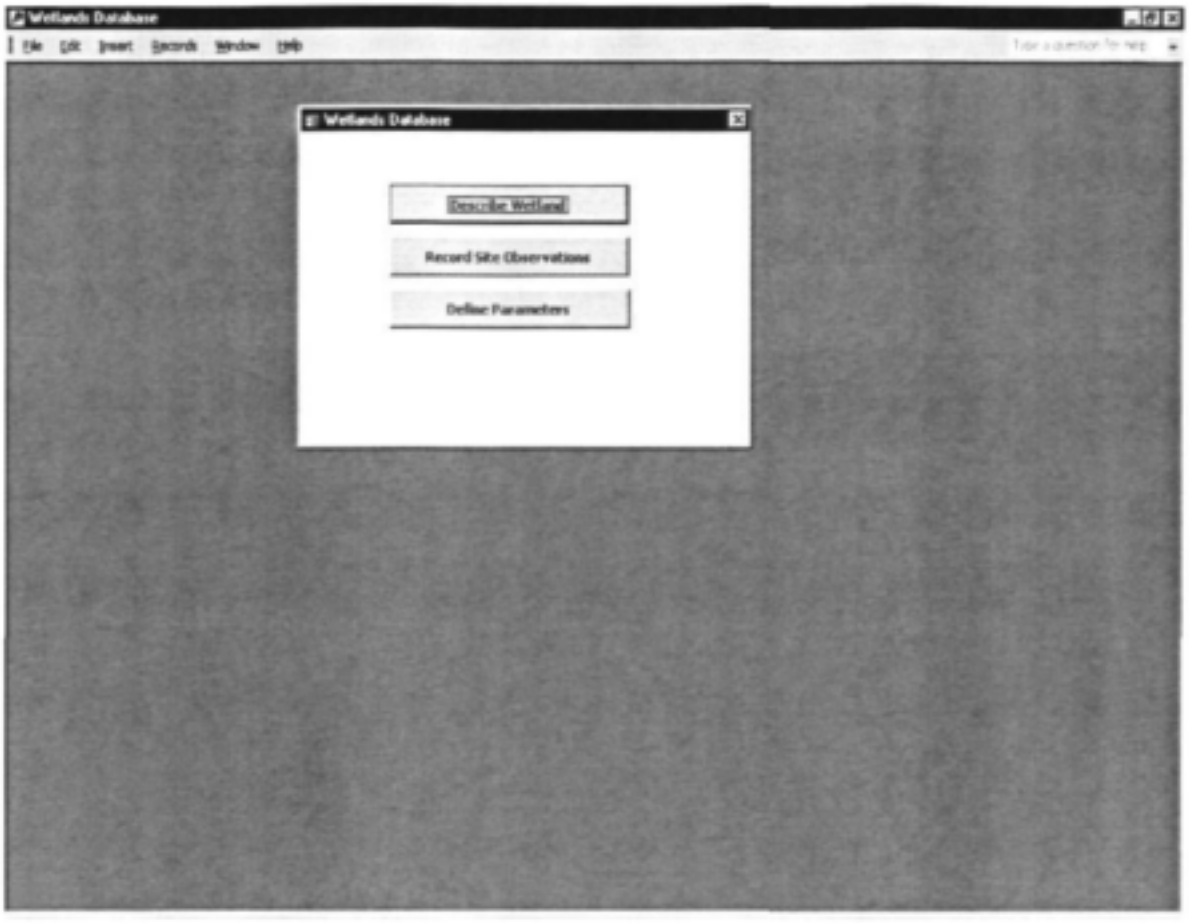
3.3.3 Description of the database

The major features of the database are described in this section.

The opening screen

The opening screen of the database is shown in Figure 3.2. Three options are available. It is the first, "Describe Wetland" and second, "Record Site Observations" options that are most commonly used. The third option "Define Parameters" is only chosen should the drop-down menus in the database need to be altered (see later).

Figure 3.2 *The opening screen of the database.*

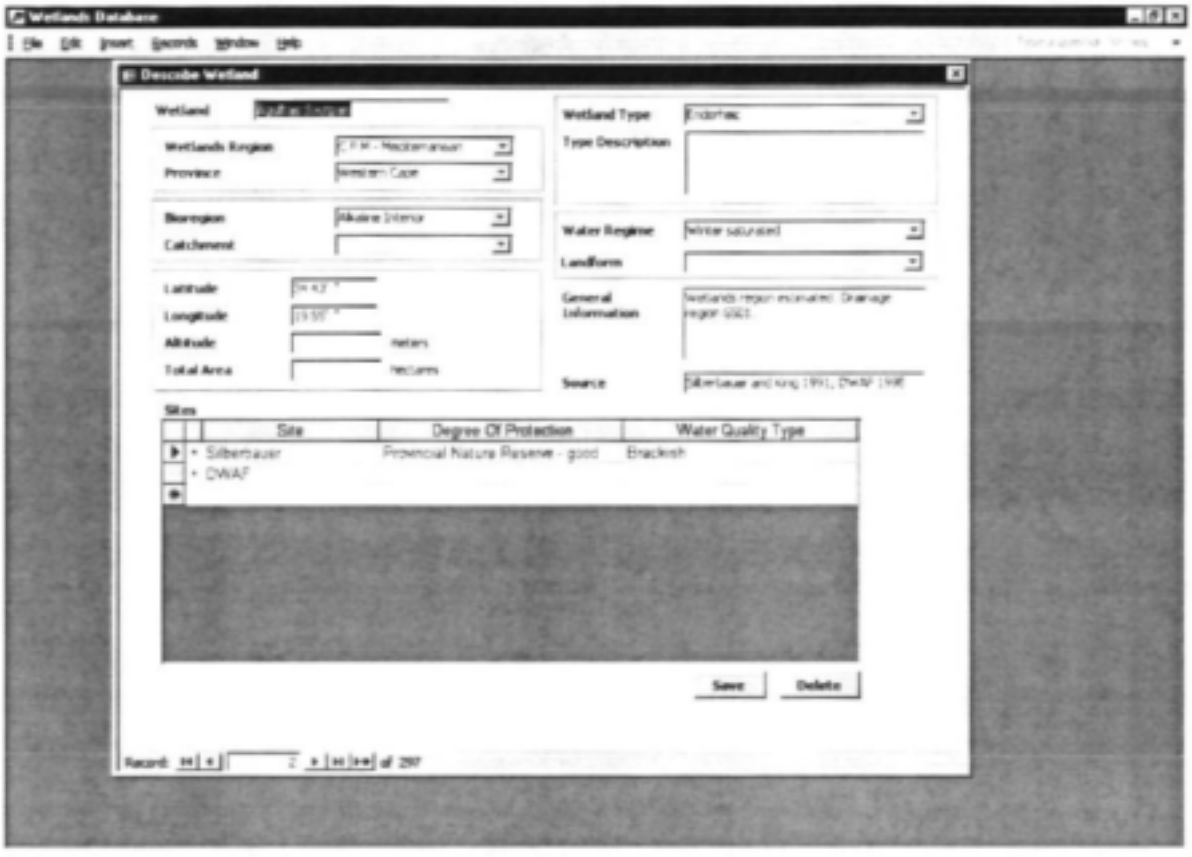


Describe Wetland

If the option "Describe Wetland" is chosen, a record form such as that shown in Figure 3.3 for the Agulhas Soutpan is brought up. By clicking on the scrolling arrow, next to the record number (bottom left hand corner) the descriptions of all the different wetlands recorded in the database can be seen. The last record form is blank and can be used to enter information for a new wetland.

As explained previously, various attributes or types of information are associated with wetlands. Figure 3.3 shows some of the information (record fields) that can be captured in the database. The choice of attributes was informed by the needs of the project and what was available in the literature. For example, from work on developing a Decision Support System to determine the water quality Reserve (section 4.4) it was recognized that information on the wetland type, the hydrological regime (water regime) and the

Figure 3.3 *The wetland description form for the Agulhas Soutpan.*



source(s) of water supplying a given wetland was useful. Each field in the wetland description record form is discussed in Table 4.1. Only the wetland name and province is, however, mandatory, the rest is useful optional information that may or may not be available for each wetland. Very often the source of water supplying a wetland was not provided in the documents from which the information was extracted. Many of the points where information can be added have drop-down lists, from which a selection can be made. As an example, for the information field "Wetland Type" a choice can be made from the types listed in Table 4.2. Any notes or comments with regard to the type of wetland, such as information about the type of drainage (endorheic or exorheic) can be recorded in the text box, "Type Description". In the lower half of this screen is a list of the sites for the wetland for which sampling data are available.

Table 3.1 *A description of each field in the "Describe Wetland Record Form."*

Describe Wetland Form	
Record field	**Comment**
Name of wetland	Sometimes this may not be unique and wetlands from different regions may have the same name. In such cases the wetlands are labeled, A, B etc. On the other hand, sometimes different authors give different names to the same wetland. Examination of the geographical position can help to sort out such confusions. If there was doubt, the data were added as two separate wetlands.
Wetland Region	The classification of wetland regions designed by Cowan (1995) was used (Figure 2.1). Note though that because the wetland regions have only been delimited at a course scale, in some cases it was not possible to determine with any accuracy in which region a wetland was located.
Province	
Bioregion	A system of dividing the country into regions (based on similar physical and biotic characteristics) after Eekhout *et al.* (1996). For example, Fynbos, Limpopo, bushveld bioregions. Used in the BIOBASE.
Catchment	In the case of riverine wetlands it was considered useful to include the catchment.
Geographic position	Latitude/Longitude. Frequently in early work the position of a wetland is given in terms of degrees and minutes (but not seconds).
Altitude	
Total area	This parameter was usually taken as the approximate area of open water (which might vary with season). Some authors reported on area covered by wetland vegetation. If necessary a qualifying statement was entered under "General information".
Wetland type	The basic DWAF system of wetland classification was used (DWAF1999d). The types are listed in Table 3.2. The types of palustrine wetland was expanded and springs added. Often a wetland was suspected to be endorheic (e.g. was called a "pan") - such wetlands were listed as "Probably endorheic".
Water Regime	A distinction was made between wetlands that became boggy (i.e. soil was saturated), and those that maintained standing water (inundated). The time of year when this happened (summer or winter), if permanently inundated or if this was variable (ephemeral) was also noted.
Landform	For example slope, basin, flat, depression (Jones 2002). This information was rarely available.
Data source	The authors of the document from which the data were accessed, or the institution that supplied the data (Appendix B).
Sites	Sampling sites at which water quality data have been collected. Where one author has collected data at more than one site, these were listed as e.g. Silberbauer 1, Silberbauer 2 etc.
Degree of protection	An attempt to assess the threat to the wetland by giving the landuse in the surrounding area e.g. National Park, Provincial Nature Reserve, farmland, municipal land.
Water quality type	An attempt to capture the broad statements of water quality type sometimes given in reports e.g. brackish, polluted, brown coloured. Also the salinity class, oligohaline, urihaline, mesohaline (Jones 2002).
Water source	The major source of water (by volume) supplying a wetland. Options are for example - groundwater, from a river, surface run-off, from the sea, or a combination of sources. (This data field is available for each site, but is not visible in Figure 4.3).

The sites were named after the author/authority who gathered the information. In the record form shown for the Agulhas Soutpan (Figure 3.3) water quality data have been collected by DWAF (1998), and by Silberbauer and King (1991a).

Table 3.2 *A list of the wetland types of that can be entered into the database (the "drop-down menu").*

Frm_DefineWetlandTypes

Wetland Type
Lacustrine
Palustrine
Endorheic
Riverine
Estuarine
Floodplain
Palustrine-floodplain
Palustrine-mountain seep
Palustrine-floodplain pan
Probably endorheic
Spring (eye)

Record site observations

Clicking on the second option from the opening screen (Figure 3.2), brings up a record form such as that shown in Figure 3.4, in which the water quality data for a particular site visit are recorded. By clicking on the bottom left hand scrolling arrows, all the different site visit records for each wetland can be perused. Water quality data have been collected by DWAF (1998), and by Silberbauer and King (1991a) for the Agulhas Soutpan (Figure 3.3). Figure 3.4 shows the data recorded in DWAF (1998), whilst Figures 3.5 and 3.6 show the data collected by Silberbauer and King (1991a) on two different sampling visits.

Date of visit

The format in which the time or date of sampling was recorded was rather variable in the literature. For example, some samples were once-off "spot samples", for which the year, month and day were given. Quite often for other samples, only the month or, perhaps the season or even less information were recorded. For input into the database the seasons of the year were defined as follows:

Spring: 1st September – 30th November

Summer: 1st December – 28th February

Autumn: 1st March – 31st May

Winter: 1st June – 31st August

Figure 3.4 *Site visit record for Agulhas Soutpan – sampling data from DWAF (1998).*

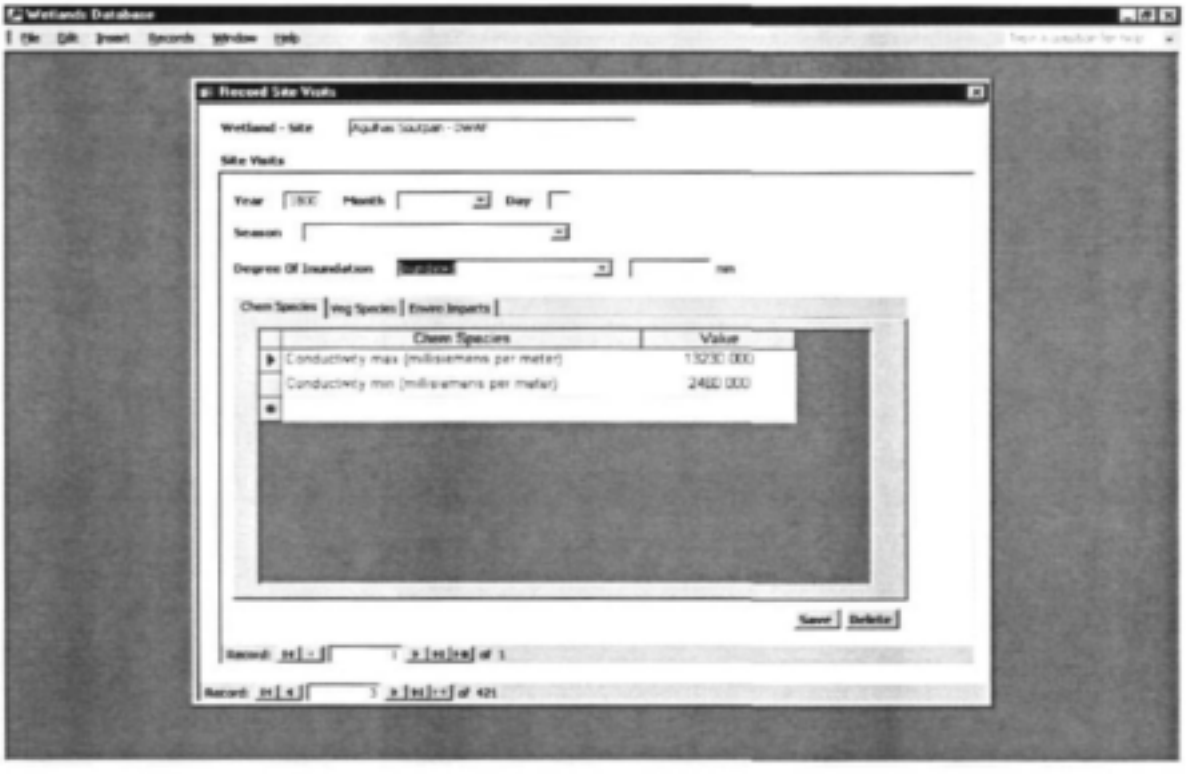
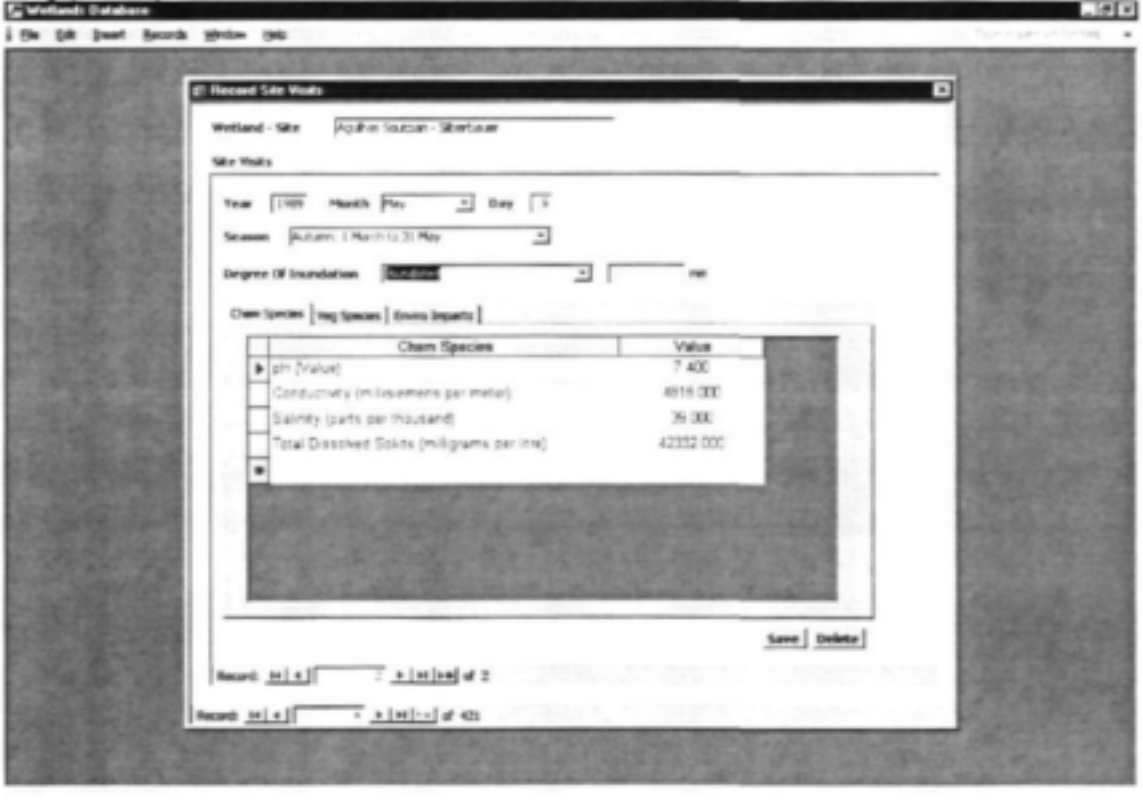


Figure 3.5 *Site visit record for Agulhas Soutpan – sampling data from from Silberbauer and King (1991a) visit 1.*



Figure 3.6 *Site visit record for Agulhas Soutpan – sampling data from Silberbauer and King (1991a) visit 2.*



In some cases, a code value for the year (either "1800" or "1900") is given in the database (for example in Figure 3.4) This indicates that in the original document from which the data were extracted, no date was given, only general values for the water quality constituents (in this case, conductivity). Where the year is represented by "1900" this indicates that the values for the water quality variables are summary statistics (usually median values calculated from a data-set). This topic is discussed further in section 3.3.2.

Degree of inundation

The water quality measured in a wetland is highly dependent on the hydrological condition at that time (see for example sections 3.3.2 or 3.6.2). Thus it was deemed important to capture any information on the hydrological status of a given wetland at the time of sampling. Provision is made in this form for recording the depth in millimeters of the water at the time of sampling – where this depth refers to the approximate average water depth of the wetland (not at the sampling point). In addition, if the degree of inundation was noted only qualitatively in the literature source, this was also captured (as inundated, flooded to capacity, dry, or saturated – i.e. the soil was wet, but there was no standing water). Unfortunately it was found that very few of the literature sources recorded linked (i.e. measured at the same place and time) hydrological and water quality data.

Chem species, Veg species, Enviro impacts

The details regarding how and what chemical species were recorded in the database are discussed in section 3.3.2. Originally it was hoped that as part of the project, preliminary investigations could be made into relationships between water chemistry and the species of plants found in a wetland. Therefore a facility was made where vegetation data (lists of plant species/vegetation types) could be entered. It was found, however, that this was too time consuming, and was not pursued further. Nevertheless, it might be useful to continue this line of research at a later stage. Where possible, a subjective estimate of the degree of ecological condition of each wetland (ranging from – none to extensive, or unknown) was made and entered in the field "Enviro Impacts".

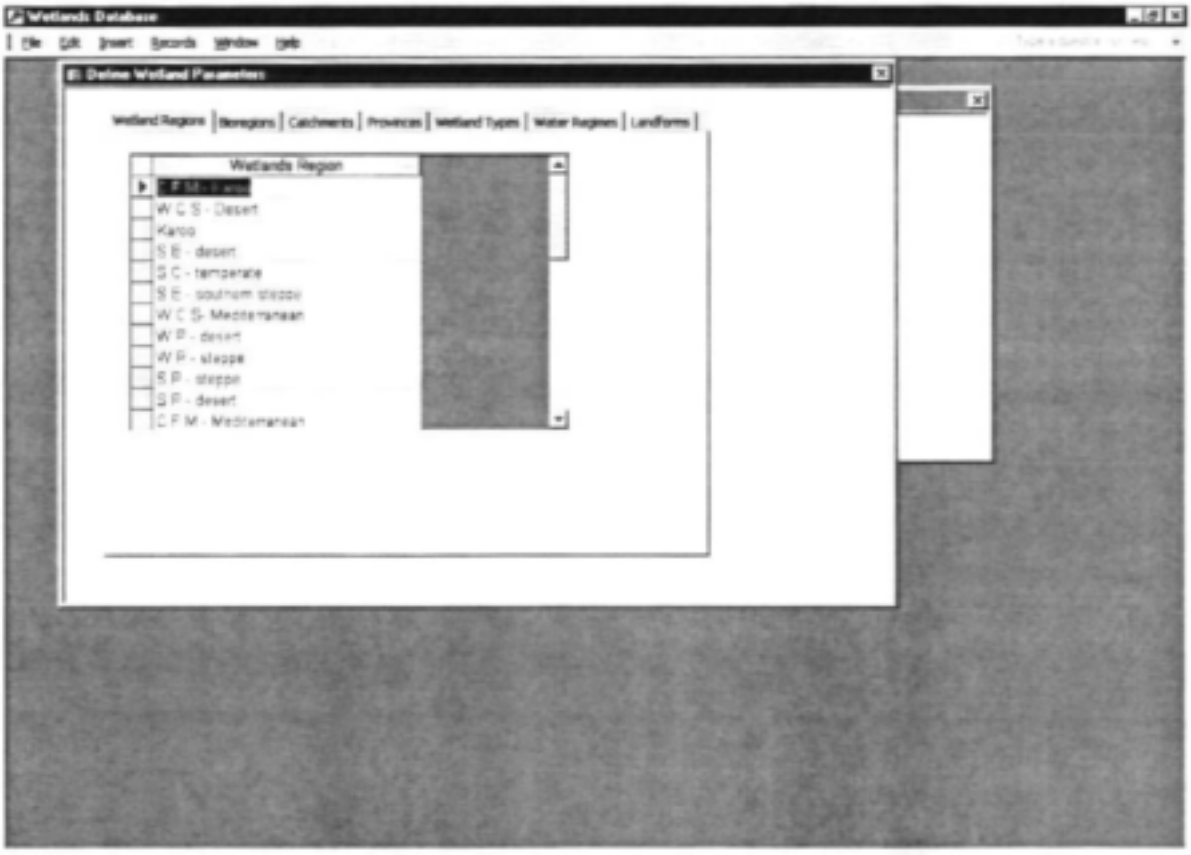
Define Parameters

As stated previously, the third option at the opening screen of the database (Figure 3.2) is used only when changes need to be made to the drop-down menu options. For example if another chemical variable or wetland type is added (Table 3.2). Figure 3.7 shows the form containing the list of parameters under "Wetland Regions".

3.3.4 Preparation of data before adding to the database

Several factors had to be taken into consideration before water quality could be entered into the database. Firstly, a wide range of water quality variables and units are recorded in the literature. The full list of water quality variables recorded in the database, as well as the units in which they are given is in Appendix A. The variables recorded for each site depended on what was measured in the investigation under consideration.

Figure 3.7 *The "Define Wetland Parameters" form showing the wetland regions.*



Appendix B lists the sources used to populate the database and gives comments, where necessary, on any manipulation carried out on the data. For instance, in some cases, values have been read off graphs in the original literature source. This process carries with it an inherent risk of error, and yet because water quality data for wetlands in general is so scarce it was felt that the information was valuable. Therefore in Appendix B, a warning is given that the data may be inaccurate.

Secondly, some of the measurements in the literature were recorded as maximum, or minimum values, and other data as spot-values, means or medians. For example DWAF (1998) reported only the maximum and minimum values of conductivity measured in the Agulhas Soutpan, and this is indicated in the Record Site Visits form by the use of "Conductivity min" and "Conductivity max" as the chemical species listed (Figure 3.4). Where authors have recorded median or mean values, this is noted in Appendix B. In addition, two sources of data namely, DWAF monitoring data and that collected by the Cape Metropolitan Council (CMC) were first condensed and then entered into the database as summary statistics for a given time-period. This is because in comparison to the rest of the data available for wetlands which was typically one or a few visits, the above two data sources represented hundreds of measurements taken over many years. Furthermore, the type of wetlands represented by the above two data sources was not particularly useful for the study being either, urban and generally impacted (CMC data) or mostly relating to lakes and springs (DWAF). Details of how the data-sets were summarized (i.e. the time range, number of measurements) are given in Appendix B.

In one or two literature sources (e.g. Snaddon and Ractliffe 1999) the concentrations of chemical constituents in the wetland sediment were recorded. The Wetland Water Quality Database currently has no orthodox way of housing this data, but this should be considered in any future work (Chapter 6).

3.4 INTERROGATION OF THE WETLANDS WATER QUALITY DATABASE

The wetlands database was used to find patterns and trends in the physico-chemical characteristics of the various kinds of wetlands. The purpose of this exercise was to be able to make informed decisions regarding the setting of water quality standards. Collating wetland data into the database is an ongoing process, and although querying the database was left as late in the project as possible, inevitably some only will be

added subsequently. At the time of interrogation the number of wetlands in the database was 297. The majority of wetlands in the database have associated water quality data, a few were dry at the time of sampling and thus no values are recorded. For quite a few wetlands (see Table 3.3) it was unclear what the wetland type was and these were marked as "Unspecified". In some cases no geographical coordinates were provided. Figure 3.8 shows the geographical distribution of the wetlands represented in the database.

Table 3.3 *The number of each wetland type in the database at the time of interrogation.*

Wetland type	Number of wetlands in database in Feb 2004
Springs	40
Riverine	1
Lacustrine	35
Palustrine	45
Endorheic	53
Probably endorheic	57
Unspecified	62

3.4.1 Preparation of the data before interrogation

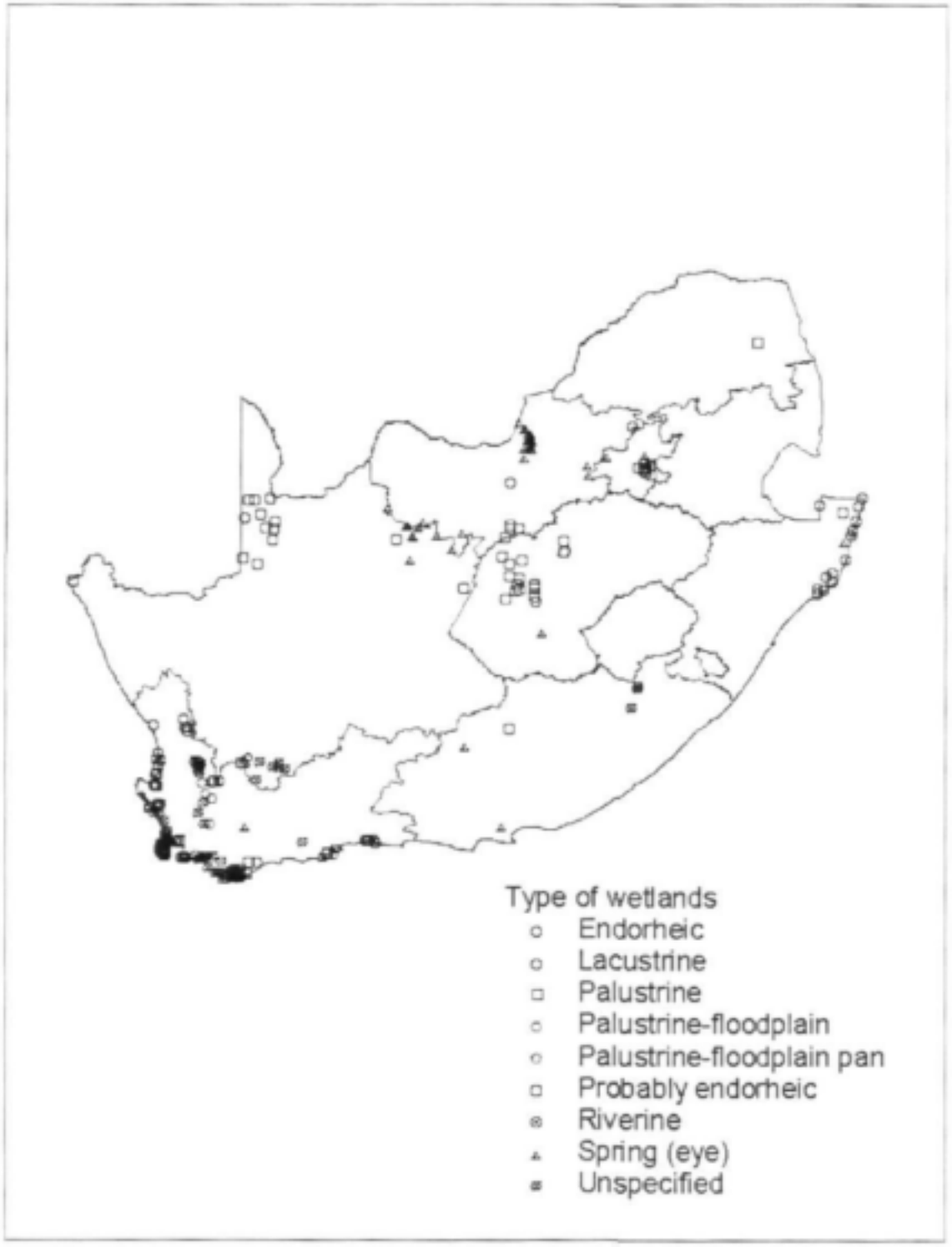
The database was used as a tool to help guide the derivation of water quality guidelines, thus it was important that whilst interrogating the data, impacted sites be specifically excluded. It was, however, difficult to assess exactly which sites were impacted. Frequently in the literature source no indication was given as to the pollution status of a given wetland. Furthermore, because little is known about wetland water quality, it was difficult to identify polluted sites from an examination of the values of water quality variables. In work carried out by Dallas, Day, Musibono and Day (1998) to derive regional water quality guidelines for rivers, the rapid biomonitoring technique, SASS (South African Scoring System) was used to separate impacted from least-impacted sites. Unfortunately an effective biomonitoring method for assessing the water quality status of South African wetlands has not yet been developed (DWAF 2004). Therefore, the procedure followed in this project was to include all wetlands in the database for

which water quality data were available. During the interrogation process, however, known polluted sites were excluded. Impacted sites were identified by examination of 1: 250 000 maps. If the wetlands were in urban/sub-urban areas they were deleted/separated from the spreadsheet used for calculating the statistical parameters for non-impacted systems. This approach is only likely to identify the most grossly polluted sites, however. Twenty wetlands were considered to be impacted and included: Germiston lake, Liesbeeck/Black River wetland, Princessvlei and Little Princessvlei, Langvlei (Cape Town), ZeeKoevlei, Zoarvlei.

The statistical parameters of the impacted wetlands, where relevant, were calculated and reported separately (as for example in Table 3.4). The water quality variables that were investigated were; conductivity (and TDS), pH, phosphorus (SRP/ortho-phosphate) and nitrogen (nitrate and ammonium). A much wider range of water quality variables are recorded in the database, however. One of the major variables of interest was the amount of dissolved material in the water column. This is indicated by "salinity", "conductivity" and "Total dissolved solids" in the database. Total Dissolved Solids (TDS) values were transformed to an approximate conductivity value by division by a factor of 6.5 (DWAF 1996). Many authors reported salinity values. Although in fresh and brackish waters TDS and salinity are very similar, at high salinities and in systems (such as wetlands) where there is likely to be a high dissolved organic carbon content in the water column, they may be a considerable discrepancy between TDS and salinity (J. Day, FRU UCT *pers. obs.* 2003). For that reason median salinity values are reported separately (Table 3.4), or were omitted from the analyses if there were few data.

Minimum and maximum values were not included in the interrogation process. Two sources of data namely, DWAF monitoring data and those collected by the Cape Metropolitan Council were entered into the database as summary statistics. These were separated from the main body of records during the interrogation process, and were analysed separately where appropriate (e.g. Springs – Table 3.4). This approach was taken because it was felt that the very large number of samples for a relatively few wetlands would dominate the results.

Figure 3.8 *The geographical distribution of the wetlands represented in the database.*



3.4.2 Results from interrogation of the database

Results from the different queries run on the database are recorded in Table 3.4 and onwards. The calculated median, minimum and maximum values are shown. The number of records (sample size) as well as the total number of wetlands represented by the records are noted. In addition, because it was felt that a better description of the spread of the data was needed than would be given by the range, the standard deviation is also given. It should be noted that the reported number of wetlands for each query (e.g. column 6, Table 3.4) is an approximate value. Because of the problem of different authors calling the same wetland by different names, and because exact geographical co-ordinates were not always available, it was not always clear if some records were measured for different wetlands or for the same wetland. In such cases to be conservative, they were treated as different wetlands and entered into the database accordingly.

Before comparing the median values obtained from analysis of the data, it was first necessary to test if the assumptions for parametric statistical tests were valid. Therefore the water quality data were first tested visually for normality by constructing normal probability plots. Homogeneity of variance was examined using the Levene and Brown-Forsythe tests. The data were not normally distributed and the variances were found to be heterogeneous, hence it was decided to check for significance of difference between samples using the non-parametric Mann-Whitney U test. The null hypothesis (that the difference between two population medians is zero) was rejected if $p < 0.05$.

The interrogation process was divided into three phases:

- Phase 1; the effect of wetland type on water quality;
- Phase 2; the effect of geographical region on water quality;
- Phase 3; a multivariate approach.

3.4.3 Phase 1: The effect of wetland type on water quality

All the wetlands in the database (irrespective of region or the season of sampling) were divided into wetland type (Table 3.3) and the median, maximum, minimum values of each water quality variable were calculated.

3.4.3.1 Wetland type and conductivity levels

The results from an examination of conductivity and salinity are summarized in Table 3.4a, and the tests for significance in Table 3.4b.

1. The lowest median EC values (51 mS/m) were shown by the **springs** ("eyes"). This wetland type also exhibited the smallest range (12 – 121mS/m) and the smallest standard deviation from the mean. From Table 4.3b, it can be seen that using the non-parametric Mann-Whitney test springs were significantly different from all the other wetland types. This is perhaps to be expected because the data are all from DWAF monitoring sites.

Such sources would often be used for potable water and thus would be expected to have relatively low levels of dissolved salts. Note that all these records represented summarized data (Appendix B) and that no salinity measurements were recorded at these sites.

Table 3.4a *The variation in* ***conductivity*** *(mS/m) and* ***salinity*** *(parts per thousand) values exhibited by the different wetland types in the Wetland Water Quality database. Salinity values given in bold.*

Wetland Type	Median	Minimum	Maximum	Sample Size (N)	No. Of Wetlands	Comments
Springs	51	12	121	40	40	Mean = 53 mS/m; SD = 22. All records = summary statistics.
Riverine	1	1	1	1	1	
Lacustrine	125 **5**	12 **0**	2 400 **60**	63 **47**	15 **18**	Mean = 522 mS/m; SD = 645.
Palustrine	460 **4**	2 **0**	8 850 **80**	83 **26**	26 **15**	Mean = 853 mS/m; SD = 1431
Endorheic	559 **7**	8 **0**	9310 **101**	65 **37**	43 **29**	Mean = 1362 mS/m; SD = 2 026
Endorheic + "Probably endorheic"	868 **9**	8 **0**	46 523 **302**	101 **77**	75 **69**	Mean = 3 082 mS/m; SD = 6 659
Impacted lacustrine	103	34	1 370	59	7	Mean = 206 mS/m; SD = 287
Impacted Endorheic + "Probably endorheic"	180	16	5 354	29	9	Mean = 441 mS/m; SD = 972

Table 3.4b *Significant differences in* ***conductivity*** *(p > 0.05) between different wetland types as indicated by the Mann-Whitney test. Conductivity values given in mS/m. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).*

	Endorheic	Endorheic +"Probably endorheic"	Palustrine	Lacustrine	Impacted lacustrine	Impacted Endorheic +"Probably endorheic"
Springs	**Significant** (0.0000)	**Significant** (0.0000)	**Significant** (0.0000)	**Significant** (0.0006)	**Significant** (0.0000)	**Significant** (0.0000)
Endorheic		**Significant** (0.0247)	N. S.	**Significant** (0.0447)	**Significant** (0.0006)	N. S.
Endorheic +"Probably endorheic"			**Significant** (0.0003)	**Significant** (0.0000)	**Significant** (0.0000)	**Significant** (0.0001)
Palustrine				**Significant** (0.0093)	**Significant** (0.0000)	**Significant** (0.0050)
Lacustrine					N. S.	N. S.
Impacted lacustrine						**Significant** (0.0100)

2. It was not always easy to tell from many of the literature sources what type of wetland was represented. **Riverine** wetlands were especially difficult to identify and thus only one wetland was listed as this type in the database at the time of interrogation. It is highly likely that riverine wetlands (i.e. those that are "wetlands contained within a channel, that periodically or continuously contain flowing water" – section 2.2.3) have been classed in the database under other wetland types. The data for the Nylsvlei system (Chapter 5) were not available when interrogation of the database was undertaken. No further examination of riverine wetlands was carried out.

3. Lacustrine systems exhibited a median EC of 125 mS/m, but also a wide range of EC values ranging from 12 to 2 400 mS/m. Similarly, the median salinity was relatively low (5 ppt) but wide range of values (from 0 to 60 ppt) have been recorded. Only 15 non-impacted lacustrine systems were available for inclusion in the calculations, a reflection perhaps of the scarcity of this type of aquatic ecosystem in this country. Lakes were significantly different from all other wetland types, except for the two impacted

ecosystem types. Lakes were thus higher in conductivity than springs, but lower in conductivity than endorheic (and "Probably endorheic") and palustrine systems.

4. All the different **palustrine** wetland sub-types (floodplain, mountain seeps, floodplain-pan) were combined. Palustrine systems exhibited a median conductivity of 460 mS/m, a value that is higher than lacustrine systems. This may be a consequence of the increased dilution capacity in the case of the latter. An even wider range of values was also exhibited by palustrine systems (2 – 8 850 mS/m) than for lakes.

5. In the case of the **endorheic** wetlands, two approaches were used. Wetlands that were confirmed (from local knowledge or from the literature source) to be inwardly-draining were classed at "endorheic" in the database. Wetlands that were suspected to be endorheic (e.g. were named "pan") were classed as "**probably endorheic**" in the database. In the first approach, only endorheic wetlands were analysed. In the second approach, "endorheic" and "probably endorheic" were lumped together. The endorheic wetlands showed the highest median conductivity value of all the wetland types and this was also reflected in the highest median salinity, and was linked with a wide range of values. Adding the "probably endorheic" wetland data resulted in an even higher median value for EC (and salinity) and a wider range. Conductivity ranged from 8 to over 46 000 mS/m. Although palustrine wetlands were significantly different to the "Endorheic and Probably endorheic" group in conductivity, if the wetlands classed as "Probably endorheic" were omitted, there was no significant difference. The most saline of all the wetlands for which data are available is Oranjemund Pan in the Northern Cape, for which an EC of 46 523 mS/m was recorded (Seaman *et al.* 1991).

6. Impacted wetlands fell into one of two wetland types, namely lacustrine or endorheic/probably endorheic. The median EC of impacted lakes (103mS/m) was lower than that of the natural systems, although this difference was not statistically significant. Similarly the values of impacted "endorheic and probably endorheic" pans was also lower than their non-impacted counterpart (p=0.0001). Although the number of wetlands for which data are available is not very high (7 and 11 respectively), this does suggest that for some wetlands, anthropogenic impacts have taken the form of dilution, rather than elevation of some water quality constituents. This may well be due to the discharge of effluents or storm-water into previously seasonal wetlands.

3.4.3.2 The effect of wetland type on pH

The results from an investigation into the effect of wetland type on pH are summarized in Table 3.5a and 3.5b.

1. In general, the range in median values for the different wetland types in the largely non-impacted state was not very great and pH tended to be circumneutral. If riverine wetlands are ignored (because of the small sample size) the median values of un-impacted wetlands ranged only from 7.55 – 8.0 and there were no significant differences between the different wetland types.

Table 3.5a *The variation in **pH** values exhibited by the different wetland types in the Wetland Water Quality Database.*

Wetland type	Median	Minimum	Maximum	Sample size (n)	No. of wetlands	Comments
Springs	8.00	4.72	8.30	40	40	Mean = 7.84; SD = 0.60
Lacustrine	7.77	3.82	9.73	63	20	Mean = 7.43; SD = 1.58
Palustrine	7.55	3.7	11.21	52	25	Mean = 7.39; SD = 1.77. Most acidic wetland = Suurdam. The most alkaline = Honing Rug.
Endorheic	7.89	5.10	10.40	63	43	Mean = 7.81; SD = 1.32
Endorheic + "Probably Endorheic"	7.96	5.10	10.40	67	46	Mean = 7.82; SD = 1.31
Impacted lacustrine	8.98	6.1	10.8	63	5	Mean = 8.81; SD = 1.04
Impacted Endorheic + "Probably Endorheic"	8.4	4.5	10.0	24	2	Mean = 8.19 ; SD = 1.22

Table 3.5b *Significant differences in* ***pH*** *(p > 0.05) between different wetland types as indicated by the Mann-Whitney test. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).*

	Endorheic	Endorheic +"Probably endorheic"	Palustrine	Lacustrine	Impacted lacustrine	Impacted Endorheic +"Probably endorheic"
Springs	N. S.	N. S.	N. S.	N. S.	**Significant** (0.0000)	**Significant** (0.0211)
Endorheic		N. S.	N. S.	N. S.	**Significant** (0.0000)	N. S.
Endorheic +"Probably endorheic"			N. S.	N. S.	**Significant** (0.0000)	N. S.
Palustrine				N. S.	**Significant** (0.0000)	**Significant** (0.0258)
Lacustrine					N. S.	**Significant** (0.0490)
Impacted lacustrine						**Significant** (0.0347)

2. There was no significant difference in the statistical results obtained if "probably endorheic" systems are added to the endorheic systems.

3. Once again springs showed the lowest range in values (approximately 3 pH units). The median pH was 8.0 for this type of wetland.

4. Palustrine systems exhibited the lowest median pH (7.55) of all the wetland types. This type of wetland also exhibited the widest range in pH values. These ranged from 3.7 (measured at the aptly named "Suurdam", on the Cape Peninsular) to 11.2 (measured at Honing Rug on the Agulhas Plain).

5. Although the number of wetlands that were considered to be impacted was small (5 lacustrine and 2 endorheic/probably endorheic systems), the median pH values for these wetlands were the highest in the entire data-set. Impacted lakes exhibited a significantly higher median pH value than all other groups (Table 3.5b).

This is likely to be a consequence of the eutrophic state of these wetlands, since extreme rates of photosynthesis commonly cause very high pH values in standing waters (Dallas and Day 2004). Impacted "endorheic and probably endorheic" systems were significantly different in pH to springs, palustrine systems and lakes. This may also be a consequence of eutrophication.

3.4.3.3 The effect of wetland type on phosphate concentration

The results from an investigation into the effect of wetland type on the concentration of soluble reactive phosphorus (SRP) (ortho-phosphate) are summarized in Tables 3.6a and b.

Table 3.6a *The variation in **SRP** (ortho-phosphate) values (in mg/L) exhibited by the different wetland types in the Wetland Water Quality Database.*

Wetland type	Median	Minimum	Maximum	Sample size (n)	No. of wetlands	Comments
Springs	0.01	0.005	0.127	39	39	Mean = 0.014 mg/L. SD = 0.019
Lacustrine	0.029	0.001	0.137	44	13	Mean = 0.031; SD = 0.028
Palustrine	0.160	0.002	2.2	46	17	Mean = 0.313; SD = 0.57
Endorheic	0.024	0.003	0.81	35	24	Mean = 0.066; SD = 0.143
Endorheic + Probably Endorheic	0.020	0	0.81	36	25	Mean = 0.064; SD = 0.142
Impacted Lacustrine	0.04	0.005	0.16	17	4	Mean = 0.047; SD = 0.039
Impacted Endorheic + Probably Endorheic	0.06	0.05	0.07	4	1	Mean = 0.058; SD = 0.01

Table 3.6b *Significant differences in **SRP** (p > 0.05) between different wetland types as indicated by the Mann-Whitney test. SRP values given in mg/L. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).*

	Endorheic	Endorheic +"Probably endorheic"	Palustrine	Lacustrine	Impacted lacustrine	Impacted Endorheic +"Probably endorheic"
Springs	**Significant** (0.0217)	**Significant** (0.0412)	**Significant** (0.0001)	**Significant** (0.0003)	**Significant** (0.0000)	**Significant** (0.0020)
Endorheic		N. S.	**Significant** (0.0095)	N. S.	N. S.	N. S.
Endorheic +"Probably endorheic"			**Significant** (0.0058)	N. S.	N. S.	N. S.
Palustrine				**Significant** (0.0011)	N. S.	N. S.
Lacustrine					**Significant** (0.0351)	**Significant** (0.0153)
Impacted lacustrine						N.S.

1. Of the non-impacted wetlands, the springs showed the lowest median concentrations of soluble reactive phosphorus in the water. This was accompanied by the lowest maximum concentration. The phosphate concentration was significantly lower in springs than in all the other wetland types (Table 4.6b).

2. The highest phosphate level exhibited by non-impacted wetlands was in the palustrine type (median = 0.16 mg/L, maximum 2.2 mg/L). Palustrine systems were significantly different with regard to phosphate levels from all other natural wetland systems. The difference between palustrine and *impacted* wetlands, however, was not significant. It is interesting that the soluble phosphate levels exhibited by endorheic (and probably endorheic systems) were not the highest. Considering that these are "closed systems" it might have been expected that the highest values be exhibited by these systems. For a more definitive examination of situation, however, phosphate levels in the sediments would need to be examined.

3. Although the number of impacted wetlands that were examined is low, the results suggest that the levels of phosphate in the water column were not particularly high. This may be a result of rapid uptake by macrophytes or phytoplankton in these systems. Nevertheless, as is to be expected, the phosphate concentrations in the impacted systems, in general were significantly higher than in their non-impacted counterparts.

4. The highest phosphate concentrations (7.0 mg/L) were recorded for the Pretoria salt pan (Ashton and Schoeman 1983). Very high levels of nitrate, nitrite and ammonia were also measured in the system. This is considered by the project team to be an atypical wetland and was excluded from the calculations of median phosphate concentration. The system with the next highest phosphate levels (2.2 mg/L) was Rocher Pan in the Western Cape (Coetzer 1981).

5. Total phosphate, as opposed to ortho-phosphate concentrations in the water column were also examined. Unfortunately few measurements of total phosphorus were recorded in the database (n = 28; number of wetlands = 15). These results are therefore not presented here.

3.4.3.4 The effect of wetland type on nitrogen concentration

The concentration of various forms of nitrogen were recorded in the database. The chemical species of nitrogen, the number of records for each (sample size), and the number of wetlands represented at the time of interrogation, are summarized in Table 3.7.

Table 3.7 *Summary of the data available for nitrogen in the database (Feb 2004)*

Chemical species	Sample size	No. of wetlands represented
Ammonium	219	101
Nitrate + nitrite	105	55
Nitrate only	103	49
Total inorganic nitrogen	2	2
Total nitrogen	93	46
Kjeldahl nitrogen	29	17

Nitrate and ammonium results were examined, because of the relatively large data-sets and because the widest range of wetland types were represented by these chemical species. The results from an investigation into the effect of wetland type on the concentration of nitrate are summarized in Tables 3.8a and b.

1. No data were available for springs, nor were any "Probably endorheic" systems represented. The number of wetlands representing the lacustrine and palustrine systems was low (9 and 3 respectively).

2. The median nitrate concentrations exhibited by the non-impacted wetlands (lacustrine, palustrine and endorheic systems) did not vary significantly between the different types. This may have been the result of the low number of wetlands examined for some wetland types.

Table 3.8a *The variation in **nitrate** values (in mg/L) exhibited by the different wetland types in the Wetland Water Quality Database.*

Wetland type	Median	Minimum	Maximum	Sample size (n)	No. of wetlands	Comments
Springs	-	-	-	-	-	No nitrate values recorded (all nitrate + nitrite)
Lacustrine	0.013	0.002	0.187	24	9	Mean = 0.031; SD = 0.049
Palustrine	0.019	0.007	0.08	6	3	Mean = 0.03; SD = 0.03. Note small no. wetlands
Endorheic	0.014	0.003	0.034	28	21	Mean = 0.015; SD = 0.008
Endorheic + Probably Endorheic						Same as for endorheic. No additional records.
Impacted Lacustrine	0.40	0.1	1.5	11	2	Mean = 0.51; SD = 0.45
Impacted Endorheic + Probably Endorheic	1.75	1.45	3.6	7	1	Mean = 2.09; SD = 0.77

Table 3.8b *Significant differences in* ***nitrate*** *(p > 0.05) between different wetland types as indicated by the Mann-Whitney test. Nitrate values given in mg/L. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).*

	Palustrine	Lacustrine	Impacted lacustrine	Impacted Endorheic +"Probably endorheic"
Endorheic	N. S.	N. S.	**Significant** (0.0000)	**Significant** (0.0001)
Palustrine		N. S.	**Significant** (0.0009)	**Significant** (0.0027)
Lacustrine			**Significant** (0.0000)	**Significant** (0.0001)
Impacted lacustrine				**Significant** (0.0010)

3. In the case of impacted wetlands, although again the number of wetlands represented is too low to make definitive inferences, the results suggest that the level of nitrate in the water column of these systems is relatively high. Thus, impacted wetland systems exhibited significantly higher nitrate concentrations than non-impacted systems. The highest nitrate concentration recorded for an un-impacted wetland was 0.19 mg/L (for Rondevlei, Wilderness Lakes National Park, Jones 2002) and the highest for an impacted system was 3.6 mg/L (for Germiston Lake, Vermaak *et al.* 1981).

The results from an investigation into the effect of wetland type on the concentration of ammonium are summarized in Table 3.9a and 3.9b.

1. The median values of ammonium for the non-impacted wetlands did not vary greatly. The highest median values were exhibited by the endorheic systems and the lowest by the springs. The springs were significantly lower in ammonium levels than all other wetland types.

2. Some of the highest values were exhibited by the impacted wetlands, particularly the freshwater lakes, and these were statistically higher than non-impacted lakes.

Table 3.9a *The variation in* ***ammonium*** *values (in mg/L) exhibited by the different wetland types in the Wetland Water Quality Database.*

Wetland type	Median	Minimum	Maximum	Sample size (n)	No. of wetlands	Comments
Springs	0.04	0.02	0.105	40	40	Mean = 0.04; SD = 0.017
Lacustrine	0.08	0.005	0.207	44	14	Mean = 0.08; SD = 0.045
Palustrine	0.07	0.04	2.5	27	16	Mean = 0.26; SD = 0.59
Endorheic	0.09	0.02	1.3	36	25	Mean = 0.19; SD = 0.28
Endorheic + Probably Endorheic						Same as for endorheic. No additional records.
Impacted Lacustrine	0.119	0.005	0.9	36	3	Mean = 0.16; SD = 0.18
Impacted Palustrine	0.09	0.05	2.3	14	4	No impacted endorheic or "probably endorheic" records for ammonia.

3.4.3.5 Comparison of the results obtained with the Target Water Quality Range

A comparison was made of the median values described above, with the Target Water Quality Range (TWQR) recommended in the guidelines for aquatic ecosystems (DWAF 1996). As noted in section 2.7, recommended values of physical variables and chemical concentrations that should not be exceeded in aquatic resources are given in the guidelines. In the guidelines, the following approaches to setting values are taken:

- **conductivity and salinity** are specified as percentage deviation from natural;
- **pH** is specified as percentage deviation from natural;
- **phosphate** guidelines are expressed as "Average summer inorganic phosphorus" concentration. Less than 0.005 mg/L is considered to represent

oligotrophic conditions, between 0.005 – 0.025 mg/L represents mesotrophic conditions, and 0.025 to 0.25 mg/L, eutrophic conditions;

- **nitrogen** concentrations are expressed in the guidelines as "Average summer values", in this case for inorganic nitrogen which includes nitrate, nitrite, un-ionized ammonia and ionized ammonium. Less than 0.5 mg/L is considered to represent oligotrophic conditions, between 0.5 and 2.5 mg/L mesotrophic conditions, and from 2.5 – 10.0 mg/L, eutrophic conditions;
- **ammonium:** The guidelines are expressed in terms of the proportion of un-ionised ammonia to total ammonia. This is dependent on the temperature and pH of individual wetlands.

Table 3.9b *Significant differences in **ammonium** (p > 0.05) between different wetland types as indicated by the Mann-Whitney test. Ammonium values given in mg/L. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).*

	Endorheic	**Palustrine**	**Lacustrine**	**Impacted lacustrine**	**Impacted Endorheic +"Probably endorheic"**
Springs	**Significant** (0.0000)	**Significant** (0.0000)	**Significant** (0.0006)	**Significant** (0.0000)	**Significant** (0.0000)
Endorheic		N. S.	N. S.	N. S.	N. S.
Palustrine			N. S.	N. S.	N. S.
Lacustrine				**Significant** (0.0043)	N. S.
Impacted lacustrine					N. S.

Because the guidelines for conductivity/salinity/TDS and pH are set as allowed percentage deviation from the natural condition, it was not possible to assess these variables. With regard to the nutrient levels, however, none of the wetland types, not

even springs could be considered from the guidelines to be oligotrophic with regard to phosphorus. The phosphate medians for springs, endorheic and probably endorheic systems fell into the mesotrophic range. The lacustrine and expecially the palustrine systems fell into the eutrophic range. Individual wetlands, however, from all wetland types spanned the range of trophic states (Table 3.6) with some palustrine wetlands being even hypertrophic (inorganic phosphate levels greater than 0.25 mg/L).

The nitrogen median values of all the wetland types, in addition to the maximum values, are all below 0.5 mg/L in Table 3.7, placing all the wetlands in a trophic status of "oligotrophic" for inorganic nitrogen. As a gross approximation the values of ammonium were added to the nitrate medians for the appropriate wetlands. Even then, the combined medians for nitrate and ammonium for each wetland type for which there were data (lacustrine, palustrine and endorheic) were less than 0.5 mg N/L and were thus still oligotrophic. It must be kept in mind that the guidelines represent summer averages, whereas the given values are from the entire year. Furthermore, the guidelines are for aquatic ecosystems in general, and not specifically for wetlands. This work would indicate that the guidelines need to be refined further for wetlands.

3.4.4 Phase 2: Effect of geographical region on water quality

Because climate, topology and geology vary across the country, it is quite likely that the characteristic water quality of wetlands will also vary. When compiling the Wetlands Water Quality Database, where available, information on the geographical province, the wetland area (*sensu* Cowan 1995 – see section 2.2.1) and the bioregion (Brown *et al.* 1996) were recorded. Preliminary efforts were made to establish if water quality differed for the different wetland areas or bioregions. These areas are relatively small however (for instance there are 26 wetland regions defined by Cowan (1995)), and the distribution of wetlands in the database (as well as in reality) is not evenly spread across the groups, so that some regions contain many wetlands, whilst others have none. Furthermore, because a variety of water quality variables are routinely measured (see for example Table 3.7 for nitrogen) finding measurements that are common to all also limits the sample size. Consequently, the effect of geographic region on water quality was examined, but at a coarser resolution than the wetland area or bioregion. It was decided, as a preliminary attempt, to investigate the effect of rainfall region on wetland water quality. Once the extent of the database has been increased, with a more even

distribution across the country, it may be possible to examine wetland grouping within bioregions, wetland areas, water quality management regions (which have been derived using river water chemistry data (Day, Dallas and Wackernagel 1998)), ecoregion (Kleynhans and Hill 1999) or broad climatic regions.

Rainfall regions in the country were demarcated using standard maps (Fullard 1987). The wetlands represented in the database were allocated to one of five rainfall regions as shown in Figure 3.9. These rainfall regions were; winter or summer rainfall regions (represented by the Western Cape and High Veld areas respectively, where most of the rainfall occurs during those seasons), "All year rain" which represents the Garden Route area, where precipitation is evenly distributed throughout the twelve months of the year. The arid western areas of the country are represented by the "Little rain, summer" region – indicating that whatever limited rainfall occurs in that area occurs during summer. The final group is the mesic "All year, summer maximum" region of the eastern seaboard where rain occurs all months of the year, with a maximum in summer. The effect of rainfall region on the five key water quality variables identified in Phase 1 of the interrogation process was first examined without dividing the wetlands in to wetland type (palustrine, endorheic etc.). Data representing all seasons of the year were combined. Because it was known, however, that the "springs" represented an anomalous group, being low in conductivity and other chemical constituents (probably an artifact due to the fact that they were selected for use as drinking water – section 3.4.4), springs were not included in the data-set. Impacted wetlands were also excluded from the analysis.

3.4.5.1 Rainfall region and conductivity levels

The results for conductivity are summarized in Table 3.10a, and the tests for significance in Table 3.10b. Because of the small sample size, salinity was not examined.

1. The lowest median EC values (26 mS/m) were exhibited by the wetlands that fell into the "All year, summer maximum" region of the country (the eastern coastal area), this group also showed the smallest range of values and was statistically different ($p < 0.05$) from all the other rainfall regions. This region of the country is one of the most mesic in the country, with annual mean rainfall of over 1 000 mm in some places (Fullard 1987) and thus the "dilution capacity" is likely to be high. Many of the wetlands in this area and

recorded in the database, are freshwater lakes which were one of the wetland types that exhibited the lowest EC values (Table 3.3a).

2. The highest median EC values (1 985 mS/m) were exhibited by the wetlands of the summer rainfall region and this value is significantly higher than all other wetland groups. Quite why this region should exhibit a higher median conductivity value than the arid region ("Little rain, summer") is not clear. The reason why the latter should have a lower conductivity median than the areas with rain evenly distributed through the entire year ("All year rain") is also not clear. These apparent anomalies may be a consequence of the small sample size for some regions (especially for the number of wetlands sampled). The different wetland types (lacustrine, pallustrine, etc.) are not distributed evenly throughout the regions. This is partly because of limited data in the database and also because some types do predominate naturally in some areas (e.g. endorheic pans in the Free State, and freshwater lakes along the eastern coast).

3. The wetlands of the arid western half of the country ("Little rain, summer" group) exhibited the widest range in conductivities. This may well be a consequence of the fact that these are largely endorheic pans which are ephemeral. Therefore conductivities could be expected to range from that more or less of rainwater (just after filling) to hypersaline conditions (after a long period of no rain and intense evaporation).

4. The median EC values exhibited by the "All year rain", "Little rain (summer maximum)" and the winter rainfall wetlands were intermediate in value. Despite this, most rainfall groups were significantly different from each other with regard to conductivity (apart from the "All year rainfall" and the "Little rain (summer maximum)" groups) and the "Winter rainfall" and "Little rain (summer maximum)" groups.

Figure 3.9 *The rainfall regions of the country and the geographical position of wetlands in the database.*

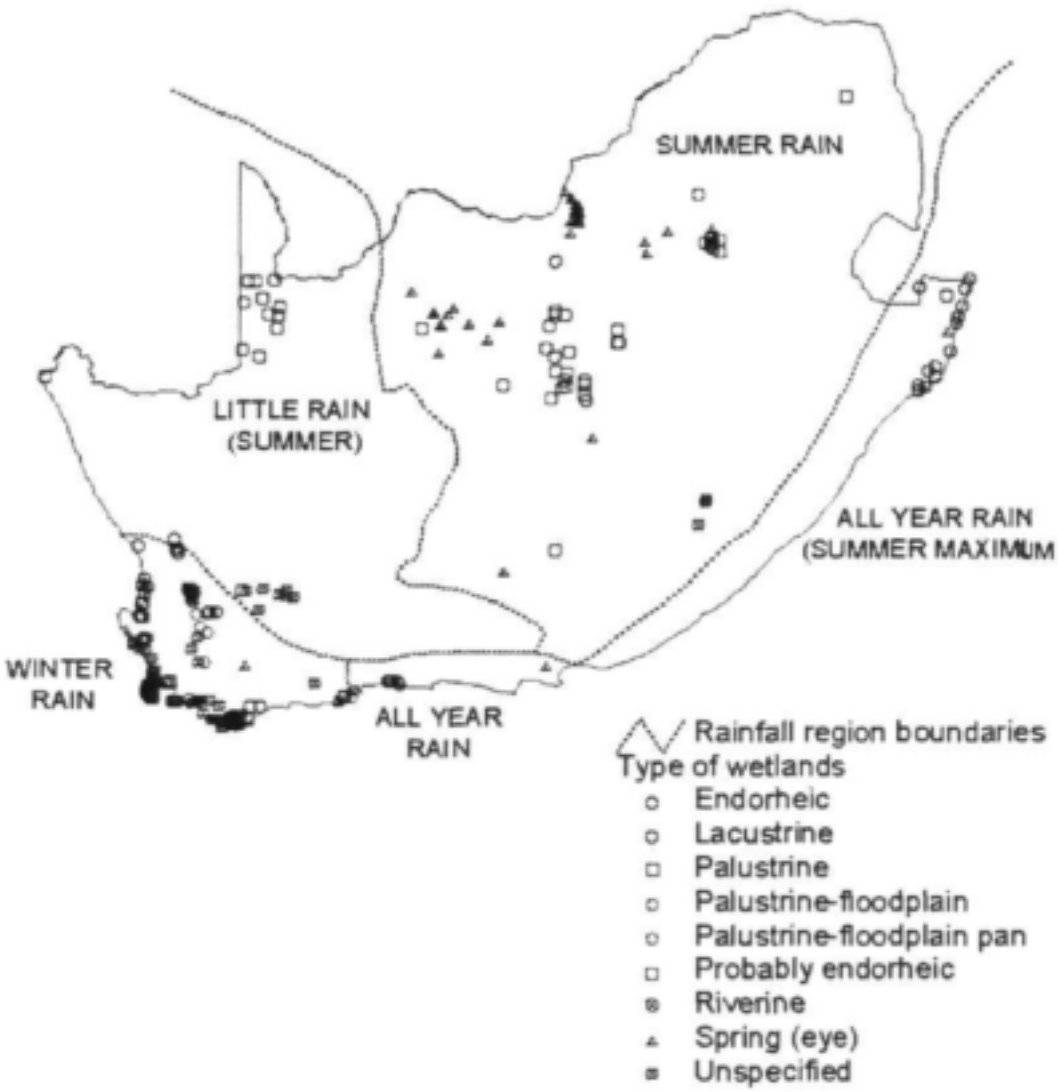


Table 3.10a *The variation in* ***conductivity*** *(mS/m) values exhibited by wetlands in the different rainfall areas.*

Rainfall region	Median	Minimum	Maximum	Sample size (n)	No. of wetlands
All year rain	944	6.2	1737	24	8
Little rain	523	4	46 523	29	26
All year (summer maximum)	26	8	569	19	13
Summer	1985	4	30 354	35	34
Winter	339	1	9 310	236	103

Table 3.10b *Significant differences in* ***conductivity*** *($p > 0.05$) between wetlands in different rainfall regions as indicated by the Mann-Whitney test. Conductivity values given in mS/m. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).*

Rainfall region	Little rain	All year (summer maximum)	Summer	Winter
All year rain	N.S.	Significant (0.0000)	Significant (0.0031)	Significant (0.0023)
Little rain		Significant (0.0015)	Significant (0.0042)	N.S
All year (summer maximum			Significant (0.0000)	Significant (0.0000)
Summer				Significant (0.0000)

3.4.5.2 Rainfall region and pH levels

The results from an examination of pH are summarized in Table 3.11a, and the tests for significance in Table 3.11b.

Table 3.11a *The variation in **pH** values exhibited by wetlands in the different rainfall area.*

Rainfall region	Median	Minimum	Maximum	Sample size (n)	No. of wetlands
All year rain	8.73	3.82	9.73	21	6
Little rain	7.9	7.60	10.70	11	9
All year (summer maximum)	7.1	5.8	8.3	11	11
Summer	7.40	5.60	10.40	15	12
Winter	7.49	3.70	11.21	207	112

Table 3.11b *Significant differences in **pH** ($p > 0.05$) between wetlands in different rainfall regions as indicated by the Mann-Whitney test. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).*

Rainfall region	Little rainfall	All year (summer maximum)	Summer	Winter
All year rain	**Significant** (0.04954)	**Significant** (0.0001)	**N.S.**	**Significant** (0.0001)
Little rainfall		**Significant** (0.0007)	**N. S.**	**N. S.**
All year (summer max.)			**N. S.**	**N. S.**
Summer				**N. S.**

1. The lowest median pH values (7.49) were exhibited by the wetlands of the mesic eastern seaboard area ("All year, summer maximum"). This was unexpected as it was thought that the winter rainfall (fynbos-dominated) area would be the lowest.

2. The highest median pH values (8.73 mS/m) were exhibited by the wetlands of the Garden Route region in which rain is evenly distributed throughout the year ("All year rain"). The wetlands of this area represented in the database are almost exclusively freshwater lakes.

3.4.5.3 Rainfall region and phosphate levels

The results from an investigation into the effect of rainfall region on the concentration of soluble reactive phosphorus (ortho-phosphate) are summarized in Tables 3.12a and b.

In general most of the data were for wetlands in the winter rainfall region, few wetlands in other regions had measurements for SRP. The highest values of SRP were exhibited by the winter rainfall wetlands (and this value was significantly higher than that for the "all year rain" or the "all year (summer maximum)" groups.

Table 3.12a *The variation in **SRP** (in mg/L) values exhibited by wetlands in the different rainfall areas.*

Rainfall region	Median	Minimum	Maximum	Sample size (n)	No. of wetlands
All year rain	0.015	0.001	0.14	18	4
Little rain	0.065	-	-	1	1
All year (summer max)	0.008	0.001	0.639	14	7
Summer	0.030	0.003	1.12	3	1
Winter	0.037	0.002	2.20	108	45

Table 3.12b Significant differences in **SRP** ($p > 0.05$) between wetlands in different rainfall regions as indicated by the Mann-Whitney test. SRP values given in mg/L. Where the difference between a pair of wetland types is significant, the p value is given in parenthesis. (N.S. = not significant).

Rainfall region	All year (summer maximum)	Winter
All year rain	N.S.	**Significant** (0.0052)
All year (summer maximum)		**Significant** (0.0055)

3.4.5.4. Rainfall region and nitrogen levels

The results from an investigation into the effect of rainfall region on the concentration of nitrate are summarized in Table 3.13.

Table 3.13 *The variation in* ***nitrate*** *(in mg/L) values exhibited by wetlands in the different rainfall areas.*

Rainfall region	Median	Minimum	Maximum	Sample size (n)	No. of wetlands
All year rain	0.020	0.006	0.18	12	3
Little rain	No Data				
All year (summer max)	0.009	0.002	0.016	2	2
Summer	No Data				
Winter	0.014	0.001	2.91	70	43

Table 3.14 *The variation in* ***ammonium*** *(in mg/L) values exhibited by wetlands in the different rainfall areas.*

Rainfall region	Median	Minimum	Maximum	Sample size (n)	No. of wetlands
All year rain	0.070	0.014	0.16	16	4
Little rain	0.121				
All year (summer max.)	0.106	0.005	0.207	4	4
Summer	0.06				
Winter	0.06	0.015	2.50	86	46

In general, as for SRP most of the data were for wetlands in the winter rainfall region, few wetlands in other regions had measurements for SRP. There were no data for the arid ("Little rain, summer maximum") or summer rainfall regions. There was no significant difference between the winter rainfall median value and the medians obtained for the other two groups. A similar result was obtained for ammonium (Table 3.14).

From the above results it was concluded that whilst there were ranges of water quality variables that were characteristic of different rainfall regions, it was difficult to pick out clear patterns. Other than the mesic "all year, summer maximum" region which is likely to have lower conductivity and pH values.

Water quality also varies with wetland type (section 3.4.4) and season, and this is likely to be confusing the pattern of regional variation.

The only rainfall region and water quality variables for which there was a fairly even distribution of wetland types represented was conductivity and pH in the winter rainfall region. In section 3.4.4.2 it was found that there was not a great deal of variation in pH between wetland types, therefore only conductivity was investigated further.

3.4.5.5 The effect of wetland type on conductivity values in the winter-rainfall region

The conductivity data for wetlands in the winter rainfall region only were examined. Data from wetlands known to be polluted were excluded. The results obtained are shown in Table 3.15. All three wetland types were significantly different from each other and a similar ranking of wetland type was obtained as shown in Table 3.4a. In other words, overall (lumping all rainfall regions together) lacustrine wetlands exhibited the lowest median conductivity, palustrine an intermediate value and the endorheic and probably endorheic wetlands the highest values. Within the winter rainfall region, a similar ranking of wetland types was obtained. This is perhaps to be expected because the proportion of wetlands from the winter rainfall region making up the total population in the database is high.

Table 3.15 *Variation in* ***conductivity*** *(in mS/m) values for wetlands in the winter rainfall region*

Wetland type	Median	Minimum	Maximum	Sample size (n)	No. of wetlands
Endorheic Probably endorheic	665	9.7	9310	52	30
Palustrine	462	2	8850	86	21
Lacustrine	66	12	1751	46	8

3.4.5 Phase 3: A multivariate approach

(This section was compiled by Dave Miller, Dept. Zoology, UCT)

One of the aims of this chapter was to examine regional patterns in water quality. In section 3.4.4 the effect of rainfall region on water quality was examined without any

allowance for the possible effects of the *type* of wetlands in each region or the rainfall *season* (wet or dry) in which the water quality data were recorded.
It was decided to take this further by using a multivariate approach to examine, which regions differed significantly from one another, and which water quality variables were accounting for these differences. This was done by first removing the effect of rainfall season and then by comparing wetlands of the same type in different rainfall regions.

As mentioned before, at present there is a bias towards the wetlands in the Western Cape and therefore the "winter rainfall" region. Likewise, most of the wetlands that have complete water quality variable data are from this region with almost all of the phosphate (SRP) and nitrate readings from the "winter rainfall" wetlands. As before, springs and impacted systems were not included in the analyses. While the data are limiting, the multivariate analysis was still performed in order to show the potential of such interrogation techniques on the information from the database. However, the scope of the tests was not as broad as could have been achieved using a complete data-set.

It was decided to perform a discriminant function analysis (DFA). Discriminant function analysis can be used to determine which variables best discriminate between a number of different groups. It is computationally very similar to analysis of variance (ANOVA) and utilises a Wilks' Lambda *F*-test to identify significant differences between groups or significant contributions by individual predictor variables. Canonical analysis was also performed to assess the relationship between predictor variables. During the DFA 'roots' are developed that combine the different variables to try explain differences in the multidimension location of each case in the analysis. Eigenvalues for each root describe what proportion of the variance the root describes. The standardised co-efficients from the canonical analysis explain how much each variable contributed to each root. The standardised co-efficients and the eigenvalues for the roots were used to calculate the relative importance of each water quality variable in distinguishing between rainfall regions.

For the DFA the water quality variable data were standardised. While the data were not quite normally distributed, DFA is fairly robust against this so this was not considered to be problematic. Other assumptions that need to be met for DFA include: homoscedastic variances (improved by standardising the data), no redundant variables (none of the

variables significantly correlated with any of the others), and no correlation between the means and variances for variables across groups.

In this case, there was a slight relationship between the means and variances, brought on mainly due to the "Winter rainfall region" having a large number of outliers. As said before, this was only a preliminary analysis to demonstrate what multivariate work could be done using the database. While the data were not perfectly suited to the DFA, no major problems were found so the results can still be accepted with a reasonable amount of confidence.

3.4.6.1 Comparison of rainfall regions during wet and dry seasons

Unfortunately, due to gaps in the database there was a trade-off between the number of water quality variables that could be used and the number of rainfall regions that could be compared. Wetlands with pH, conductivity, phosphate (SRP) and nitrate data were only found in the "All year rainfall region", "Summer rainfall" (only two records) and the "Winter" rainfall regions. If only pH and conductivity were considered, however, all five rainfall regions could be compared. Therefore, initially only conductivity and pH were used as predictor variables.

A Wilks' Lambda test showed that these two water quality variables could significantly differentiate rainfall regions during the wet or dry season ($p < 0.0086$). Of the individual variables however, only pH (explaining 65% of the variance) was shown to be a significant contributor to differences between groups. The results of Wilks' Lambda tests done on the Mahalanobis distances ('multidimensional distances') between the rainfall regions during the wet or dry season are shown in Table 1. Within each region no significant differences between water quality in the dry and wet season are found. The arid ("little rain, summer maximum") region only differed significantly from the "all year, summer maximum" region and this was only during the dry season. The "all year, summer maximum" region, which had data only for the wet season, also differed significantly from the "all year" region and the "summer" region during the dry season.

Table 3.16 *Significant differences in conductivity (in mS/m) and pH ($p < 0.05$) between rainfall regions in the wet or dry season as indicated by the Wilks' Lambda test on Mahalanobis distances between groups. Where the difference between a pair of wetland types is significant, the p-value is given in parenthesis. (N.S. = not significant).*

	Arid, Wet	**All year, summer maximum Wet**	**Summer Dry**	**Summer Wet**	**Winter Dry**	**Winter Wet**	**All year**
Arid, Dry	N.S.	Significant (0.0436)	N.S.	N.S.	N.S.	N.S.	N.S.
Arid, Wet		N.S.	N.S.	N.S.	N.S.	N.S.	N.S.
All year, summer max, Wet			Significant (0.0018)	N.S.	N.S.	N.S.	
All year, summer max Dry	NO DATA						
Summer, Dry				N.S.	Significant (0.0027)	Significant (0.0017)	N.S.
Summer, Wet					N.S.	N.S.	N.S.
Winter, Dry						N.S.	N.S.
Winter, Wet							N.S.

The only other significant difference was between the "summer rainfall region" during the dry season and any season in the "winter rainfall region". As season had no significant effect on water quality within rainfall regions, the remainder of the analyses only consider rainfall regions, incorporating readings from throughout the year. Table 2 shows the results of Wilks' Lambda tests done on the Mahalanobis distances between the rainfall regions. Again pH explains the majority of the variance (64%). The "all year, summer maximum" region appears to be the most unique being significantly different from all but the "winter" region. The "winter" rainfall region was also shown to differ significantly from the "arid" (i.e. "little rain, summer" area) and "summer rainfall regions", but not from the "all year rainfall" region of the garden route area.

When conductivity, pH, phosphate and nitrate were used as predictor variables, the same results were found, even though only "all year", "summer" and "winter" rainfall regions could be compared. However, the canonical analysis showed nitrate to be the most important predictor variable (64%). Phosphate explained 22% of the variance while pH only explained 13% and conductivity barely explained any of the variance at all.

Table 3.17 *Significant differences in conductivity (in mS/m) and pH ($p < 0.05$) between rainfall regions as indicated by the Wilks' Lambda test on Mahalanobis distances between groups. Where the difference between a pair of wetland types is significant, the p-value is given in parenthesis. (N.S. = not significant).*

Significant amounts

	Arid	**All year, summer max**	**Summer**	**Winter**
All year	N.S.	**Significant** (0.0051)	N.S.	N.S.
Arid		**Significant** (0.0142)	N.S.	**Significant** (0.0413)
All year, summer max			**Significant** (0.0073)	N.S.
Summer				**Significant** (0.0095)

Therefore, while it was shown in section 4.4.4 that conductivity varied greatly between wetland types, it appears that pH and nitrate are more important between rainfall regions. This needs to be investigated further once more data are available for phosphate and nitrate in regions other than the "winter" rainfall region.

3.4.6.2 Comparison of wetland types in different rainfall regions

Whilst it could be shown that significant differences in water quality do exist between the rainfall regions, it is also of importance to show whether or not wetlands of the same type differ in terms of the water quality variables.

Endoheic wetlands

Sufficient data to compare endorheic wetlands were found only for pH and conductivity in the "winter", "summer" and "all year, summer maximum" rainfall regions. All three rainfall regions were shown to differ significantly from each other ($p < 0.05$). Only pH significantly differentiated between the endoheic wetlands of the different regions, explaining 62% of the variance.

Lacustrine wetlands

Sufficient data to compare lacustrine wetlands were found only in the "all year" and "winter" rainfall regions. However data were available for conductivity, pH, phosphate and nitrate. While it was shown in section 4.4.6.1 that water quality in general in these two rainfall regions did not differ significantly, the DFA for lacustrine wetlands did show a significant difference. Unlike the endorheic wetlands where pH was shown to differ significantly from region to region only conductivity (explaining 61% of the variance) differed significantly between lacustrine systems in the two regions. pH and nitrate explained 21% and 18% of the variance, respectively, while phosphate practically none of the variance.

There were not sufficient data to compare any of the other wetland types. However, while this was a fairly incomplete data-set, it suggests that acceptable water quality standards of wetlands types vary between rainfall regions. Also, different water quality variables explain regional differences between each wetland type. In endorheic wetlands pH was shown to be quite variable while in lacustrine wetlands conductivity was the important distinguisher. These results also confirmed previous thoughts that differences between wetland types confused comparisons of rainfall regions (section 4.4.5) as the "all year" and "winter" rainfall regions did not differ in general but did differ when only lacustrine wetlands were examined.

3.5 CONCLUSION

In this chapter a database containing water quality measurements for many South African wetlands was presented. Interrogation of the database yielded useful information on the ranges of values of water quality variables exhibited by different wetland types. Further preliminary examination of the characteristic water quality for different climatic regions were not so successful. This is likely to be a consequence of the fact that the water quality in a wetland is the result of many factors including the wetland type, climate, geology, topography and season. Once extra data have been added to the database, it may be possible to elucidate these factors more clearly.

CHAPTER 4
WATER QUALITY OBJECTIVES FOR WETLANDS IN SOUTH AFRICA

4.1 INTRODUCTION

In the literature review for this project (Chapter 2) the importance of wetlands and the need for protecting these ecosystems by setting limitations to the pollution loads entering them was explained. The major focus of this chapter is to present a proposed Decision Support System (DSS) for setting the water quality component of the ecological Reserve, or in other words, for establishing the allowable maximum concentrations of pollutants in a given wetland. This is necessary because, although wetlands are efficient at removing pollutants from water flowing through them, this capacity is not infinite. For example, if a toxic effluent were discharged into a wetland and the resulting concentrations of pollutants in the water were sufficiently high – this could cause the death of micro-organisms or even emergent plants in the system. A decrease in or even total loss of, the water purification ability of the wetland and other ecological functions may well then follow.

The current method for determining the water quality component of the ecological Reserve for wetlands (section 2.5.4) is incomplete. The task of setting the appropriate water quality objectives is complicated by the fact that the objectives need to be linked to the ecological classification system (Category A, B, C etc. – or alternatively Natural, Good, Fair etc.) used in the Reserve methodology. This is because it is recognized in the National Water Act (1998) that whilst resources need to be protected, they also need to be utilized, and therefore it is not possible to keep all aquatic resources in a pristine state (DWAF 1999a). By making an assessment of (amongst other factors) the present ecological status of a wetland and its ecological and social importance, a decision can be made as to Ecological Category for which a given system will be managed (DWAF 1999b). The more sensitive and conservation-worthy a wetland is, the higher the level of protection from pollution that should be given, the higher the desired level of ecosystem "health" and the more stringent the water quality specifications. At the same time, for other wetlands that are not so ecologically important, a higher level of impact may be allowed. As shall be seen in this chapter, it is the linking of the desired level of

ecosystem health and the water quality conditions that will achieve this state that is one of the most challenging aspects of this project.

4.2 THEORETICAL APPROACHES TO SETTING WATER QUALITY OBJECTIVES

From an examination of the literature and of local and international practices (section 2.7) for setting water quality specifications for aquatic ecosystems, there appears to be at least five different approaches that have some merit for the present project. These are discussed below.

4.2.1 Emission versus immission standards

Since water quality in wetlands is inherently variable and it is difficult to set realistic water quality specifications – there is perhaps merit in not setting immission specifications (also known as "receiving water quality objectives") at all. Rather efforts could be directed to limiting the amount and type of pollutants entering the wetland, as has been used in some states in the USA (WDNR 1992). The argument concerning the merits of resource directed measures compared to source-directed controls has been widely discussed for aquatic resources in general (Mance 1986), and to some extent in relation to wetlands (Coetzee 1995). It is generally recognized that a combination of both approaches is needed and that both have their advantages and disadvantages (van der Merwe and Grobler 1990; Lijklema 1995; Jirka, Bleninger *et al.* 2004). The vulnerability of wetlands in general and endorheic systems in particular, to diffuse sources of pollutants makes the use of receiving water objectives necessary. In addition, strict controls for effluent emissions in order to bring about the required receiving water quality objectives are essential.

4.2.2 Extrapolation from other types of aquatic ecosystems

The water quality of some other aquatic ecosystems in South Africa (especially rivers) has been monitored and studied for much longer than wetlands. As a result, water quality specifications for different Ecological Categories in rivers can be set reasonably confidently (DWAF 2002). As part of the on-going development of methods to set the ecological Reserve, water quality ecospecifications for groundwater (Colvin, Cavé and Saayman 2003) and estuaries (DWAF1999c) are also being developed. A possible approach for setting water quality specifications for a given wetland is to determine the major water source (by volume) supplying a wetland and extrapolate, if possible, from

the appropriate standards (i.e. for rivers or groundwater). This is the type of approach suggested in the RDM manual (DWAF 1999b) for setting the ecological Reserve for wetlands (section 2.5.4). As part of the process of setting the ecological Reserve (the water quantity component), a site visit is made and the source of water feeding any wetlands is determined (section 2.5.4) This approach has been expanded and is used as the basis for the Decision Support System (DSS) to derive wetland water quality objectives in this chapter (section 4.4).

4.2.3 Extrapolation from international standards

In the case of developing countries such as South Africa, a sensible way to derive water quality standards might be to examine those used in other countries and extrapolate to suit local conditions. Sediment guidelines in the United Kingdom, for example, were drawn-up in such a manner (section 2.7.4). This approach was used to derive the "best-guess" default values of permissible maximum concentrations of chemical constituents used in this project. The default values form part of the DSS and are discussed further on in this chapter (section 4.5). It should be stressed though, that because of the poor understanding of the ecology of wetlands in South Africa, adapting values used in other countries to suit local conditions is likely to carry a risk of inaccuracy.

4.2.4 Extrapolate from other wetlands or from the Reference Condition

Wetlands with long-term sets of monitoring data can be used to extrapolate to similar wetlands which are data-poor. Care needs to be taken that the wetlands are comparable in type, degree of impact and that they will be managed at a similar level of ecological functioning (i.e. that the future Ecological Category is the same). A similar method to this is recommended in the Australian and New Zealand Water Quality Guidelines (ANZECC/ARMCANZ 2000), which makes use of the 80th percentile of the variable in question, calculated from at least two years data from the "comparable/target wetland" (section 2.7.2). In this regard, the Wetlands Water Quality Database developed during this project could be useful in helping to identify similar wetlands.

Alternatively, extrapolation can be made from the Reference Condition (RC) if these data are available. The reference data may either be historical data for the wetland under consideration or from a nearby wetland of the same type. For example, it was mentioned in section 2.7.1.3 that the US EPA recommends using the 75th percentile of a distribution

of nutrient measurements from reference sites (minimally impacted) as the recommended target concentration (water quality standard). If information on minimally impacted sites on a local scale is not available, they recommend that the 25th percentile of a distribution of samples from the entire population of water bodies (within, for example, a given ecoregion) be used (EPA 1998b). Research would need to be carried out to see if this last approach could be used in South Africa. It is likely that because of the heterogeneity amongst wetlands in this country, a more site-specific approach is required. In the Australian and New Zealand Water Quality Guidelines (ANZECC/ARMCANZ 2000), the RC may also be set using data from other sources, such as from the literature, modeling or expert opinion.

4.2.5 Narrative criteria

Even though it is difficult to put numerical values to the permissible maximum concentration of chemical constituents that should be allowed in a wetland, frequently, enough is known to be able to make qualitative statements. The wetland water quality standards developed by the US EPA contain narrative statements such as an "anti-degradation statement" (section 2.7.1.3). Such a statement ensures that if the water quality in a resource is better than the level established by standards, deterioration of water quality is still not permitted. Frequently the deleterious ecological consequence is known, but the quantitative water quality conditions that may trigger that response are not. Until there is a better understanding of cause and effect, narrative statements such as "salinity should not increase so that >10% of *Phragmites* reed beds die back" may also be useful as a management tool in South Africa.

4.3 GENERAL CONSIDERATIONS CONCERNING WATER QUALITY OBJECTIVES

When deriving water quality standards for aquatic ecosystems the following aspects need to be considered.

- Some water quality indicators are likely to differ significantly across the country, and therefore ecoregional values should be specified. Other indicators are not likely to differ and thus a set value is given that is applicable to all. In a similar manner, some variables are likely to vary depending on the type of wetland (e.g. salinity). A common assumption is that aquatic ecosystems are adapted to site-specific water quality and that any significant change from this will lead to impacts to the system (Liston and Maher 1997). These authors state that in the case of

water quality variables that are likely to differ, such differences can be dealt with *explicitly* by specifying a range of values (i.e. for each region or type of aquatic ecosystem). Ecoregional or ecosystem-type difference can also be dealt with *implicitly* by specifying that a future change not exceed a certain percentage of the long term mean. This latter approach has an inherent disadvantage in that the future condition of a water body will be judged from its present state. Thus if the condition is already degraded, there is a strong possibility that it will not improve. In the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF 1996), and in the first assessment method for deriving Resource Quality Objectives for the ecological Reserve for rivers (DWAF 1999a) some variables (e.g. pH) were specified as the allowable percentage deviation from the Reference Condition. Whilst using the Reference Condition rather than the present state might help to arrest a tendency towards worsening water quality, it is often difficult, especially for wetlands for which so little information is available, to establish what the Reference condition should be.

- Usually, formal statistical procedures are employed when deriving water quality standards for water bodies. Values may be derived from an analysis of the statistical distribution of chemical monitoring data (e.g. the nutrient guidelines for reservoirs and lakes in different ecoregions of the USA, Table 2.4). If no field data are available, the results from ecotoxicological studies may be used to give insight into the problem. Such studies relate a given concentration of a chemical substance to the effects (lethal or sub-lethal) on a population of test organisms under controlled conditions. Ecotoxicological results are often used in the case of toxic substances for which long-term, widespread field data are usually lacking. Examples of this method for deriving criteria include; copper (Carlson, Nelson and Hammermeister 1986); selenium (Van Derveer and Canton 1997); and several chemical entities in the South African Water Quality Guidelines for Aquatic Ecosystems (DWAF 1996). The results from ecotoxicological experiments are most commonly used for toxic or anthropogenic substances. The boundary values for various naturally occurring inorganic salts, as used in the RDM water quality assessment method for rivers, have also been derived from an international database of ecotoxicological data (DWAF 2002). Unfortunately in the case of South African wetlands, not enough is known about the biota living in them to use ecotoxicological data in the above manner. More

research on the distributions, tolerances and community structure of wetland biota is first required.

- Because water quality in aquatic ecosystems is variable, there is a movement internationally towards setting objectives that are risk-based rather than hazard-based (Claasen, Strydom, Murray, and Jooste 2001). Thus US EPA guidelines require that a stream segment to be listed as impaired when greater than 10% of the measurements of water quality exceed the given numeric criteria (Smith, Keying, Hughes and Shabman 2001).
- Once impacts due to pollution have occurred it is often difficult and expensive to rehabilitate wetlands (EPA 2002). Furthermore, it is important that site-specific water quality standards be developed (Peeters and Gardeniers 1998). For this reason it is important that water quality objectives are conservative and will not allow significant impacts to occur. In a similar vein, as emphasized in section 2.7.4, since sediments tend to accumulate in wetlands compared to rivers, there is a very real threat of sediment-associated pollutants such as phosphates or metals also accumulating (Fields 1992). Thus any standards that are developed need to be especially conservative with regard to pollutants that adsorb to sediments.

4.3.1 Water quality objectives and South African wetlands

In addition to the general considerations discussed above, in connection with setting water quality objectives for wetlands in South Africa the following points are of concern:

- Wetlands in South Africa and other arid regions are especially vulnerable to pollution because of the low dilution capacity (section 2.4.2). Furthermore, many wetlands, especially in the more arid parts of the country are endorheic and thus prone to accumulation of pollutants.
- Investigation of the water quality from impacted wetlands in the database collated during this project (section 3.4) has indicated that in some systems the levels of selected chemical constituents, and in particular inorganic salts are lower than in non-impacted systems. This is likely to be the effect of increased run-off in urban areas. Thus permitted *minimum* concentrations of some naturally-occurring chemical constituents as well as *maximum* concentrations need to also be considered, since such conditions are also likely to affect the species of biota that can live in such wetlands.

- Aquatic ecosystems are frequently linked within the landscape and perturbations upstream can affect downstream functioning. Thus it is necessary to consider the integrated hydrological system when considering water quality, and not just individual wetlands. Because of this (and from the point of view of efficiency of implementation) it was recognized that any method that was developed for setting the water quality component of the ecological Reserve for wetlands must be compatible with other relevant sections namely;
  - the water *quantity* Reserve for wetlands themselves, rivers, groundwater and estuaries.
  - the water *quality* Reserve for rivers, groundwater and estuaries.

4.4 A DECISION SUPPORT SYSTEM FOR SETTING WATER QUALITY OBJECTIVES FOR WETLANDS

4.4.1 Wetland water quality types

The Decision Support System developed to aid in setting water quality objectives is based primarily on the major source (by volume) of water supplying a wetland, in addition to the drainage pattern. Whether the water feeding a wetland originates from an underground spring, river floodwater or from overland run-off will obviously influence the type and concentration of chemical constituents in that water. In addition, the residence time of that water is also important since this affects evaporation and exchange processes between the water column and sediments. The drainage pattern of a wetland and specifically if it is endorheic (a "closed system") or exorheic (an "open system") is also likely to influence water quality. Exorheic systems tend to have a higher dilution capacity and thus the concentration of chemical constituents in such water bodies is likely to be lower than in endorheic systems.

From consideration of the above factors, it is proposed that for non-marine-influenced (athalassic) wetlands there are four basic wetland **water quality types**, namely:

- riverine systems;
- runoff-fed systems;
- groundwater-fed systems;
- freshwater lakes.

For a wetland to be classed as "riverine" in terms of water quality, the system should receive most of its water from a river, be exorheic and the retention time of that water be reasonably short. In such systems, evaporation is not likely to be excessive, and a reasonable approximation of water quality is to use the specifications for rivers. Some wetlands may be exorheic only part of the year and expert judgement would be needed to determine if the system should be most correctly classified as endo- or exorheic. Runoff-fed systems may be exorheic but are typically endorheic and represent the collection point of the run-off from the surrounding landscape. Thus run-off is the major water source. Groundwater-fed systems are those whose major water source is from springs, seepage, or a high-water table. They may be endorheic or exorheic. If the former, a conservative approach must be taken to setting water quality objectives. A reasonable manner to setting water quality objectives for such systems, until more site-specific information becomes available, is to extrapolate from the hydrogeochemical data. The suggested protocol for dealing with freshwater lakes is tentative. Depending on hydrology, depth (and thus stratification regimes) these systems can be very diverse and water quality objectives that are set need to be site specific. The question of setting realistic water quality objectives for lakes is discussed further on in this chapter.

4.4.2 The Decision Support System

The DSS (Figures 4.1 – 4.4) gives a structured method for determining water quality objectives for a wetland. Which specific method is chosen depends on the water quality type of the wetland under consideration (as discussed in the previous section). Nevertheless, the basic method is very similar for each water quality type and differs only in the details. Each water quality type is discussed below.

4.4.2.1 Riverine systems

A wetland that is classed as "riverine" in terms of water source should receive most of its water from a river, be exhorheic and the retention time of that water should be reasonably short. Figure 4.1 shows the proposed DSS for such wetlands.

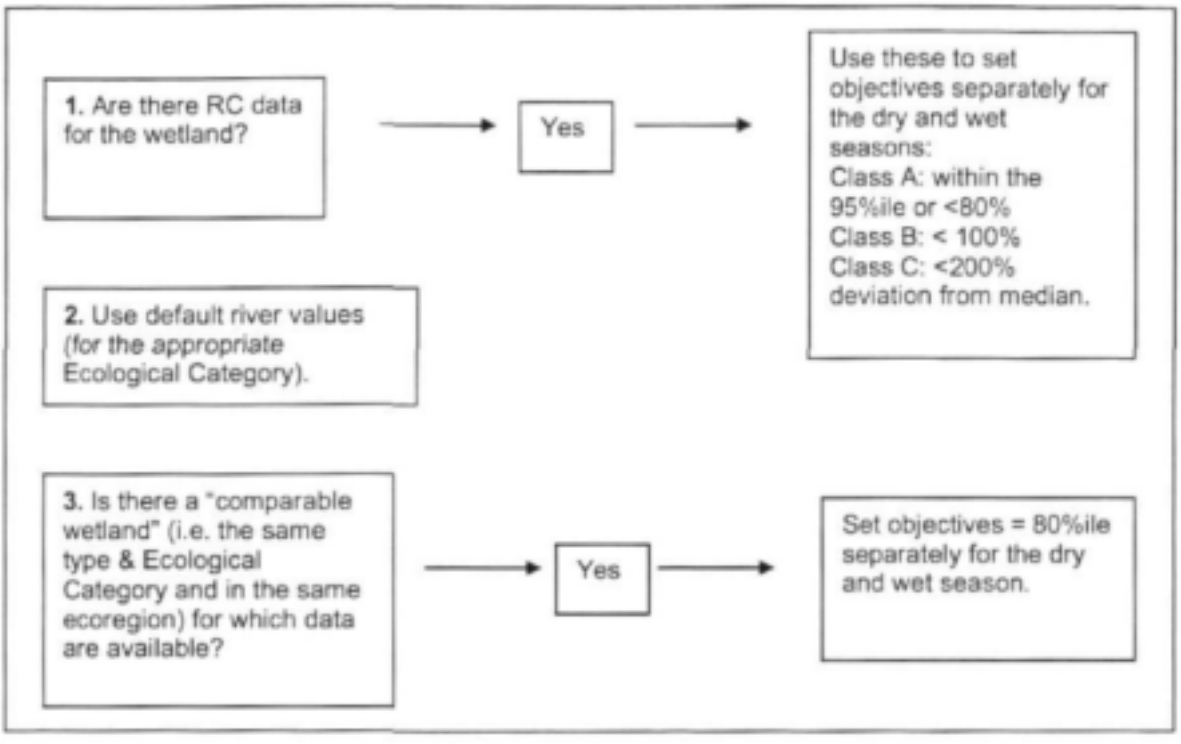


Figure 4.1 *The DSS for riverine wetlands*

Three different approaches can be used to set the water quality objectives. Which method (or methods) is/are actually employed depends on data availability. The first step is to see if any data are available which could be used indicate what the water quality of the wetland in the natural state should be. If there are Reference Condition (RC) data these can be used as the starting point from which objectives for the different Ecological Categories can be derived. If the wetland is to be managed as "more or less natural" (Category A), the concentrations of chemical constituents should not exceed either the 95%ile of the RC data, or the median value + 80% (whichever is the most conservative). For less protected wetlands managed in a lower Ecological Category, a greater deviation away from RC could be allowed. The median values for key water quality variables would be calculated separately for the dry and wet seasons. If a wetland is to be managed as Ecological Category = B class (i.e. slightly impacted), the objective for each variable should be set at 100% deviation from the seasonal median. This means that objective value should not be higher than the seasonal median RC value + 100%. *Similarly for Class C, a deviation greater than 200% should not be allowed.* For wetlands managed in a Class D condition, no values are given because it is considered

that given the current knowledge of water quality and wetlands in South Africa not enough is known to specify these values.

If no data for the wetland in question are available, the water quality objectives derived for rivers (termed "ecospecs") can be used (DWAF 2002). A third approach is to look for a "comparable" system. A check is made to see if water quality data are available from a nearby, similar wetland, that is of the same type and same level of impact for which the test wetland will be managed. If such data are available, they can be used to set the objectives. It is suggested that the 80%ile value for each month (or season) be calculated and used as the water quality objective. This value is the same as that recommended in the Australian and New Zealand Water Quality guidelines (ANZECC/ARMCANZ 2000), but needs to be verified for South African conditions.

4.4.2.2 Runoff-fed systems

In the case of wetlands that are largely fed by surface water, three approaches are suggested. The first two methods are the same as discussed for riverine water quality systems namely, extrapolation from the Reference Condition or extrapolation from a comparable site.

If none of the more site-specific approaches are applicable, then until more information becomes available, the "best-guess" default values should be used (section 4.5). Results obtained in this manner represent a low level of confidence and therefore this method should only be used if no water quality data are available. A monitoring programme would need to be implemented to ensure that water quality objectives were appropriate and to ensure compliance. This is especially true in the case of wetlands where the objectives have been derived using the generic default values.

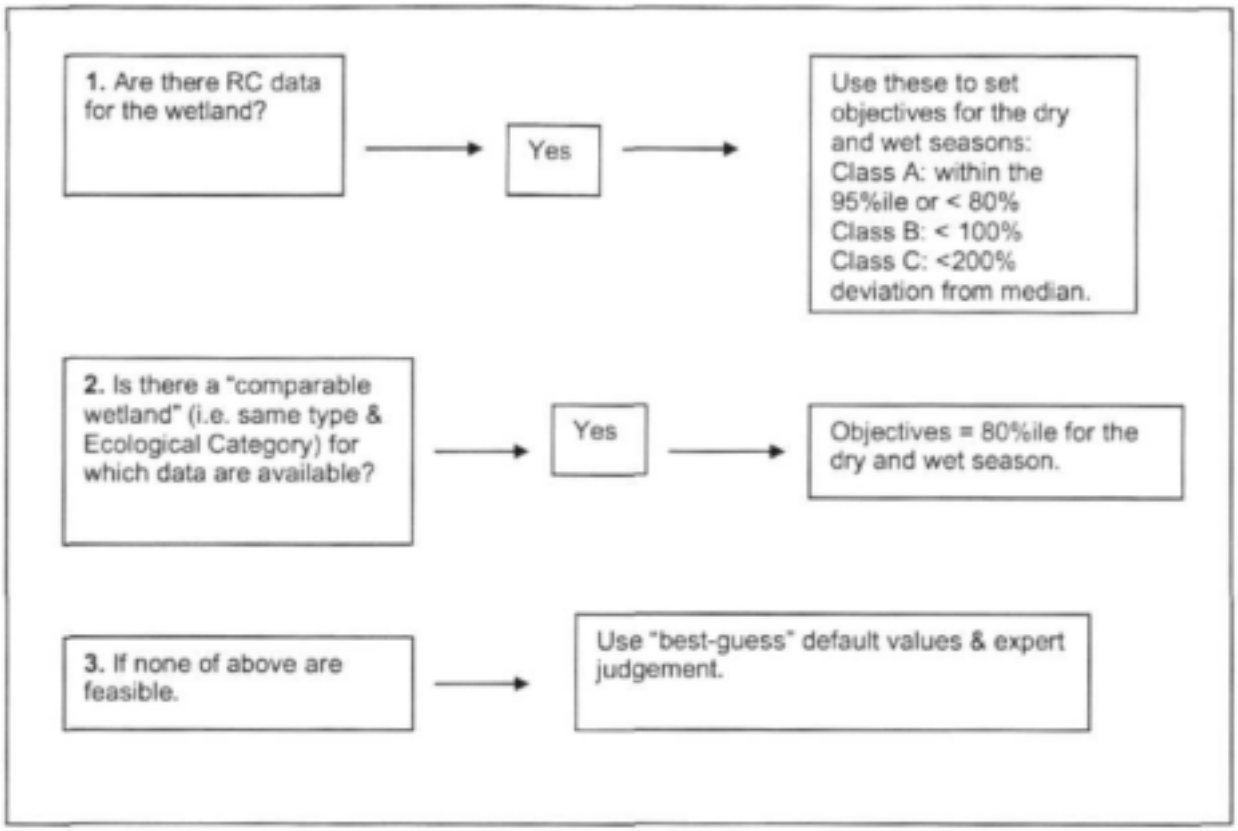


Figure 4.2 *The DSS for runoff-fed wetlands*

4.4.2.3 Groundwater-fed systems

In the case of wetlands that are largely fed by groundwater (due to springs, seepage, high-water table etc.) four different approaches can be used. In addition to the three methods discussed above for runoff-fed systems, it may be possible to extrapolate from the specifications for the groundwater Reserve. If the groundwater is judged to be un-impacted then the values of concentrations of chemical constituents can be considered to be indicative of the Reference Condition for the wetland. These values can then be used to extrapolate from the Reference Condition. If none of the above more site-specific approaches can be applied, then the default generic values should be used.

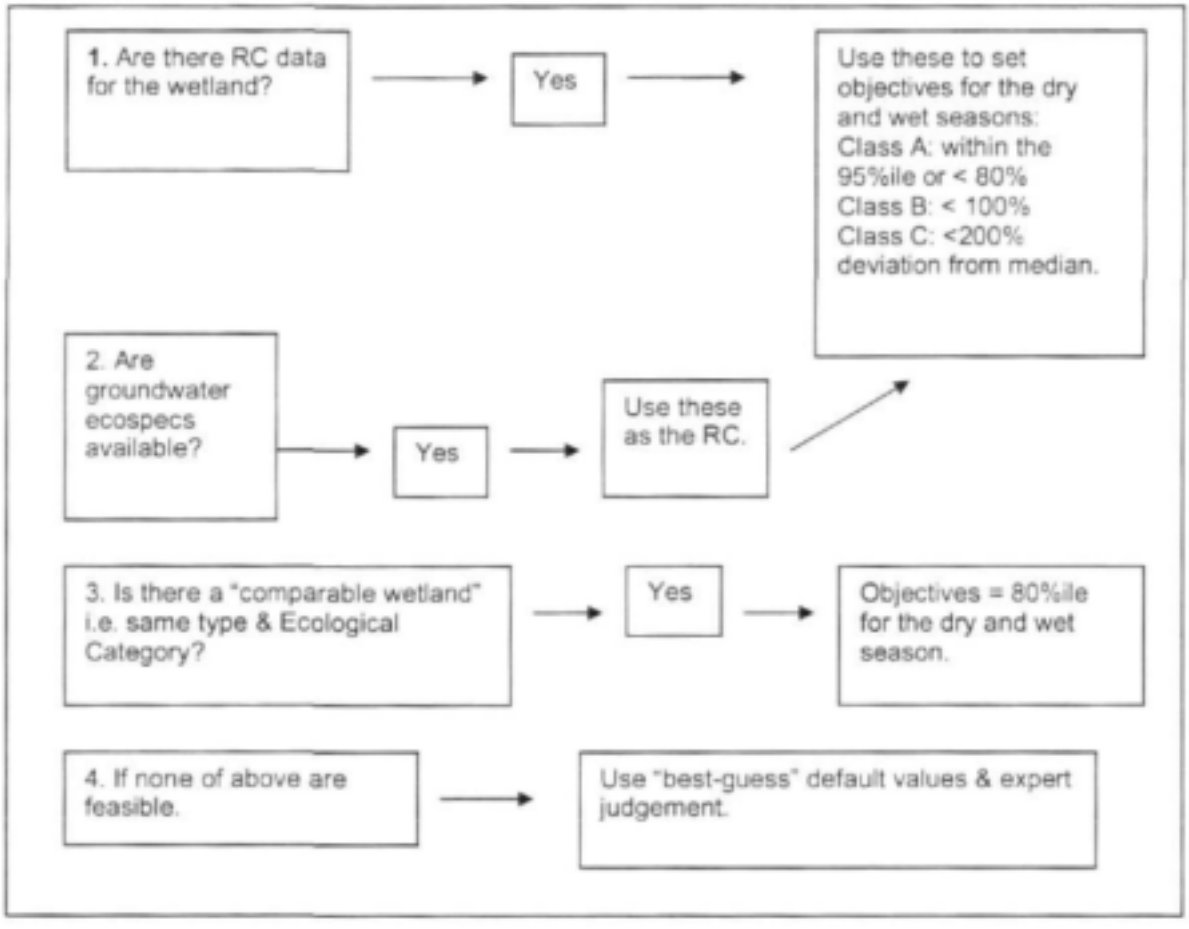


Figure 4.3 *The DSS for groundwater-fed wetlands*

4.4.2.4 Freshwater lakes

In the case of freshwater lakes it is felt that these systems should be treated in a site-specific manner. Hart (1995) concluded that with respect to the South African coastal lakes, it is difficult to generalize with regard to management and that this should be targeted to specified objectives. Therefore the use of maximum deviational values from the median are not recommended in the DSS for lakes. Thus, if the system is to be managed as "natural" (Class A), the water quality specifications can be set as the 95%ile of the RC data-set. If the Ecological Category is to be set lower (Class B, C, or D), however, a different approach would need to be used.

Management objectives could be derived by extrapolation from a comparable wetland. If this is not possible, until further research has been carried out on methods for setting water quality objectives for freshwater lakes, the default boundary values for rivers as defined in (DWAF 2002) should be used. These values should be modified using site-specific information where available. Such an approach was used in setting the water quality component of the ecological Reserve for the lakes of the Mhlatuze system (J. Rossouw, Ninham Shand, *pers. com.* Dec 2003).

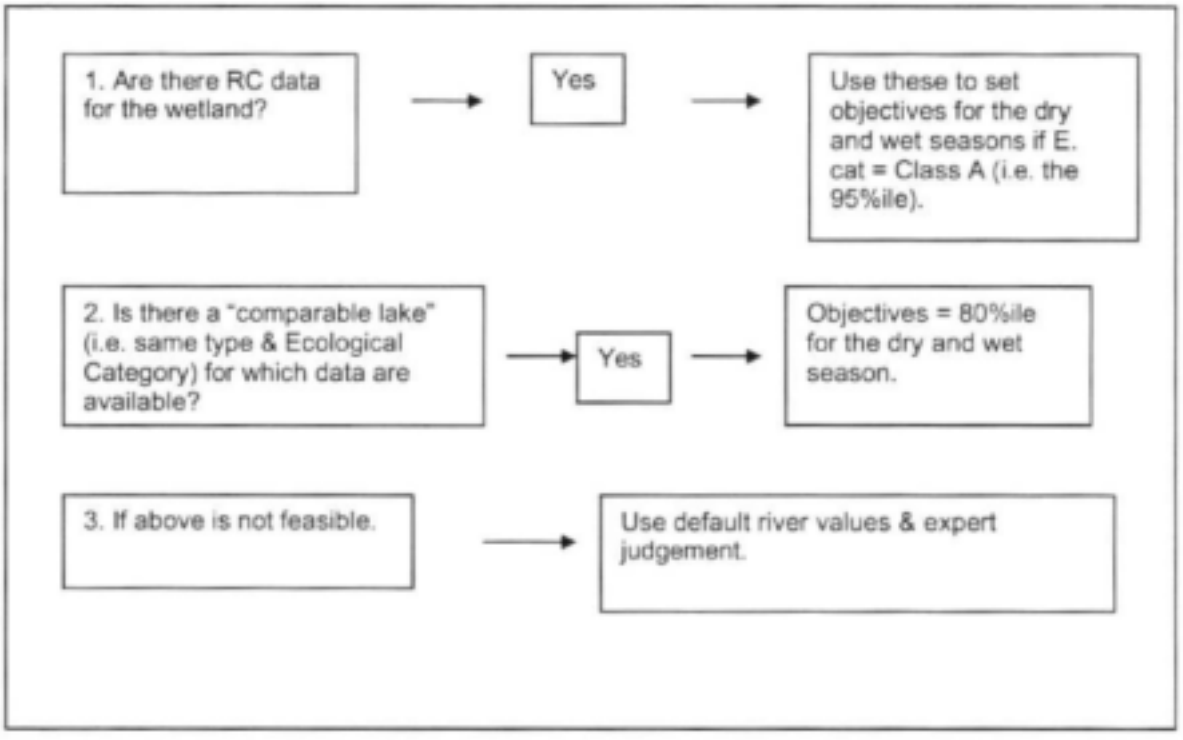


Figure 4.4 *The DSS for freshwater lakes.*

4.4.3 Points to note in the DSS for setting wetland water quality objectives

The following considerations have relevance to the DSS.

- Over and above the approaches suggested in the DSS, if there are site-specific data linking water quality and ecological response these should be used to inform the process of setting water quality objectives. For example, in some cases, the nutrient budget for an urban lake may have been quantified and the concentrations in the water column that are likely to trigger a cyanobacterial bloom known. Research has shown that lacustrine environments in South Africa have a threshold of approximately 0.035 mg/L total phosphorus beyond which

algal growth begins to increase dramatically (Harding, W. DH *pers .comm.* April 2004). In such cases this information can be used to adjust the water quality objectives for nutrients derived using the DSS.

- The maximum allowable deviation from the RC median values (e.g. Approach 1, Figure 4.1) were derived from examination of the water quality data for actual wetlands (Chapter 5). Because little is known about this stage about water quality in wetlands and the effects on the biota, these **values are tentative** and require further research. The values were derived from an analysis of the statistical distribution of water quality data for selected wetlands. They are not linked to ecological response and are therefore merely preliminary steps in the evolution of meaningful water quality objectives that will protect wetlands, but that will at the same time, allow a predefined level of utilisation. As a consequence of the uncertainty in setting water quality objectives, it is essential that once the water quality component of the Reserve is determined, a monitoring process be put in place. Based on the results of the monitoring programme the water quality objectives may need to be revised.
- In the case of ecologically sensitive and important wetlands that will be managed in a largely natural state (Category A) the DSS stipulates that the maximum allowable concentration should be set at the 95%ile of the RC (or less than the median+80%, whichever is the lower value) for the dry and wet season. These values are in line with the results found for the case-study wetlands (Chapter 5) The 95%ile is also used in the Australian and New Zealand Water Quality Guidelines (ANZECC/ARMCANZ 2000). It is considered that these objectives are stringent enough to protect wetlands of high value. Nevertheless more research and monitoring of water quality and ecological response is required in order to confirm this supposition.
- It will be seen from the results in Chapter 5 that water quality in wetlands varies temporally. The natural intra-annual variation in the concentration of key chemical constituents and values of physical variables is an important factor for maintaining biodiversity in aquatic ecosystems (Dallas and Day 1993) including wetlands (Coetzer 1987). For that reason in the DSS, the water quality objectives, where possible, should be derived separately for the wet and dry seasons.

- Narrative statements should be used, in conjunction with numerical water quality specifications to guard against specific adverse ecological outcomes. For example, it may be specified that nutrient levels be minimised (in conjunction with the provision of a suitable hydrological regime) in order to prevent excessive growth of *Typha* in a particular wetland.
- It was noted in section 4.3.2 that the *minimum* values of inorganic salts might need to be stipulated in some situations, because some biota are specially adapted to saline conditions and cannot tolerate freshwater. In the DSS at the moment, this concept is not formally incorporated because not enough is known about the physico-chemical requirements of wetland aquatic organisms in this country. If there is expert knowledge about tolerance ranges of biota for a given wetland, however, this information can be used to set the minimum allowable concentrations (and adjust the upper concentration limits if applicable). The topic of setting the lower limits of water quality objectives for wetlands specifically for salinity requires further research.
- The water quality variables considered in the DSS are the suite of variables recommended for Reserve determinations in rivers (DWAF 2002) and include; Total dissolved solids (TDS), individual salts, pH, dissolved oxygen, temperature, nutrients, toxins (pesticides, metals, organics) and sediments.
- In the case of toxic substances (pesticides, metal, organic compounds etc.) the values given in the South African Water Quality Guidelines for Aquatic Ecosystems should be used. These guidelines are for inland waters and were derived largely from data for rivers (DWAF 1996). Until wetland-specific guidelines have been developed, however, they should be used. Toxic substances that are likely to be adsorbed on to sediment particles should be treated conservatively.
- The proposed DSS is designed to be flexible and to use alternative approaches depending on the data that are available. This is necessary due to the lack of data and the poor understanding of water quality-ecological responses in most South African wetlands. In data-rich situations, where more than one method is possible, it is suggested that results be derived using all the possible approaches. These results can then be compared and evaluated using expert judgement.

- As noted previously, a conservative approach would also need to be used in the case of endorheic systems. Thus if the availability of data permits more than one approach to deriving the water quality objectives for a given wetland, they should all be used. The lowest calculated values of water quality objectives should then be employed to manage the wetland. Furthermore, it is suggested that such wetlands should not be managed at an Ecological Category lower than Class B (slightly impacted).
- The DSS is based on the major water source of a given wetland. Determining the water source is part of the field-work that accompanies high-confidence Reserves (section 2.5.4). For lower levels of Reserve determinations, i.e. desk-top estimations, however, it may not be possible to deduce the major water source. For this reason, the DSS may not be suitable for low-confidence ecological Reserve determinations.
- A possible problem with deriving objectives for groundwater-fed wetlands is that geochemical variation has not been well-studied in this country (Colvin *et al.* 2003). Although broad regional values of water quality variables are available for sub-surface water resources (Simonic 2003), this resolution may not be fine enough to be suitable for setting water quality objectives for surface-water ecosystems. Nevertheless it is felt that this is a useful approach that will become increasingly relevant as more becomes known about the hydrogeochemistry of groundwater resources in South Africa.
- Wetlands, as for most other aquatic ecosystems, are especially vulnerable to diffuse sources of pollutants. Vegetation buffer zones have been shown to be effective in prevention of chemicals, especially phosphates from entering into aquatic systems (Croke 2002). In order to limit the levels of pollutants that enter into wetlands, it is suggested that there should be more stringent enforcement of existing legislation (e.g. Conservation of Agricultural Resources Act – act 43 of 1983) protecting natural vegetation adjacent to wetlands.
- At the moment, it is suggested that for freshwater lakes, in the absence of a comparable wetland, the water quality ecospecifications (boundary values) that have been derived for rivers be employed (DWAF 2002). From a limnological point of view, lakes are, however, very different to rivers and processes, such as stratification frequently influence water quality in the former (Hart 1995), but

rarely in the latter. Eventually, site-specific water quality ecospecifications for individual lakes would need to be developed.

- The DSS is not suitable for wetlands that are known to have a strong marine influence (e.g. estuaries, lagoons).
- The hydrological status of a wetland at a given point in time has a great effect on the water quality status; the two are intimately linked. It is recommended that the discharge of effluent, or the abstraction of water should not change the hydrological nature of the wetland. i.e. seasonal wetlands should not become perennial and perennial wetlands should not become seasonal or temporary. The question of linking hydrological requirements of wetlands with water quality requirements is discussed further in section 4.6.

4.4.4 The Reserve determination method and the DSS

Setting the water quality objectives for an aquatic ecosystem is a sub-section of the larger process of determining the ecological Reserve. In section 2.5.4 the Reserve determination process was described in detail and a flow diagram of the procedure given (Figure 2.3). Consequently, only a brief description is given here with the major emphasis on how the DSS could fit into the overall Reserve determination process.

Step 1: Definition of the study area and Step 2: Determine ecoregional type

The Reserve determination starts with delineating the study area (this may be an individual wetland, or the wetland(s) may be part of a larger system). The ecoregions into which the study area falls is determined (Kleynhans and Hill 1999) and the water resource is divided into units (water quality units) which would be expected to be more or less homogeneous with regard to water quality. Sites that are representative of each unit ("IFR sites") are selected. The DSS would normally be applied to individual wetlands, if the wetland is large, and the water quality data are detailed enough, it could be applied to different water quality regions within a single wetland system. Information and data concerning the water quality of the system is collated, including land-use maps, chemical and biological information. The project team would collect samples and data for other aspects of the wetland, including hydrology and the water source, vegetation, fish, social importance. Information on the water source and drainage pattern is crucial for the DSS.

Step 3: Determine the Reference Condition

The next step is to establish the expected Reference Condition water quality. This can be potentially deduced from several sources of data – which may, or may not be available. These are:

- Historical data for the wetland in question (i.e. prior to present-day impacts).
- From a nearby, minimally impacted wetland of the same type (a "reference wetland"). This maybe either historical or present-state data.
- In the case of groundwater-fed wetlands from regional geochemical concentrations.
- In the case of riverine wetlands fed by water from, from values derived as part of the water quality Reserve for these ecosystems (DWAF 2002).

Whichever source is used, the data need to be scrutinized to ensure that within the limited knowledge we have of water quality in South African wetlands, they do represent the Reference Condition. This is likely to be a difficult and inexact process until more research has been carried out, especially because there is no biomonitoring method for wetlands in South Africa (section 4.6). An examination of landuse in the catchment, and the presence/absence of point sources of effluents can be useful in this regard.

Step 4. Determine the Present Ecological State

The next task is to make an assessment of the present status of (amongst other factors) the water quality of the wetland, and hence determination of whether there has been an impact on the system. Frequently, no monitoring data will be available and only a few measurements will be made during the course of the Reserve determination project. Currently, the degree of impact - for water quality, but mainly for other aspects of wetland ecological status, is assessed qualitatively using a scoring system. The present day status may either be natural (Category A), slightly modified (Category B) down to severely impacted (Category E or F). Because of the lack of links between water chemistry and resultant wetland health it is not possible at the moment to accurately specify what class a wetland is in with regard to water quality alone.

Step 5. Determine the Ecological Category

Using a similar scoring system the Ecological Importance and Sensitivity of a wetland (as well as the rest of the aquatic system) is assessed and in conjunction with the

present ecological status, is used to determine the Ecological Category. The Ecological Category represents the minimum acceptable level of ecological functioning for which the system will be managed and is a crucial consideration when deriving water quality objectives. The Ecological Category may be similar to the Present Ecological State, or it may be lower (and thus represent a reduced level of ecosystem health). If the system is currently in a poor status ecologically, a higher Ecological Category may be recommended than the Present Ecological State, in which case rehabilitation measures would need to be implemented, and water quality would need to be improved.

Step 6. Set the Reserve

The final task is to set the wetland Reserve water quality component i.e. to recommend water quality objectives. It is in this part of the process, using information obtained from the previous steps that the DSS is used.

4.5 DEVELOPMENT OF THE "BEST GUESS" DEFAULT VALUES

4.5.1 Introduction

It is recognized in the DSS that for some runoff-fed or groundwater-fed wetlands there may be no water quality data available, for either the wetland under consideration, or for a comparable system. In such situations none of the site-specific approaches are feasible, and as a last resort, default ("best-guess") values would have to be used. Such values have been derived as part of this project and are presented and discussed in this section.

Several issues needed to be considered when deriving the "best-guess" default values. For example, the results from interrogation of the database (section 3.4) as well as investigation of individual case-study wetlands (Chapter 5) showed that there are differences with regard to characteristic water quality depending on the:

- Wetland type
- Climatic region
- Season of the year (and hence hydrological status).

Consideration was given as to whether different default values should be assigned for these different situations. For example, objectives for nutrient concentrations in the USA

are assigned depending on the ecoregion (Table 2.4) and in Australia, the default values for wetlands are based upon the climatic region (Table 2.5). In the case of wetland type, lakes are frequently specified separately from rivers, or as in the Australian situation, criteria are given separately for rivers, lakes and wetlands (the latter incorporating palustrine systems and endorheic pans). Furthermore, there are often distinct intra-annual variations in the water quality exhibited by non-impacted wetlands (Chapter 5) and thus separate default values should optimally be set for the dry and wet seasons. A further consideration is that wetlands in South Africa will be managed for different levels of ecological functioning (i.e. as different Ecological categories). Thus different default water quality objectives might be set for these different levels of protection.

After considering the above issues and taking into account the current lack of data, and ecological knowledge, it was decided by the project team that differentiating the default values in terms of climatic region, Ecological Category, or wetland type, in a meaningful way, is too ambitious at this stage. More information is needed about the physico-chemical conditions in South African wetlands before a finer resolution can be given to the default values. This is, however, a goal towards which research efforts should be urgently directed. The "best-guess" default values (Table 4.1) are generic water quality objectives applied to wetlands that fall into the ground-water and runoff-fed water quality type categories (Figures 4.2 and 4.3). At this stage they should be used across the entire country and for all seasons. The "best-guess" default values are so named in order to emphasize that they are at this stage only informed estimates. To be pragmatic, this preliminary version of the "best-guess" default values also does not differentiate with regard to the Ecological Category for which the wetland under consideration will be managed. The values given as water quality objectives are considered to be fairly conservative and should maintain systems in a "slightly impacted" state. It should be emphasized again, however, that until the knowledge base concerning wetland water quality and ecological response is increased, the future ecological status of a wetland under a given water quality regime cannot be predicted with any certainty.

4.5.2 How the "best-guess" default values were derived

The "best-guess" default values, presented in Table 4.1, were derived as follows:

1. The numerical values for water quality standards used by other countries (Tables 3.4 - 3.6) and published in the literature (Hart, Campbell, Angehrn-Bettinazzi *et al.* 1993; Liston and Maher 1997) were critically examined. In some cases an explanation was included in the literature of how these values were derived, and this information was studied for relevance to the South African situation.
2. The values found in the international arena were compared with the results obtained from interrogation of the Wetland Water Quality Database (Tables 3.4 – 3.15) and thus from measurements obtained from South African wetlands.
3. Using expert judgement and the information from the sources described above, maximum values (the water quality objectives) were assigned for each key water quality variable.
4. These values were circulated amongst wetland specialists in South Africa for comment and were subsequently modified where appropriate.

4.5.3 How the DSS and "best-guess" default water quality values should be used

The "best-guess" default values are shown in Table 4.1. Before using these values to set the water quality specifications for a given wetland, the Decision Support System should be applied. Thus, as the minimum requirement, at least one site visit will have been carried out (and preferably more, over several months or seasons), and a spot-value of water quality variables taken. From these measurements and from information gathered during site visits (used to establish the present ecological state and the ecological importance and sensitivity), several features can be deduced:

1. The continuum of salinities exhibited by wetlands can be divided into four ecological "bands" (Williams 1985).
   - Hypersaline systems: > 40 g/l
   - Saline systems: 40 – 12 g/l
   - Brackish systems: 2 – 12 g/l
   - Freshwater systems < 2 g/l

Measurements of salinity, in addition to ecological clues such as the presence of brine shrimps (*Artemia* spp. – indicate a wetland is saline) can help to establish what broad salinity type the wetland is, and thus the upper water quality objective for salinity. The actual boundary values are, however, imprecise and expert judgement should be used if

measurements indicate that the salinity of a wetland is borderline between two ecological bands.

2. The broad pH range into which the wetland falls should be established. In other words whether the system is:
    - Circum-neutral.
    - Naturally acidic.
    - Naturally alkali.

In the case of circum-neutral systems, the pH should be maintained between pH 6 -8. For naturally acidic or alkali systems, the pH should not be allowed to deviate more than 1 pH unit from the measured value. In saline/hypersaline systems, it is considered that pH is not an important ecological driver and thus this water quality variable is not critical.

Table 4.1 *"Best-guess" default water quality objectives for wetlands.*

Water quality variable	Value	Comments
Salinity	Hypersaline systems: > 40 g/l Saline systems: 40 – 12 g/l Brackish systems: 2 – 12 g/l Freshwater systems < 2 g/l	Must remain within the salinity band. Systems considered as fresh must not exceed deviation of 100% from the spot value.
pH	Circum-neutral systems: 6-8 Naturally very acidic: < 1 pH unit Naturally very alkali: < 1 pH unit deviation from median	In saline/hypersaline systems, pH is not a driver (i.e. not critical). In systems with low salinity/conductivity, it is important to maintain pH in specialized systems (i.e. naturally acidic/alkali).
Phosphate (SRP)	0.08 mg/L	
Total inorganic nitrogen	0.8 mg/L	

Effects on the biota due to the high concentration of salts are more important than pH in such systems. In systems with low salinity/conductivity, however, it is important to maintain pH in specialized systems - those that are naturally acidic or alkali. Results from interrogation of the Wetland Water Quality Database (section 3.4.4.2) show that most un-impacted wetlands in South Africa tend to be circum-neutral.

Ecological clues such as humic-acid stained waters may indicate that a system is naturally acid. The presence of stoneworts (*Chara* spp.) is often indicative of alkaline conditions.

3. Water quality objectives for nutrients should be set according to Table 4.1.

The project team would like to emphasize that the "best-guess" default values are preliminary water quality objectives, and should be used only when no site-specific data are available.

Points to note.

- As noted previously, the values are considered to be moderately conservative. Because of the uncertainty associated with specifying numerical values with no site-specific information, for all regions and for all wetland types, the same values are applicable to all Ecological categories.
- As noted earlier in section 4.4.3, a monitoring programme should be put in place to establish the water quality status of the wetland and to identify the trends in key water quality variables with time. This is especially in the case of wetlands where water quality objectives have been derived using the default values.
- Default values have not been set for dissolved oxygen, because not enough is known about the range in values naturally exhibited by this factor in wetlands. It is known, however, that values can be very low (Maltby *et al.* 1994; Batchelor *et al.* 2002). Similarly, very little data are available for turbidity, or suspended sediments, and thus it was not possible to set default values for this variable. Therefore, should an application be made to discharge an effluent high in dissolved organic carbon or likely to result in high turbidity, site-specific investigations would need to be performed.

- In the case of toxic substances, the values specified in the South African Water Quality Guidelines should be applied.
- Water quality objectives for nutrient values are expressed as concentrations in the default table. It may be more ecologically meaningful to express these as loads per unit time (Harding, W. DH *pers .com.* April 2004). This is especially considering that phosphate, in particular, tends to accumulate in sediments which can, under certain circumstances, be re-suspended. Attention should be directed towards investigating the quantitative relationship between loads and concentrations of nutrients and ecological effects (for example nutrient blooms) in South African wetlands. This information should then be used to refine the default values. Ultimately, the management strategy for a wetland should be underpinned by an understanding of the pollutant loads and the resultant concentration in the water body.

4.6 LINKING WATER QUANTITY, WATER QUALITY AND ECOLOGICAL RESPONSE IN WETLANDS

4.6.1 Water quality and ecological response

Section 2.5.5.1 described the default boundary values (delimiting each Ecological Category) as used in the revised water quality Reserve determination procedure for rivers (DWAF 2002). These values are based on extensive data-sets, since for some rivers water quality data go as far back as the early 1970's and samples have been collected each season, each month or sometimes even more frequently. Compared to many countries in the world, the South African water quality data-set for rivers is extensive. Understanding of how these ecosystems function (brought about to some extent by the National Water Act and the need to determine the ecological Reserve), whilst not exhaustive, has developed to a reasonable level. In addition, there are well established biomonitoring methods for rivers, such as SASS (Chutter 1998) and other biotic indices of ecological integrity (e.g. Fish Assemblage Integrity index, Riparian Vegetation index; River Health Programme 2003). Consequently, in the case of rivers, links can be made between water quality and predicted ecological response with some degree of confidence (Malan and Day 2002b & 2003).

This in turn, facilitates the following tasks which are crucial to setting meaningful water quality objectives:

- the water quality and ecological condition characteristic of the Reference condition can be identified;
- the current ecological condition (the "Present Ecological state") of a river stretch can be assessed;
- the ecological consequences that are likely to arise from a given water quality scenario can be predicted with some degree of certainty.

In the case of wetlands in South Africa, there are limited water quality data and very little work has been carried out concerning the abiotic and biotic conditions in our wetlands. At the moment the methods for assessing wetland condition in general (e.g. the method used in the RDM manual – DWAF 1999b) are rather crude, and there are no verified biomonitoring techniques for assessing wetland water quality in this country. According to Hart, Maher and Lawrence (1999) effective management of ecosystems, and in particular the development of numerical biocriteria (Robb 1992), requires a good understanding of how that system works. It is therefore essential that future research efforts are directed towards understanding wetland ecological functioning. The lack of assessment methods for wetlands in South Africa (using biological data, or possibly other indices) has been recognized as a major constraint to effective management, and moves are afoot to remedy this situation. The Department of Water Affairs and Forestry have commissioned research to address this problem (B. Madikizela, DWAF, *pers. com.* May 2004). In addition, the Water Research Commission has included assessment techniques as a key thrust in their recently-established wetland research programme (S. Mitchell, WRC, Pretoria, *pers. com.* 2004).

Wetland specialists are usually able to identify wetlands that are in a reasonably natural ecological condition, and similarly can recognize those that are highly impacted. It is the graduations of ecological condition between these two extremes that are difficult to assess. In particular, linking Ecological Categories or levels of ecological functioning, with water quality objectives in a sensible manner is very difficult in the absence of a thorough understanding of the abiotic stressor and the biotic response relationships. Consideration should be given to making the Ecological Categories for wetlands broader and less refined.

For example, three (rather than 4/6) categories could be recognized.

- Largely natural
- Moderately impacted
- Highly modified.

These classes could be refined at a later stage when the scientific knowledge-base is enlarged. A similar approach is used to define aquatic ecosystems in the Australian and New Zealand Water Quality Guidelines (ANZECC/ARMCANZ 2000).

4.6.2 Water quantity and water quality

It was recognized by Malan and Day (2002b) that there is a chain of cause and effect in rivers between flow, water quality and biotic response. Wetlands are similar in that water quantity (i.e. the hydrological status of a wetland – the extent of inundation) is likely to have a major influence on water quality, and this in turn will affect the biota. The relationship between concentration of a given chemical constituent and hydrology (as shown by season) is evident from examination of the water quality data for individual wetlands (e.g. Figure 5.3). When compiling the Wetland Water Quality Database scope was provided for recording the degree of inundation of a wetland at the time of sampling. It was found, however, that very few records measured water depth when taking water quality samples. Furthermore water depth needs to be linked to concentration of chemical constituents or the values of physical variables on a site-specific basis. For example, the relationship between water depth and hence water volume in an endorheic pan, would depend on the topography of the particular system.

If the quantity of water flowing into a wetland is decreased, particularly if the pollutant loads entering into a wetland remain the same, the water quality of the wetland may well deteriorate. Thus it is important that when determining the water quantity requirements of a wetland (i.e. setting the water quantity Reserve) quantitative predictions are made as to what water quality conditions will result. It then needs to be established if the water quality objectives will be met under the current pollution loading. Linking water quantity and water quality in wetlands has not been addressed explicitly in this project. To some extent the issue has been addressed implicitly in that water quality objectives are set separately for the dry and wet seasons.

Thus season is used as a surrogate of hydrological condition. The entire question of linking water quantity and water quality requirements in wetlands is another important topic of research that requires attention.

4.7 CONCLUSION

For effective protection of water quality in wetlands it is essential that reasonably stringent pollution limits be set because wetlands are frequently sinks and accumulate material. At the same time, to enable optimum utilization of those systems for economic development, the water quality objectives should not be too conservative. A method for determining the optimum level of protection/utilization for wetlands with regard to water quality was presented in this chapter. These objectives are, at best, estimates. At the same time these ecosystems are already under considerable pressure with regard to development, and by the time more is known about our wetlands, many might have been irretrievably damaged or lost if steps are not taken to protect them. Despite the dangers of enforcing water quality objectives that may be too lenient (or alternatively too restrictive) it is felt by the project team that the DSS presented here is at least a step in the right direction. More research is needed to refine and develop the methods for setting the water quality Reserve for wetlands. Finally, it should be remembered that the setting of water quality objectives is only part of the process. Those objectives need to be implemented and the ecological effects monitored and evaluated in order to check that the desired outcome is attained.

CHAPTER 5
APPLICATION OF THE DECISION SUPPORT SYSTEM FOR SETTING THE WATER QUALITY ECOLOGICAL RESERVE FOR WETLANDS

5.1 INTRODUCTION

This chapter discusses the results from application of the Decision Support System (DSS) developed in Chapter 4 to several existing wetland systems. The DSS was designed to provide a protocol of steps to be followed when determining the water quality component of the ecological Reserve for a given wetland. The results from application to three different wetland systems are reported, namely:

- Nylsvlei (an ephemeral flood-plain wetland in the summer rainfall region).
- Rocher Pan (a seasonal endorheic system in the winter rainfall region).
- Verlorenvlei (a coastal lake and riverine wetland system in the arid area of the winter rainfall region).

The objective of this chapter was to identify any problems or inconsistencies when applying the DSS and to examine the results obtained. Development of the DSS and its application to real wetlands was carried out in an iterative manner. In particular the percentage deviation from the median that represents the boundary value between Ecological Categories was refined after examination of the statistical distribution of water quality variables.

5.2 APPLICATION OF THE DSS TO NYLSVLEI

5.2.1 Site description

The Nyl River floodplain is located approximately 140 km north-east of Pretoria and a short distance from the town of Nylstroom. It is a Ramsar site and is considered to be one of southern Africa's conservation, eco-tourism and agricultural assets (Higgins, Coetzee, Marneweck *et al.* 1996). Despite this, the system is under considerable threat from up-stream water abstraction (Higgins, Rogers and Kemper 1997) and effluent from mines and from sewage treatment works (STWs).

According to Higgins *et al.* (1996), the system falls into the summer rainfall region with most precipitation occurring from October to April. Average annual rainfall is 500-600 mm per year with an average of 600-700 mm in the main catchment area of the Waterberg. The vegetation is typical bushveld and savanna, and the soils of the region are moderate to deep, flat and clayey (Midgeley, Pitman and Middleton 1994). Because of the clay deposits, the wetland base is almost impermeable, so that ground-water is not an important water source in the hydrology of this system (K. Rogers, CWE, University of the Witwatersrand, *pers. com.* May 2004).

Nylsvlei is a system in which water usually flows (for 7 out of 10 years) in one or more channels. The wetland areas are ephemeral, and from time-to-time (although not every year), the river comes down in flood, inundating the flood-plains and oxbow lakes (Higgins *et al.* 1996; Higgins *et al.* 1997). Tarboton (1987) reported that the Nyl River had flooded 10 times in the previous 25 years, and that at maximum flooding *approximately 16 000ha of land were inundated. Recently, a WRC-funded project* (K5/1258: "A biophysical framework for the sustainable management of wetlands in the Northern Province, with Nylsvley as a reference model") was conducted to gather further understanding of the ecological functioning of the system (W. Vlok, University of the North, *pers. com.* 2004). A hydrologic and hydraulic model of the floodplain system has also recently been completed (Birkhead, James and Kleynhans 2004).

5.2.2 The Decision Support System

Of the four water quality types discussed in section 4.4.1 and from the discussion above, Nylsvlei wetland can be considered to be a "riverine" system. The DSS for a riverine wetland system was presented in Figure 4.1. From the figure, it can be seen that there are three different approaches that can be used for setting the water quality objectives for this wetland depending on the data available. These are using:

- The Reference condition, and/or
- Default river values, and/or
- Making use of a comparable wetland.

Of the above approaches, the last is not feasible because the Nylsvlei wetland system is unique. Large floodplain wetlands are not common in this country (Walmsley 1988), and the nearest, or most comparable is the Pongola floodplain system, which is in a different

ecoregion and rainfall area. Therefore only the first and second approaches were pursued further.

5.2.3 Approach 1: Deviation from the Reference Condition

Apart from the data gathered by the Department of Water Affairs and Forestry (DWAF) virtually no historical water quality data for the Nylsvlei wetlands themselves are available (W. Vlok, University of the North, *pers. com.* 2004). The DWAF, however, do have several gauging and water quality monitoring stations in the Nyl river catchment. Three monitoring sites (although there are several more in the catchment) were considered from their geographical location to be the most promising for inferring Reference Condition water quality. These were:

- A6H002-Q01: Nyl River at Deelkraal
- A6H011-Q01: Great Nyl River at Modderpoort
- A6H012-Q01: Olifantspruit at Olifantspoort

The Nyl River site at Deelkraal was found to be a useful monitoring station, since A6H002-Q01 is only (approximately) 10 km upstream of the wetland nature Reserve. The water quality of this site can therefore be considered to be more-or-less representative of the water quality for the wetland. This site is downstream of the town of Nylstroom and therefore impacts from the town are likely to be reflected in the water quality. The data-set for this site goes back to 1983, but unfortunately ends in 1991. There are almost weekly readings for conductivity, and monthly or bimonthly readings for ammonium, and ortho-phosphate. The values given for combined nitrate and nitrite were irregular and very low (mostly recorded as below 0.04 mg/L), and no measurements of total phosphorus are recorded.

Figure 5.1 *Trends in selected water quality variables at A6H002 (Nyl River at Deelkraal) with time.*

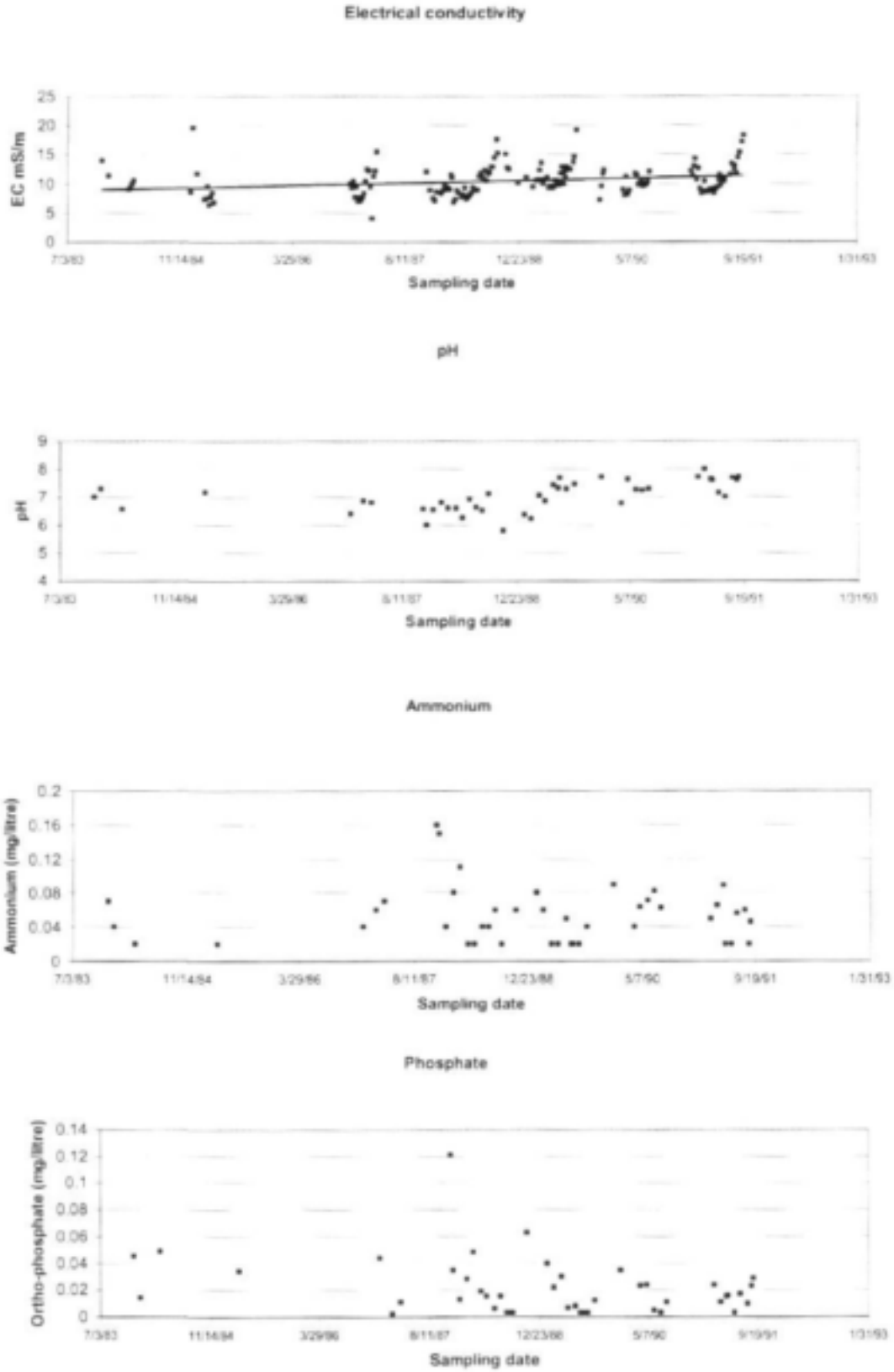


Figure 5.1 shows the trend with time in selected water quality variables at Deelkraal (A6H002-Q01) just upstream of the Nylsvlei Reserve. Values of conductivity are relatively low (annual median of 10 mS/m) and there is only a slight increase in conductivity over the entire time-series. The median value for phosphate over the entire data-set is 0.015 mg/L. The change in ammonium concentration from 1983 to 1991 is also shown in the figure. Neither phosphate nor ammonium, appear to have increased significantly. The only water quality variable that seems to have changed with time is pH, and this is most likely an artefact due to the change in the method used by DWAF for measuring pH around the end of the 1980s.

The data for the other two water quality monitoring sites (A6H011-Q01 on the Great Nyl River and A6H012-Q01 on the Olifantspruit) were also examined. No significant increases in time in any of the water quality variables of concern (pH, conductivity, nitrate and nitrite, ammonium, sulphate) were found (data not shown). The median values for each site for the entire time-period are given in Table 5.1. Also shown are details of the sample size (number of records, *n*) and the time-period over which sampling was carried out. Because the value of *n* that is recorded was frequently not the same for all water quality variables, the smallest (usually for temperature or ammonium) and largest (usually for pH, conductivity) are given.

It can be seen that, with regard to nitrate and nitrite, ammonium, phosphate, and sulphate, the instream values are all relatively low. Furthermore, there is generally no significant difference between the median values at the different sites. With regard to conductivity, the median value of this constituent is higher at A6H002-Q01 compared to the other two sites. This slightly elevated value may be a consequence of the fact that the other two sites are in the headwaters of streams, whereas A6H002-Q01 is further down the system. The higher value may also be due to the fact that, as mentioned previously, the town of Nylstroom is upstream of this site. It is surprising in this regard that nutrient levels are also not elevated at A6H002-Q01, however, see the analysis of more recent data (below).

From consideration of the above factors it was decided to use the data-set for A6H002-Q01 as the Reference Condition for all variables including conductivity. Water entering the floodplain area can be expected to increase in conductivity (as well as other

chemical constituents) on contact with the soil (Kingsford 2000). Thus, the values for conductivity exhibited by A6H002-Q01 are considered to be in line with values expected for the wetland.

Table 5.1 *Median values of selected water quality variables from three DWAF monitoring stations (considered to be Reference Sites) in the Nyl catchment. The time-period represented by the data is shown and the sample size (n).*

Monitoring station	EC mS/m	pH	Nitrate +nitrite mg/L	Ammonium mg/L	Phosphate mg/L	Sulphate mg/L	Temp °C	Time-period
A6H002-Q01	10.1	7.06	0.02	0.05	0.015	2.0	18.2	11/83 – 9/91 n = 43 - 157
A6H011-Q01	4.7	6.88	0.07	0.02	0.01	2.4	20	2/72 – 11/03 n = 352 - 485
A6H012-Q01	6.0	7.14	0.02	0.06	0.011	2.8	18.1	2/72 – 5/03 n = 106 - 443

5.2.3.1 The Present Ecological State for water quality

As part of the recently completed WRC-funded study on the Nylsvlei system, water quality data were gathered (R. Greenfield, RAU, *pers. com.* May 2004). A variety of water quality variables were measured, including microbiological parameters and metals, in addition to the normal system variables and nutrients. Several sites in the catchment were monitored over a period of almost two years. The results for the Nylsvlei Reserve are summarised in Table 5.2. A comparison of Table 5.1 (the data for A6H002-Q01) and 5.2 shows that, with regard to pH and sulphate, the median values are very similar. Thus there has been little change in these water quality variables over the past 15 years or so. Median conductivity values have increased slightly (from 10 to 13 mS/m), as has temperature. The most marked increase, however, has been in the level of nutrients. Both the levels of phosphate and combined nitrate and nitrite have increased ten-fold.

The increased level of nutrients (and conductivity) is likely to be a result of effluent from the STW at Nylstroom. Caution should be applied when interpreting the data since the present state is inferred from data collected in the wetland Reserve area itself, whereas the data used to infer the Reference condition are from the Nyl River, several kilometres

upstream. As mentioned previously, water from the wetland may be naturally higher in chemical constituents. Water temperature is highly dependent on hydrology (especially water depth) and so it is difficult to interpret the change for this variable without knowing the hydrological conditions during sampling.

Table 5.2 *Median values of selected water quality variables representing the Present State for water quality for Nylsvlei. The time-period represented by the data is shown and the sample size (n). Data provided by R. Greenfield, RAU, June 2004.*

EC (mS/M)	pH	Dissolved Oxygen (mg/L)	TDS (mg/L)	Temperature °C	Time-period
13	7.04	5.4	65.3	22.9	1/4/01 – 3/1/03 n = 7
Nitrate +nitrite (mg/L)	**Phosphate (mg/L)**	**Sulphate (mg/L)**	**Ammonium (mg/L)**		**Time-period**
0.2	0.2	5	No data		1/4/01 – 1/7/02 n = 5

5.2.3.2 Derivation of water quality objectives

The Reference Condition water quality was derived from DWAF monitoring station A6H002-Q01 (Nyl River at Deelkraal). The entire data-set was divided into wet season and dry season for each water quality variable. From the monthly rainfall distribution (Higgins *et al.* 1996), the seasons were differentiated as follows:

- Wet season: October - March
- Dry season: April – September.

Figure 5.2 *Seasonal variation in water quality variables for the Nyl River at Deelkraal (A6H002-Q01)*

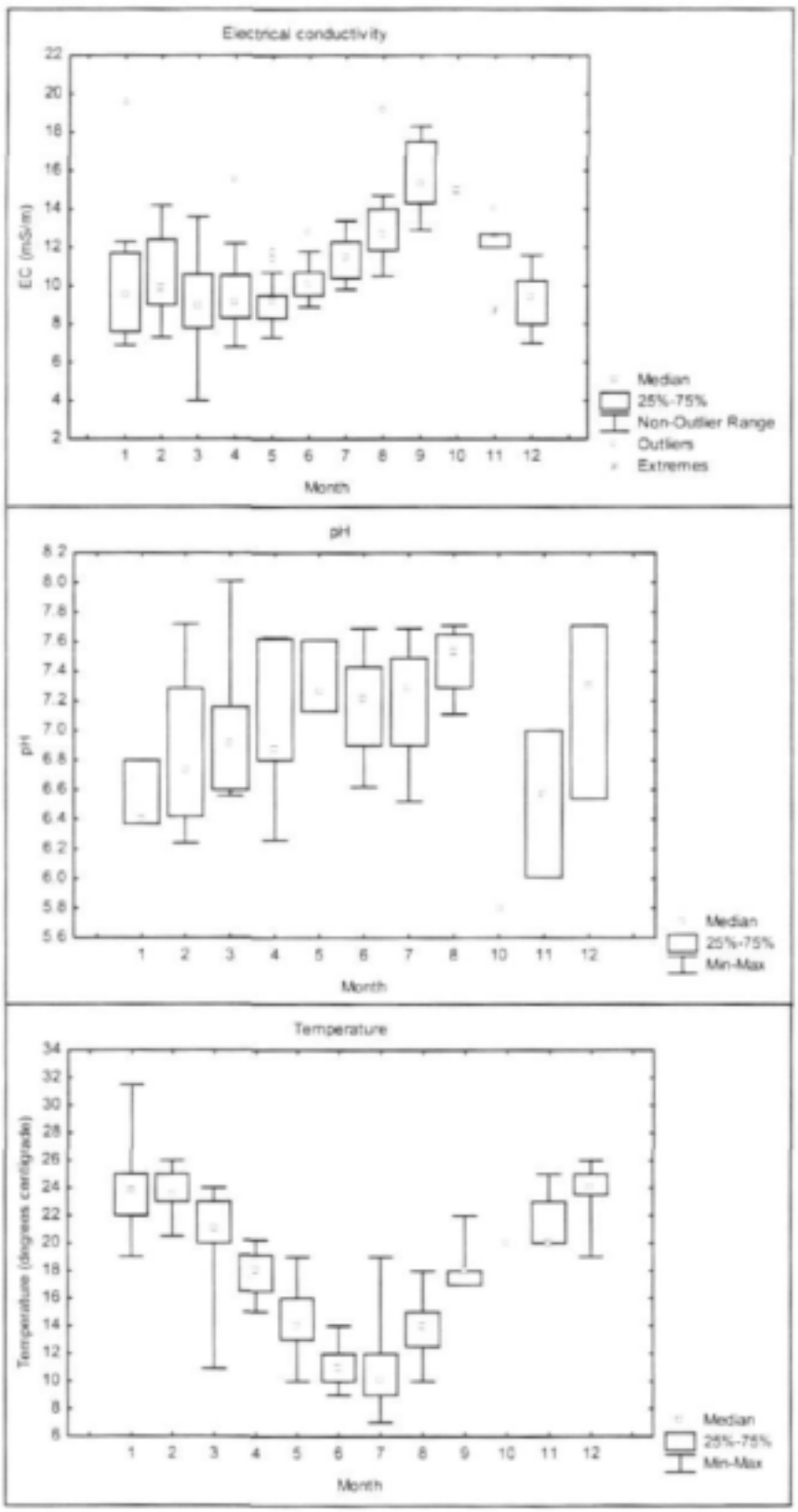
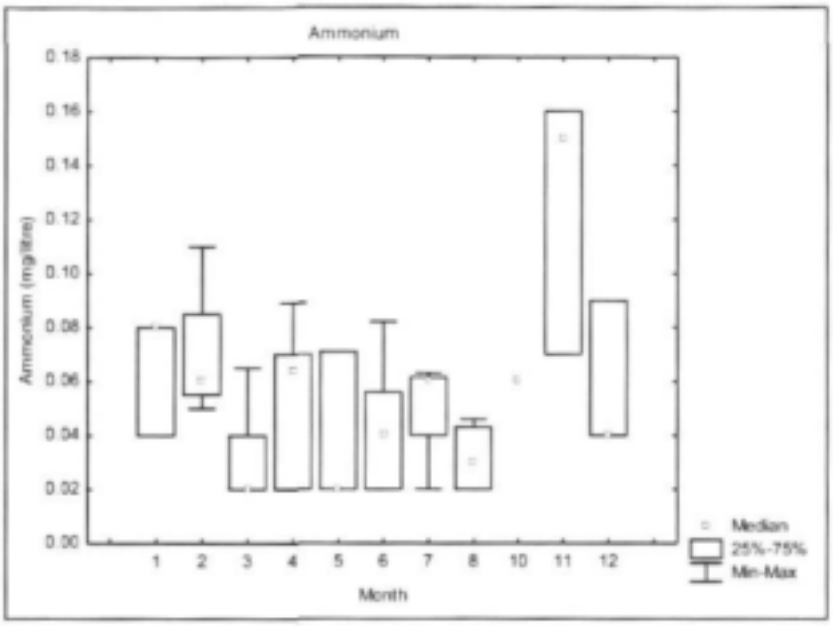


Figure 5.2 *Seasonal variation in water quality variables for the Nyl River at Deelkraal (A6H002-Q01) continued.*



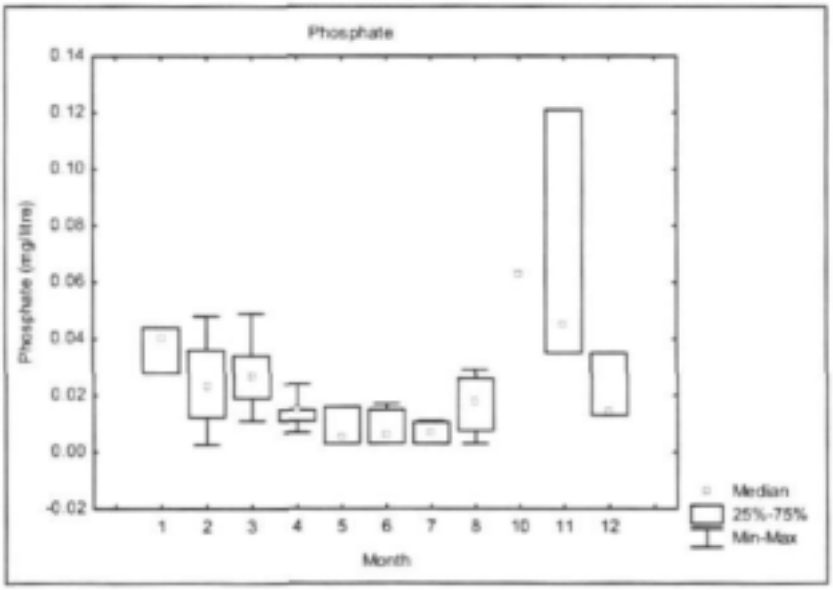


Figure 5.2 shows a box-and-whisker plot of the monthly medians for selected water quality variables at A6H002-Q01. It can be seen that both conductivity and temperature exhibit seasonal differences. Conductivity levels are elevated during the dry season and lower during the wet season, whereas temperatures are higher in the summer than the winter. Compared to the rainfall distribution however, there is a lag, by approximately two months in conductivity. Electrical conductivity values start to increase in June (not April) and return to the low values (below 10 mS/m) characteristic of the high-flow period in December. Water temperature starts to decrease from March and reaches a minimum in July. The seasonal variation of nutrients (especially ammonium) is not so clear cut. Both phosphate and ammonium show peaks in October/November (before the onset of the summer rains). The also both show a peak in January. The question arose, therefore, of how to define the "dry" and "wet" seasons (i.e. what months), in particular for conductivity and temperature, which are not synchronised with rainfall. This is discussed below.

5.2.3.3 The water quality objectives

Table 5.3 shows, for the key water quality variables, the median value and the sample size for the wet and dry season respectively. In the case of conductivity, two values are shown. The figures in light print were calculated using the definition of dry and wet season as defined by the rainfall pattern (Wet season: October – March; Dry season: April – September). The figures in bold were calculated by defining a "high EC" season and "Low EC" season. Measurements of EC taken during June-November were included in the former and measurements taken during the rest of the year in the latter. It can be seen from the table that there was not a large difference in medians for conductivity between the two methods.

The water quality objectives derived using the Reference Condition approach of the DSS, are also shown in Table 5.3. Values for three different levels of Ecological Category are shown to illustrate the method, although because Nylsvlei is a Ramsar site, it would be managed as "natural". The objectives for Class A (i.e. a wetland managed as "Natural") are shown using either the 95%ile or 80% deviation from the seasonal median. The value that should actually be implemented is the value that is lowest (in this case the 95%ile).

Table 5.3 *The water quality objectives (and other statistical parameters) for key water quality variables for the Nylsvlei wetlands derived from the Reference Condition. The figures in bold for conductivity are derived using different months for the dry and wet seasons (see explanation in text).*

Water quality variable	Season	Sample size (n)	Median	Water Quality Objectives			
				95% ile (Class A)	Median +80% (Class A)	Median +100% (Class B)	Median +200% (Class C)
Conductivity	Wet	69	9.4 9.25	13.8 12.7	16.9 16.7	18.8 18.5	28.2 28.2
	Dry	88	10.5 11.8	15.4 17.1	18.9 21.2	21 23.6	31.5 35.4
Ammonium	Wet	20	0.06	0.151	0.11	0.12	0.18
	Dry	23	0.04	0.081	0.072	0.08	0.12
Phosphate	Wet	20	0.032	0.066	0.058	0.064	0.096
	Dry	23	0.01	0.024	0.018	0.02	0.03
Sulphate	Wet	20	2.0	8.6	3.6	4	6
	Dry	23	2.0	11.5	3.6	4	6
Nitrate + nitrite	DATA NOT SUITABLE						
Temperature	Wet	69	23	26	-	-	-
	Dry	88	14	20		-	-
pH	Wet	20	6.69	7.7	-	-	-
	Dry	23	7.28	7.7		-	-

It is necessary to specify two alternative methods for deriving the Class A boundary because for some water quality constituents (e.g. ammonium and sulphate) depending on the statistical distribution of the water quality data, the 95%ile can fall outside of the median + 100% value. Thus, in such a situation the Class A water quality objective would be higher than the Class B, and thus the median + 80% value should be used.

As noted above for conductivity, the objectives were calculated using two different approaches to deriving the seasons. The results are shown in light and bold print respectively. In the case of Nylsvlei there is not a big difference between the dry and wet season conductivity levels. Thus the values calculated using the rainfall season (values in normal/light print) were used as the water quality objectives for conductivity.

Figures 5.3 - 5.7 show the monthly distributions of the key water quality variables in relation to the proposed water quality objectives.

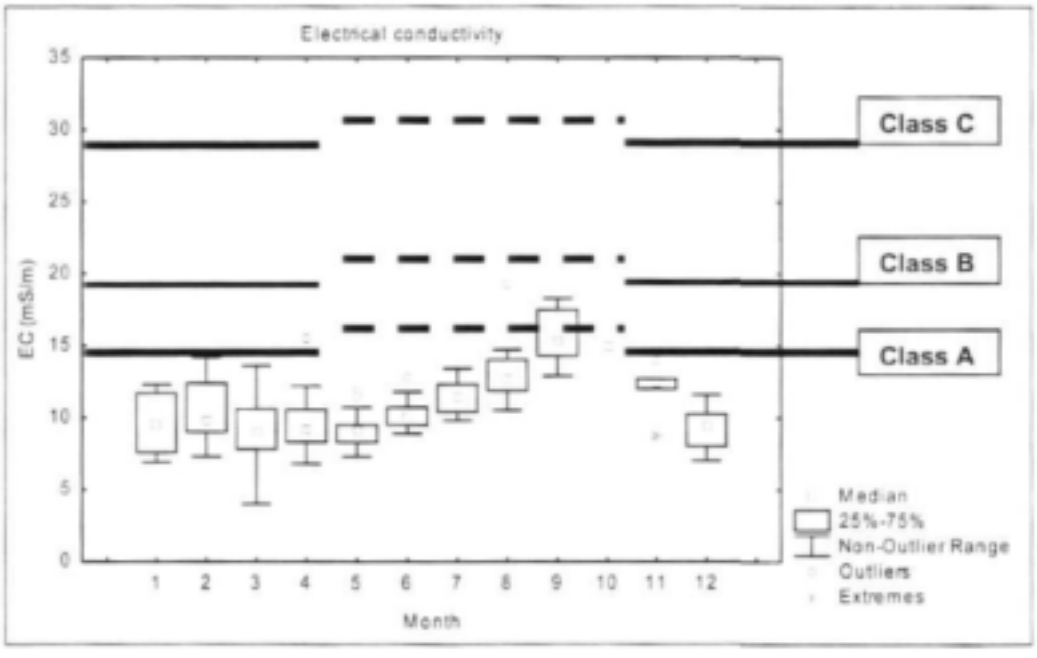


Figure 5.3 *The water quality objectives for conductivity in the Nylsvlei wetland as calculated using Approach 1 in the DSS. The wet season objectives for each Ecological Category are indicated by solid lines, the dry season by broken lines.*

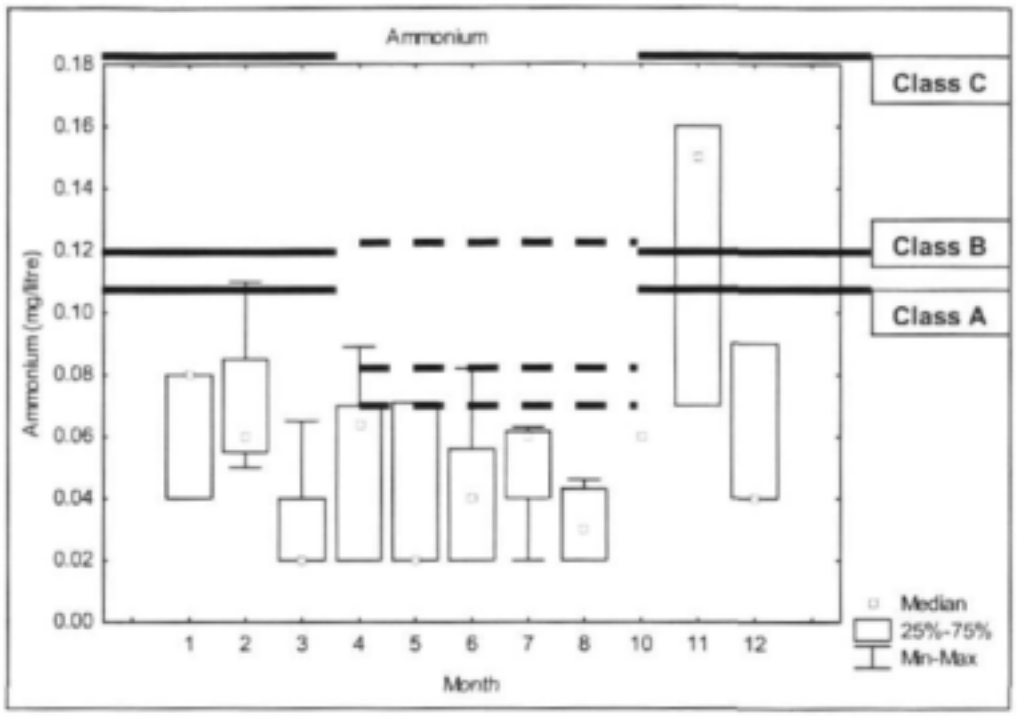


Figure 5.4 *The water quality objectives for ammonium in the Nylsvlei wetland as calculated using Approach 1 in the DSS. The wet season objectives for each Ecological Category are indicated by solid lines, the dry season by broken lines.*

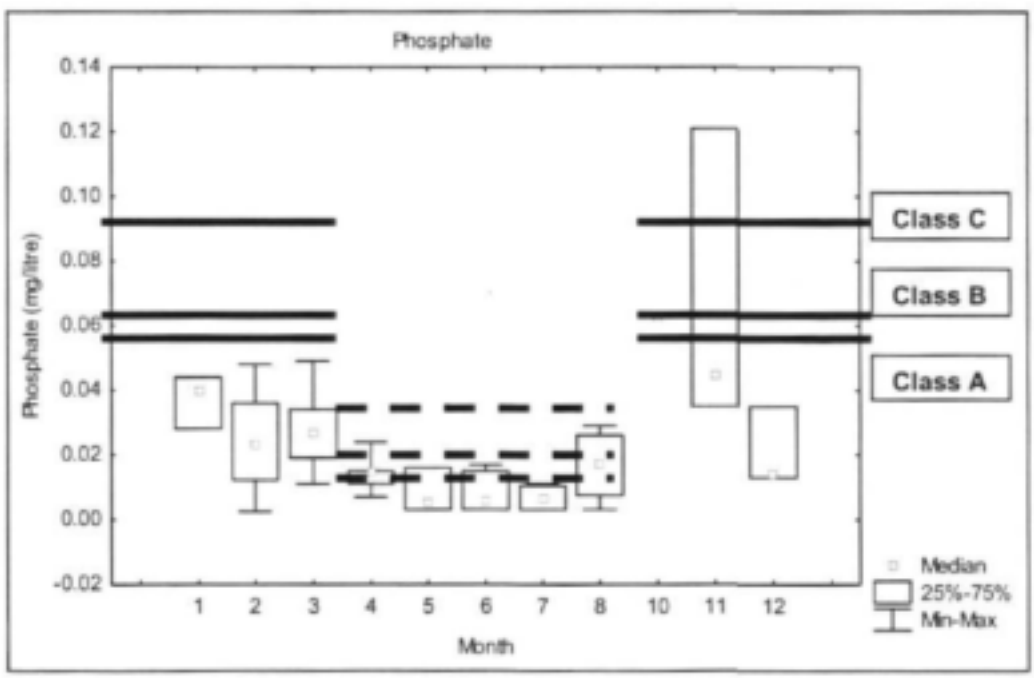


Figure 5.5 *The water quality objectives for phosphate in the Nylsvlei wetland as calculated using Approach 1 in the DSS. The wet season objectives for each Ecological Category are indicated by solid lines, the dry season by broken lines.*

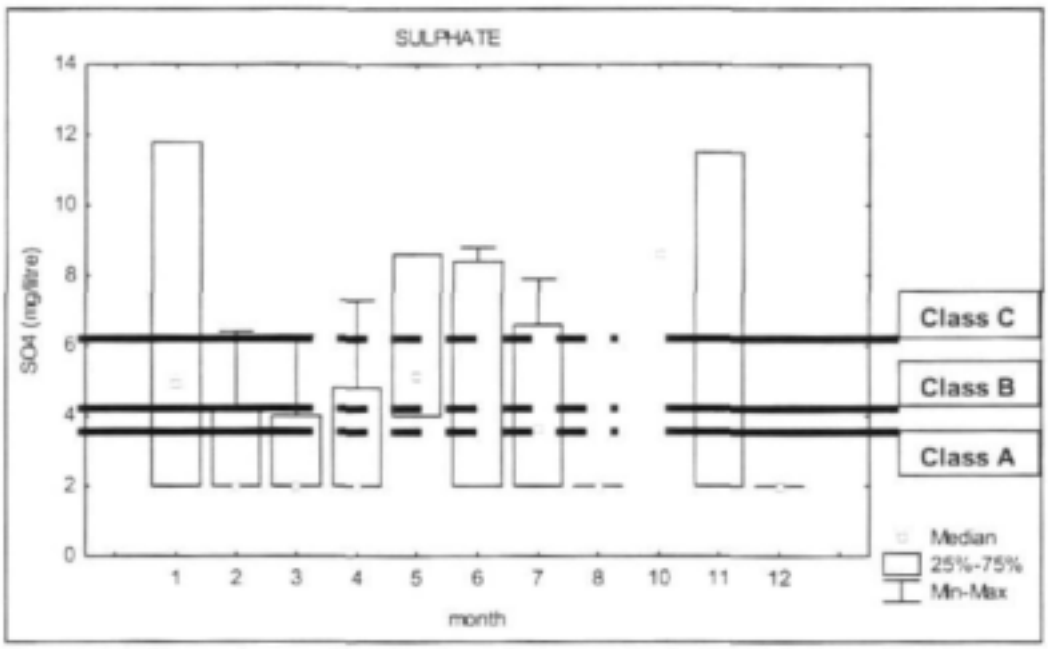


Figure 5.6 *The water quality objectives for sulphate in the Nylsvlei wetland as calculated using Approach 1 in the DSS. The wet season objectives for each Ecological Category are indicated by solid lines, the dry season by broken lines.*

5.2.4 APPROACH 2: EXTRAPOLATION FROM RIVERS

The methods used for setting water quality objectives (now referred to as "ecospecs") for rivers have undergone considerable changes in the last few years. This is especially true for salinity (conductivity, TDS). The methods have varied from that derived by Bath *et al.* (DWAF 1999a), to using site-specific ecotoxicology (*e.g. Palmer and Rossouw 2000*), to the latest approach (Jooste and Rossouw 2002) which is based on the ecotoxicological effect of individual ions rather than an overall measure of salinity. Individual salts have been considered because, according to the authors, evidence indicates that the effects of individual salts outweigh the overall effect attributable to TDS. Often, however, only conductivity or TDS data are available, rather than individual ions, therefore a method using TDS/conductivity has recently been developed for rivers (UCEWQ-IWR 2004). This latter method was used to derive objectives for conductivity in this study. Table 5.4 shows the water quality objectives for rivers.

Table 5.4 *Water quality objectives for Nylsvlei as derived using the default values for rivers.*

Water quality variable	Ecological Category				
	Natural (Class A)	Good (Class B – upper C)	Fair (Class lower C – upper D)	Poor (Class lower D – E/F)	Source
Conductivity (mS/m)	30	55	85	-	UCEWQ-IWR (2004)
TIN (mg/L)	≤0.25	0.251-1.0	1.01 – 4.0	> 4.0	DWAF (2002)
Phosphate (mg/L)	≤0.005	0.0051 – 0.025	0.0251-0.125	>0.125	DWAF (2002)
Ammonium (µg/L)	15	58	100	-	DWAF (2002).
pH	6.5 - 8.0	5.75–6.46 and 8.05-9.00	5.0-5.70 and 9.05-10.00	<5.00 or >10.00	DWAF (2002)
DO (mg/L)	-	6	4	-	DWAF (2002). No value given for natural boundary.

5.2.5 Comparison of the two approaches

A comparison of the water quality objectives yielded by the two different approaches (Tables 5.3 and 5.4) shows that quite different values were obtained for some variables. It was also not always easy to compare the results from the two approaches. Sometimes different variables were specified by the two methods – for instance values for DO are given in the riverine water quality methods, but no DO data had been collected by DWAF for Nylsvlei. Another complicating factor is that using the Approach 1 (deviation from the Reference Condition), values are specified separately for the dry and wet seasons. The default river values apply to the entire year. The objectives derived from Approach 1 are site-specific, whereas those from Approach 2 are default values (which, for high-confidence Reserve determinations would require adjustment with measured data).

It is therefore recommended by the project team that the objectives derived using Approach 1 are the most appropriate. Objectives for TIN and DO, for which there were not enough data, should be taken from Table 5.4. Implementation of the water quality component of the ecological Reserve should, for a given wetland include monitoring readjustment of the water quality objectives if necessary.

The DSS needs to be as simple and pragmatic as will yield useful and realistic results. In the case of Nylsvlei, there is little difference in magnitude between conductivity in the summer and winter. Consequently, how the seasons are defined and the water quality objectives calculated there from are not very different. In other wetland systems however, this might be important. This is an aspect that requires further investigation. In the case of some water quality variables there may be no difference between dry and wet season objectives (e.g. sulphate for Nylsvlei) or the wet season values may be higher for the wet season than the dry season (e.g. phosphate for Nylsvlei), or the difference in seasonal medians may be very small. Expert judgement should always be used to check if the objectives appear to be sensible. For instance in some cases it may be better to set objectives that are the same for both seasons.

5.3 APPLICATION OF THE DSS TO ROCHER PAN

5.3.1 Description of the site

Rocher Pan is on the West Coast of South Africa (32°36' 17"S 18°18' 08"E) and lies north of Saldhana, but south of Verlorenvlei. It is in a relatively arid part of the winter rainfall region. A description of the wetland is given in Coetzer (1981) and Coetzer (1987) and seasonal patterns were discussed in section (2.6.2.2). It is a fairly simple wetland and consists of an elongated, freshwater lake lying parallel to the coast. The water source is the Papkuils River, which flows into the system during winter. Sea water does not appear to enter the wetland despite the proximity to the sea. The vlei is endorheic, and is usually seasonal. According to Coetzer (1987), the vlei usually starts to fill in April/May and is deepest in September, but by Febuary/March the next year it is usually dry again. In wet years, however, the vlei may remain inundated the entire year. A large proportion of the wetland lies within a nature reserve (the area is well-known for birds). The northern portion of the lake, as well as the catchment area of the Papkuils

River, lie outside the boundary of the Reserve, and are subject to impacts from agriculture.

5.3.2 Examination of the available data

Several investigations have been carried out at this site (Coetzer 1981 and 1987) and more recently Jones (2002) however there are no long-term monitoring data. The data that are available were downloaded from the Wetlands Water Quality Database to a spreadsheet for further manipulation. The total number of sampling occasions for Rocher Pan in the database is 33. Note though that Coetzer took samples from up to four different sites on one day, so that to some extent there is pseudo-replication. All the data values, however, were used in the analysis. There is no DWAF water quality monitoring station on the Papkuils River.

Time-series of conductivity, SRP and nitrate were examined to see if any trends in water quality could be discerned (Figure 5.7). The conductivity values showed distinctly higher values in summer than in winter, although a clear seasonal pattern was not exhibited by nutrients. The values obtained in 2000/2001 by Jones (2002) – considered to represent the PES, are shown below:

	Winter 2000	**Summer 2001**
Conductivity (mS/m)	438	1202
Nitrate (mg/L)	0.014	0.014
Phosphate (mg/L)	0.38	0.002

From an examination of the data, the current water quality does not appear to have changed since the late '70s. The data were divided into wet season and dry season as indicated below:

Dry season: Oct, Nov, Dec, Jan, Feb, March.
Wet season: April, May, June, July, Aug, Sept.

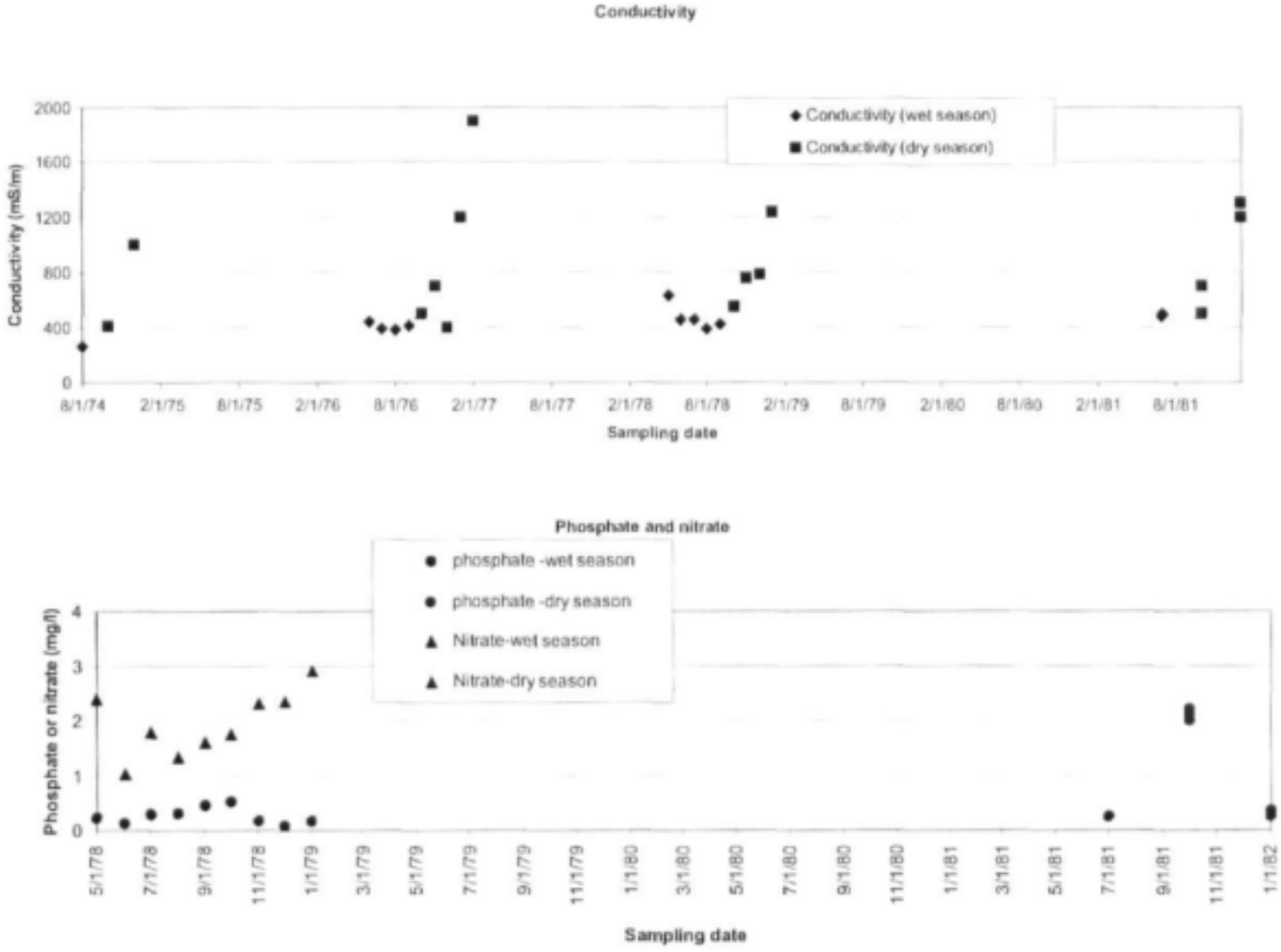


Figure 5.7 *Trends in conductivity, soluble reactive phosphorus and nitrate measured at Rocher Pan (data from Coetzer 1981 and 1987).*

5.3.3 The Decision Support System

Although Rocher Pan is fed largely by a river (the Papkuils River) it is an endorheic system. Therefore in terms of water quality type, it is more correct to treat it as a "runoff-fed" rather than a riverine system, as there is no outflow. The DSS for runoff-fed wetlands is shown in Figure 4.2. It can be seen that there are three different approaches that could be used to derive the water quality objectives, using namely:

- the Reference condition, and/or;
- making use of a comparable wetland, and/or;
- default "best-guess" values.

Of the three approaches, the second is not feasible since there is no comparable wetland in the area for which more data are available. The Reference Condition approach was therefore used, since this is the most site-specific method.

5.3.4 APPROACH 1: Deviation From The Reference Condition Water Quality

As can be seen from the above discussion, the water quality data that are available for Rocher Pan are fairly limited. Because the system is in a nature reserve (although the river entering into the system flows through some areas where dry-land irrigation is practiced) it was assumed that the pan is in a relatively unimpacted condition. High levels of nutrients were recorded on occasion, which may be natural, or possibly be due to eutrophication. Nevertheless, not enough is known to verify if this was the case. Therefore all the sampling data were combined and were used to define the Reference Condition (i.e. they were used to calculate the median and 95% values). The water quality objectives for the variables for which data are available are shown in Table 5.5. Once again the Class A (natural) boundary value was calculated using two methods. The lowest value (indicated by an asterisk) should be used. The boundary values for each Ecological Category in comparison to the measured values are shown in Figures 5.8 to 5.10.

It is interesting to compare the results shown in Table 5.5 with the default ("best-guess") values in Table 4.1. The objectives for nutrients that were calculated from the Reference Condition are considerabley higher than those in the default table. This indicates that site-specifically derived objectives may be very different from the generic values.

Table 5.5 *The water quality objectives (and other statistical parameters) for key water quality variables for Rocher Pan derived from the Reference Condition. Also shown are the default values for nutrients as recommended in Table 4.1.*

Water quality variable	Season	Sample size (n)	Median	Water Quality Objectives			
				95% ile (Class A)	Median +80% (Class A)	Median +100% (Class B)	Median +200% (Class C)
Conductivity (mS/m)	Wet	14	439	548*	790	878	1317
	Dry	18	728	1390	1310*	1456	2184
Nitrate (mg/L)	Wet	6	1.47	2.25*	2.65	2.94	4.41
	Dry	5	2.3	2.80*	4.14	4.6	6.9
Default ("best guess") value for TIN = 0.8 mg/L							
Phosphate (mg/L)	Wet	8	0.25	0.41*	0.45	0.5	0.75
	Dry	11	0.38	2.15	0.68*	0.76	1.14
Default ("best guess") value for phosphate (SRP) = 0.08 mg/L							

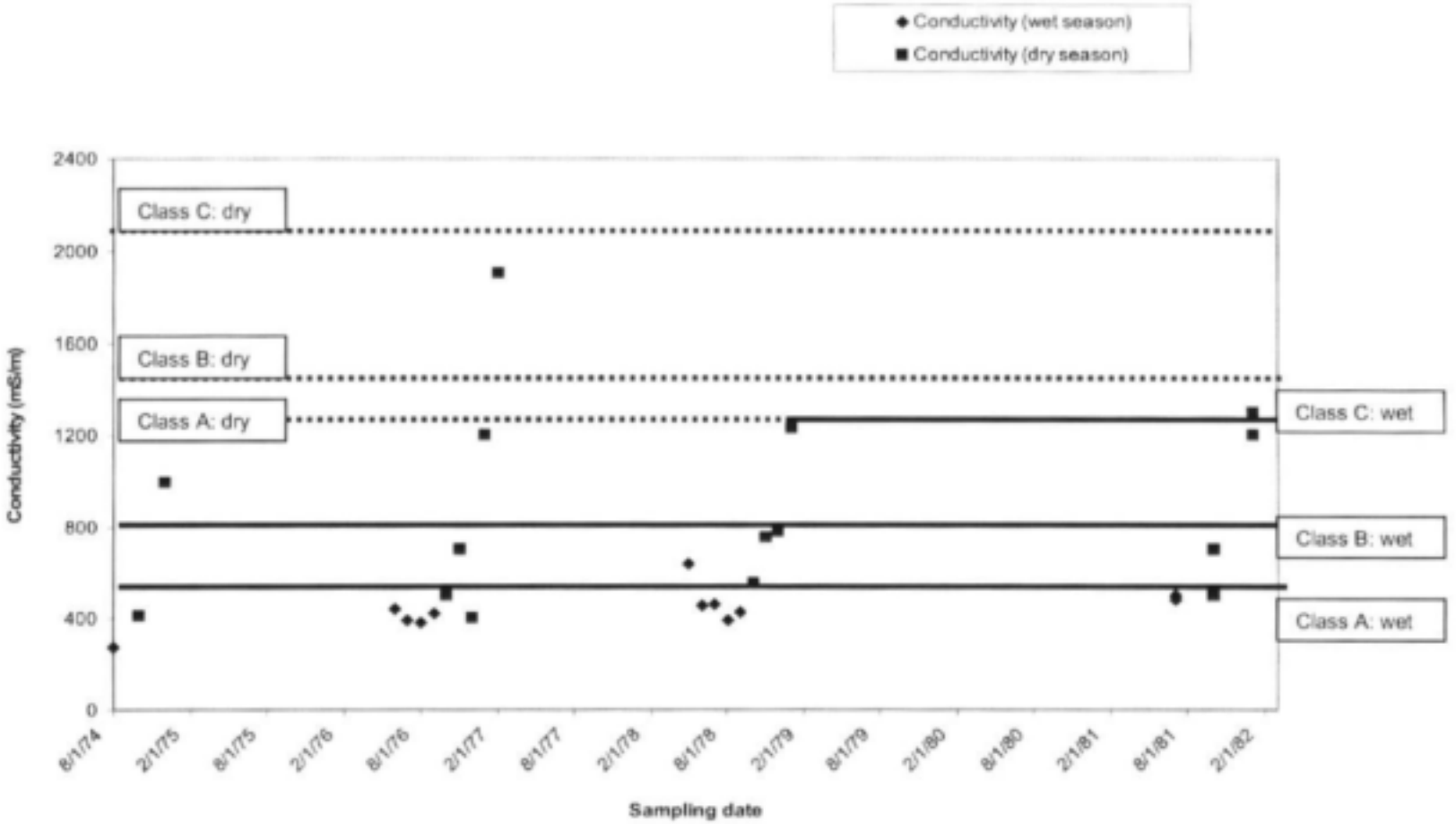


Figure 5.8 *Water quality objectives for* ***conductivity*** *at Rocher Pan as derived in Table 5.5. Objectives for the wet season shown as solid bold lines, objectives for the dry season shown as stippled lines.*

Figure 5.9 *Water quality objectives for* ***phosphate*** *at Rocher Pan as derived in Table 5.5. Objectives for the wet season shown as solid lines, objectives for the dry season shown as stippled lines.*

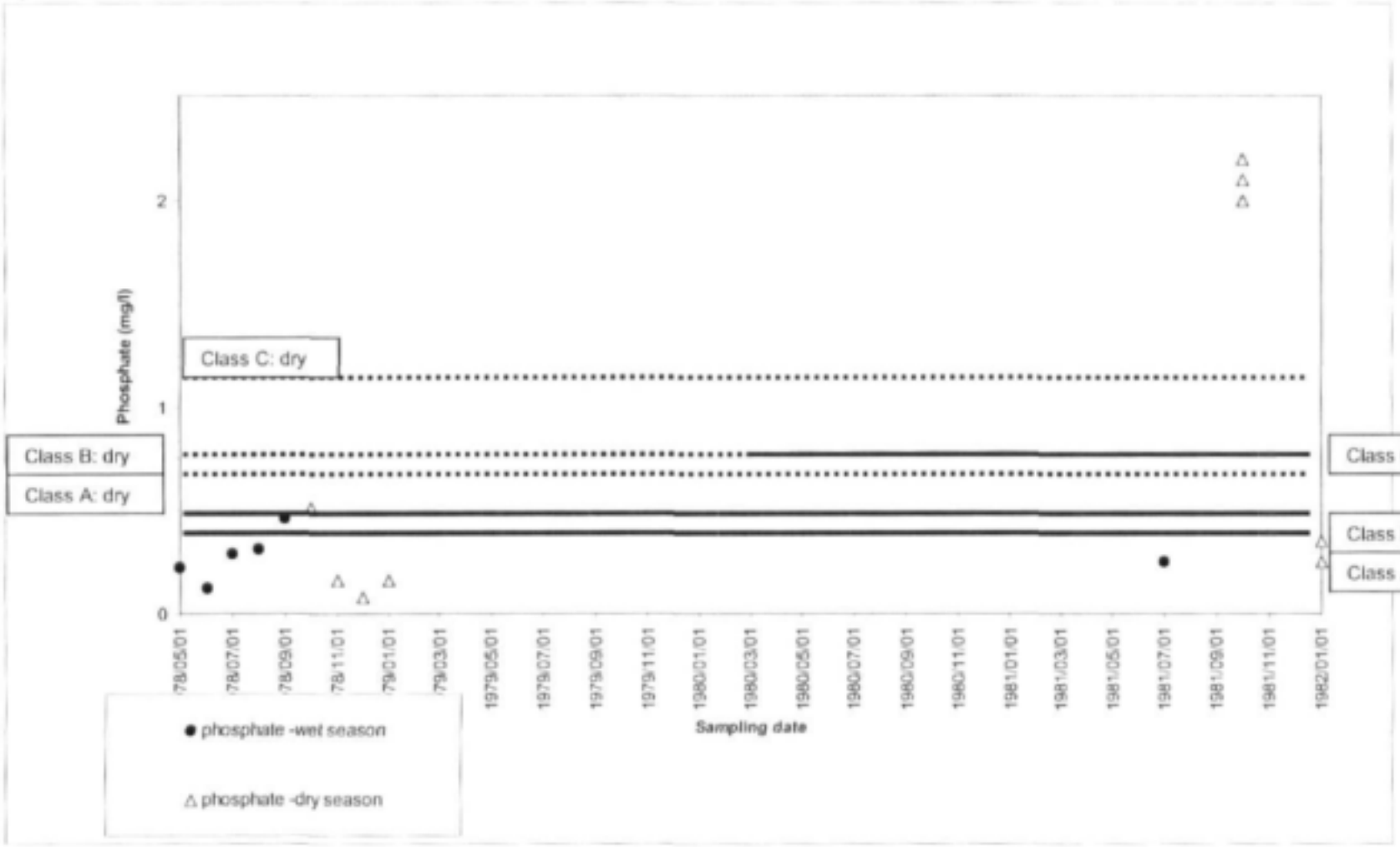
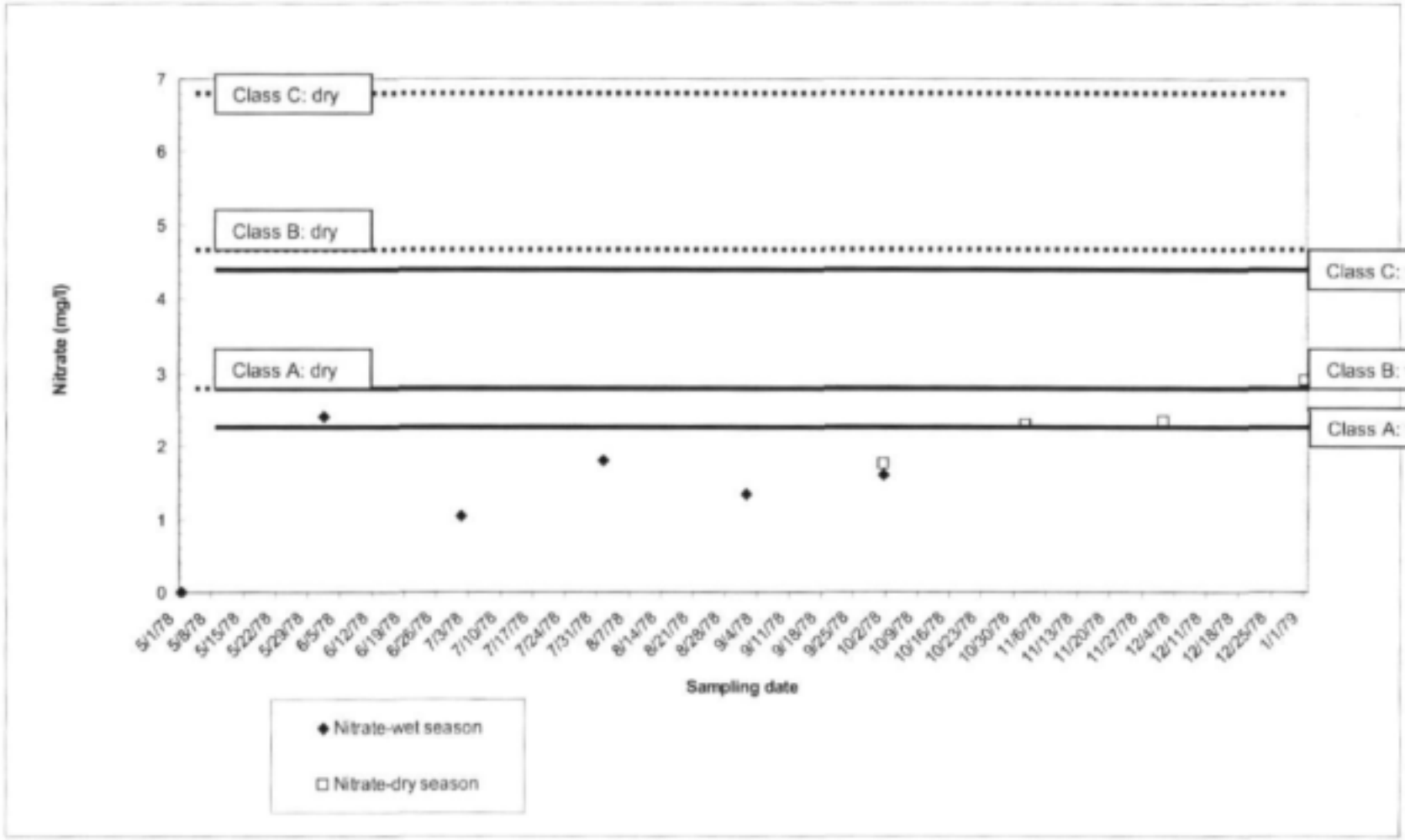


Figure 5.10 *Water quality objectives for* ***nitrate*** *at Rocher Pan as derived in Table 5.5. Objectives for the wet season shown as solid bold lines, objectives for the dry season shown as stippled bold lines.*



5.4 APPLICATION OF THE DSS TO VERLORENVLEI

Verlorenvlei is situated in the arid Sandveld region of the west coast of South Africa, to the north of Rocher Pan. The system lies in the winter rainfall area, and receives less than 300 mm rain per annum, with more than 95% of the runoff occurring between April and September (Gale 2002). The mouth of Verlorenvlei reaches the coast at Elands Bay (Figure 5.14). The system is fed largely by the Kruis River (named the Verlorenvlei River in the reach just above the lake) and is a perennial lacustrine system and a designated Ramsar site. The mouth of the system is periodically open to the sea (due to natural or artificial breaching) and under such conditions there is a tidal influence extending about 5 km from the mouth (Grindley and Grindley 1987). The Verlorenvlei River from the top of the open water of Verlorenvlei to the confluence with the Hol River (just downstream of the gauging weir) is composed of extensive reed-beds with little open water.

Verlorenvlei (in addition to the Jakkalsvlei/Jakkals River, and Wadrif/Langvlei River systems) was studied as part of a preliminary Reserve study carried out by DWAF (DWAF 2003b). There is increasing demand for water for irrigation of potatoes and sweet potatoes in the area, in addition to supplying the water needs of local towns such as Lamberts Bay (Sinclair, Lane and Grindley 1986; J. Conrad, GEOSS, *pers. com.* September 2004). At the same time, there is concern that groundwater resources (the principle source of water in the area) are being used unsustainably and that this is impacting on the ecological integrity of the wetlands in the area. The number of birds visiting, especially Wadrif Pan, has noticeably diminished in recent years (A. Williams, WCNCB & ADU-UCT, *pers. com.* 2004). The results of the Reserve determination study are given in DWAF (2003b), including a detailed description of the hydrology, water quality (surface and groundwater), topography, and aspects of the biology, and hence are not duplicated here.

5.4.1 Data availability

Despite the fact that Verlorenvlei is a Ramsar site, relatively little research has been carried out on the system. Sources of relevant water quality data are listed below:

i) DWAF water quality data from the following stations:
G3H001-Q01 (Kruis River @ Tweekuilen) 70/05/08-02/08/21, n=276. Good Reference Condition data-set (for time-period 1977-1983), but sampling subsequently less frequent (less than once-per-month).

G3H002-Q01 (Verlorevlei River @ Redelinghuys) 70/05/08 – 72/09/08, n = 6.
G3H003-Q01 (Verlorevlei @ Bontehevel) '72-'89, n=2
G3H004-Q01 (Verlorevlei @ Elandsbaai) 70/07/15 – 97/04/22, n= 7.
G3H005-Q01 (Hol River @ Wittewater/Papkuilsvlei) 78/05/08 – 90/09/12, n=101. This is a tributary of the Kruis River.
G3R001-Q01 (Verlorevlei Lake) 95/08/24 – 95/10/10, n=2

ii) Spot values of various water quality variables and background information are reported in *Grindley (1978), Robertson (1980), Sinclair, Lane and Grindley (1986), Wishart (1998), and Jones (2002).*

iii) Data collected as part of the ecological Reserve determination for the Sandveld (W. Harding, DH Environmental consulting, *pers. com.* 2003). These were, however, collected over a relatively short period of time (July 2002 – January 2003).

5.4.2 Hydrology

According to Southern Waters (2002) with the exception of the extreme upper reaches of the Verlorenvlei catchment, surface flows are limited to event-driven, short-duration episodes. Groundwater plays a very strong role both in the Kruis River-Verlorenvlei, as well as that of the Langvlei River-Wadrif Pan and the Jakkals systems. Extensive lengths of the rivers in this arid area are characterised by hyporheic flow. The rivers are characterized by multiple intersections of groundwater (springs and seeps) and the water quality of the numerous eyes that feed the systems, appear to vary quite considerably (DWAF 2003b). Water quality, and in particular salinity, in the lower part of the vlei is profoundly influenced by the status of the estuary mouth and whether seawater can enter the system or not. Thus, from the point of view of water quality, this system is extremely complex and given the paucity of water quality data at this stage, only tentative water quality objectives can be prescribed.

5.4.3 Trends in the water quality of the Kruis River

The water quality data from the DWAF monitoring station G3H001-Q01 (Kruis River @ Tweekuilen) were examined for trends in the levels of water quality constituents over the measured time-period. This exercise was carried out because the Kruis River is a major source of water entering Verlorenvlei and from this, changes in water quality that may

have affected the lake can be inferred. The station at Tweekuilen is also the only sampling point for which there is a relatively good time-series of water quality data.

The Reference Condition was derived using the data from the time-period 1978-1982. The present ecological state (PES) was derived using the data from July 1996-July 2002. The median values for each key water quality variable for which there were data are shown in Table 5.6. Time-series graphs for key water quality variables are shown in Figures 5.11 – 5.13. It should be noted that impacts in the catchment are likely to have occurred before the start of monitoring and thus the even the earliest data are likely to show some impacts. Thus the Reference Condition should more accurately be referred to in this case as the "less impacted condition". Southern Waters (2002) did not use these data to define the Reference Condition for the Kruis River, but rather used expert judgement to derive generalised values, because it was considered that even the early data from this station were reflective of impaired water quality.

From Table 5.6 and from the time-series plots the following trends can be detected:

- **Electrical conductivity:** This variable does not seem to have increased over the last 25 years – in fact it may have decreased slightly. This is shown by the median EC values and is supported by the decreased median concentrations of Na^+, Cl^- and SO_4^{-2}.
- **pH:** Although the data seem to show that there has been an increase in pH since the late 1980s, this is likely to be an artefact due to the change in sampling/measuring technique.

Table 5.6 *The median Reference Condition and Present Ecological State values for key water quality variables at Tweekuilen (G3H001-Q01, Kruis River). Also shown are the median values for the same data, split into dry and wet season, and the standard deviation for the median (both seasons combined).*

REFERENCE CONDITION (DATA FROM 1978 – 1982)										
	pH	**NO_3+NO_2 (mg/L)**	**NH_4-N (mg/L)**	**Na (mg/L)**	**Si (mg/L)**	**PO_4-P (mg/L)**	**SO_4 (mg/L)**	**Cl (mg/L)**	**K (mg/L)**	**EC (mS/m)**
Overall median	7.3 ± 0.4	0.170 ± 0.38	0.04 ± 0.11	673 ± 212	4.3 ± 1.5	0.018 ± 0.06	134 ± 46.5	1361 ± 439	9 ± 4.12	437 ± 133
Wet season median	7.3	0.23	0.09	677	4.7	0.022	133	1365	9.7	446
Dry season median	7.2	0.06	0.04	662	2.2	0.012	135	1299	7.3	397
PRESENT ECOLOGICAL STATE (DATA FROM 1996 – 2002)										
	pH	**NO_3+NO_2 (mg/L)**	**NH_4-N (mg/L)**	**Na (mg/L)**	**Si (mg/L)**	**PO_4-P (mg/L)**	**SO_4 (mg/L)**	**Cl (mg/L)**	**K (mg/L)**	**EC (mS/m)**
Overall median	7.9 ± 0.2	0.281 ± 0.46	0.02 ± 0.03	563 ± 293	3.1 ± 1.8	0.025 ± 0.03	108 ± 81	1129 ± 579	8.09 ± 5.7	394 ± 180
Wet season median	7.9	0.353	0.043	606	4.2	0.027	120	1147	11.3	402
Dry season median	8.0	0.118	0.02	554	1.8	0.015	93	1104	5.6	393

- **Nitrogen:** Combined nitrate and nitrites do seem to have increased at this site, although this is not accompanied by an increase in ammonium. According to the South African Water Quality Guidelines, average *summer* inorganic nitrogen (nitrates, nitrites, ammonia and ammonium combined) should be less than 0.5 mg/L. From an examination of the time-series it can be seen that nitrate and nitrite show a periodic increase during low-flow conditions (summer) and that the concentrations regularly do go above this value during the summer. The Kruis River, at least in the region of Tweekuilen can therefore be classified as mesotrophic with regard to nitrogen.

- **Phosphate:** Soluble Reactive Phosphate levels (there are no total phosphate data) also appear to have increased slightly over the measured time period. According to the South African Water Quality Guidelines, average *summer* inorganic phosphorus concentrations should be less than 0.025 to be considered mesotrophic. A cursory examination of the data shows that the data are very

variable and that highest concentrations are not during summer, but in autumn (possibly due to wash-off when the rains begin). If not presently eutrophic, the system is verging on this state since the median level is 0.025 mg/L.

Separate values for wet and dry season were calculated using the Tweekuilen data-set. The seasons were differentiated as follows:

Dry season: October - March

Wet season: April - Sept

Distinct seasonality for conductivity, TIN and phosphate was not exhibited. The median values that were calculated, even for the Reference Condition exhibited high standard deviations, and there was no significant difference between seasons (Table 5.6). Median values were often higher in the wet season than in the dry season. This is likely to be a consequence of the wash-off of chemical constituents from the surrounding catchment.

There has been extensive increase in crop cultivation in the upper area of this catchment since the 1970's (Southern Waters 2002). It would be expected that the increase in nutrients would be even higher than that shown above. Possibly the reed beds above the site are responsible for the lower than expected values. According to B. Low (pers. *com.* 2002) the extensive *Typha* beds above this site thrive in mesotrophic, freshwater environments. With regard to Verlorenvlei itself, although it is considered that the lake should be oligotrophic (Robertson 1980) blooms of phytoplankton have been reported in the vlei (Southern Waters 2002). The village of Redelinghuys is downstream of Tweekuilen and may well be contributing to the nutrient load entering the vlei.

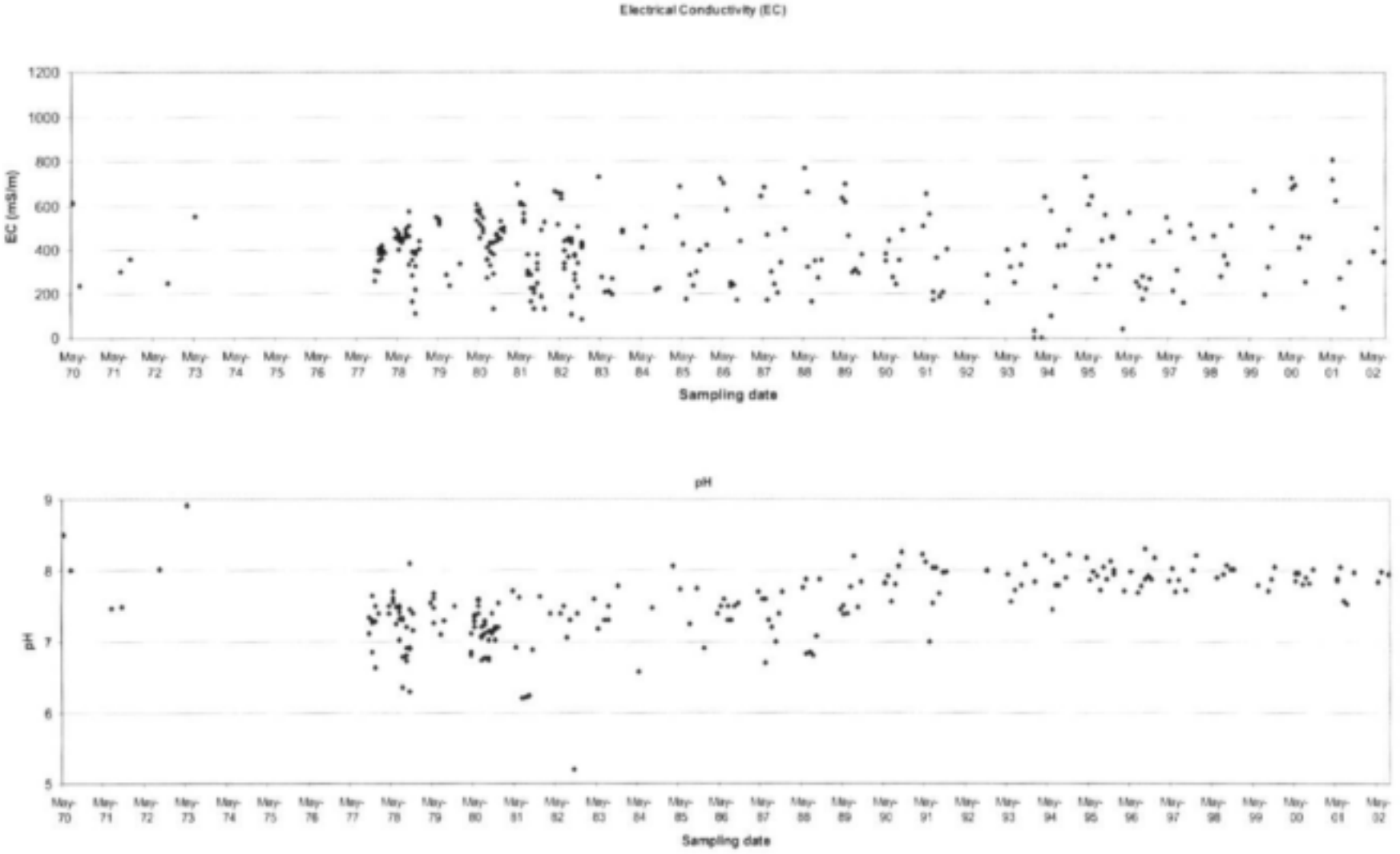


Figure 5.11 *Time-series plot of conductivity and pH from the DWAF monitoring station G3H001-Q01 on the Kruis River at Tweekuilen.*

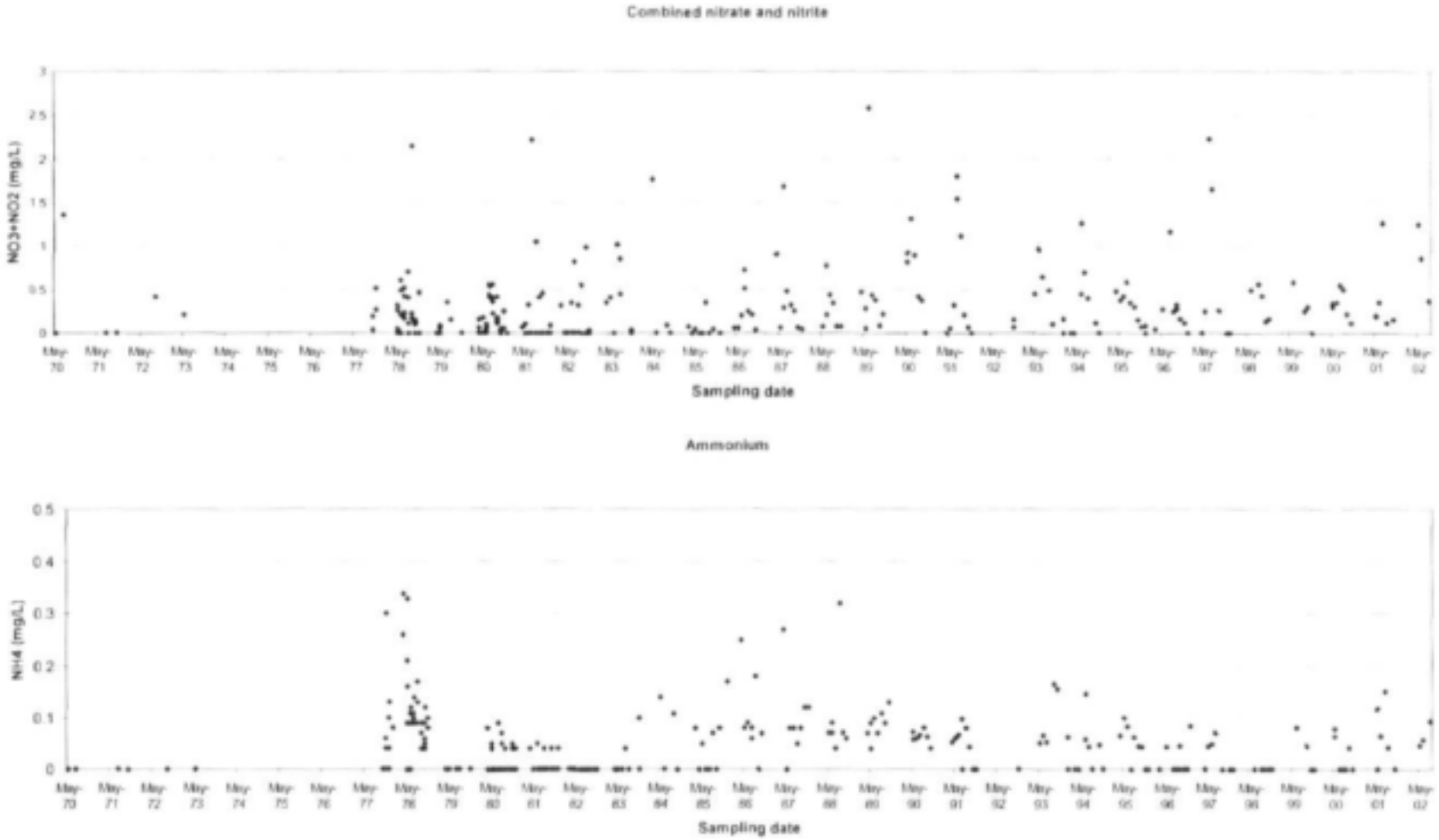


Figure 5.12 *Time-series plots of combined nitrate and nitrite, and ammonium from the DWAF monitoring station G3H001-Q01 on the Kruis River at Tweekuilen.*

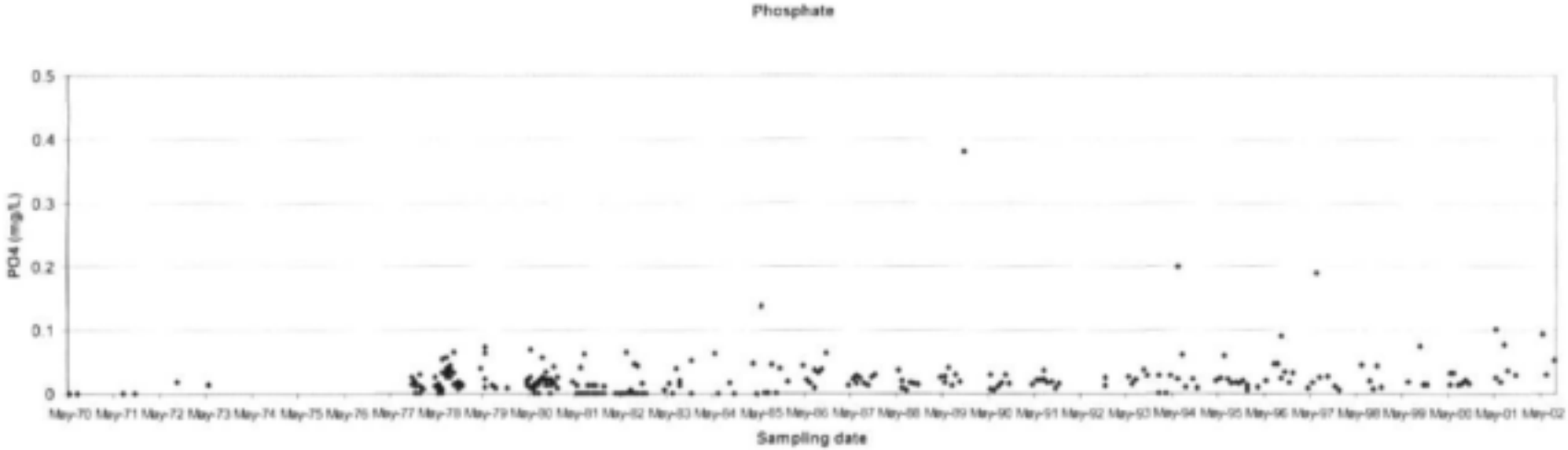


Figure 5.13 *Time-series plot of phosphate from the DWAF monitoring station G3H001-Q01 on the Kruis River at Tweekuilen.*

5.4.4 The Decision Support System

Verlorenvlei itself is a coastal lake and therefore, according to the DSS (Figure 4.4) there are three different approaches that could be used to setting the water quality objectives.

- If there are Reference Condition data for the lake, use these to derive the water quality objectives.
- Use a "comparable system" for which data are available.
- Use the water quality boundary values for rivers.

In the case of Verlorenvlei, there is no comparable system in the same ecoregion. With regard to using the Reference Condition, there are data for the Kruis River, however, those for the vlei are very limited. The approach used in this study was to use the Reference Condition data for Kruis River and to adjust these, using expert judgement after examination of the data for the vlei itself, as well as of the default boundary values for rivers (DWAF 2002). Furthermore, because Verlorenvlei is a Ramsar site, by default it will be maintained in a "natural" condition (Class A Ecological Category). Because of the paucity of the data water quality data, objectives were derived only for Class A (natural). It is felt that there was too much uncertainty to extrapolate to lower Ecological Categories (management classes). If the system were to be managed at a lower class, more data would first need to be gathered.

5.4.5 Determination of the water quality objectives

The Kruis River-Verlorenvlei system was divided into five water quality reaches that were considered to be relatively homogeneous with regard to water quality. These reaches are shown in Figure 5.14 and consist of:

- Reach 1: The Kruis River around TweeKuilen, an area of surface-flowing water in winter (although flow may cease in summer).
- Reach 2: The wetland area around Redlinghuys – characterized by extensive reed beds and limited open water.
- Reach 3: The upper portion of Verlorenvlei lake – not subjected to tidal influence when the mouth is open (i.e. more than 5 km from the estuary mouth).
- Reach 4: The lower portion of Verlorenvlei lake – subject to tidal influence (if the mouth is open).
- Reach 5: The estuary section of Verlorenvlei.

Water quality objectives were not derived for the last two water quality reaches as these areas are periodically open to the sea. The DSS developed in this study has not been designed for wetlands that exhibit a strong marine influence (section 4.4.3).

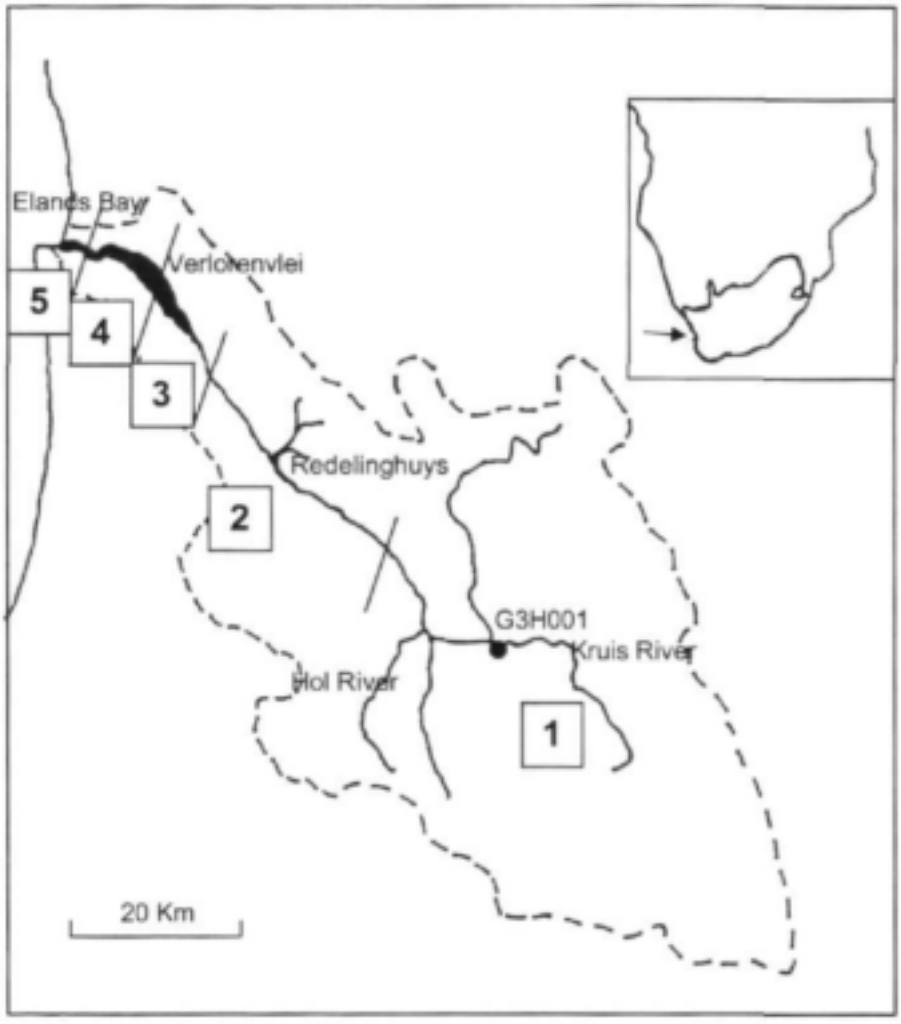


Figure 5.14 *Map of the Verlorenvlei area showing the five different water quality reaches used in the assessment. The catchment boundary is delineated by a dashed line.*

Table 5.7 *Proposed objectives for* ***electrical conductivity*** *(mS/m) for the different water quality reaches of the Verlorenvlei system. The default water quality objective for rivers is shown and the Reference condition and Present state.*

Water Quality reach	Reference Condition	Present state	Default river boundary value (Natural)	Proposed WQ objective (A Class)
Kruis River	437* (n=125)	394 (n=41)	30 (UCEWQ-IWR 2004)	500 (605=95%ile of RC data-set)
Middle wetland area (Redlinghuys)	178* (n=6)	234**s (n=4)		500
Upper vlei area (Verlorenvlei)	273t (n=2)	265**sV (n=8)		500
Lower vlei area (Verlorenvlei)	530t (n=2)	354**sg (n=2)		Not determined
Estuary (below causeway)	Not considered	Not considered		Not determined

*DWAF (G3H001 Kruis R., or G3H002 Redlinghuys); **measured by Malan 2002, s=measured by Southern Waters (2002/2003); g=Jones (2002); t=Sinclair *et al.* (1986), V=DWAF G3R001.

Table 5.8 *Proposed objectives for* ***total inorganic nitrogen*** *(mg/L) for the different water quality reaches of the Verlorenvlei system. The default water quality objective for rivers is shown and the Reference condition and Present state.*

Water Quality reach	Reference Condition	Present state	Default river boundary value (Natural)	Proposed WQ objective (A Class)
Kruis River	0.21* (n=80)	0.30* (n= 41)	0.25	0.4 (median +80%)
Middle wetland area (Redlinghuys)	0.4* (n = 2)g only NO3+NO2	<0.04s (n=1)		0.4
Upper vlei area (Verlorenvlei)	0.17 (n=2) g only NO3+NO2g	0.07jsV (n=6)		0.4
Lower vlei area (Verlorenvlei)	No data (0.05 =NO3+NO2)g			Not determined
Estuary (below causeway)	Not considered	Not considered		Not determined

*DWAF (G3H001 @ Kruis R, or G3H002 @ Redlinghuys); **measured by Malan 2002, s measured by Southern Waters (2002/2003); j=Jones (2002); t Sinclair *et al.* (1986).g=Grindley (1978), V=DWAF G3R001.

Table 5.9 *Proposed objectives for SRP (mg/L) for the different water quality reaches of the Verlorenvlei system. The default water quality objective for rivers is shown and the Reference condition and Present state. Sources from which the values were calculated are given at the bottom of the table.*

Water Quality reach	Reference Condition	Present state	Default river boundary value (Natural)	Proposed WQ objective (A Class)
Kruis River	0.02* (n=80)	0.03*	0.005	0.032 (median+80%)
Middle wetland area (Redlinghuys)	0.027g (n=1)	0.02s (n=1)		0.032
Upper vlei area (Verlorenvlei)	0.031g (n=1)	0.023js (n=4)		0.032
Lower vlei area (Verlorenvlei)	0.145g	No data		Not determined
Estuary (below causeway)	Not considered	Not considered		Not determined

*DWAF G3H001, or G3H002; **measured by Malan 2002, s measured by Southern Waters (2002/2003); j= Jones (2002); t Sinclair *et al.* (1986). g=Grindley (1978), V=DWAF G3R001.

Tables 5.7 - 5.9 show for EC, TIN and ortho-phosphate, the Reference Condition and Present state values for each of the water quality reaches. These values were derived by calculating the median of all available values in the literature sources. Separate values for dry and wet seasons were not set due to lack of data. Also shown is the default river boundary value for the natural class and the proposed water quality objectives for each variable.

Water quality objectives were set as either the 95%ile of the Reference condition data for the Kruis River @ Tweekuilen (G3H001) or at the median value + 80% (whichever was the lowest) and then adjusted using expert judgement. The default riverine EC value for the "natural boundary" is very low (30mS/m) compared to values of conductivity normally found in the Kruis River. The former value was derived in part from a consideration of all the conductivity data for rivers in the Western Cape (UCEWQ-IWR 2004) but not specifically for sites in highly arid areas and for rivers that are seasonal. For this reason it was felt to be more appropriate to use the 95%ile value of the Reference Condition data-set for the Kruis River (which was then decreased due to the fact that the RC data are more representative of the "slightly-impacted" state). The water quality objective for Redelinghuys and the upper part of the vlei have also been set at 500 mS/m.

With regard to TIN, the median + 80% value was found to be equal to 0.38 mg/L for the Reference Condition data-set (and this was lower than the 95%ile). Thus a water quality objective of 0.4 mg/L for Total inorganic nitrogen is proposed (Table 5. 8). A similar approach was used to set the objective for phosphate.

As noted previously in this chapter, the Kruis River-Verlorenvlei system is extremely complex from the point of view of hydrology and water quality and the data are extremely limited. The proposed water quality objectives are therefore tentative and more data urgently needs to be gathered in order to refine these. This is especially relevant in view of the fact that this is a Ramsar site.

CHAPTER 6
MAJOR RESULTS, CONCLUSIONS AND RECOMMENDATIONS

6.1 MAJOR RESULTS OF THIS PROJECT

The major results obtained were those arising from interrogation of the Wetland Water Quality Database (section 3.6) and are summarized below:

The effect of wetland type on water quality:

- The lowest median values for electrical conductivity (EC), phosphate and ammonia in all the types of wetland examined, were found in springs ("eyes"). This is perhaps to be expected because the data are all from DWAF and many are used as sources of potable water in arid areas.
- Lakes were significantly different from all non-impacted wetland types with regard to EC, but were not different to impacted wetlands. Lakes were thus higher in conductivity than springs, but lower in conductivity than endorheic and palustrine systems.
- Endorheic wetlands (pans) showed the highest median conductivity of all the wetland types.
- Preliminary results indicate that for some wetlands, anthropogenic impacts have taken the form of dilution, rather than concentration of some water quality constituents (although these results were not statistically significant).
- The range in median pH values for the different wetland types in the non-impacted state was small and pH tended to be circumneutral. There were no significant differences between the non-impacted wetland types. Impacted lakes, on the other hand, exhibited a significantly higher median pH value than all other groups. This is likely to be a consequence of the eutrophic condition of these wetlands.
- The highest phosphate level exhibited by non-impacted wetlands was found in palustrine systems and these were significantly different with regard to all other natural wetland systems. The difference between palustrine and *impacted* wetlands, however, was not significant.
- The median nitrate concentrations exhibited by the non-impacted wetlands (lacustrine, palustrine and endorheic systems) did not vary significantly between

the different types. Nitrate levels in the water column of impacted wetlands on the other hand, were significantly higher than in natural systems. The number of wetlands examined, however, was very low for some wetland types. Similarly, some of the highest values of ammonium were exhibited by the impacted wetlands, particularly the freshwater lakes, and these were statistically higher than non-impacted lakes.

The effect of climatic region on wetland water quality:

- The lowest median EC values were exhibited by the wetlands that fell into the "All year, summer maximum" region of the country. This area was statistically different from all the other rainfall regions. This region is one of the most mesic in the country and thus the "dilution capacity" is likely to be high. Linked to this is the fact that many of the wetlands in this area are freshwater lakes which were one of the wetland types that exhibited the lowest EC values.
- The highest median EC values (1 985 mS/m) were exhibited by the wetlands of the summer rainfall region and this value is significantly higher than all other wetland groups.
- It was concluded that whilst there are ranges of water quality variables characteristic of different rainfall regions, it is difficult to pick out clear patterns. Water quality also varies with wetland type and this is likely to confuse any regional patterns.

6.2 CONCLUSIONS ARISING FROM THIS PROJECT

The following major conclusions were reached during the course of this project.

- There is a lack of water quality monitoring data for wetlands, in conjunction with limited research that has been carried out into the fundamental understanding of ecological functioning of these systems. This is a serious obstacle to deriving realistic water quality objectives and ultimately, to providing effective protection for these systems. Linking the desired level of ecosystem health and the water quality conditions that will achieve this state was found to be one of the most challenging aspects of the project. Until the knowledge base concerning wetland water quality and ecological response is increased, the future ecological status of a wetland under a given water quality regime cannot be predicted with any certainty.

- The 'best-guess" default values should be used only when no site-specific data are available since objectives defined in this manner represent a low level of confidence. Whatever the approach used, a monitoring programme would need to be implemented to ensure that water quality objectives were appropriate and to ensure compliance.
- Useful approaches to setting water quality guidelines for wetlands were obtained from examination of international practices, especially those of the USA and Australia/New Zealand. Largely because of the lack of monitoring data and the poor understanding of wetland ecological functioning in South Africa, no single international approach was suitable on its own, but rather a combination of methods were considered to be the most useful.
- Because of the limited understanding of wetland ecology, it is often difficult to establish numerical relationships between the concentration of a given chemical constituent or value of a physical variable and the effect on wetland biota. Narrative statements such as "salinity should not increase so that >10% of *Phragmites* reed beds die back" may therefore be useful as a management tools.
- The database MedWet (3.0) was found to be unsuitable for housing the water quality data collated during the course of the project. This was largely because multiple sampling visits could not be linked to a particular wetland site. It was also difficult to interrogate the data and obtain statistical parameters. Some of the difficulties experienced may have been a consequence of faulty installation of the database program.
- It was concluded by the project team that for non-marine-influenced wetlands there are four basic wetland water quality types, based on drainage pattern and the water source. These are riverine, runoff-fed, and groundwater-fed systems, and freshwater lakes.

6.3 MAJOR PRODUCTS ARISING

The major products emanating from this part of the project are:

- a literature review of water quality and wetlands,
- a database (Access 2000) of water quality and other information for South African wetlands,
- an analysis of some of the major trends and patterns in water quality in South African wetlands,

- a Decision Support System to aid in setting the water quality component of the ecological Reserve for wetlands.

6.4 RECOMMENDATIONS AND FUTURE RESEARCH AREAS

The following recommendations and topics for further research are proposed:

- Development of assessment techniques, and if possible a rapid biomonitoring system for water quality in wetlands should be a high research priority. This should be underpinned by research into abiotic stressor-biotic response relationships, where water quality is one of the abiotic stressors included. Thus investigations of water quality-biotic response, distribution, tolerances and community structure of wetland biota are also required. In this way a more profound understanding of ecological functioning in these ecosystems will be obtained. It should then be possible to refine the method for defining water quality objectives, using information from chemical measurements, biomonitoring and ecotoxicology.
- The DSS developed in this project to facilitate setting water quality objectives for wetlands needs to be refined through use by wetland specialists. The particular aspects that need refinement are:
    - the recommended % deviation from the median (for each management class);
    - the use of the 80%ile from a comparable wetland;
    - the default "best-guess" values;
    - the classification system of wetland water quality types.
- It would be useful to continue adding data to the Wetlands Water Quality Database, and in particular, wetlands from regions other than the winter rainfall region should be included. This would aid in making the statistical analysis of wetland water quality more robust. In order to ensure compatibility, however, and to maximize the value, cognizance needs to be taken of the final form of the database used to house the National Wetlands Inventory, so that the two can be merged.
- Very few riverine wetlands were recorded in the database. It would be useful to look at these systems more closely and to see how (and if) they differ in terms of water quality from the rivers feeding them.

- During interrogation of the Wetland Water Quality Database, the correlation between rainfall areas and water quality was investigated. It would be useful to extend this work to ascertain if a link between for example, bioregions or water management areas and water quality could be found. This would further our understanding of water quality in the wetlands of this country. Furthermore, if strong correlations were found it would also facilitate in the setting of water quality objectives, since then the same objectives could be set for similar wetlands in the same area. The above work, however, could only be done if the database was first extended in the number of wetlands it covers.
- It was found that very few of the literature sources that were used to compile the Wetlands Water Quality Database recorded linked (i.e. measured at the same place and time) hydrological and water quality data. The question of linking water quantity and water quality requirements in wetlands is another important topic of research that requires attention.
- It would be useful if during the compilation of the National Wetland Inventory or during extension of the Wetlands Water Quality Database a system of Reference Conditions and/or sites could be identified. These would be wetlands of different types and regions that were un-impacted with regard to water quality (and other aspects such as habitat).
- It may also be useful to extend the database to the inclusion of biotic data e.g. the plant, fish or macroinvertebrate species present at a wetland site. These could then be correlated with water quality so that eventually an understanding of the tolerances and relationships between water chemistry and the species of plants could be established.
- In order to limit the levels of pollutants that enter into wetlands, it is suggested that there should be more stringent enforcement of existing legislation (e.g. Conservation of Agricultural Resources Act – act 43 of 1983) protecting natural vegetation adjacent to wetlands.
- The project team is assisting in refinement of the data that are collected by volunteers as part of CWAC (Coordinated Waterbird Counts). The project is run by the Avian Demography Unit at UCT. Thus, in addition to counting water birds, simple observations on wetlands (for example on water depth, colour, presence of algae) would also be collected. Successful volunteer monitoring programmes

are running in other countries (notably USA) and should be encouraged in South Africa.

- The topic of setting the lower limits of water quality objectives for wetlands, specifically for salinity requires further research. Minimum concentrations of inorganic salts need to be incorporated into the DSS.
- Ultimately sediment quality guidelines should be drawn up for aquatic ecosystems, and particularly those such as wetlands in which these materials accumulate. At the moment, however, not enough is known about water quality in general and that of sediments in particular to draw up such guidelines. To this end, monitoring programmes that are initiated to assess wetland water quality should also include analyses of the sediments. Similarly, databases that are developed in future to store wetland water quality data (e.g. that arising from the National Wetlands Inventory) should also be able to store sediment data.

6.5 CONCLUSION

For effective protection of water quality in wetlands it is essential that reasonably stringent pollution limits be set. At the same time, to enable optimum utilization of those systems for economic development, the water quality objectives should not be too conservative. A method to help determine suitable water quality objectives was developed in this project. The DSS is the result of preliminary deliberations and needs verification by application to many wetlands. It is recognized that "expert knowledge" currently plays a large role in applying this protocol and that the process is not definitive. There is, however, intense pressure on wetlands and many licence applications have, and are being received by DWAF for the discharge of effluents, or for activities that might impact on the water quality of wetlands (R. Stassen, Directorate of Resource Directed Measures, DWAF, *pers. com.* October 2003). Although not perfect, it is considered by the authors that DSS should at least help to set some limits to protect these assets. Wetlands have long been neglected in South Africa in terms of research and monitoring of all aspects, not just water quality. In a country where fresh water is limited it is important that attention now be directed towards these vitally important resources. Once more is known about these systems the methods of setting water quality objectives can be improved so that a greater confidence can be placed in the values obtained. Otherwise there is a very real danger of some wetlands being irretrievably damaged if steps are not taken to protect them.

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Vermaak, J.F., Swanepoel, J.H. and Schoonbee, H.J. (1981) The phosphorus cycle in Germiston Lake. 1. Investigational objectives and aspects of the limnology of the lake. *Water SA* **7**: 160-165.

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APPENDIX A
THE WATER QUALITY VARIABLES INCLUDED IN THE WETLANDS WATER QUALITY DATABASE

(max = maximum value, min = minimum value)

Chemical Species	Unit Of Measure (mg/L unless indicated otherwise)	Chemical Species	Unit Of Measure (mg/L unless indicated otherwise)
Al		Dissolved oxygen	% saturation
Al max		Dissolved oxygen	
Al min		Dissolved oxygen max	% saturation
Ammonium		Dissolved oxygen min	% saturation
Ammonium max		Euphotic zone depth	meters
Ammonium min		F	
B		F max	
B max		F min	
B min		Fe	micrograms per litre
Ba		Fe max	
Ba max		Fe min	
Ba min		HCO	
Ca		K	
Ca max		K max	
Ca min		K min	
Carbon (dissolved organic)		Kjeldahl nitrogen	
Carbon max		Kjeldahl nitrogen max	
Carbon min		Kjeldahl nitrogen min	
Cd		Mg	
Cd max		Mg max	
Cd min		Mg min	
Chlorophyll a		Mn	
chlorophyll a max		Mn max	
chlorophyll a min		Mn max	
Cl		Mn min	
Cl max		Mn min	
Cl min		Mo	
Co		Mo max	
Co max		Mo min	
Co min		Na	
CO3		Na max	
Conductivity	millisiemens per meter	Na min	
Conductivity max	millisiemens per meter	Ni	
Conductivity min	millisiemens per meter	Ni max	
Cr		Ni min	
Cr max		Nitrate	
Cr min		Nitrate and nitrite	
Cu		Nitrate and nitrite max	
Cu max		Nitrate and nitrite min	

Chemical Species	Unit Of Measure (mg/L unless indicated otherwise)	Chemical Species	Unit Of Measure (mg/L unless indicated otherwise)
Cu min		Nitrate max	
Nitrate min		SRP (Phosphate)	
Nitrite		SRP (Phosphate) max	
Nitrite max		SRP (Phosphate) min	
Nitrite min		Temp max	Degrees Celsius
Pb		Temp min	Degrees Celsius
Pb max		Temperature	Degrees Celsius
Pb min		TIN	
pH	Value	TIN max	
pH max	Value	TIN min	
pH min	Value	Total alkalinity	
Salinity	parts per thousand	Total alkalinity max	
Salinity		Total alkalinity min	
Salinity max	parts per thousand	Total Dissolved Solids	
Salinity min	parts per thousand	Total Nitrogen	
Secchi disc depth	meters	Total Phosphorus	
Secchi disc depth max	meters	Total phosphorus max	
Secchi disc depth min	meters	Total phosphorus min	
Si		TSS	
Si max		TSS max	
Si min		TSS min	
SO4		Turbidity	NTU
SO4 max		Turbidity max	NTU
SO4 min		Turbidity min	NTU
Sr		Zn	
Sr max		Zn max	
Sr min		Zn min	

APPENDIX B
SOURCES OF WATER QUALITY DATA USED IN THE WETLANDS WATER QUALITY DATABASE

The wetland information and water quality data entered into the database were obtained from several different types of sources, namely:

- Published papers
- Unpublished reports and theses
- DWAF chemical monitoring programme
- Municipalities

A list is given below of the individual sources used. Notes on the form and quality of the water quality data are included where these are relevant, including assumptions or possible inaccuracies in the data.

Published papers

Akhurst, E.G.J. and Breen, C.M. (1988) Ionic content as a factor influencing turbidity in two floodplain lakes after a flood. *Hydrobiologia* **160**: 19-31.

Allanson, B.R. and Hart, R.C. (1975) The primary production of Lake Sibaya, KwaZulu, South Africa. *Verh. Internat. Verein. Limnol.* **19**: 1426-1433.
WARNING POSSIBLE INACCURACIES: *Values read off graph. Only chlorophyll a recorded. Converted from mg/m^3 to mg/L.*

Ashton, P.J. and Schoeman, F.R. (1983) Limnological studies on the Pretoria salt pan, a hypersaline maar lake. *Hydrobiologia* **99**: 61-73.
WARNING POSSIBLE INACCURACIES: *Some of the values read off graphs. Samples taken every 30 days over a 28 month period (Feb 1978-May 1980). "Typical values" reported in paper. For some of the data, no dates given – therefore year recorded as "1800" in database. Measurements taken at different depths below surface, but only surface values entered into database. Values for nutrients considered by the project team to be abnormally high and therefore excluded when interrogating the database.*

Bruton, M.N. and Appleton, C.C. (1975) Survey of Mgobezeleni lake-system in Zululand, with a note on the effect of a bridge on the mangrove swamp. *Transactions of the Royal Society of South Africa* **41**: 283-294.
Water quality variable cited as "alkalinity" in paper recorded under "Total alkalinity" in database (reported pH of the lake less than 8.3, therefore all carbonate in the HCO_3^- form).

Coetzer, A. (1981) Some limnological observations of a temporary West Coast pan in southern Africa: Rocher Pan. *Hydrobiologia* **78**: 161-167.
This paper records measurements of inflow water and in the pan. Only the latter values recorded. Paper notes that samples from the last period of sampling (1978/1979) were analysed using a different method to 1974 and 1976/1977, and that there could be discrepancies. The variable cited as "Alkalinity ($CaCO_3$)" in the paper was recorded as "Total alkalinity" in the database. No pH values were given.

Coetzer, A.H. (1987) Succession in zooplankton and hydrophytes of a seasonal water on the west coast of South Africa. *Hydrobiologia* **148**: 193-210.
WARNING POSSIBLE INACCURACIES: *Values read off graphs. Four transects were taken across the vlei and water samples extracted from the middle point (recorded as Coetzer B1-B4 in the database). Orthophosphate values in the paper recorded under "SRP (Phosphate)".*

Coetzee, D.J. (1985) Zooplankton and some environmental conditions in De Hoop Vlei, a southern Cape coastal lake. *Bontebok* **5**: 10-16.
WARNING POSSIBLE INACCURACIES: *Values read off graphs. Values for surface samples recorded only. Percentage oxygen saturation recorded.*

Cowan, G.I. (1995) Wetlands of South Africa. South African Wetlands conservation programme series, Department of Environment Affairs and Tourism, Pretoria.

Cowan, G.I. and van Riet, W. (1998) A directory of South African wetlands. South African wetlands conservation programme series, Department of Environment Affairs and Tourism, Pretoria.

Day, J.A. (1993) The major ion chemistry of some southern African saline systems. *Hydrobiologia* **267**: 37-59.
This paper reports the cation and anion concentrations for wetlands in the Western Cape carried out by Silberbauer and King (1991). These values were converted from meq/l to mg/L. Salinity was converted from g/l to mgL. The original paper was consulted for data from Seaman et al. (1991), in addition to original DWAF data.

De Moor, F. and Barber-James, H.M. (1994) A survey of the freshwater invertebrates of northern Eastern Cape rivers. Contract report, 65pp.
Values sent by one of the authors. Conductivity converted from µS/cm to mS/m.

Hamer, M.L. and Appleton, C.C. (1991) Physical and chemical characteristics and phyllopod fauna of temporary pools in north-eastern Natal, Republic of South Africa. *Hydrobiologia* **212**: 95-104.
Conductivity values converted to mS/m.

Harding, W.R. (1992) A contribution to the knowledge of South African coastal vleis: The limnology and phytoplankton periodicity of Princess Vlei, Cape Peninsular. *Water SA* **18**: 121-130.
WARNING POSSIBLE INACCURACIES: *Values of variables read-off graphs. Recorded at tri-monthly intervals. "Inorganic nitrogen" recorded as nitrate.*

Harding, W.R. and Quick, A.J.R. (1992) Management options for shallow hypertrophic lakes, with particular reference to Zeekoevlei, Cape Town. South African *Journal of Aquatic Science* **18**: 3-19.
Mean values of water quality variables for the period April 1989 to March 1991 entered into the database. Year recorded as "1900" in the database. Sample size and standard deviations given in original paper. Values for both "soluble-P (mg/L= total filterable phosphorus" and "ortho-P (mg/L)" are given in the paper. Only the former are recorded in the database as "SRP (Phosphate)".

Hart, R.C. (1995) South African coastal lakes. Cown, G.I. (ed) Wetlands of South Africa. pp 103-130.
Values represent mean or spot values for various lakes. No dates given – therefore year recorded as "1800" in database. Values for nutrients converted from µg/L to mg/L. Where the values were indicated in the document as being questionable these were excluded from the database. The data are given under "Hart 1995" in the database but is summarized by Hart from many authors.

Harrison, A.D. (1962) Hydrobiological studies on alkaline and acid still waters in the Western Cape Province. *Transactions of the Royal Society of South Africa* **36**: 213- 243.
Conductivity values converted from micromhos to mS/m. It is not clear from the paper whether or not the conductivity was adjusted to 25˚C.

Howard-Williams, C. (1980) Aquatic macrophyte communities of the Wilderness Lakes: Community structure and associated environmental conditions. *Journal of the Limnological Society of Southern Africa* **6**: 85-92.
Values given are the means for the study period March – May 1979. Sample size (n) not reported.

King, J.M. (1987) Inventory of South-Western Cape wetlands. pp 76-90. *in* Walmsley, R.D. and Botten, M.L. (Eds.) *Proceedings of a symposium on the ecology and conservation of wetlands in South Africa.* Occasional Report Series No. 28. Ecosystems Programmes, FRD. Pretoria.
No dates given – therefore year recorded as "1800" in database.

Meintjes, S., Seaman, M.T. and Kok, D.J. (1994). Duration of inundation and change in physical and chemical characteristics of small temporary pans in South Africa. *Hydrobiologia* **281**: 79-90.
Minimum and maximum values recorded of various water quality variables over a period of several months. No dates were attached to values in the paper and so the mid-point of the specified time period was noted and recorded as the sampling date in the database.

Milstein P. le S. (1975) The biology of Barberspan, with special reference to the avifauna. The *Ostrich* (Supplement No. 100: 1-22).

Muspratt, J. and Henning, S.F. (1983) Mosquitoes of an inland salt marsh in the eastern Transvaal region of South Africa. *South African Journal of Science* **79**: 378- 379.
Water quality variable cited as "alkalinity" in paper recorded under "Total alkalinity" in database (pH less than 8.3, therefore all carbonate in the HCO_3^- form).

Prior, B.A. (1978) Properties of two halophilic bacteria from a salt pan. *Water SA* **4**:119-123.
No details of location or name of wetland given. Recorded as "Soutpan – Free State 1". Concentrations converted from µg/mL to mg/L.

Quick, A.J.R. and Harding, W.R. (1994) Management of a shallow estuarine lake for recreation and as a fish nursery: Zandvlei, Cape Town, South Africa. *Water SA* **20**: 289-297.
Mean, minimum and maximum values recorded for 1992-1993. Year recorded as "1900" in database.

Rogers, K.H., Ellery, W.N., Winternitz, N.L. and Dohmeier, R. (1989) Physical, chemical and biotic responses to decreasing water depth in a highveld pan following wet and dry summers. *South African Journal of Aquatic Science* **15**: 67-90.
WARNING POSSIBLE INACCURACIES: *Depth of water at time of sampling read off a graph. Values given in µg/l converted to mg/L and µS/cm converted to mS/m.*

Scott, H.A. and Hamman, K.C.D. (1988) Recent fish records from De Hoop Vlei, a southern Cape coastal lake. *Bontebok* **6**: 30-33.
No dates given – therefore year recorded as "1800" in database.

Seaman, M.T. (1987) Ecological diversity in Orange Free State pans. in Walmsley, R.D. and Botten, M.L. Symposium on the ecology and conservation of wetlands in South Africa. CSIR conference centre, Pretoria, 15-16 October 1987.
General ranges of water quality variables given. No site details or dates - therefore year recorded as "1800" in database. Conductivity values converted from µS/cm to mS/m.

Seaman, M.T., Ashton, P.J. and Williams, W.D. (1991) Inland salt waters of southern Africa. *Hydrobiologia* **210**: 75-91.
Wetlands cited in this paper were listed as "Probably Endorheic" under wetland type in the database. Only wetlands whose geographical coordinates indicated they were in South Africa were included. No dates given – therefore year recorded as "1800" in database.

Silberbauer, M.J. and King, J.M. (1991) Geographical trends in the water chemistry of wetlands in the south-western Cape Province, South Africa. *South African Journal of Aquatic Science* **17**: 82-88.
Salinity recorded as parts per thousand. Conductivity values also reported. Total Dissolved Salts determined for some sites, but not all. These values were not added to the database because of possible confusion with the parameter "Total Dissolved Solids". Values for anions and cations taken from Day (1993).

Vermaak, J.F., Swanepoel, J.H. and Schoonbee, H.J. (1981) The phosphorus cycle in Germiston Lake. 1. Investigational objectives and aspects of the limnology of the lake. *Water SA* **7**: 160-165.
Conductivity values converted from µS/cm to mS/m. "BOD" assumed to be the BOD after 5 days (APHA 1971 cited as reference for methods).

Unpublished reports

Burgers, C.J. (1984) Ramsar data: De Hoop Vlei. Jonkershoek Nature Conservation station, Stellenbosch. (No further details of report provided).

DWAF (1998) Overberg coastal water resources Volume III. Report on the groundwater resources, current and future water requirements of the coastal strip and adjoining hinterland between the Potberg and Quoin Point. Compiled by:Toens & Partners, T&P report no. 980145, Wynberg, Cape Town, 7824.
Only maximum and minimum conductivity values recorded. No dates given – therefore year recorded as "1800" in database.

EGS (2001) Environmental management plan for the Khayelitsha Wetlands Park. Department of Environmental and Geographical Science, University of Cape Town.

Measurements of water quality taken from lower boundary of the wetland system. General values given. Date recorded as "1800" in database.

Gardiner, A.J.C. (1988) A study on the water chemistry and plankton in blackwater lakelets of the southwestern Cape. Ph.D thesis, Dept. Zoology, University of Cape Town.
WARNING POSSIBLE INACCURACIES: *Values read-off graphs. Values given as µg/l converted to mg/L. Concentrations recorded as mM/litre or µM/litre converted to mg/L.*

Gardiner, A. and Day, J.A. (1982) The groot vleie of Bettys Bay: Grootwitvlei and Grootrondevlei - present and projected status. (No further details of report provided).
Only general ranges given. No dates given – therefore year recorded as "1800" in database. Chlorophyll a converted from µg/L to mg/L.

Jones, M.G.W. (2002) Developing a classification system for Western Cape wetlands. MSc. Thesis, Dept Zoology, University of Cape Town.
Where necessary concentrations converted from mM/L to mg/L.

Kyle, R. and Ward, M.C. (1988) Ramsar data: Kosi Lake System, KwaZulu-Natal. Summary of Ramsar designated sites. (No further details of report provided).
No dates given – therefore year recorded as "1800" in database.

McQuaid, J. (1997) Primary productivity of De Hoop Vlei, South Africa. (No further details of report provided).
Electrical conductivity reported as mS l^{-1}. Assumption made that the units were actually mS/m, since the values were consistent with this supposition.

Morant, P.D. (1981) Wetland classification: a review of existing wetland classification systems and their applicability in Southern Africa. MSc thesis, Environmental Studies, University of Cape Town.
Only wetlands entered into database that were listed in the above as "no marine influence". Conductivity measured in micromhos – converted to mS/m. Systems that were larger than 8 ha (or smaller than 8 ha and deeper than 2m) were classified as lacustrine and entered accordingly in the database (according to the classification system of DWAF 1999b). No effort was made to check if all the wetlands in the thesis were still in existence. The following "water quality types" were listed in the thesis and are recorded in the database.

- *Clear and colourless*
- *Clear and stained (i.e. typical brown staining)*
- *Turbid and mineral (i.e sand or silt – brown)*
- *Turbid and organic (i..e. phytoplankton – green)*

It was assumed (from similarity of the geographical position, surface area etc) that:
"Grootvlei west" (Morant) was "Groot Rondevlei 2"
"Grootvlei east" (Morant) was "Groot Witvlei"
"Heuning Rugt" (Morant) is "Honing Rug" in database.

Scott, H. A. (1986) De Hoop Nature Reserve. Cape Conservation series 7. Cape Department of Nature and Environment Conservation, Cape Town.
Just general values given in document. Date recorded as "1800".

Snaddon, C.D. and Ratcliffe, S.G. (1999) An ecological assessment of the Zoarvlei wetlands, Paarden Eiland, Western Cape Province. Report for Blaauwberg Municipality. *Some values of nitrate and nitrite, ammonia and phosphate were recorded as "<0.1". These were recorded in the database half of this value i.e. "0.05 mg/L". The exact days of sampling were not recorded but between May- June 1999. All samples were recorded under June 1999 in the database. Total suspended solids were reported separately as "organic" and 'inorganic" fractions. These were combined in the database under "TSS". Depth recorded in database as the deepest value of the wetland.*

Swart, D.R. (1992) Ramsar data: Barberspan. Chief Directorate: Nature and Environmental Conservation, Pretoria. (No further details of report provided). *Just general values given in document. Date recorded as "1800".*

Ward, M.C. and Kyle, R. (1990) Ramsar data: Sibaya Lake, KwaZulu-Natal. Summary of Ramsar designated sites. (No further details of report provided). *Just general values given in document. Date recorded as "1800".*

DWAF Chemical monitoring programme

The Department of Water Affairs and Forestry do not have a water quality monitoring programme dedicated to wetlands (R. Stassen, RDM Directorate, DWAF, *pers. com.* July 2002), although some wetlands appear to be monitored on somewhat *ad hoc* basis. Possible sources of data were identified by browsing through the list of stations covered under DWAF's monitoring programme (both chemical and hydrological). Wetlands were identified from the station name (i.e. names that included the term "wetland", "lake", "vlei", "marsh" or "pan") and where the station code was designated with an "R", rather than "H", indicating that the water is standing, rather than flowing. These wetlands (a total of 21) are listed in the table below (Table B.1). In the case of several wetlands, only hydrological information has been collected, but no chemical data. For many of the sites (e.g. Swartvlei or the Lake St Lucia system), although chemical data were available they were not entered into the database because of the known strong marine influence, and because the sampling site is close to the sea. If a wetland source was considered to be not suitable for the database (a total of 9) the reason for this is indicated in the table.

Of the sites remaining, for some (e.g. Verlorenvlei), less than a handful of samples had been recorded, whereas for others (e.g. Barberspan) long-term data-sets were available. In the latter data-rich situation, the number of samples (n), and the time-period over which the record extends, is recorded in the table. Because the value of *n* that is recorded was frequently not the same for all water quality variables, the smallest (usually for temperature, chlorophyll *a* or turbidity) and largest (usually for pH, conductivity) are given. The median value of each water quality variable, and the maximum and minimum values were entered into the database. In some wetlands, samples were taken from more than one site at a given wetlands. These are indicated by the code for the site (-Q01, -Q02 etc.). The year is recorded as "1900" in the database.

There are also over 40 springs, or eyes listed by DWAF as monitoring stations. The majority of these (but not all, see Table B.2) are, or have been at some stage been monitored for water quality. It was decided to include these in the database as a separate "wetland type" called "eyes". The rational for including these wetlands is that they represent a unique type of ecosystem that deserves to be protected. No attempt

was made, however, to determine to what extent the eyes have been impacted and thus to what extent they still maintain functioning ecosystems and vegetation.

Table B.1 *List of wetlands monitored by the Department of Water Affairs and Forestry*

Wetland and station code	Site code, time-period and sample size
Abiekwasputs pan - D4R002-Q01 Molopo River, Northern Cape.	Q01; 1/7/63 – 2/9/97; n = 2-1.
Barbers (Babers) pan - C3R003-Q01 North West province.	Q01; 14/3/72 – 1/9/2003; n = 356-10.
Bot river wetlands - G4R003-Q01 Kleinmond, Western Cape.	Q01; 25/4/78 – 9/6/03; n = 421–2.
De Hoop wetland - G5R001-Q01 Western Cape.	Q01; 30/7/65 – 30/7/02; n = 181-85.
Eilandvlei (Onder-langvlei) - K5R005-Q01, -Q02, -Q03, -Q04, -Q05. Wilderness lakes area.	Q01; 1/12/77 – 17/6/03; n = 297-122. Q02; 1/7/96 – 16/12/96; n = 6-4. Q03; 19/2/98 – 13/1/99; n = 12-6. Q04; 19/2/98 – 13/1/99; n = 12-4. Q05; 19/2/98 – 13/1/99; n = 11-6.
Groenvlei - K4R001-Q01, -Q02, -Q03, -Q04. Wilderness lakes area.	Q01; 5/4/78 – 17/6/03; n = 359-13. Q02; 5/5/97 – 12/1/99; n = 21-2. Q03; 4/2/98 – 5/1/99; n = 11-3. Q04; 6/2/98 – 5/1/99; n = 12-7.
Klein River wetland - G4R004 Hermanus	No water quality data.
Kosi lake (Ku-hlange) - W7R003 Kwa-Zulu Natal	Not included – strong marine influence.
Langvlei (Bo-langvlei) - K3R004-Q01, -Q02, -Q03, -Q04. Wilderness lakes.	Q01; 20/2/70 – 17/6/03; n = 313-2. Q02; 8/2/98 – 13/1/99; n = 11. Q03; 18/3/98 – 13/1/99; n = 11. Q04; 18/3/98 – 15/1/99; n = 10.
Mgobezelani lake - W7R002-Q01 Sodwana	Q01; 23/2/83 – 3/7/03; n = 207-3.
Mkuze marsh – W3H018 Kwa-Zulu Natal	No water quality data.
Msingazi lake - W1R004-Q01 Richards Bay	Q01; 12/5/83 – 3/7/03; n = 346-8.
Nsezi lake - W1R003-Q01 Empangeni, Kwa-Zulu Natal	Q01; 12/8/82 – 16/7/03; n = 276-2.
Onrus river wetland - G4R011 Western Cape	No water quality data.
Palmiet river wetland - G4R009 Western Cape	No water quality data.
Rondevlei – K3R003-Q01, -Q02, -Q03, -Q04. Wilderness lakes area.	Q01; 1/12/77 – 15/7/03; n = 341-2. Q02: 3/2/98 – 15/1/99; n = 11-3. Q03; 18/3/98 – 15/1/99; n = 11. Q04; 18/3/98 – 15/1/99; n = 11-10.
Sibaya lake - W7R001-Q01, -Q03. Kwa-Zulu Natal	Q01; 31/7/69 – 1/12/99; n = 213-3. Q03; 3/11/99 – 3/6/03; n = 37.

Sout pan - C3R004	No water quality data.
St Lucia lake - W3R002 Kwa-Zulu Natal	Not included – strong marine influence
Swartvlei - K4R002 Wilderness lakes area.	Not included – strong marine influence
Verlorevlei - G3R001-Q01. West Coast, Western Cape	Q01; 24/8/95 – 10/10/95; n = 2
Wadrift sout pan. - G3R002	No water quality data.

Table B.2 *List of springs (eyes) monitored by the Department of Water Affairs and Forestry*

Eye/spring and station code	Site code, time-period and sample size
Aslaagte eye (at Lichtenberg) C3H011	No water quality monitoring
Badfontein eye (at Aan de Cogmanskloof) H3H007	Q01; 4/3/70 - 23/9/97 n = 3
Bloemhof-eye (at Bloemhof) N1H014	Q01; 2/8/72 – 6/11/91 n = 175
Boplaas eye (at Boplaas) C9H013	Q01; 3/7/95 – 23/5/03 n = 27
Bothetheletsa eye (at Botheletsa loc.) D4H010	Q01; 30/5/77 – 4/8/99 n = 132
Bovenste eye (at Bovenste eye of Mooi River) C2H172	Q01; 7/1/97 – 16/4/03 n = 101
Buffelshoek eye (at Buffelshoek) A3H009	Q01; 26/7/77 – 26/2/03 n = 106
Doornfontein eye (at Doornfontein) A3H015	Q01; 27/7/77 – 14/18/95 n = 91
Doornplaat eye (at Doornplaat) A3H026	Q01; 25/7/77 – 26/7/93 n =60
Elandsfontein eye (at Elandsfontein) A2H008	Q01; 29/1/80 – 17/10/98 n =241
Eye of Elandsfontein C2H052	No water quality monitoring
Great Boetsap eye (at Boetsap loc.) C3H009	Q01; 15/3/72 – 15/7/93 n = 61
Great Koning eye (at Kono loc.) D4H009	Q01; 4/11/72 – 22/5/03 n = 137
Groot Marico eye (at Grootfontein) A3H004	No water quality monitoring
Kafferkraal eye (at Kafferkraal) A3H013	Q01; 28/9/71- 11/3/91 n = 45

Kareebosch eye (at Kareebosch) A3H020	Q01; 6/2/78 – 20/8/92 n = 53
Klaarstroom eye (at Kaarstroom) A3H012	Q01; 28/9/71 – 28/9/71 n = 1
Koloniesplaas-eye (at Sevenfontein) N1H009	No water quality data
Kraalkop eye (at Kraalkop) C2H051	No water quality data
Kuruman eye A (at Kuruman) D4H005	Q01; 16/3/72 – 19/05/03 n = 50
Kuruman eye B (at Kuruman, Rensville) D4H006	Q01; 11/8/76 – 24/9/98 n = 25
Little Koning eye (at Kono loc.) D4H008	Q01; 3/10/77 – 3/9/02 n = 159
Lower eye at Tweefontein (at Tweefontein road) A1H004	Q01; 28/7/77 – 26/8/92 n = 86
Lower eye Dinokana (at Dinokana at dam) A1H002	Q01; 28/7/77 – 9/9/02 n = 116
Malmanie eye Lower (at Pardenvallei) A3H022	Q01; 19/12/79 – 8/10/03 n = 101
Malmanie eye Upper (at Pardenvallei) A3H021	Q01; 29/9/71 – 8/10/03 n = 74
Maloney's eye (at Steenekoppie) A2H010	Q01; 9/2/78 – 8/9/03 n = 188
Manyeding eye (at Manyeding loc.) D4H007	Q01; 13/6/77 – 19/5/03 n = 246
Molopo eye (left overflow gate) D4H014	Q01; 29/6/83 – 10/09/02 n = 70
Mostershoek eye (at Mostershoek) C5H021	Q01; 29/10/75 – 6/8/99 n = 152
Nouklip eye (at Hartbeeshoek) A2H003	No water quality data
Olivendraai eye (at Olivendraai) D4H017	Q01; 28/9/71 – 4/8/99 n = 49
Polfontein (at Matlabes loc.) D4H019	Q01; 31/3/81 – 3/9/91 n = 23
Rietfontein eye (at Rietfontein) B2H002	No water quality monitoring
Rietpoort eye at (at Rietpoort) A3H014	Q01; 28/9/71 – 14/10/96 n = 51
Skilpad eye (at Moilwas) A1H005	Q01; 26/9/71 – 6/11/96 n = 101
Stinkhoutboom eye (at Stinkhoutboom) A3H010	Q01; 24/8/77 – 29/11/00 n = 110
Thabasikwa eye (at Taung loc.) C3H010	Q01; 23/10/75 – 2/10/88 n = 129
Tsineng eye (at Lower Kuruman, Tsineng) D4H011	Q01; 14/1/88 – 20/1/96 n = 13
Turfontein-eye-under (at Turfontein) C2H094	No water quality monitoring

Uitenhague Springs (at Uitenhague) M1H003	Q01; 4/9/68 – 21/2/91 n =67
Upper eye at Tweefontein (at Tweefontein) A1H003	Q01; 25/10/77 – 22/6/93 n = 87
Upper eye Dinokana (at Dinokana) A1H001	Q01; 26/9/71 – 6/10/03 n = 135
Vergenoegd eye (at Vergenoegd) A3H011	Q01; 27/9/71 – 26/8/92 n =55
Vlakfontein eye (at Metsematshwe loc.) C3H012	Q01; 15/3/71 – 15/7/93 n = 203
Welgedacht eye (at Welgedacht) D4H016	Q01; 29/6/83 – 18/8/97 n = 37
Weltevrede eye (at Weltevreden) D4H018	No water quality monitoring
Wonderfontein eye C2H128	No water quality monitoring

CMC wetland water quality data

The Cape Metropolitan Council has a water quality monitoring programme at twelve wetlands in the Cape Town area. The data-set is quite extensive since some wetlands have been monitored since 1981, and the larger wetlands are sampled at more than one site (although not all sites have been entered into the Wetlands Water Quality Database). The suite of variables that are monitored is given below. Note that at a few sites, not all variables are measured.

Chemical oxygen demand	Soluble nitrate and nitrite
Chlorophyll *a* (corrected for phaeophytin)	Soluble phosphorus
Dissolved oxygen	Soluble reactive silica
Electrical conductivity	Total alkalinity
Ortho-phosphate	Total kjeldahl nitrogen – (no longer measured)
Oxygen saturation	Total nitrogen (alkaline persulphate oxidation)
Phaeophytin	Total suspended solid
Salinity	Water temperature
Secchi transparency	
Soluble ammonia	

The sampling data were statistically summarized before being entered into the database. The complete time-series was broken into 5 year intervals and the median, minimum and maximum values for each sub-period calculated and entered in the database. The middle year of the time-period was recorded as the date. Table B.3 shows for each wetland, the total time-period for which monitoring data are available. For each five year sub-period, the maximum and minimum sample size is given (not all variables are measured at all sites, so that the sample size varies). The year entered into the database is also shown. Where data for more than one site at a given wetland are recorded in the database, the site name is given.

Table B.3 *List of wetlands monitored by the Cape Town Municipal Council.*

LITTLE PRINCESS VLEI (TIME-PERIOD OF MONITORING: 11/08/89 – 1/4/04)			
Site name	**Time-period**	**Sample size**	**Year in database**
CMC- North	11/08/89 – 3/17/03	66 - 24	1991
	1/17/95 – 12/9/99	41 - 6	1997
	3/1/00 – 3/17/03	39 - 1	2001
CMC- South	10/11/89 – 12/20/93	69 - 24	1991
	1/3/94 – 12/9/99	62 - 1	1996
	3/1/00 – 1/4/04	49 - 1	2002
LANGEVLEI (TIME-PERIOD OF MONITORING: 7/1/91 – 1/4/04			
Site name	**Time-period**	**Sample size**	**Year in database**
	7/1/91 – 11/13/95	93 - 38	1993
	1/11/96 – 12/6/00	59 - 34	1998
	1/9/01 – 1/4/04	37 - 34	2002
PRINCESS VLEI (TIME-PERIOD OF MONITORING: 8/11/82 – 3/4/03)			
Site name	**Time-period**	**Sample size**	**Year in database**
CMC-North	8/11/82 – 12/18/86	25 - 13	1984
	2/24/87 – 12/17/91	85 - 5	1989
	1/7/92 – 11/28/96	81 - 12	1994
	1/23/97 – 12/10/01	52 - 18	1999
	1/28/02 – 3/4/03	14 - 8	2002
CMC-South	8/11/82 – 8/1/85	21 - 11	1983
	10/18/88 – 12/12/92	64 - 7	1990
	1/18/93 – 12/11/97	48 - 3	1995
	1/29/98 – 3/4/03	47 - 9	2000
RONDEVLEI (TIME-PERIOD OF MONITORING: 3/20/98 – 1/22/04)			
Site name	**Time-period**	**Sample size**	**Year in database**
	3/20/98 – 12/11/02	60 - 19	2000
	1/22/03 – 1/22/04	11 - 9	2003
GLENCAIRN VLEI (TIME-PERIOD OF MONITORING: 6/2/98 – 1/22/03)			
Site name	**Time-period**	**Sample size**	**Year in database**
	6/2/98 – 1/22/03	47-13	2000
ZANDVLEI NORTH (TIME-PERIOD OF MONITORING: 3/2/78 – 5/29/03)			
Site name	**Time-period**	**Sample size**	**Year in database**
	3/2/78 – 12/22/82	109 - 69	1980
	1/6/83 – 12/29/87	109 - 19	1985
	1/12/88 – 12/18/92	112 - 30	1990
	1/11/93 – 12/9/97	79 - 49	1995
	1/20/98 – 5/29/03	50 - 1	2000

WEST LAKE WETLAND (TIME-PERIOD OF MONITORING: 3/9/94 – 1/29/04)			
Site name	**Time-period**	**Sample size**	**Year in database**
	3/9/94 – 12/2/98	61-12	1996
	2/2/99 – 1/29/04	53 - 2	2001
ZEEKOEVLEI (TIME-PERIOD OF MONITORING: 8/25/81 – 1/13/04)			
Site name	**Time-period**	**Sample size**	**Year in database**
CMC-Yacht club	8/25/81 – 11/12/85	66 - 25	1983
	1/7/86 – 12/27/90	114 - 18	1987
	1/4/91 – 11/6/95	100 - 27	1993
	2/8/96 – 12/13/00	64 - 20	1998
	1/9/01 – 1/13/04	37 - 1	2002
CMC-Aquatic club	8/25/81 – 11/12/85	65 - 26	1983
	1/7/86 – 12/20/90	115 - 21	1988
	1/4/91 – 11/6/95	109 - 43	1993
	2/8/96 – 12/13/00	50 - 24	1998
	1/9/01 – 1/13/04	37 - 17	2002
WILDEVOELVLEI (TIME-PERIOD OF MONITORING: 11/27/97 – 1/21/04)			
Site name	**Time-period**	**Sample size**	**Year in database**
	11/27/97 – 12/11/01	75 -40	1999
	1/22/02 – 1/21/04	25 -1	2003
ZOARVLEI (TIME-PERIOD OF MONITORING: 2/23/88 – 1/27/04)			
Site name	**Time-period**	**Sample size**	**Year in database**
	2/23/88 – 12/2/92	45 - 31	1990
	5/5/93 – 1/8/97	20 - 2	1995
	7/2/98 – 1/27/04	42 - 36	2000
DIE OOG (TIME-PERIOD OF MONITORING: 7/7/98 – 1/29/04			
Site name	**Time-period**	**Sample size**	**Year in database**
	7/7/98 – 1/29/04	60 - 5	2001
SCHUSTERS WETLAND (TIME-PERIOD OF MONITORING: 10/17/01-12/18/02)			
Site name	**Time-period**	**Sample size**	**Year in database**
	10/17/01 – 12/18/02	10 - 8	2001
RIETVLEI (TIME-PERIOD OF MONITORING: 8/31/99-12/22/03)			
Site name	**Time-period**	**Sample size**	**Year in database**
	8/31/99 – 12/22/03	55 - 34	2001

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Dallas HF; Day JA

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