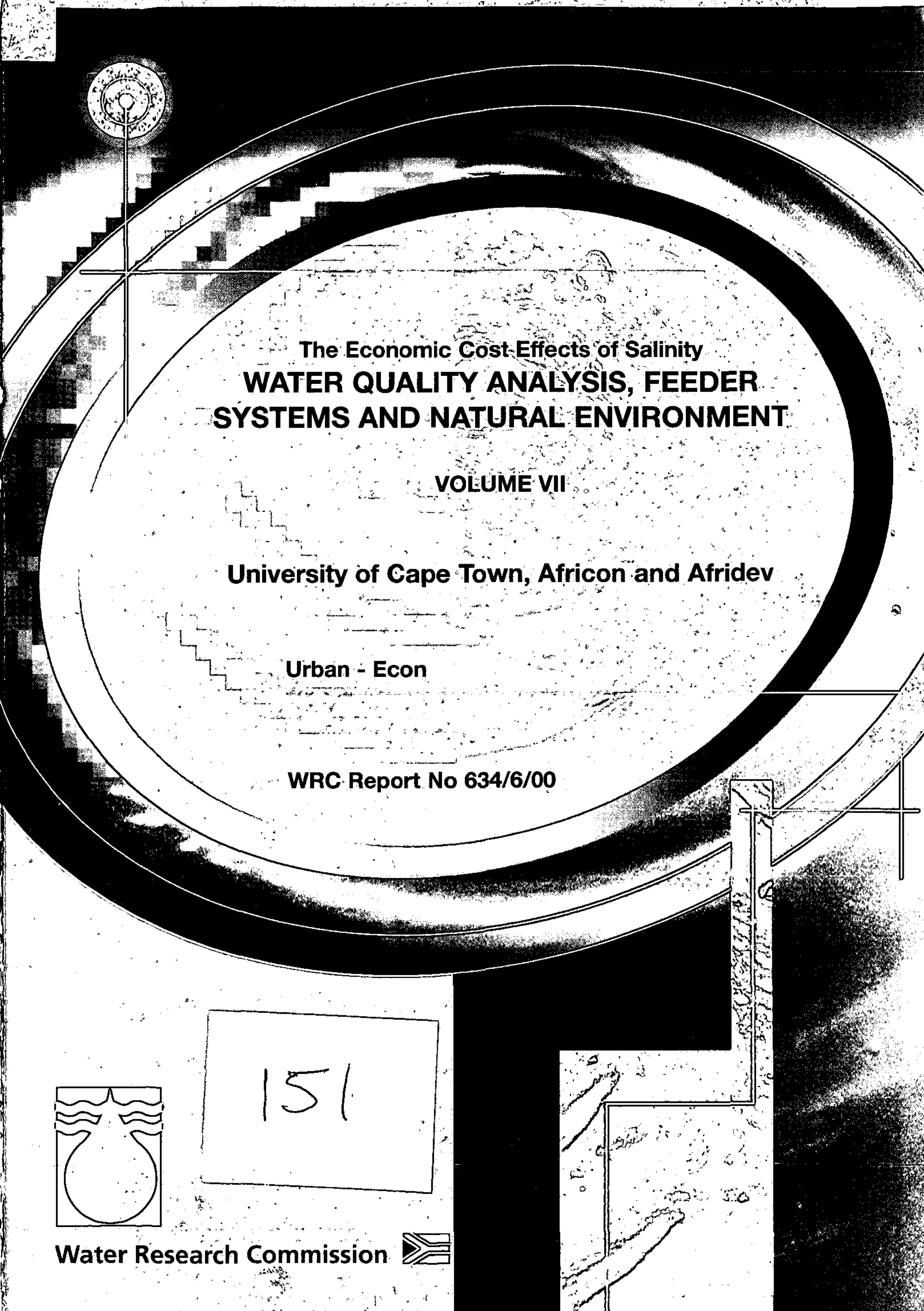

The Economic Cost Effects of Salinity

WATER QUALITY ANALYSIS, FEEDER SYSTEMS AND NATURAL ENVIRONMENT

VOLUME VII

University of Cape Town, Africon and Afridev

Urban - Econ

WRC Report No 634/6/00

Water Research Commission

VOLUME VII

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WATER QUALITY ANALYSIS, FEEDER SYSTEMS AND NATURAL ENVIRONMENT

by the

University of Cape Town, Africon and Afridev

Report to the Water Research Commission and
the Department of Water Affairs and Forestry
on the project
Determining the impact of the salinisation of
South Africa's water resources
with respect to economic effects

Project Leader: Urban-Econ

WRC Report No.: 634/6/00
ISBN No. : 1 86845 742 7
ISBN Set No. : 1 86845 595 5

EXECUTIVE SUMMARY

1 PURPOSE OF THE STUDY

The Water Research Commission and the Department of Water Affairs and Forestry commissioned an investigation into the economic, social and behavioural impacts that would result due to changes in the salinity of South Africa's water resources.

The aim of the study was, primarily, to develop a generalised methodology model to determine the generic impact of changes in the total salt concentration found in South African rivers and to interpret these impacts in financial, economic and social terms. The resultant model was required to be:

i. comprehensive with respect to addressing the salinity problems
ii. applicable to any salinity situation in any water sector in South Africa.

An important role of the study was to verify the generalised model. This was achieved by applying it to a specific geographic area, namely the Middle Vaal River area. In order to achieve this, actual data gathering exercises were conducted and applied in the conceptual model. Based on this, a process of verification and model adjustments was undertaken, to incorporate the distinctive circumstances pertaining specifically to the Middle Vaal River area.

A generic model, making provision for all possible conceptual elements applicable to salinisation, has thus resulted. The model comprises separate equations representing the different sectors of the economy as well as the natural environment and water feeder systems. An outstanding feature of the model is that it is a generalised model and as such is applicable to any salinisation situation in South Africa.

The value of the study lies in applying the findings of the study in a policy environment. This means that the study results can provide motivation to formulate new policy directives for utilising water resources. in a given area.

2 BACKGROUND TO STUDY

There has been a steady increase in the salt content of the Vaal River since 1935. This increase has accelerated markedly since 1965 with a further pronounced effect caused by the droughts prior to 1996. This increase in the salt content affects all water use components exposed to such water.

A major salinity problem exists in the Middle Vaal River area, between the Barrage and Bloemhof Dam. Various options for solving the problem have already been identified. All the options are, however, costly and it is important to quantify the benefits of a reduction in salt concentration in order to justify expenditure on measures to reduce the salinity.

3 *OBJECTIVES*

Prior to deciding how salinity in the water supply could be managed, it is necessary to determine the total cost of salinity to the economy, namely its direct, indirect and induced cost effects. Costs borne by the various sectors in the economy have to be determined, including the identification of behavioural impacts. The study addresses the impacts of increased level of salinity throughout the economy.

In order to address the uncertainties with respect to the economic implications of salinity, the Water Research Commission identified the need to develop a methodology that can be utilised in difficult salinity situations.

The project was divided into two phases:

- The development of a generalised methodology for the determination of the generic impact of salt concentration of South African rivers and the interpretation of these impacts.
- The application of the methodology to an investigation of the impact of increased salt concentration in the Middle Vaal River.

4 FORMAT OF RESULTS

The research conducted to determine the economic effects of salinity is based on a sectoral approach. The economy had been classified into different sectors and research was conducted separately for each sector. These results were integrated to determine the total economic effects on the economy. On account of the volume of research results, the sectoral research is presented in separate volumes to support the integrated results contained in the main report (Volume I).

Each of the sectoral reports, combining its initial inputs for the generalised model with its findings in the case study, has been separately bound. These are individually available as:

Volume II : Household Sector
Volume III : Agriculture Sector
Volume IV : Mining Sector
Volume V : Industrial Sector
Volume VI : Services Sector
Volume VII : Water Quality Analysis, Feeder Systems, Natural Environment.

As the main report is an integration and interpretation of the background research, variations may occur. The background research should be interpreted as the development of a reference framework by the different specialists and during the course of the study, research findings were continually refined.

5 STUDY APPROACH

The approach followed with the study is based on economic theory by conceptualising sectoral behaviour within the economy. In quantifying these conceptual formulae, surveys were undertaken in the Middle Vaal River study area to obtain the direct costs related to salinity. These direct costs represent only a partial estimate of the total costs of salinity. In order to determine the indirect costs and other spin-off effects, an integrated costing framework had to be set up. This was done by utilising the input-output (IO) technique and a combination of IO applications.

Despite the inherent limitations of the IO technique, it is a very versatile and flexible model to simulate real-world situations. Furthermore, its ability to determine the indirect and induced cost effects, renders the approach as well as the results unique and comprehensive.

The sectors analysed are households, agriculture, mining, industry, services and feeder systems, as well as the natural environment. Conceptual cost formulae were formulated to determine the direct costs and behavioural impacts on costs for different levels of salinity. Based on this background research to set develop these formulae, the research results indicated that both the feeder systems and the natural environment would not incur significant (incremental) costs within the specified salinity range of

200 mg/ℓ to 1200 mg/ℓ Total Dissolved Solids (TDS). These two sectors were therefore not incorporated into the integrated model.

Upon conducting surveys in the study area to determine the direct sectoral costs, a variety of problems was encountered. The most important of these is the fact that many respondents (i.e. sectoral water users) are not aware of the costs of salinity and therefore assigning costs to behaviour becomes rather presumptuous. Behaviour does, however, play an important role in the household and agricultural sectors. With the other sectors, behaviour is driven by technology and production factors.

The survey results obtained in the Middle Vaal River study area were analysed and transformed where necessary, to be integrated into the IO modelling framework. The following approaches were followed:

- Conducting a multiplier analysis that provides a first approximation of the additional costs of salinity due to a change in the TDS and using this to rank sectoral sensitivities with respect to the impact of salinity.
- Setting up a pricing model that simulated the cost increases of different levels of salinity in terms of price changes being passed on as price increases. These price changes are passed on as price increases to all sectors of the economy and can be interpreted as proxies for changes in the Consumer Price Index (CPI) and Producer Price Index (PPI).
- Running an augmented IO model to estimate total additional resource usage as salinity rises. To cost this, a new industry was postulated to enter the economy to combat salinity. A new row and column representative of this industry was inserted into the IO table.

Each of these approaches focused on a different aspect in determining the total cost effects of different levels of salinity.

6 INTERPRETATION OF RESULTS

The results obtained with the IO analyses indicated that the total costs of salinity are significant in the Middle Vaal River study area.

6.1 Direct Cost Effects

The direct costs of salinity to the entire economy of the case study area are established from the mathematical combination of the survey data collected within each individual sector. There are constraints with much of the data, since most interviewees were unable to supply data for any conditions other than those currently being experienced and were generally rather uninformed about salinity and its potential effects.

Despite the drawbacks, the data provided some indication of the direct economic effects of increased salinity. The collected data was centred around 500 mg/ℓwhich is the average salinity level presently experienced in the study area. Data for salinity levels below 500 mg/ℓ implies a corresponding saving at these lower salinity levels. A 100 mg/ℓ increase in the TDS to 600 mg/ℓ is expected to effect a R26 million increase in annual direct costs in the study area (refer to Table 1). Increasing the TDS to the highest limit (1200 mg/ℓ) is expected to result in a direct cost of R183 million/a to the region. Conversely, a saving of R80 million/a is anticipated should the salinity drop from current levels to 200 mg/ℓ.

Table 1. Direct Costs of Salinity, (1995 Values in Millions of SA rands)

SALINITY SECTOR	mg/ℓ TDS						% Contribution at 600 mg/ℓ
	200	400	600	800ℓ	1000	1200	
Mining	(7.309)	(2.212)	0.844	4.863	10.209	17.816	3.17
Business and Services	(1.843)	0.487	1.211	1.707	2.209	2.697	4.55
Manufacturing 1	(0.145)	0.028	0.086	0.123	0.160	0.198	0.32
Manufacturing 2	(2.825)	0.294	1.351	1.993	2.635	3.278	5.07
Agriculture	0.000	0.000	0.439	0.439	0.427	0.503	1.65
Households (suburban)	(35.121)	(11.707)	11.707	35.121	58.535	81.949	43.94
Households (township)	(27.927)	(9.309)	9.309	27.927	46.544	65.162	34.94
Households (informal)	(5.081)	(1.694)	1.694	5.081	8.469	11.855	6.36
TOTALS	(80.251)	(24.113)	26.640	77.253	129.225	183.457	100.00

In considering these direct cost changes the effects can be equated to changes in prices in the economy. The percentage direct impact of salinity abatement on CPI and PPI at a salinity level of 600 mg/ℓ TDS, amounts to 0.0013% and 0.0016% respectively. In effect this implies a relatively small change in these indices which can be equated to changes in inflation.

The greatest direct cost implications occur to the household sector. The direct costs to the households comprise approximately 85% of the total direct costs within the economy under investigation. This is not unexpected, since the household sector comprises the largest group of treated water users in the economy even though the per capita cost increases are not the highest. Conversely, the sectors that use very little water and those using predominantly untreated water are expected to have lower direct cost effects.

Manufacturing 1, where water requires no treatment, has a relatively low water consumption and experiences less than 0.5% of the direct cost of salinity increases at 600 mg/ℓ. By way of contrast, business and services, a relatively large sector within the economy, can be attributed with 4.5% of the total direct costs, while Manufacturing 2 (which treats its water) will face cost increases owing to the costs of treatment. Thus, unsurprisingly, this latter sector experiences 5% of the direct costs to the economy.

Although the mining sector uses large volumes of water in terms of production, much of the water employed is used in re-circulating circuits. Further, this water does not, in general, require a high degree of purification and thus the costs are lower than might otherwise be expected (3%).

Similarly, most of the water employed in the agricultural sector is drawn directly from the river itself. The water costs to agriculture are low, and agriculture is a small sector, occupying a fairly narrow band along the Vaal River. Thus, agriculture occupies a small niche in the economy and its contribution of 1.5% of the total direct costs of the study region, is not unexpected.

6.2 Indirect and Induced Effects

The models employed for the case study calculated the direct, indirect and induced costs to the economy. Since the IO table was closed with respect to households, an allowance was made for the reciprocal relationships between income and consumption, as well as the impact on the economy, resulting from the interdependence of industries in their production process and the behaviour of households. The closing of the IO table effectively added another industry to the economy. Households have a large impact on the economic processes in the region of study and wider, resulting in the expectation of larger impacts than would have been anticipated if the table had been in its open format, considering direct and indirect effects alone.

Ratios of the direct, indirect and induced costs to the direct costs (Direct Cost Multipliers, DCM) determined by means of the multiplier analysis, range from 1 to about 3.3. This implies that the spin-off effects of increased salinity are significant and the direct costs alone are a poor reflection of the cost impacts of salinity.

The ranking of the sectors researched, based on the salinity multipliers, indicates that at relatively low levels of salinity it is the community and other service sectors which will be most adversely affected. At high levels of salinity the gold mining sector will have to incur the highest cost to combat salinity.

The results of the pricing model are expressed in terms of percentage changes in the consumer and producer price indices and essentially represent forward linkages. The impacts have been determined in terms of regional and national impacts. Considering only the impact on the productive sectors, results of the same order as the multipliers provided are found, but with less spread. The direct and indirect DCMs for PPI and CPI are found to lie between 1.36 and 1.84, whilst the direct, indirect and induced DCMs are found to lie between 1.96 and 3.5. It should be noted that the pricing model results indicate variables for a base year expressed in percentages. This implies annual changes in costs or prices.

The percentage total increases in CPI and PPI for salinity levels increases from 600 mg/ℓ to 1200 mg/ℓ can be summarised as indicates in Table 2:

Table 2: Percentage increase in price indices

Salinity abatement by :	**CPI: % change**	**PPI: % change**
Productive sectors	-0.008 to 0.01	-0.01 to 0.015
Productive sectors & households	-0.1. to 0.22	-0.11 to 0.26

These changes seem to be small but are significant when related to Rand values in regional and national context. This had been done and the regional and national annual impacts are summarised as indicated in Table 3:

Table 3: Impacts on price indices

IMPACTS	**CPI**	**PPI**
National increase		
600 mg/ℓ	R101.5m	R402.6m
1200 mg/ℓ	R647.5m	R2623.4m
Regional increase		
600 mg/ℓ	R7.4m	R18.0m
1200 mg/ℓ	R47.1m	R117.3m

The augmented model was executed using both regional and national IO tables to determine the total cost effects of salinity abatement. Multipliers were calculated for comparison with the other model applications.

The chief outcome was that the DCM was 3.0 for the national case, and 2.6 for the regional case. These figures did not change significantly over the salinity range of 600 mg/ℓ to 1200 mg/ℓ TDS. The difference in the national and regional DCM is due to the differences in structure between the national and the regional economies. Since the IO analysis is based upon coefficients, the actual size of the economies has no influence on the DCMs. Only changes in the size of the input (or technical) coefficients (which in turn reflects a change in the structure of the tables) would influence the outcome.

6.3 Behavioral Effects

The decisions regarding salinity changes made in the mining, business and services and the manufacturing sectors tend to be driven by technology and production regimes. These sectors are likely to make changes to combat the effects of salinity, based purely on the financial implications to the concern. As a result, there are few, if any, unexpected responses to salinity effects and the calculated costs can be accepted as reliable.

During the data collection in the agricultural sector, the cost effects of two possible scenarios, based on management decisions or behaviour, were established. These included a "best case" scenario, where the farmer would 'maintain the current levels of production, regardless of cost, and a second scenario, where the farmer would choose to allow the crop yields to be reduced. This was only done for the hybrid model and the overall costs to the economy were found to be hardly affected by the two alternatives. At the 600 mg/ℓlevel, the total costs decrease by less than R0.3 million. The variations are found to be between 0.1% and 0.3% of the overall costs, which are less significant than the probable errors in the data. Thus, the different behavioural responses available in the agricultural sector are unlikely to impact on the total costs to the economy.

The most significant behavioural effects are, however, from the household sector. The responses to increased salinity, while to some extent determined by the need to adapt to the changes, are largely driven by the availability of finances to maintain the *status quo* and overcome the problems arising from increased salinity. These behavioural responses are more likely to appear in those sectors of lower earning potential, and the informal household sector is far less likely to effect changes arising from increased salinity than suburban households. This is borne out by the variance in the data collected.

7 CONCLUSIONS

Based on the output from the model established for the Middle Vaal River region, the economic costs attributable to changing salinity, have been determined.

There exists an effective limit to the cost of salinisation. This is determined by the cost of desalinating the bulk water supply which would represent the most costly option of water treatment. Care must be taken not to allow the costs of salinisation to reach high levels. The viability of desalinating may be increased if selective desalination is applied to the consumer sectors incurring the highest relative costs, although other options should be explored first.

This is obviously a simplistic first-line approach, but it highlights the need to consider bulk or partial treatment of the water supply in the Middle Vaal area as the *status quo* is already 500 mg/ℓ. Behavioural response is particularly important as the quality of the water in the area is already perceived to be problematic.

The results of the study identified the total economic effects of increased salinity levels for the Middle Vaal River area. Based on these findings and the knowledge gained with respect to behaviour, the following observations are made:

- The application of the generic model in the Middle Vaal River area was accompanied by some limitations mainly on account of the undiversified economic structure. Undiversified, in this regard, refers to the strong reliance of the economy on the mining and services sectors. Despite this, very significant information could be obtained on the relative importance of cost effects between the various sectors. To validate these, the model should be applied in a diversified economy such as that of the Gauteng area. More insight concerning relative costs could be obtained on, for instance, the manufacturing sector.

- Differential desalination may be considered. The reason for this is that the household sector has been found to bear relatively high costs in terms of combating salinity, followed by the industrial and services sectors. It may be of value to motivate differential desalination of waters on a purely experimental basis, that is, to study the social and socio-economic benefits to be gained by households if the costs of salinity are decreased. This also implies that sectors that experience relatively low salinity costs may have to continue bearing these.

- It may benefit water users if a salinity awareness campaign were introduced. If end users were made aware of the cost effects, they might choose to behave differently and take informed decisions which may lessen the costs of salinity.

- A specialised database has been established. As part of an awareness campaign, users can contribute towards the refinement and extension of a more comprehensive database that can be utilised when the model is applied elsewhere. Since the availability of the data determines to a large extent the robustness of the model, such a database can contribute significantly to the ease of applying the model.

The interpretation of the findings of this study does not take into account alternative options with respect to water provision. This implies that the costs of salinity have not been related or compared to the situation of utilising transfer water and other allocation options. Furthermore, the results of the study are expressed in direct and spin-off effects and thus any further interpretation or comparison of these results with specific options, should be done in the same manner, namely to refer to total costs.

8 FURTHER RESEARCH

The value of the study lies in the fact that a first approximation of the spin-off effects of salinity on the economy had been determined. Furthermore, an indication of the behavioural costs for specific sectors has been obtained. On account of the specific study area chosen and the difficulties encountered in applying an integrated economic cost model to its specific considerations/circumstances, the following shortcomings may be addressed with further research:

- ☐ Application of the model in a relatively diversified economy such as that of Gauteng. In doing this, a more disaggregated model can be executed. Cost effects for more subsectors may then be identified, such as for the leather industry. Based on expectations the total costs of salinity may be higher.

- ☐ In applying the model to a chosen study area more significant costs may be identified if the study population is made aware of the problem in advance. The benefits arising from this approach, namely more accurate cost estimates and possibly more correct reporting of behaviour, could outweigh potentially over-reporting, due to increased awareness of the problem.

ACKNOWLEDGEMENTS

The research in this report emanated from a project funded by the Water Research Commission and entitled:

DETERMINING THE IMPACT OF THE SALINISATION OF SOUTH AFRICA'S WATER RESOURCES WITH RESPECT TO ECONOMIC EFFECTS

The Steering Committee responsible for this project, consisted of the following persons

Dr D van Driel	Department of Water Affairs & Forestry
Dr G C Green	Water Research Commission
Mr C Swiegers	Department of Water Affairs & Forestry
Mr F C van Zyl	Department of Water Affairs & Forestry
Mr F X Jurgens	Development Bank of South Africa
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Mr P H van Niekerk	Department of Water Affairs & Forestry
Ms A Belcher	Department of Water Affairs & Forestry

The financing of the project by the Water Research Commission and the Department of Water Affairs and the contribution of the members of the Steering Committee is acknowledged gratefully.

This project was only possible with the cooperation of many individuals and institutions. The authors therefore wish to record their sincere thanks to them.

SECTION A : INTRODUCTION

1. INTRODUCTION

The purpose of this document is to present the findings of the analyses conducted for the non-economic sectors in determining the cost impacts of salinity. The three analyses included in this volume are the following :

i. Section B : Water quality analysis
ii. Section C : Feeder system analysis
iii. Section D : Natural environment analysis.

The water quality analysis was undertaken to determine the estimation of salinity (TDS) from parameters commonly measured in water analyses, speciation of expected principal ionic matrices in terms of TDS, and assessment of the propensity for precipitation of salts from the water for the study area, ie the Middle Vaal River Area.

Conceptual formulae have been formulated to determine the potential cost impact of salinity on feeder systems and the natural environment. In both cases it was found that the impact of salinity can not be costed and therefore these two sectors have not been included in the integrated model. This implies that no incremental costs are incurred in the TDS range of 200mg/*ℓ* to 1200mg/*ℓ*.

SECTION B : WATER QUALITY ANALYSIS

SALINIZATION OF WATER IN THE MIDDLE VAAL REGION

SALINITY CHARACTERISATION

by

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August, 1995

SALINIZATION OF WATER IN THE MIDDLE VAAL REGION

SALINITY CHARACTERIZATION AND SATURATION STATE IN THE MIDDLE VAAL REGION

EXECUTIVE SUMMARY

This study forms part of an assessment of the economic impact arising from increase in inorganic dissolved salts in water in the Middle Vaal Region. In particular, the report addresses estimation of salinity (total dissolved salts, TDS) from parameters commonly measured in water analyses (eg. electrical conductivity), speciation of the expected principal ionic matrics in terms of TDS, and assessment of the propensity for precipitation of salts from the water.

With regard to quantifying salinity (TDS) in terms of electrical conductivity (E_c), regression of measured data in the region gives the following empirical relationship (for $50 < TDS < 600$ mg/ℓ)

$TDS = 7{,}30.\ E_c - 34$

E_c = electrical conductivity in mille semens/m.

Increase in salinity of a water leads to numerous adverse effects in its utilization. These effects arise from the influence of particular ionics species on processes and/or materials in the aqueous environment. Consequently, a link was obtained via regression between salinity (TDS) and concentrations to be expected of particular ionic species of interest to the economic impact analyses (i.e. Na^+, Ca^{2+}, Mg^{2+}, $C\ell^-$, SO_4^{2-}, Alkalinity and Hardness). Such relationships allow estimation of these species concentrations at any particular salinity of interest (for $TDS > 140$ mg/ℓ).
The particular relationships obtained linking salinity to the various ionic species concentrations (mg/ℓ) are :-

$Na^+ = -7{,}9 + 0.1191.TDS$
$Ca^{2+} = 3{,}3 + 0.1067.TDS$
$Mg^{2+} = -1{,}2 + 0{,}0563.TDS$
$C\ell^- = -6{,}2 + 0{,}1121.TDS$
$SO_4^{2-} = -76{,}0 + 0{,}5166.TDS$
Alkalinity = $68{,}3 + 0{,}0634.tds$

Hardness = 3,3 + 0,0563.TDS

These relationships are valid for TDS from about 150 to 900 mg/ℓ.

With regard to precipitation of salts from solution, of particular importance is the minteral $CaCO_3$. The report gives critical comment regarding use of various indices (eg. Langelier Saturation Index) used to assess propensity for precipitation/dissolution of minerals. Attention is drawn to the inadequacies of these indices and a rationale is presented for use of the "the mineral precipitation potential". This parameter is based on equilibrium chemistry and can be determined easily using either graphical techniques or computer programmes such as STASOFT.

INTRODUCTION

All natural waters contain dissolved and suspended substances in amounts which depend on their previous history. When rain falls, small quantities of oxygen, carbon dioxide and oxides of nitrogen are dissolved into the water. Impurities such as dust and bacteria are also picked up. As the rain flows over the earth or percolates through soil, a wide variety of mineral salts and some organics are dissolved. The mass and type of mineral dissolved depending on both the geochemical characteristics of the soil and the environment. For example, decomposed shales release high concentrations of sodium, chloride and sulphate species, while decomposed dolomites contribute principally calcium, magnesium and carbonate species.

River waters receive these landwashings and also street drainage, treated effluents from both domestic and industrial sources, and all manner of materials resulting from the activities of man. These factors further add to the concentrations of dissolved salts and organics. These waters usually serve as a source for (a) agricultural, (b) industrial and (c) domestic use, all of which are influenced inter alia by the nature and concentrations of dissolved minerals. Agriculture is the largest single user of water, principally irrigation. Minor farming uses are for watering purposes, care of livestock and poultry, and for cleaning. The quality of irrigation water appears to be governed principally by four characteristics:

(i) Total concentration of soluble salts: excessive salinity at root-zone level leads in succession to leaf burn, leaf drop, twig dieback and plant destruction.

(ii) Relative proportion of sodium to other cations: if the sodium content of an irrigation water is high compared with the calcium and magnesium content, sodium will be absorbed by the soil and replace the calcium and magnesium ions. High sodium concentration breaks down soil structures, seals pores and interferes with the drainage. Undesirable alkali conditions can develop in the soil.

(iii) Trace concentrations of certain elements (e.g. boron) are essential for plant growth; high concentrations of the elements, however, can be injurious to plants.

(iv) Concentration of bicarbonate ion: where there is high concentration of bicarbonate in a water there is a tendency for calcium and magnesium to precipitate as the solution becomes more concentrated in the root-zone.

The relative amount of sodium in the water is thus increased - possibly to a level where conditions in paragraph (ii) are attained.

It is particularly apparent in (iv) that the carbonate system influences the suitability of agricultural waters. At present there is little that man can do to adjust by artificial means the qualities of agricultural waters due to the large volumes that are used daily. The only adjustment practical is that involving the carbonate system in the solid phase, e.g. the addition of $CaCO_3$ or $Ca(OH)_2$ to soils.

The quality of water required for industrial purposes depends largely on the nature of the industry. The use of 'hard' waters in the textile industry can cause inferior products and uneven dyeing; water for laundering industry is required to be free of discolouration forming contaminants and 'soft' enough to avoid soap wastage; in paper and pulp industry 'hard' alkaline waters can cause problematic interferences in precipitation processes; boiler feed waters are required to be conditioned to a state that they are neither scale forming nor corrosive at high temperatures and pressures.

Water for domestic purposes is required to be free of taste, colour and odour; free of biological contaminants; non-corrosive; non-aggressive; well buffered against pH change and of such a nature that it easily forms a lather with soap. Hard waters are generally considered to be those that require considerable amounts of soap to produce a lather, and also those that form a scale in units where water temperatures are raised: e.g. boilers, kettles and hot water pipes. Corrosive waters are considered to be those that attack iron and copper materials in water distribution systems, aggressive waters those that attack cement type materials. Waters for domestic (and industrial) purposes usually are treated to minimize the adverse effects arising from these factors.

Hardness is caused by divalent metallic cations in solution: e.g. calcium, magnesium, strontium, ferrous and manganous ions. These divalent cations are associated with a number of anionic species, in natural waters these are usually SO_4^{2-}, Cl^-, HCO_3^- , CO_3^{2-}. Calcium and magnesium are the main contributors to hardness in natural waters. It is usually assumed in practice that the concentrations in solution of these two cations constitute the *total hardness* of the water. That part of total hardness which is chemically associated with carbonate species CO^-_3 and HCO^-_3 (i.e. alkalinity in the water) is called *carbonate hardness.*

Corrosion of copper and iron fittings in water distribution systems usually arises from one or more of the following. Firstly, high concentrations of Cl^- and/or

SO_4^{2-} in solution: where the concentrations of either of these species is greater than about 70mg/ the water should be regarded as potentially corrosive, and increasingly so with increase in concentration. Secondly non stabilized waters: where a water has a potential to dissolve $CaCO_3$ a corrosive situation may arise. In this regard the pH, alkalinity and calcium concentrations should be adjusted so that the water has a propensity to precipitate not more than about 5mg/l$CaCO_3$.

Aggression to cement type materials by water arises principally where the water has a $CaCO_3$
dissolving potential, i.e. waters that have not been stabilized as discussed above. However, high SO^{2-}_4 concentrations (greater than about 400 mg/l) have also been implicated, but generally such high concentrations are not found with natural terrestrial waters and arise only with recycling of waters within industrial processes.

Each utilization of water results in some impairment of its quality for re-use. This applies particularly to increase in inorganic minerals. Such affects are exacerbated in periods of drought where an increase in re-use of water is inevitable. This study forms part of an assessment of the economic impact of increase in inorganic salts on use (and reuse) of water in the Gauteng region. The particular aspects addressed here include estimation of inorganic mineral content, partitioning these minerals into principal and minor constituents, a brief assessment of stability with respect to precipitation/dissolution of minerals ($CaCO_3$), and finally linking ionic species constituting the principal ionic matrix to the total inorganic mineral content.

SALINITY

Salinity refers to the total concentration of inorganic salts in solution (usually expressed in mg/l). The concentrations of the various cationic and anionic species comprising the ionic matrix vary from trace to significant. For normal terrestrial waters the principal contributors to the ionic matrix are the cations Na^+, Mg^{2+} and Ca^{2+}, and the anions Cl^-, SO_4^{2-}, HCO_3^- and CO_3^{2-}, the latter two carbonate species comprising the proton accepting capacity of the water, i.e. alkalinity. For practical reasons the trace species contributions to the concentration of total dissolved salts (TDS) often are ignored. In this event, salinity (i.e. TDS) approximates the sum of the measured concentrations (in mg/l) of these seven ionic species.

Measurement of salinity

Various direct and indirect methods are available for measuring the concentration of dissolved salts (i.e. salinity). Direct measurement can be effected either via evaporation methods (Standard Methods, 1972), or from analyses of the concentrations of the individual species constituting the principal ionic species. Both methods are time consuming, however, the latter also is costly.

From a practical point of view it is usually sufficient for most purposes in water chemistry to use an approximation for TDS. This has been derived from correlations with specific conductance, Kemp (1971). The conductance of an aqueous solution is a measure of its ability to conduct a current. This is a property resulting from ions in solution. Electrical current is transported through solutions via the movement of ions, and conductivity increases as ion concentration increases. An empirical link between conductance and TDS (for $TDS<1000mg/l$) has been derived from the work of Russell (1976) and Langelier (1936) as

$$TDS = 6{,}4.E_c \qquad \textbf{(1a)}$$

Where E_c = specific conductant is in mille semens/m (i.e. mille mho/m).

From a practical point of view, estimation of dissolved salts using Eq(1a) has attained wide use in practice and appears reasonable for terrestrial waters. However, where possible the constant of proportionality used (i.e. 6,4 in Eq1a) should first be determined by correlation.
Regression analyses of TDS versus electrical conductivity for water derived from the Middle Vaal region is shown in Appendix 1 to this report. The analyses give the following relationship for the region for TDS from about 40 mg/l to around 600 mg/l.

$$TDS = 7{,}30.E_c - 34 \qquad \textbf{(1b)}$$

Comparing predicted TDS values using the Russell-Langelier equation (Eq1a) with those using (Eq1b), significant differences occur. These probably arise from dissimilarities between surface geology in the catchment areas of the two regions for which the equations were formulated, i.e. for the western states of USA (Russell-Langelier equation) and the Middle Vaal region (Eq1b).

PRECIPITATION OF SALTS FROM SOLUTION

Normally, the principal ionic matrix of terrestrial waters is comprised of the cations Ca^{2+}, Mg^{2+} and Na^{+} (and perhaps K^{+}) and the anions SO_4^{2-}, Cl^{-} and the carbonate species (CO_3^{2-} and HCO_3^{-}). The only mineral comprised of any of these species that may precipitate from a terrestrial water during normal use is calcite ($CaCO_3$). For this reason, and because precipitation/dissolution of $CaCO_3$ is intimately linked to corrosive/aggressive attack on water retaining materials, the propensity of a water to dissolve or precipitate this mineral (i.e. its saturation state) has attained general importance in water use.

With regard to assessing the saturation state of a water with respect of $CaCO_3$, either this can be measured directly using the marble test (Standard methods, 1972), or it can be assessed using equilibrium chemistry and certain measurements on the state of the water. Direct measurement is impractical for a number of reasons. Firstly, it is time consuming and may take several days to obtain the measurement. Secondly, the water sample may exchange CO_2 with the atmosphere before or during the test. Finally, and perhaps most important, the test does not indicate in any quantitative fashion how a water should be treated to obtain a desired final state. For these reasons the approach is used only rarely in practice.

Historically, equilibrium chemistry has been introduced into assessing saturation state via the use of certain indices (e.g. the Langelier Saturation Index and the Ryzner Index.) However, over the past decade use of these has been discouraged. This has arisen in part because of the general use of personal computers which allow more meaningful characterization of the saturation state.

Saturation state - Langelier Saturation Index

Use of equilibrium chemistry to assess the saturation state of a water originated with Langelier, 1936. He developed an index, the Langelier Saturation Index (LSI), the sign of which indicates either super or undersaturation with respect to $CaCO_3$. Despite the inadequacies of using the LSI in water characterization and treatment (see below), the index (and variations of it, for example the Ryzner Index) are still widely used in practice, for this reason it is considered briefly in this report.

The LSI is the difference between the actual pH of a water and a theoretical pH, pH_s, the water would attain by adding or removing CO_2 so that it is saturated with respect to $CaCO_3$, i.e. $LSI = pH - pH_s$.

Determination of pH_s is effected by recognising that addition/removal of CO_2 has no influence on the actual measured Alkalinity and calcium concentration of the water. Application of equilibrium chemistry leads to (Loewenthal and Marais, 1976),

$$pH_s \approx pK_2^1 - pK_s^1 + p[Ca^{2+}] + p[Alk] \qquad (2)$$

Where pK^1_2 = apparent second dissociation constant for the carbonate system.
pK^1_s = apparent solubility product for $CaCO_3$
[x] = molar concentration of species x

A positive value for LSI indicates supersaturation with respect to $CaCO_3$ and a negative value undersaturation.

A number of criticisms can be levelled at the usefulness of the index in water treatment and distribution. Firstly, it gives no indication of the mass concentration of mineral that can potentially precipitate or dissolve. For example, a water with LSI + 0,3 and pH8,4 can precipitate 2 mg/ℓ $CaCO_3$; however a water with LSI also +0,3 but pH 7,0 can precipitate over 40mg/l $CaCO_3$ (i.e. for LSI of +0,3, pipe scaling is unlikely at pH8,4 but a certainly at pH~7).
Secondly, many workers assume that a positive LSI value indicates a non corrosive water, this however is not so (Loewenthal et al, 1986). Finally, utilization of the index does not give the user any lead as to the type or mass of dosage that should be applied to minimize the adverse effects a water may have on a distribution system.

$CaCO_3$ precipitation potential

The $CaCO_3$ precipitation potential gives the saturation state of a water both qualitatively and quantitatively. However, determination of this potential requires using equilibrium chemistry and computations are difficult and time consuming. This makes direct determination totally impractical. To bypass this problem, Loewenthal and Marais (1976, 1978) developed graphical methods to be effected in computer generated equilibrium diagrams (the so called Modified Caldwell Lawrence Diagrams). These diagrams are useful not only for determining the precipitation potential but also for dosage determination in water treatment, treating underground waters and solving a broad spectrum of other water quality related problems.

The graphical methods above, however, require that the user has a thorough understanding of water chemistry and this has limited the usefulness of the approach. To broaden the use of equilibrium chemistry in both water characterization and treatment, Loewenthal and Friend (1993) developed a user friendly computer program STASOFT, that effects all the tasks that can be carried out graphically using the Modified Caldwell Lawrence Diagrams. The program is applicable to both terrestrial and industrial waters where pH is controlled by the carbonate system.

In conclusion, the various indices used to characterize waters are not recommended. Not only do they have no meaningful quantitative interpretation, but also their use has led to large scale misapplications - for example, linking the indices to the potential corrosiveness of a water to metals containing it. On the other hand, graphical or computer utilization of equilibrium chemistry and the Precipitation Potential give the user the ability to critically assess potential problems associated with the use of a water.

CHARACTERIZATION OF IONIC SPECIES WITH SALINITY

In the introduction to this report it was stated that increase in salinity of a water leads to numerous adverse effects in its utilization. These arise from the influence that particular ionic species may have on processes and/or materials in the aqueous environment. For example, increase in Cl^- and SO_4^{2-} has a pernicious effect on the corrosiveness of water to metals. On the other hand, increase in Ca^{2+}, Mg^{2+} and the carbonate species (i.e. HCO_3^{2-}) have no adverse effects on corrosive attack in the aqueous environment, in fact there is an ameliorating effect.

Clearly, any investigation into the economic implications of increase salinity in a region first needs to link salinity to the ionic constitution of the water, and thereafter, assess the effects on the industrial, agricultural and domestic sectors. Recognising that most of the principal ionic species have extensive properties (all except the carbonate species concentrations which are linked to the intensive property pH), in any geographical region one would expect these species concentrations to be linked to salinity (i.e. TDS). For the carbonate ionic species, the sum of these reflect the total proton accepting capacity of the water (i.e. Alkalinity) which also has extensive properties and consequently also should be linked to salinity.

To assess the relationships between salinity and each of the principal ionic species present in the water of the Middle Vaal Area, water quality data was obtained from sampling stations indicated widely over this area. Stations selected were the Vaal Dam (wall), Balkfontein, Klipplaatdrift, Orkney and Schoemansdrift.

Analyses were carried out in two steps. First, for each station data were analysed to obtain functional relationships linking TDS to each of the ionic components of interest. Second, comparison of these relationships between individual stations allowed statistical assessment off the feasibility of pooling data from the various stations to obtain general functional relationships for each of the ionic species versus TDS for the Middle Vaal River Area.

TDS versus ionic constitution - individual station results.

Sodium ($Na^{+)}$ data:

In figures 1 to 5 are shown plotted Na^+ concentration versus TDS values for each of the stations Balkfontein, Klipplaatdrift, Orkney, Schoemansdrift and Vaal Dam (Wall) respectively.

Equations linking Na^+ with TDS were obtained by regression. For each station a linear relationship was obtained of the form:

$$Na^{+} = A + B * TDS \qquad (3)$$

Where Na^+ = concentration of sodium, mg/ℓ.

A and B = regression constants.

Values of the constants A and B to be used with Eq(3) to obtain the sodium concentration for a known TDS for each of that stations are listed in Table 1.

Calcium (Ca^{2+}) data :

In figures 6 to 10 are shown plotted Ca^{2+} concentration versus TDS for each of the stations Balkfontein, Klipplaatsdrift, Orkney, Schoemansdrift and Vaal Dam (Wall) respectively. Again, regression yielded a linear relationship between Ca^{2+} concentration and TDS of the form depicted in Eq. (1), i.e.

$$Ca^{2+} = A + B * TDS \qquad (4)$$

Ca^{2+} = concentration of Ca^{2+} in mg/ℓ.

Values for the constants in Eq(4) were obtained by regression for each station and are shown listed in Table 1.

$C\ell^-$, SO_4^{2+}, T Alkalinity, Hardness and Mg^{2+} data :

A similar approach to that above was adopted for each of the special $C\ell^-$,SO_4^{2-}, T. Alkalinity, Hardness and Mg^{2+}. The respective plots against TDS are shown in figures 11 to 15 for $C\ell^-$, figures 16 to 20 for SO_4^{2+}, Figures 21 to 25 for T. Alkalinity, Figures 26 to 30 for Hardness, and Figures 31 to 35 for Mg^{2+}. In each case regression yielded a linear relationship between species concentration and TDS. of the form.

$$\text{Species Concentration} = A + B * \text{TDS} \qquad (5)$$

The respective regression constants A and B to be used with Eq(5) to obtain species concentrations for a particular TDS value at each of the sampling points are listed in Table 1. Very poor correlation (i.e. R squared column) was obtained for TDS versus both $C\ell^-$ and SO_4^{2-} at the Vaal Dam (Wall) station compared with the other stations. Furthermore, the constants A and B in the linear relationships linking $C\ell^-$, $SO4^{2-}$ and T. Alkalinity to TDS have very different values for the Vaal Dam Wall station compared with the remaining stations. For these reasons it was decided to disregard the data from the Vaal Dam (Wall) station in obtaining the overall relationships linking species concentrations to TDS for the Middle Vaal River Area.

TDS versus ionic constitution - Middle Vaal River Area

To obtain the relationships linking each of ionic constituents of interest in this study to TDS, all the data for each of the ionic constituents from Balkfontein, Klipplaatsdrift, Orkney and Schoemansdrift were pooled. Regression analyses yielded linear relationships for each of the ionic constituents against TDS of the form:

$$\text{Species concentration} = A + B * \text{TDS} \qquad (6)$$

The regression constants A and B for each of the ionic constituents are listed in Table 2. Units of species concentration obtained by substituting the relevant constants from Table 2 into Eq(6) are in mg/ℓ for Na^+, Ca^{2+}, Mg^{2+}, $C\ell^-$ and SO_4^{2-}, and in mg/ℓ as $CaCO_3$ for T. Alkalinity and Hardness. It should be noted that the linear relationships reported here are applicable only for TDS values greater than about 90mg/ℓ.

Inserting the relevant regression constants from Table 2 into Eq(6) gives the following relationships for each of the ionic constituents with TDS as :

$$Na^{+} = -7{,}9 + 0{,}1191 * TDS \qquad (7)$$

$$Ca^{2+} = 3{,}3 + 0{,}1067 * TDS \qquad (8)$$

$$C\ell^{-} = -6{,}2 + 0{,}1121 * TDS \qquad (9)$$

$$SO_4^{2-} = -76{,}0 + 0{,}5166 * TDS \qquad (10)$$

$$\text{T.Alkalinity} = 68{,}3 + 0{,}0634 * TDS \qquad (11)$$

$$\text{Hardness} = 3{,}3 + 0{,}5013 * TDS \qquad (12)$$

$$Mg^{2+} = -1{,}2 + 0{,}0563 * TDS \qquad (13)$$

Where TDS = total dissolved salts, mg/ℓ.
Na^+, Ca^{2+}, $C\ell^-$, SO_4^{2-} = concentrations of sodium, calcium, chloride and sulphate respectively, mg/ℓ
T. Alkalinity, Hardness = concentration of these constituents in mg/ℓ as $CaCO_3$.

*It should be noted that Eqs. (7 to 13) are valid only for TDS greater than about 90mg/ℓ.

As an example, if the TDS in the Middle Vaal region were 700mg/ℓ, then
from Eq. (7), Na = 75,5 mg/ℓ
From Eq. (8), Ca^{2+} = 78,0 mg/ℓ
From Eq. (9), $C\ell^-$ = 72,3 mg/ℓ
From Eq. (10), SO_4^{2-} = 285,6 mg/ℓ
From Eq. (11), T. Alkalinity = 112,7 mg/ℓ as $CaCO_3$
From Eq. (12), Hardness = 354,2 mg/ℓ as $CaCO_3$.
From Eq. (13), Mg^{2+} = 38,2 mg/ℓ

Table 1

Correlation of species (Na^{+}, Ca^{2-}, Mg^{2+}, Total Hardness, Alkalinity, Cl^{-} and SO_4^{2-}) with TDS for each of the stations Balkfontein, Klipplaatsdrift, Orkney, Schoemansdrift and Vaal Dam Wall (see Equation 5), i.e.

Species concentration = A + B * TDS

Note: R squared column = correlation coefficient
Points column = number of samples used in correlation

TDS vs	Location	A	B	R squared	Points
Na	Balkfontein	-10.7	0.1211	0.8837	2604
	Klipplaatdrift	-10.5	0.1251	0.8993	750
	Orkney	-2.5	0.1075	0.8714	718
	Schoemansdrift	-5.8	0.1278	0.9391	831
	Vaal Dam (wall)	-1.5	0.0961	0.8429	444
Ca	Balkfontein	5.4	0.1012	0.8738	2604
	Klipplaatdrift	5.8	0.0987	0.8845	750
	Orkney	1.4	0.1157	0.9184	718
	Schoemansdrift	-1.1	0.1176	0.9492	831
	Vaal Dam (wall)	-0.0	0.0975	0.8950	444
Cl	Balkfontein	-6.7	0.1081	0.7637	2603
	Klipplaatdrift	-12.2	0.1279	0.8192	748
	Orkney	-1.0	0.1040	0.7419	717
	Schoemansdrift	-5.8	0.1242	0.8968	831
	Vaal Dam (wall)	3.4	0.0441	0.2030	444
SO_4	Balkfontein	-84.4	0.5379	0.9339	2604
	Klipplaatdrift	-71.1	0.5026	0.9274	750
	Orkney	-72.0	0.5072	0.9319	718
	Schoemansdrift	-58.2	0.4697	0.9133	831
	Vaal Dam (wall)	3.1	0.0744	0.1999	444
T. ALK	Balkfontein	76.5	0.0496	0.1170	2604
	Klipplaatdrift	69.0	0.0650	0.2247	750
	Orkney	58.1	0.0810	0.3767	718
	Schoemansdrift	54.0	0.0834	0.3450	831
	Vaal Dam (wall)	-8.0	0.5198	0.9322	444
Hardness	Balkfontein	9.0	0.5080	0.9266	2604
	Klipplaatdrift	8.8	0.4796	0.9283	750
	Orkney	-10.5	0.5193	0.9350	718
	Schoemansdrift	-1.2	0.4684	0.9599	831
	Vaal Dam (wall)	-1.1	0.4636	0.9260	444
Mg	Balkfontein	-1.1	0.0612	0.7211	2604
	Klipplaatdrift	-1.4	0.0559	0.7323	750
	Orkney	-3.3	0.0552	0.8309	719
	Schoemansdrift	0.4	0.0418	0.8912	831
	Vaal Dam (wall)	-0.2	0.0524	0.8759	443

Table 2

Correlation of species (Na^+, Ca^{2+}, Mg^{2+}, Total Hardness, Alkalinity, Cl^- and SO_4^{2-}) with TDS for pooled data (see Equation 5), i.e.

Species concentration = A + B * TDS

Note : R squared column = correlation coefficient
Points column = number of samples used in correlation

TDS vs	Location	A	B	R squared	Points
Na	All points except those at Vaal Dam	-7.9	0.1191	0.8779	4903
Ca		3.3	0.1067	0.8922	4903
Cl		-6.2	0.1121	0.7739	4900
SO_4		-76.0	0.5166	0.9285	4903
T. ALK		68.3	0.0634	0.2033	4903
Hardness		3.3	0.5013	0.9221	4903
Mg		-1.2	0.0563	0.6983	4904

Table 3

Ionic species concentrations versus TDS predicted using the regression equations (Equations 7 to 13).

TDS value	200.0	400.0	600.0	800.0	1000.0	1200.0
Na	15.9	39.7	63.6	87.4	111.2	135.0
Ca	24.6	46.0	67.3	88.7	110.0	131.4
Cl	16.3	38.7	61.1	83.5	106.0	128.4
SO_4	27.3	130.6	233.9	337.2	440.6	543.9
T. ALK	81.0	93.7	106.4	119.0	131.7	144.4
Hardness	103.6	203.9	304.1	404.4	504.7	604.9
Mg	10.1	21.3	32.6	43.9	55.1	66.4

TDS value	Na	Ca	Cl	SO_4	T. ALK	Hardness	Mg
200	15.9	24.6	16.3	27.3	81.0	103.6	10.1
400	39.7	46.0	38.7	130.6	93.7	203.9	21.3
600	63.6	67.3	61.1	233.9	106.4	304.1	32.6
800	87.4	88.7	83.5	337.2	119.0	404.4	43.9
1000	111.2	110.0	106.0	440.6	131.7	504.7	55.1
1200	135.0	131.4	128.4	543.9	144.4	604.9	66.4

Regression of pH versus TDS at the various sampling stations. The extremely poor regression indicates no functional relationship.

	Location	average	std dev	points
pH	Balkfontein	7.781	0.4351	2601
	Klipplaatdrift	7.768	0.5093	750
	Orkney	7.801	0.5315	719
	Schoemansdrift	7.693	0.5296	832
	Vaal Dam (wall)	7.613	0.5437	445

APPENDIX I

TDS versus conductivity.

Individual stations:

Correlation of TDS with electrical conductivity (E_c in mille semens/metre for each of the stations Balkfontein, Kliplaatsdrift, Orkney, Schoemansdrift and Vaal Dam (wall) give a linear relationship of the form.

$TDS = A + B.E_c$.

Regression constants A and B are listed in Table 5.

Table 5

Regression constants A and B for change in TDS with electrical conductivity.
Note: R squared column = correlation coefficient.
Points = Number of samples used.

Location	A	B	R squared	Points
Balkfontein	-27	7,19	0,966	2604
Klipplaatsdrift	-37	7,46	0,958	750
Orkney	-51	7,58	0,953	719
Schoemansdrift	-33	7,25	0,969	832
Vaal Dam (Wall)	-14	7,87	0,878	445

Pooled data (in all stations):

For pooled data (without Vaal Dam Wall) regression analyses for TDS with electrical conductivity gives the linear relationship (see Table 6):

$TDS = -34 + 7{,}3.E_c$.

(E_c in mille semens/metre)

Table 6

Correlation of TDS with electrical conductivity for pooled data Middle Vaal Region.
R squared column = correlation coefficient
Points = number of samples used.

Location	**A**	**B**	**R squared**	**Points**
All stations except Vaal Dam Wall	-34	7,3	0.963	4905

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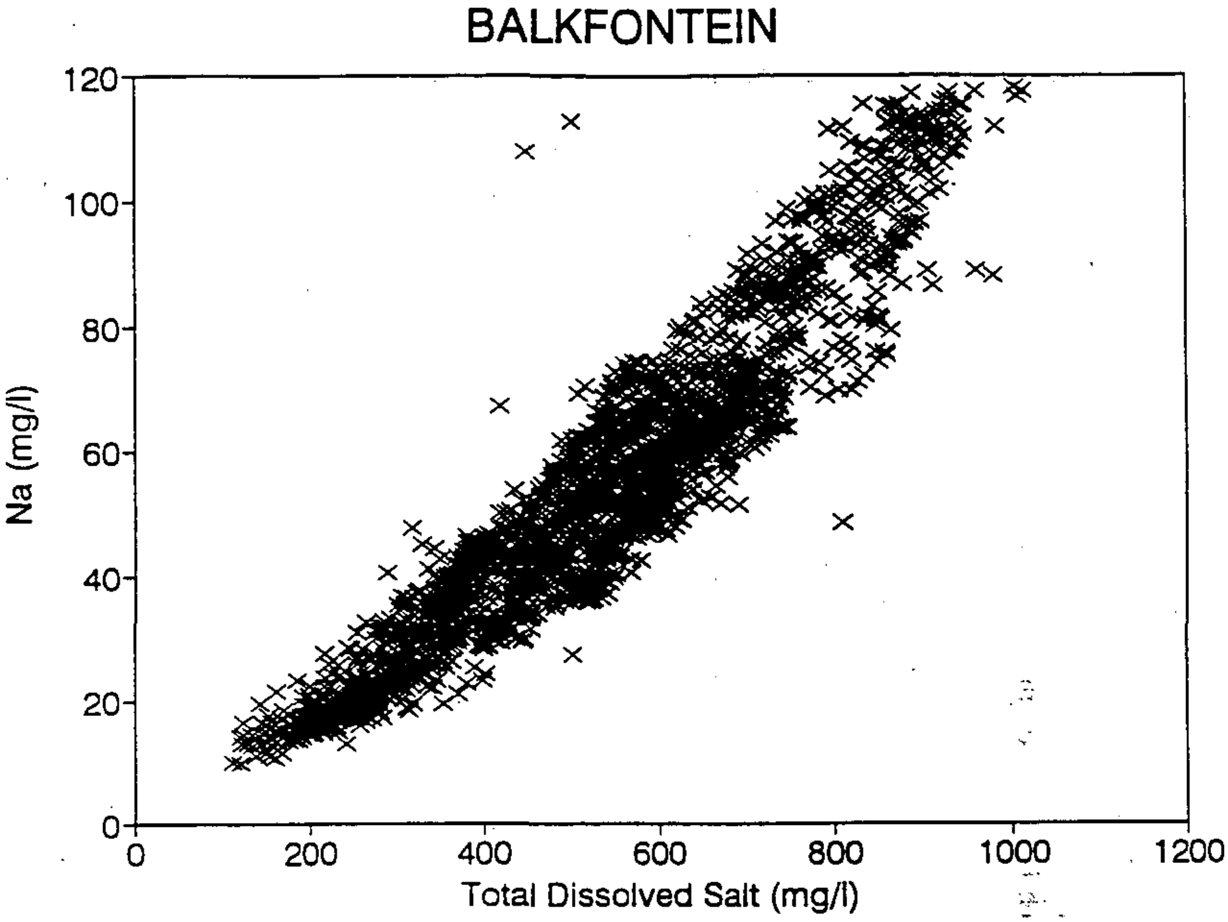

FIGURE 1: Plot of Total Dissolved Salts vs Na^+ as sampled at Balkfontein

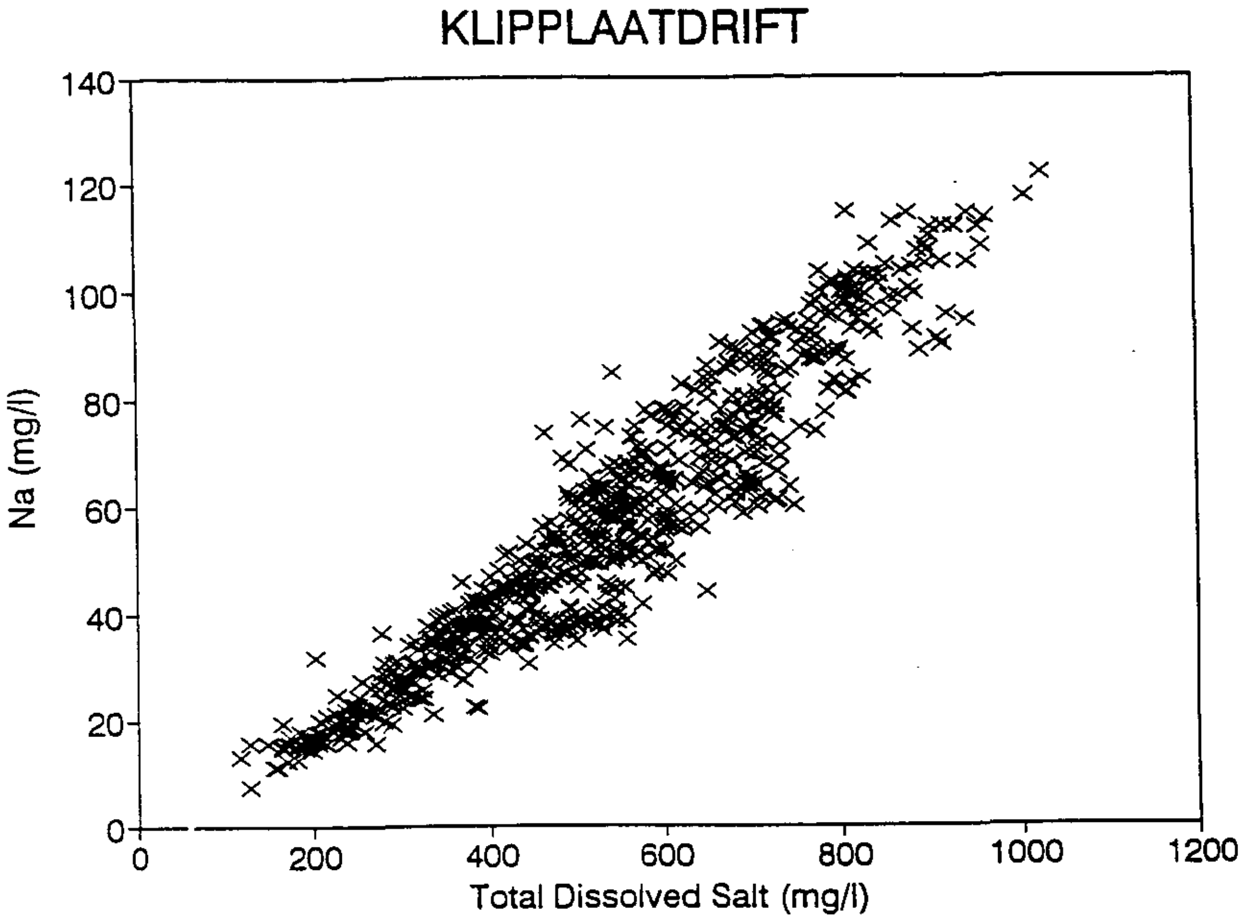

FIGURE 2: Plot of Total Dissolved Salts vs Na$^+$ as sampled at Klipplaatdrift

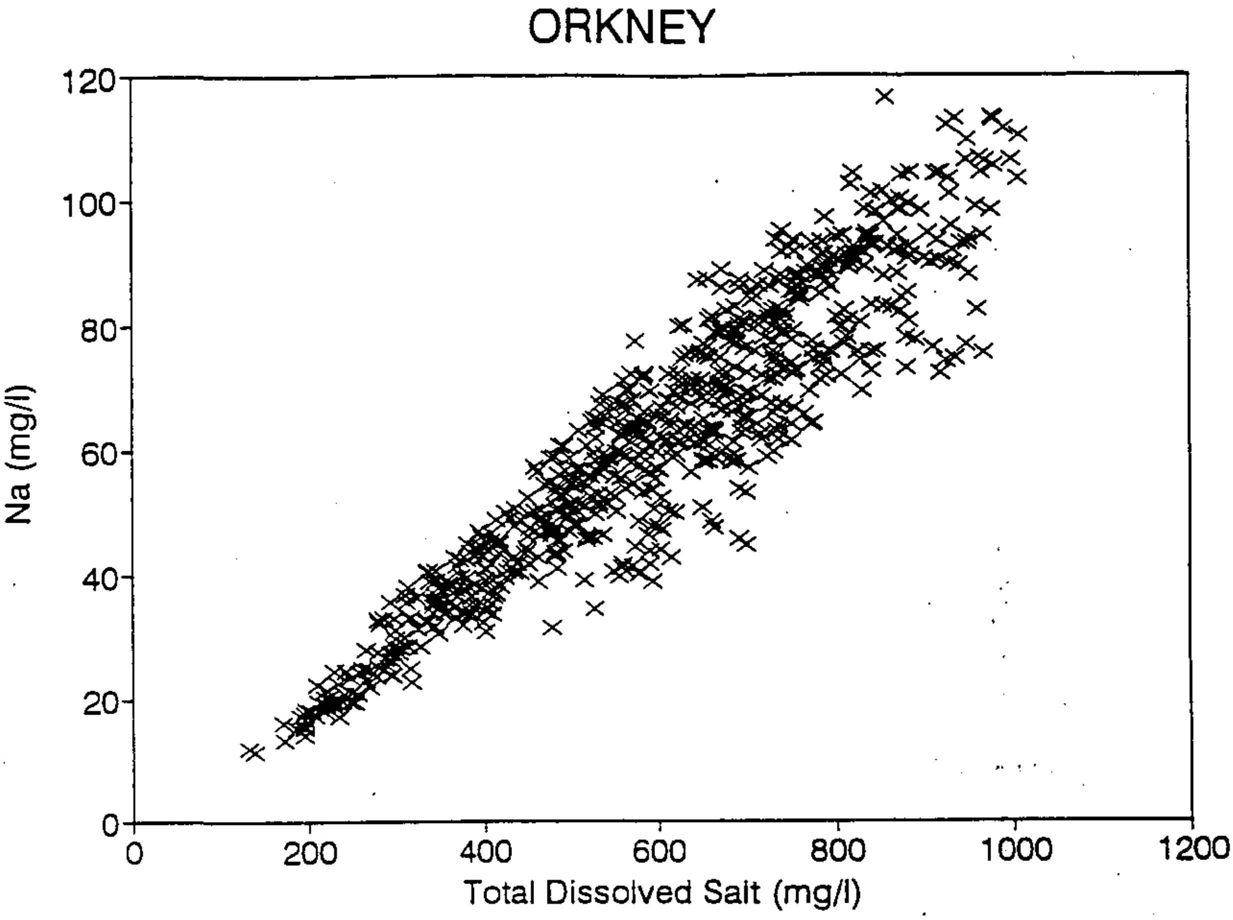

FIGURE 3: Plot of Total Dissolved Salts vs Na$^+$ as sampled at Orkney

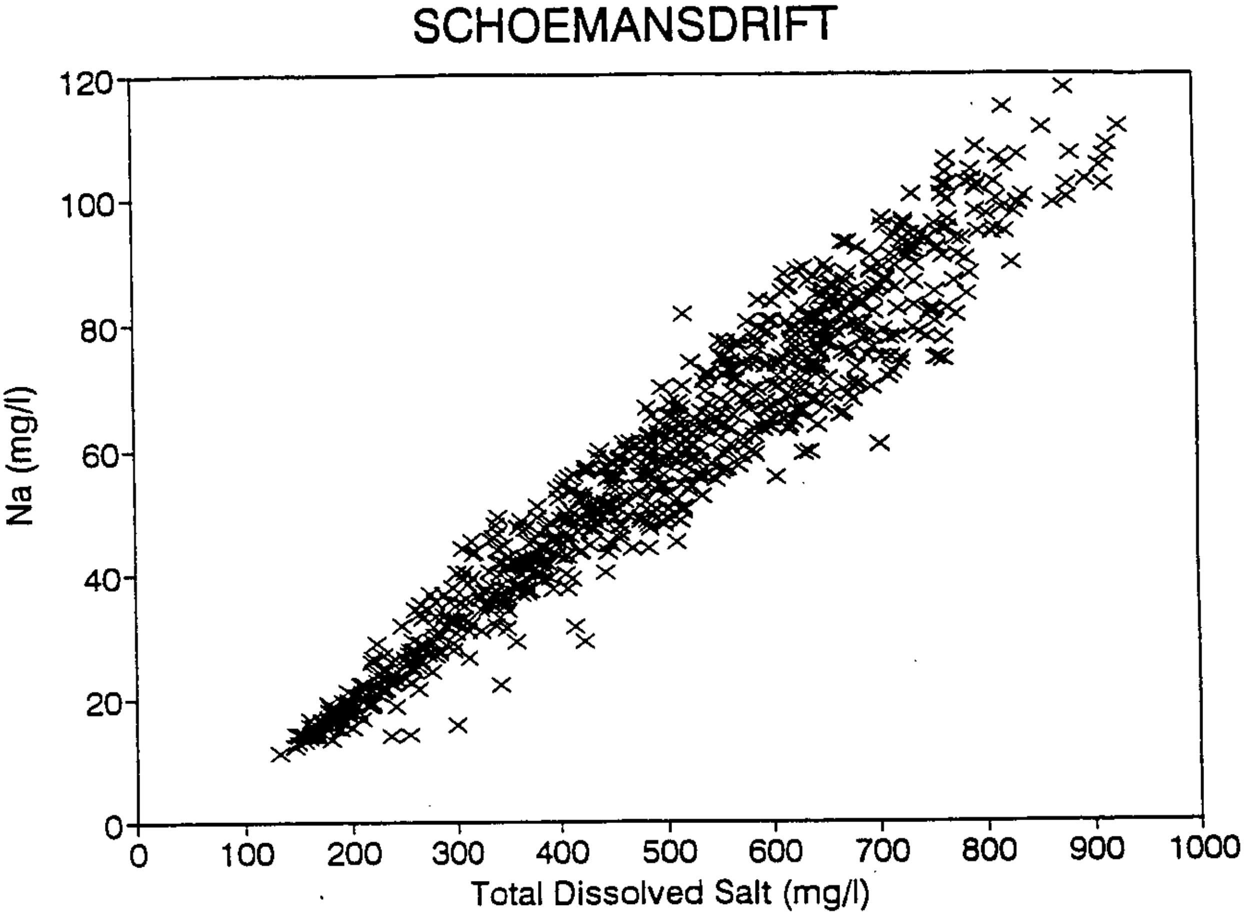

FIGURE 4: Plot of Total Dissolved Salts vs Na$^+$ as sampled at Schoemansdrift

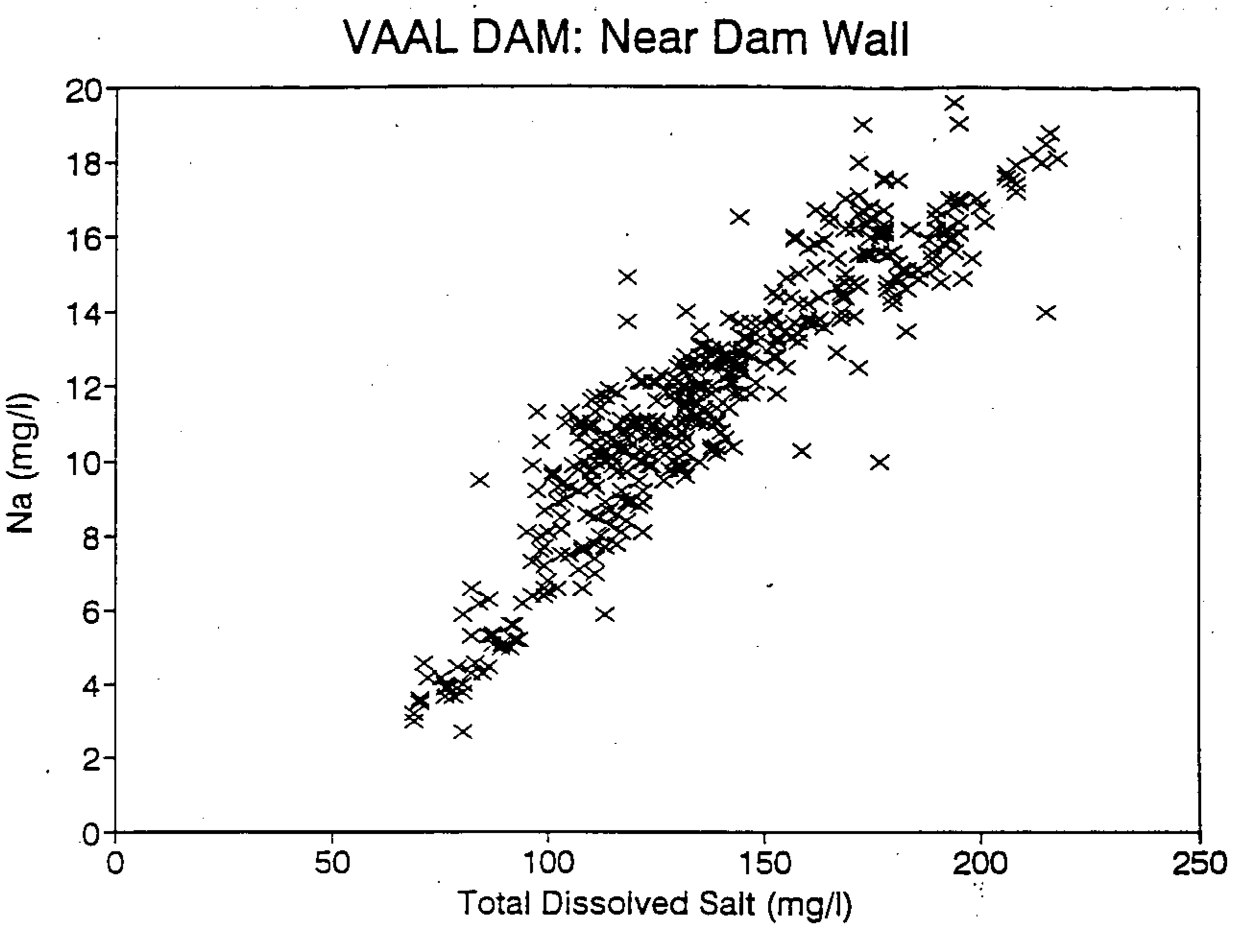

FIGURE 5: Plot of Total Dissolved Salts vs Na$^+$ as sampled at Vaal Dam (near dam wall)

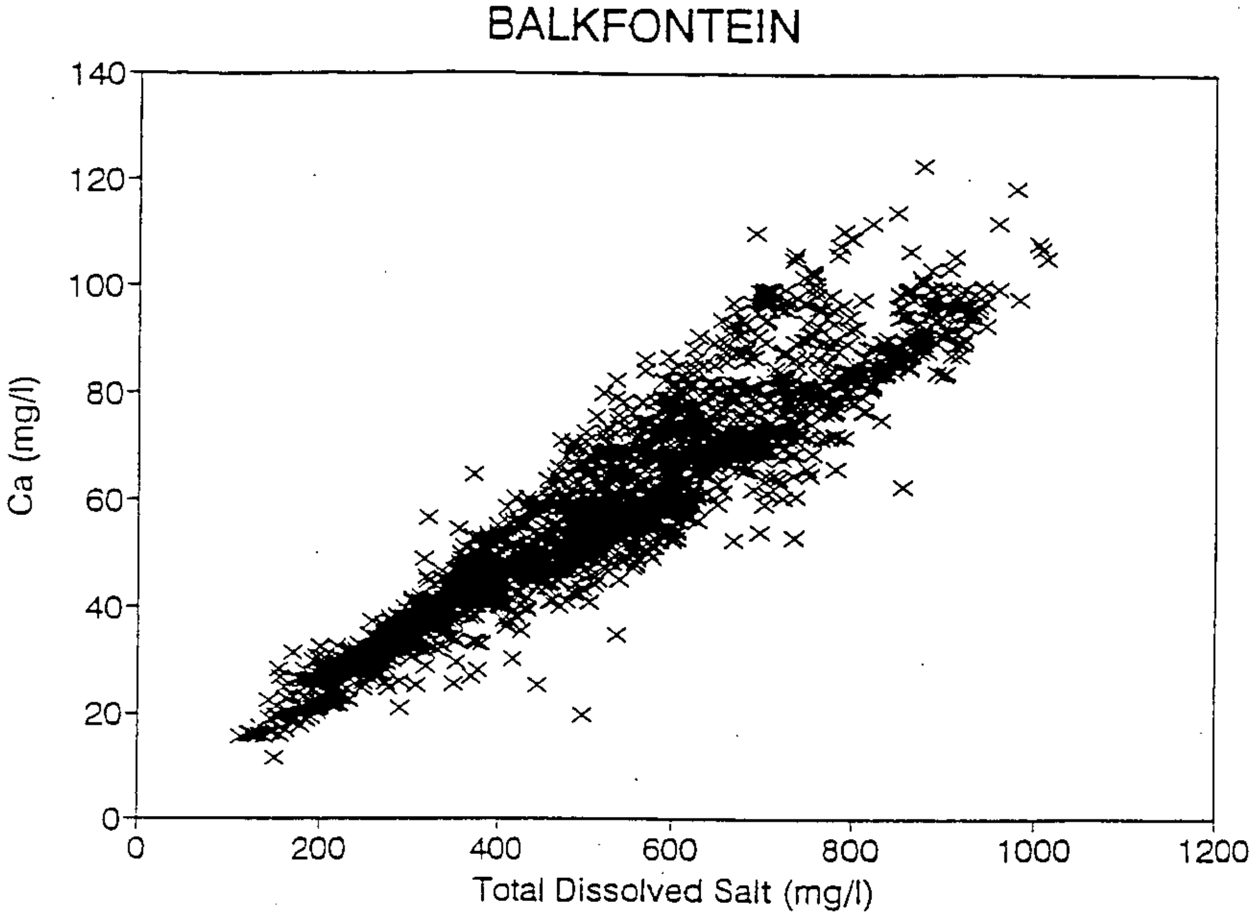

FIGURE 6: Plot of Total Dissolved Salts vs Ca^{2+} as sampled at Balkfontein

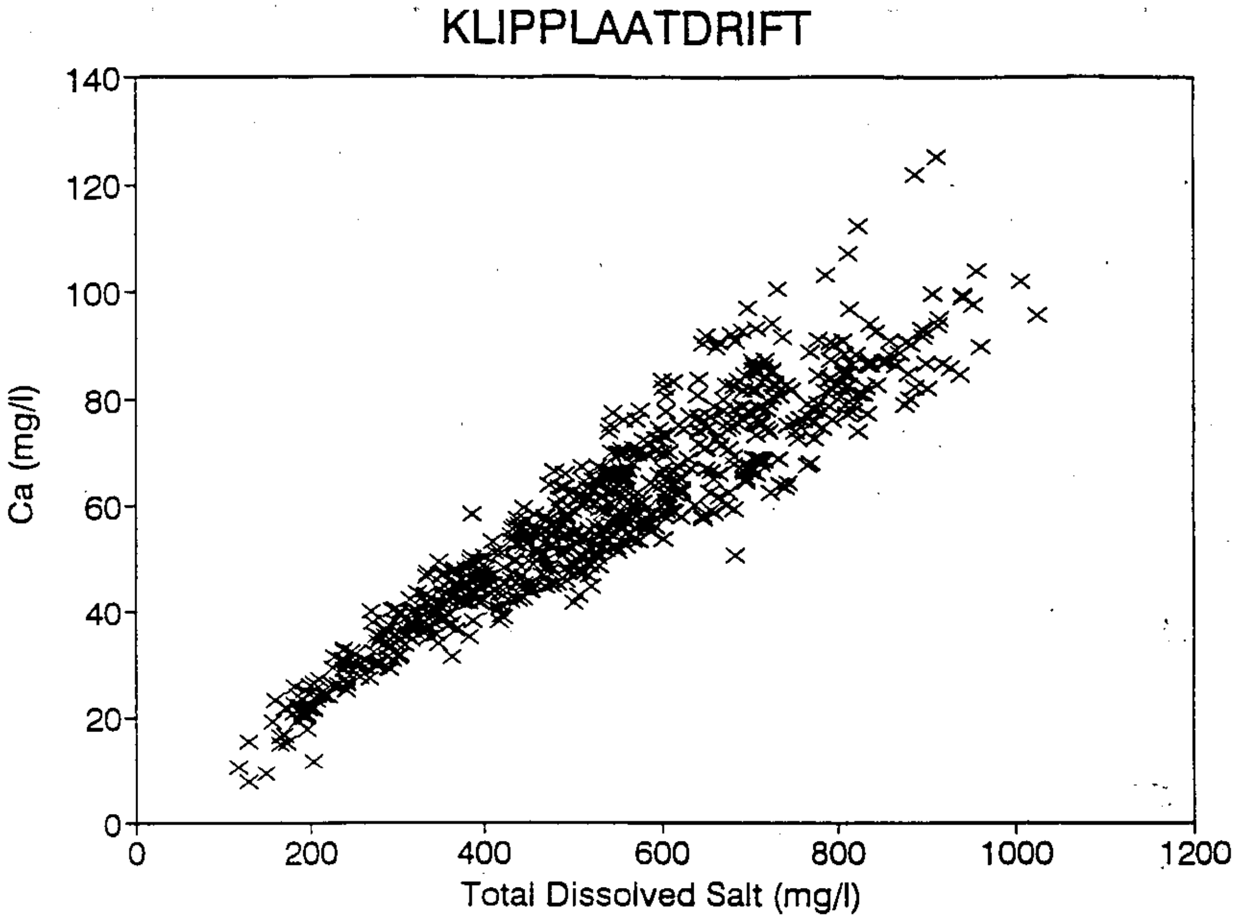

FIGURE 7: Plot of Total Dissolved Salts vs Ca^{2+} as sampled at Klipplaatdrift

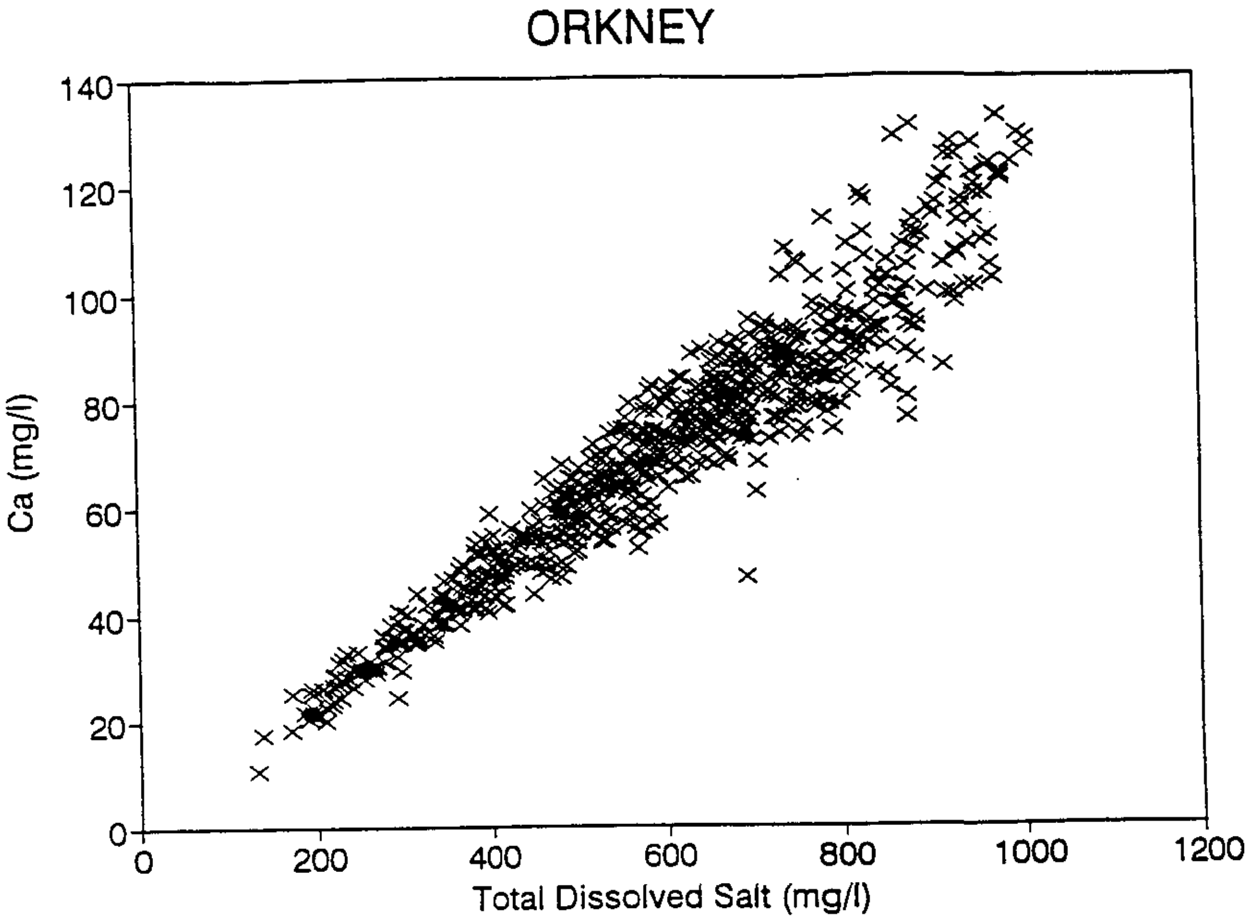

FIGURE 8: Plot of Total Dissolved Salts vs Ca^{2+} as sampled at Orkney

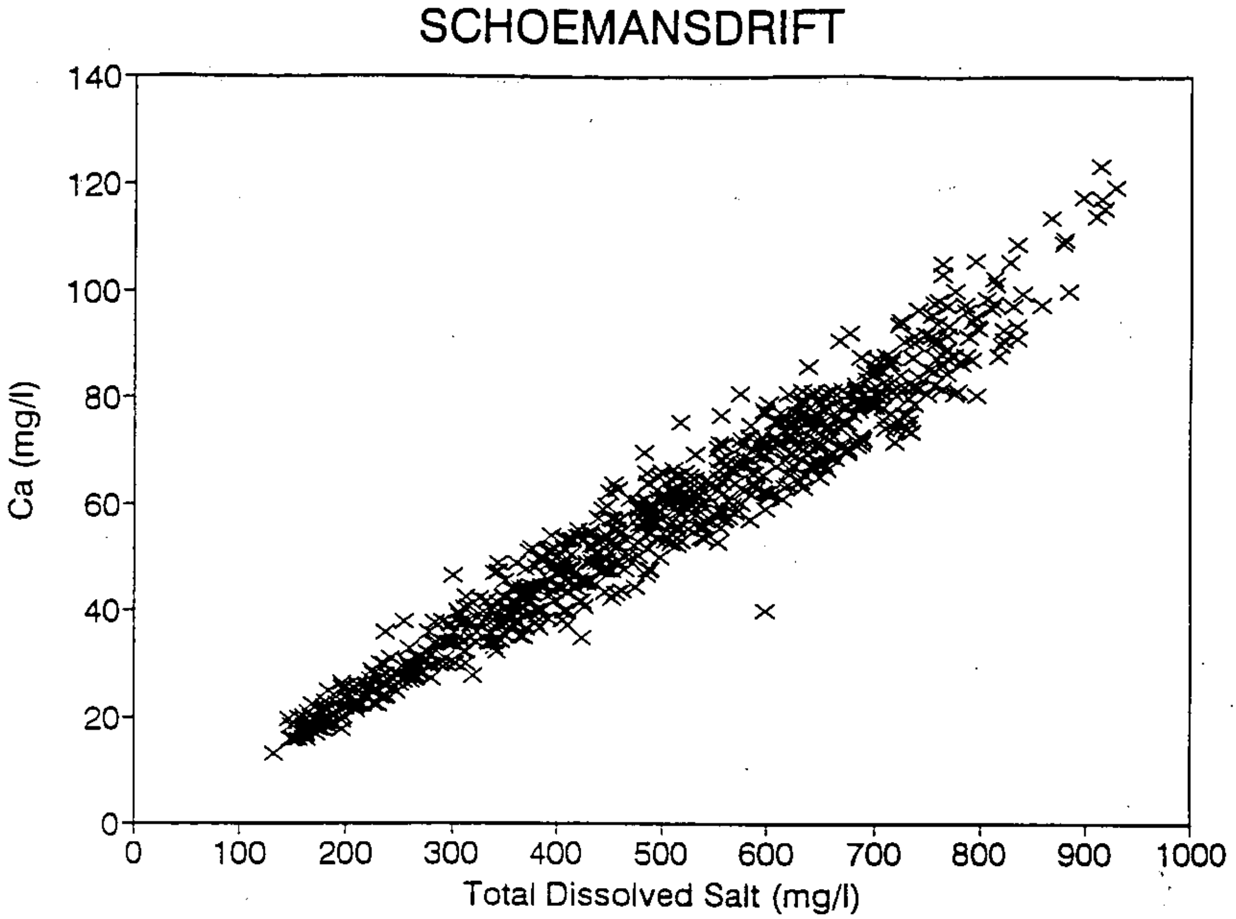

FIGURE 9: Plot of Total Dissolved Salts vs Ca^{2+} as sampled at Schoemansdrift

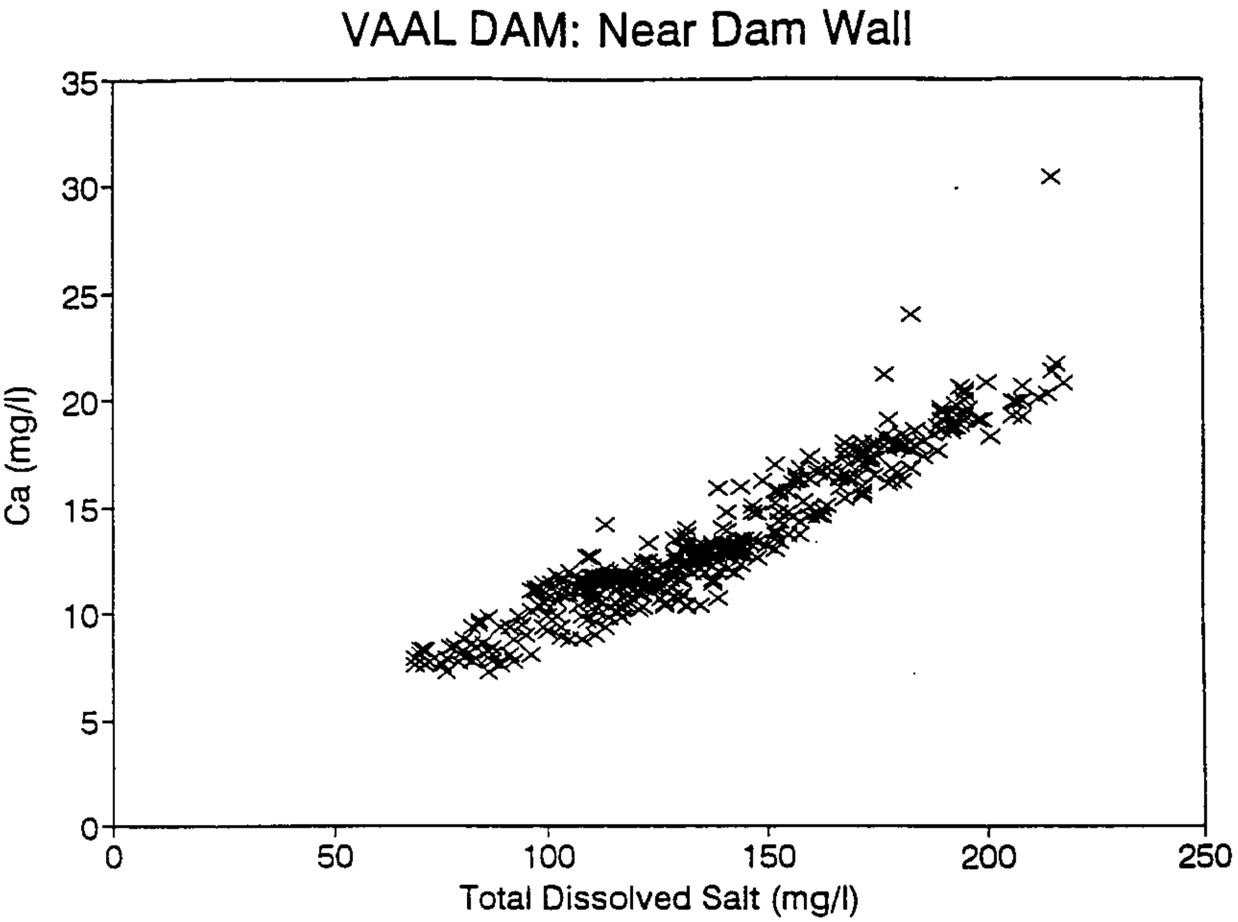

FIGURE 10: Plot of Total Dissolved Salts vs Ca^{2+} as sampled at Vaal Dam (near dam wall)

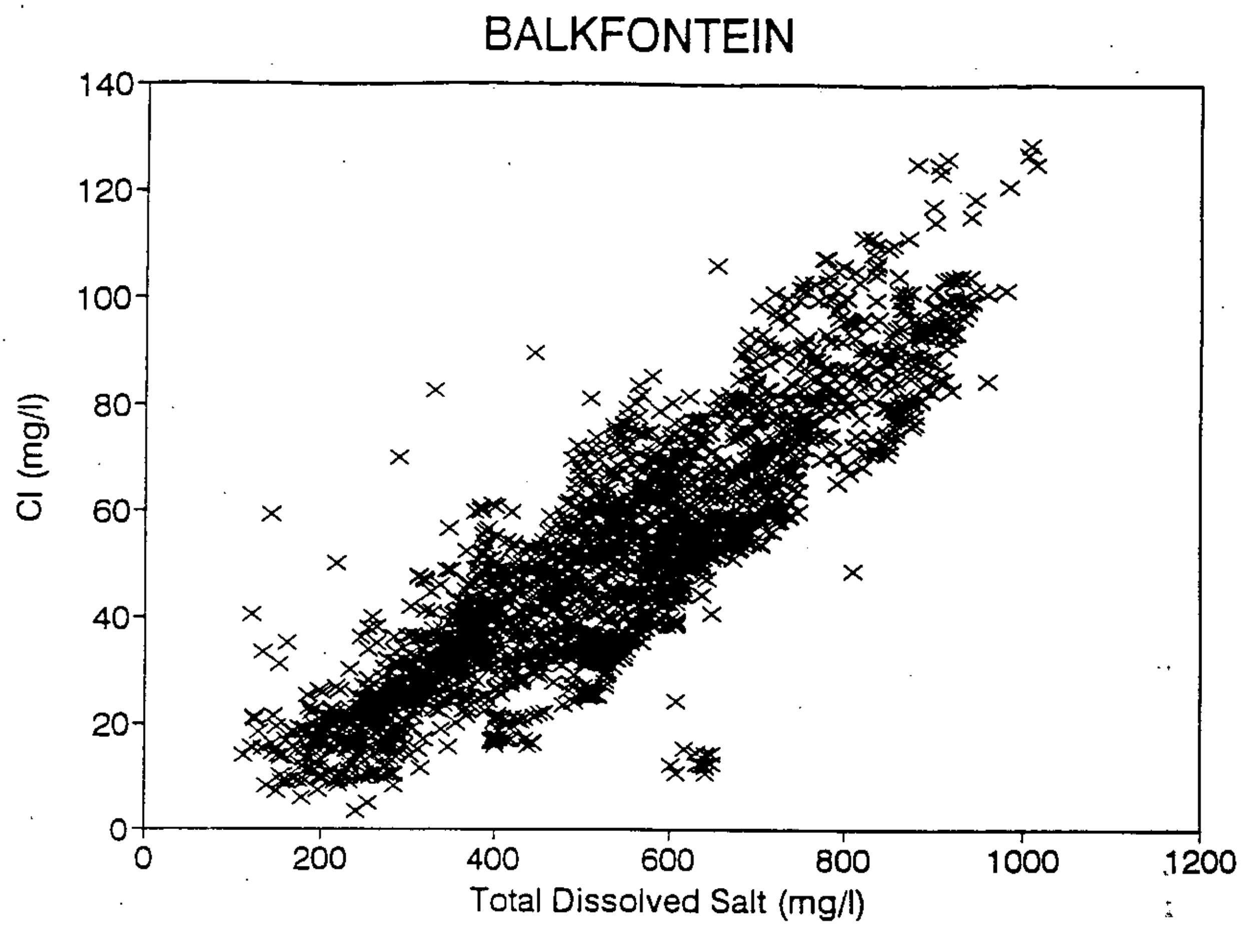

FIGURE 11: Plot of Total Dissolved Salts vs Cl^- as sampled at Balkfontein

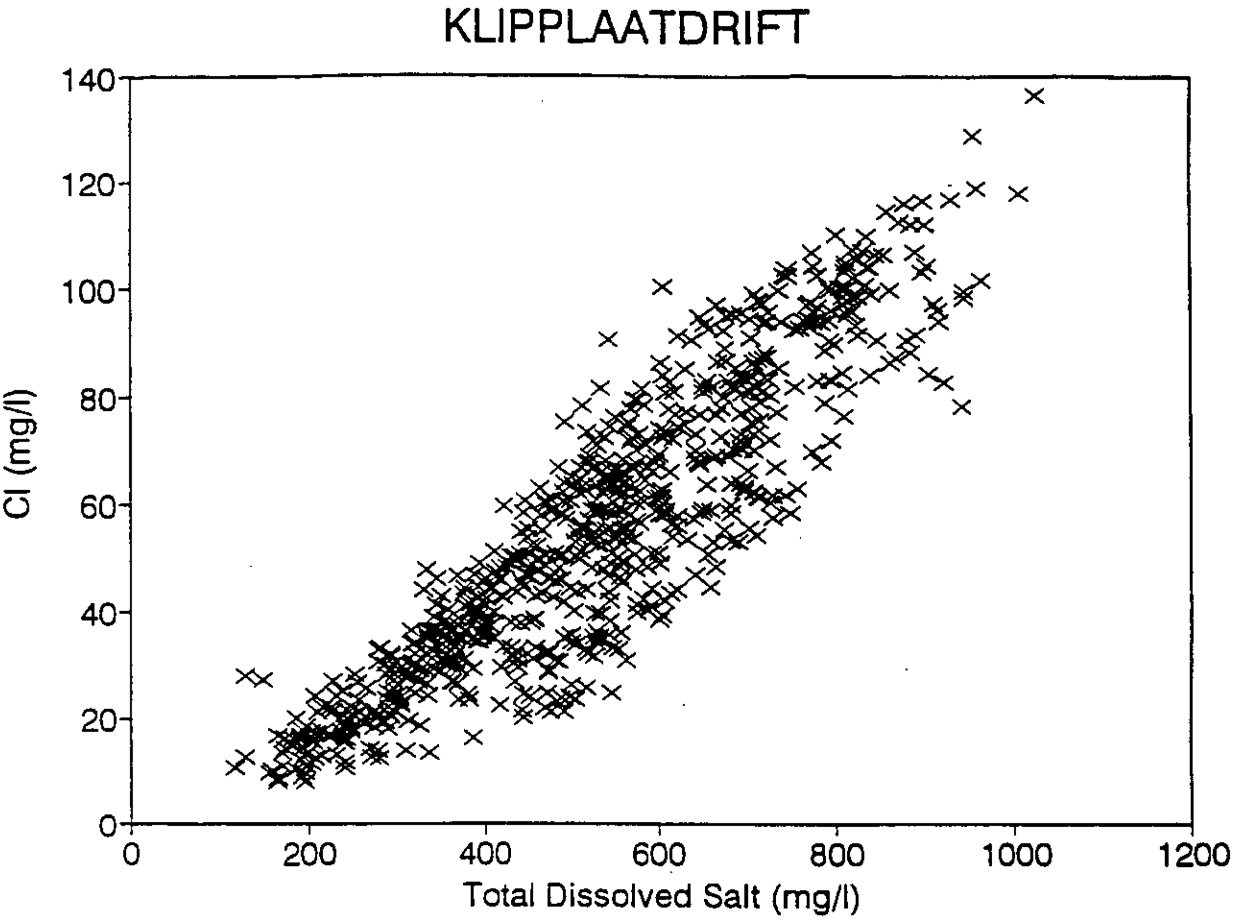

FIGURE 12: Plot of Total Dissolved Salts vs $C\ell^-$ as sampled at Klipplaatdrift

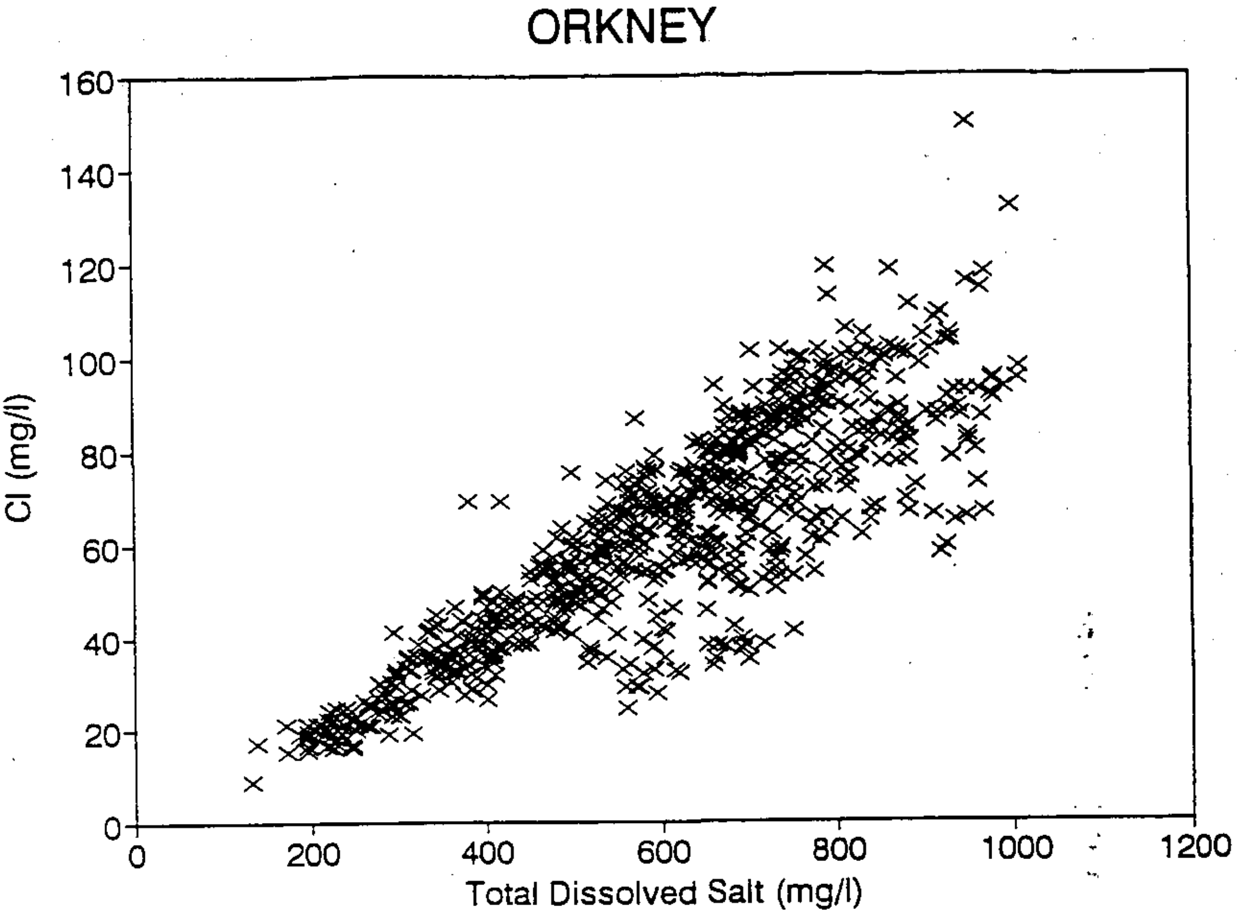

FIGURE 13: Plot of Total Dissolved Salts vs $C\ell^-$ as sampled at Orkney

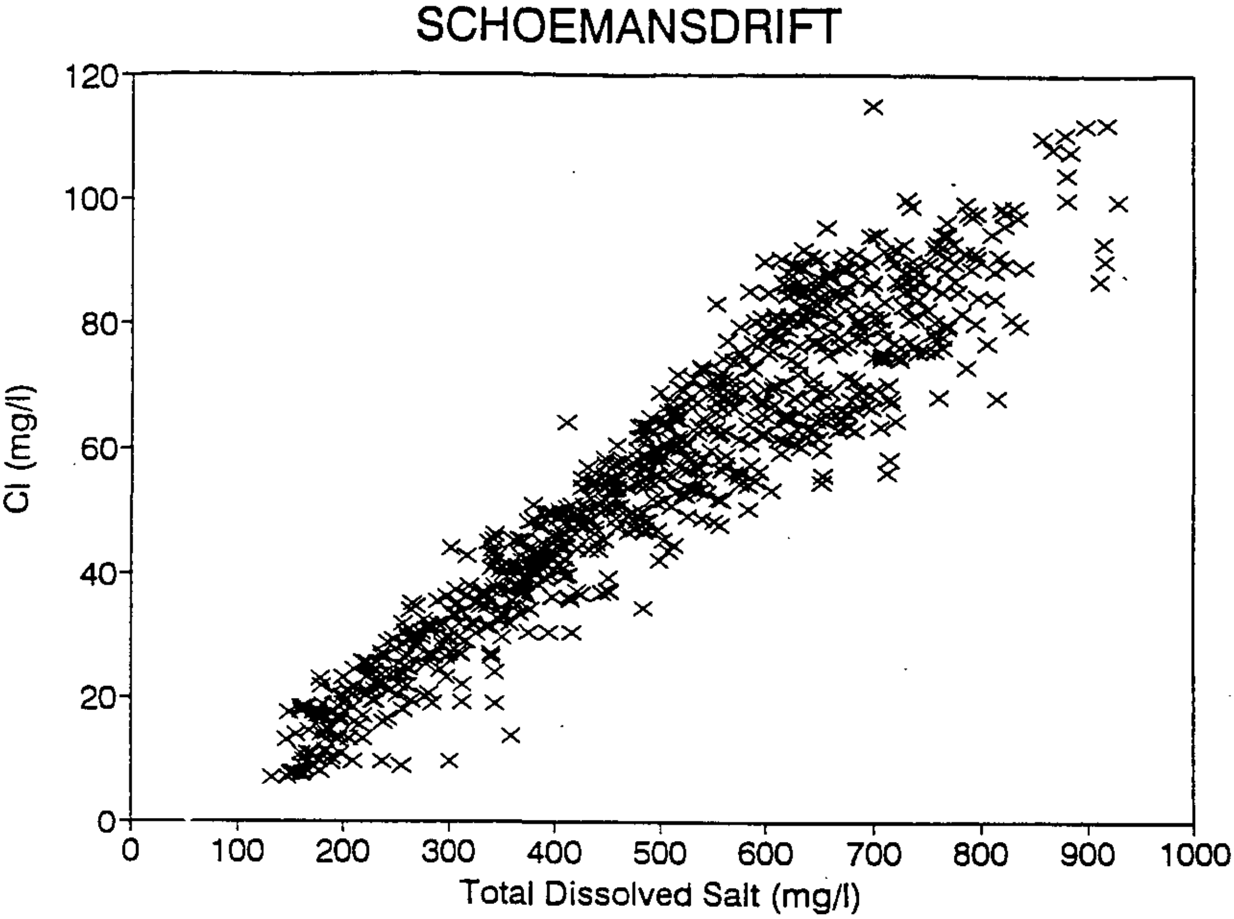

FIGURE 14: Plot of Total Dissolved Salts vs $C\ell^-$ as sampled at Schoemansdrift

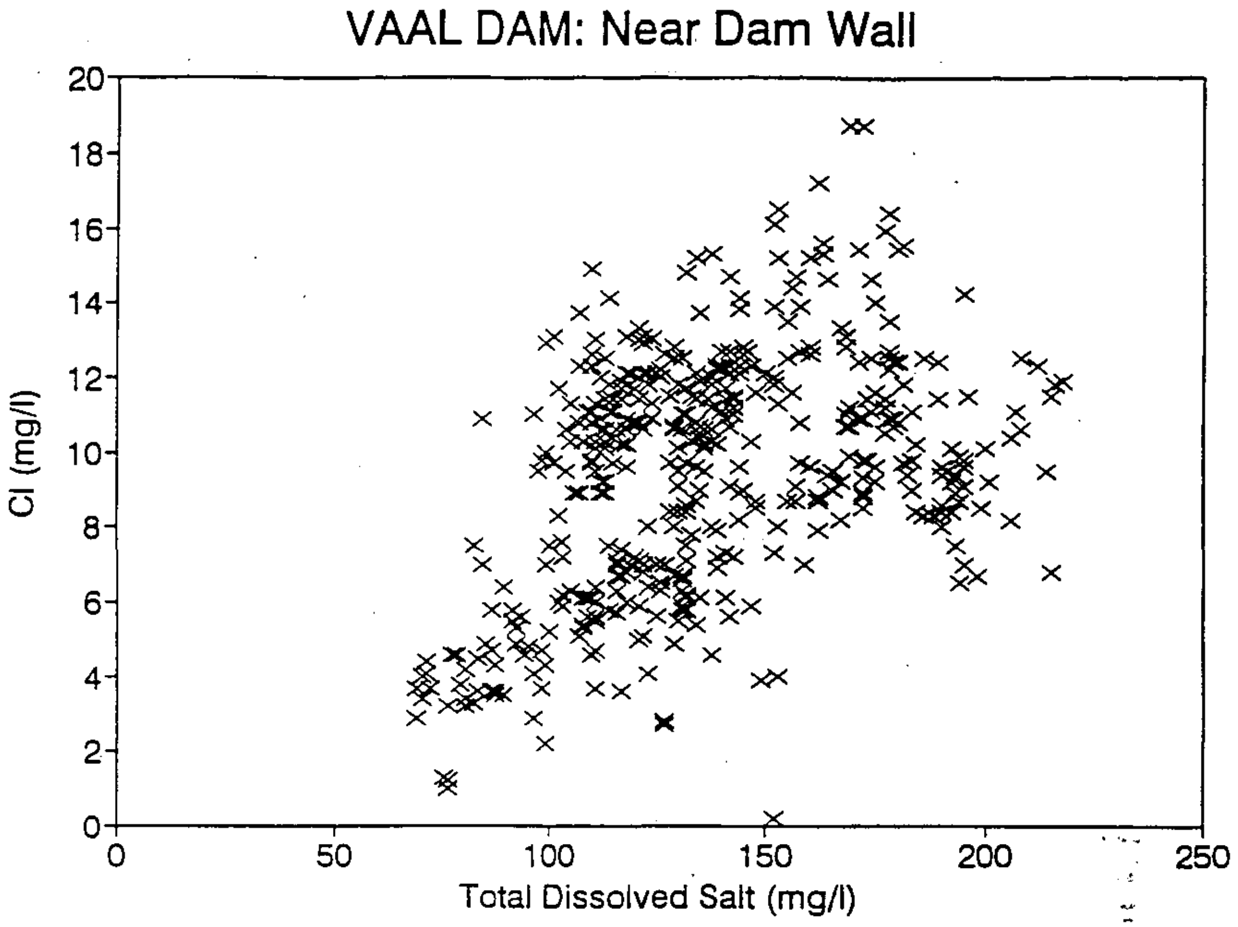

FIGURE 15: Plot of Total Dissolved Salts vs Cℓ⁻ as sampled at Vaal Dam (near dam wall)

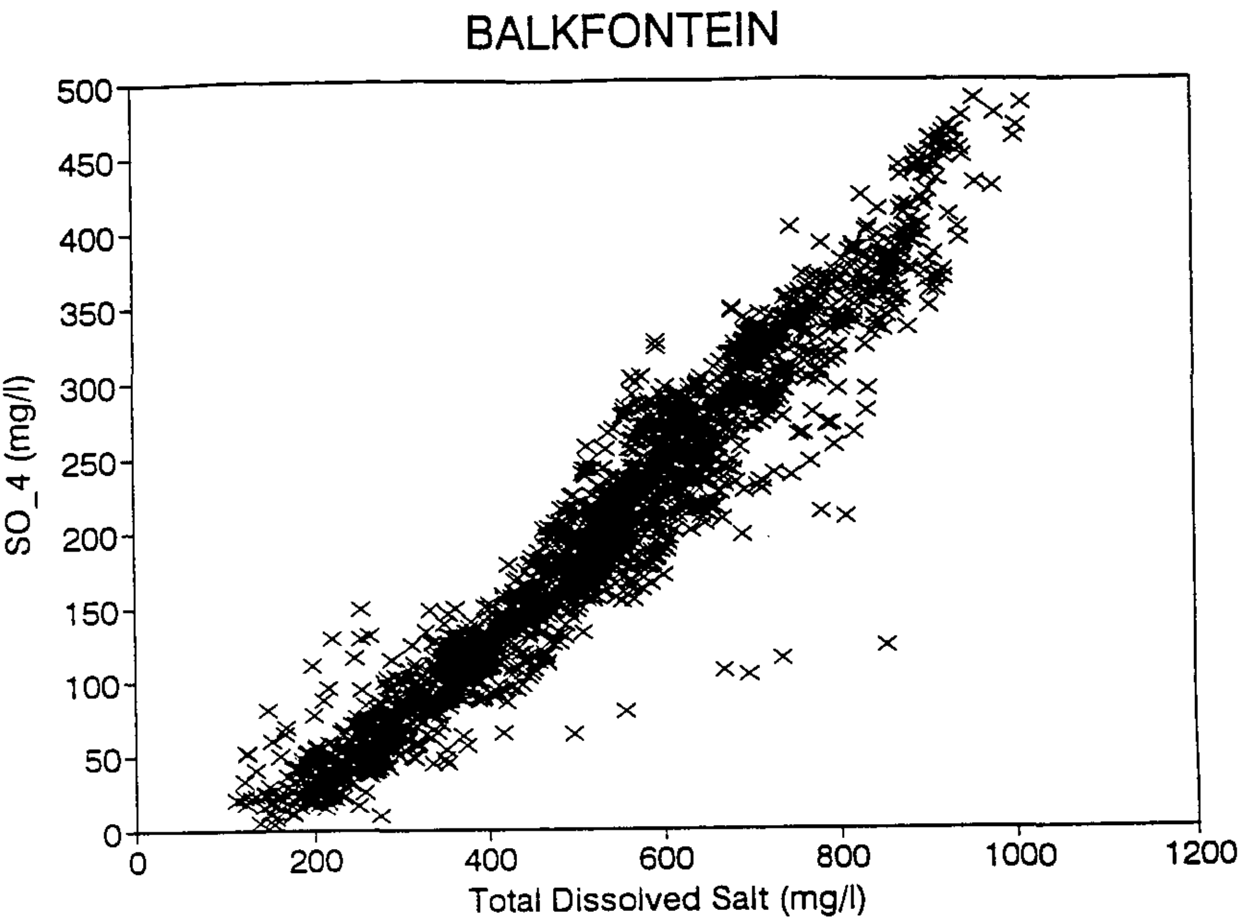

FIGURE 16: Plot of Total Dissolved Salts vs SO_4^{2-} as sampled at Balkfontein

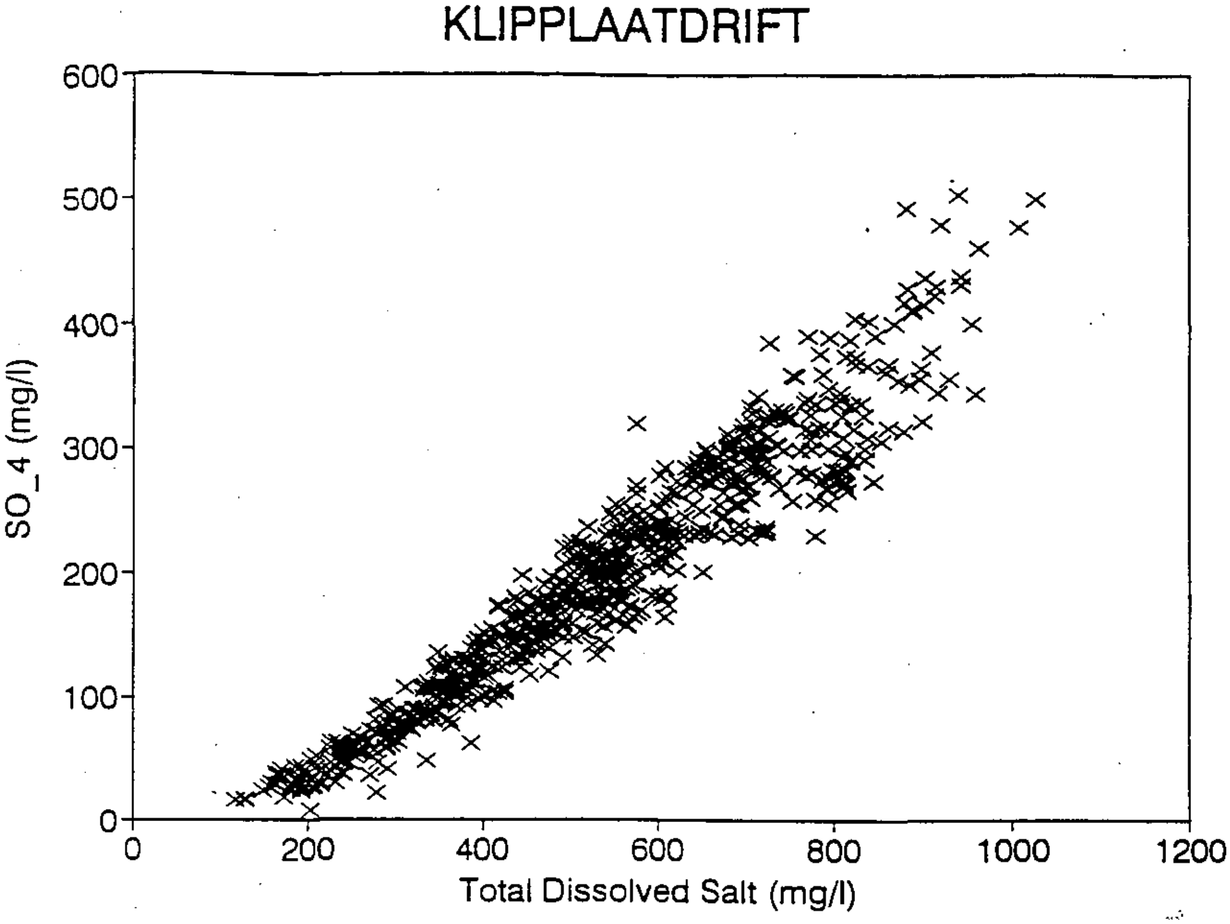

FIGURE 17: Plot of Total Dissolved Salts vs SO_4^{2-} as sampled at Klipplaatdrift

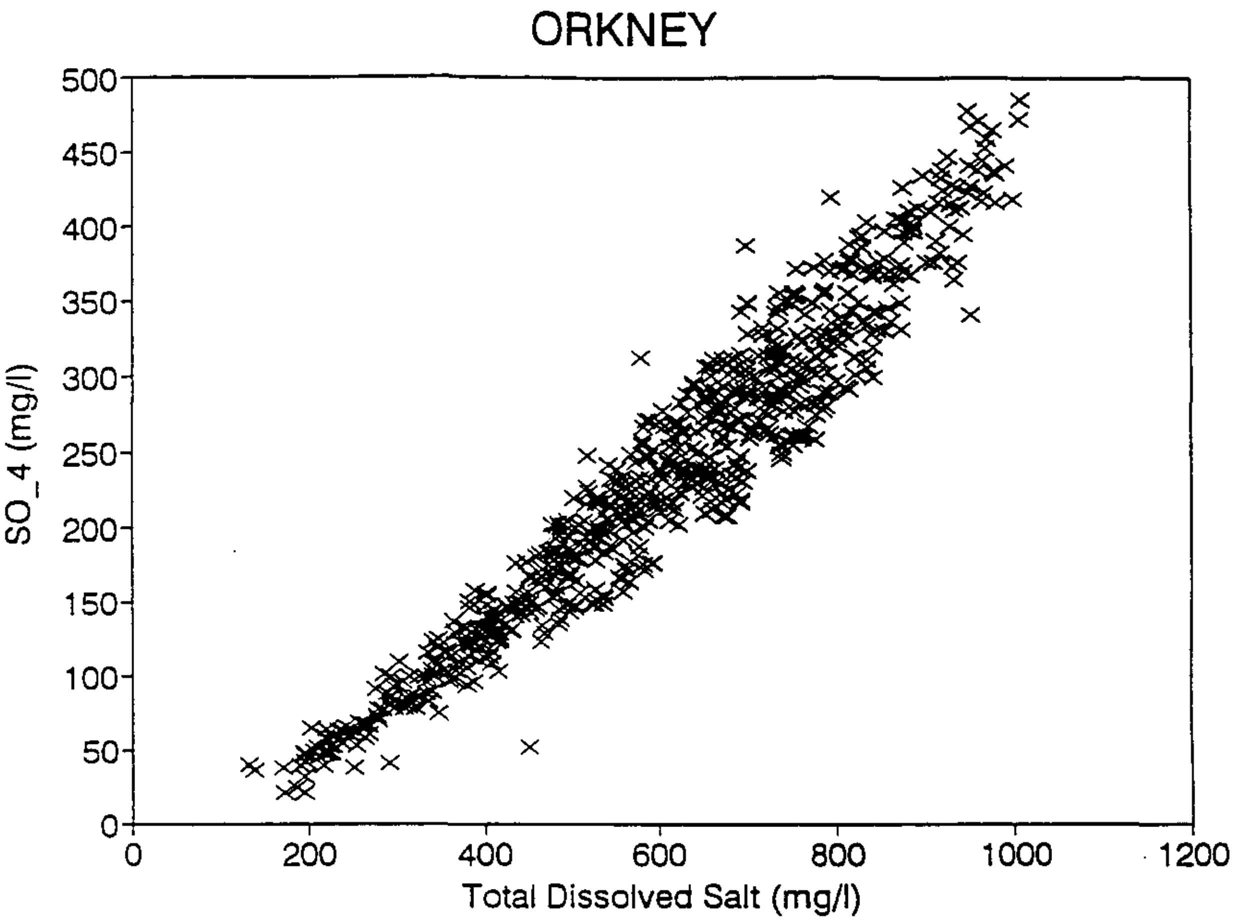

FIGURE 18: Plot of Total Dissolved Salts vs SO_4^{2-} as sampled at Orkney

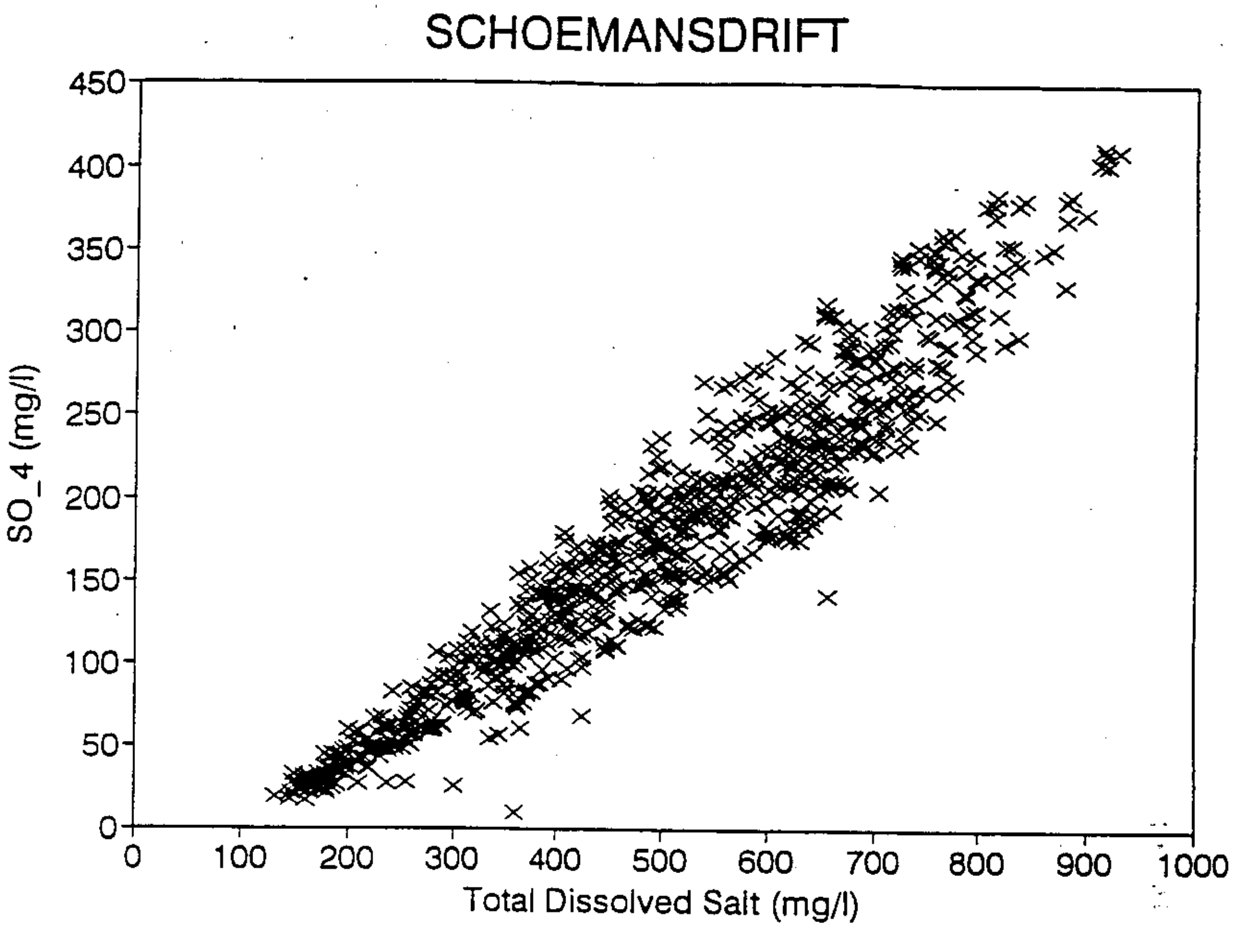

FIGURE 19: Plot of Total Dissolved Salts vs SO_4^{2-} as sampled at Schoemansdrift

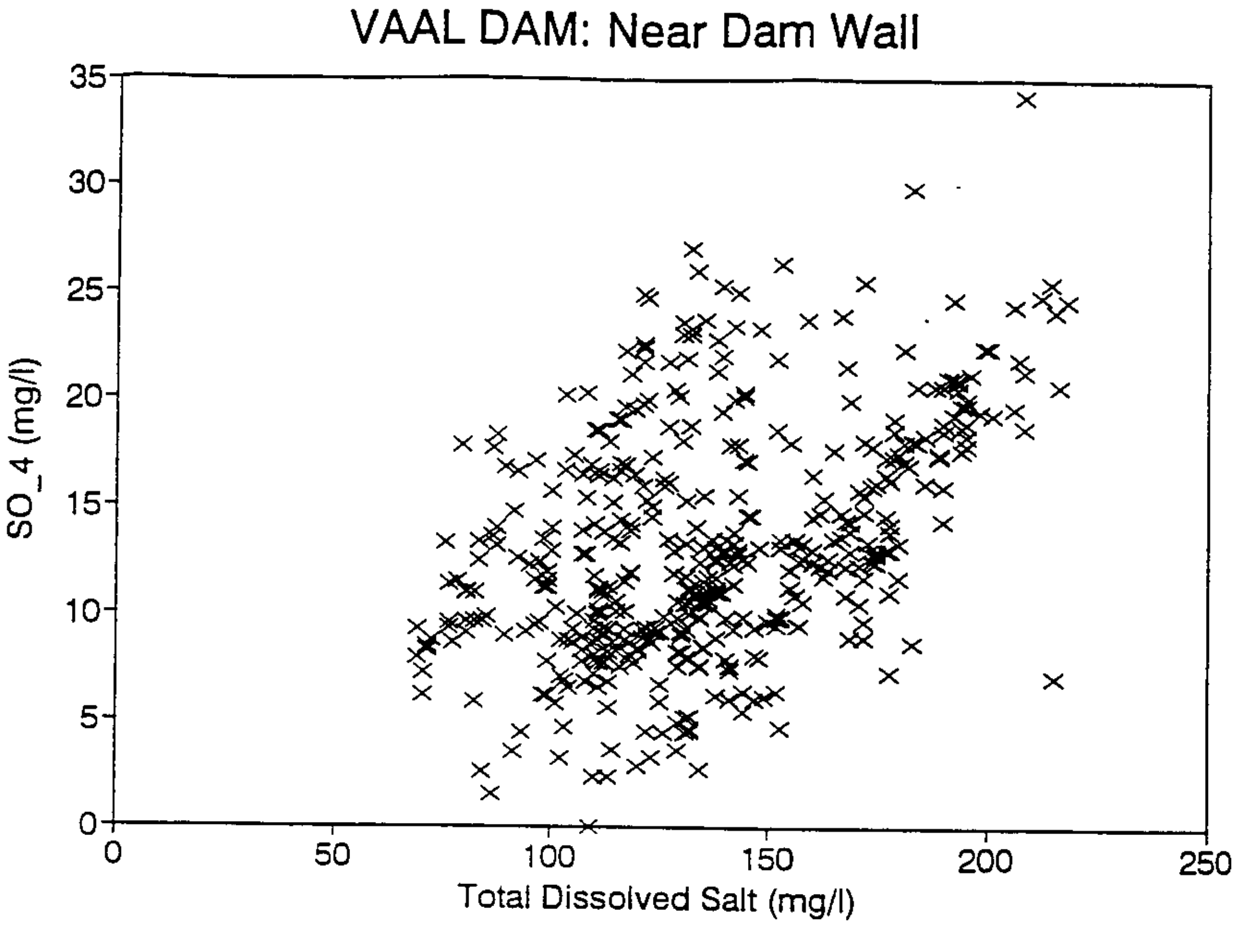

FIGURE 20: Plot of Total Dissolved Salts vs SO_4^{2-} as sampled at Vaal Dam (near dam wall)

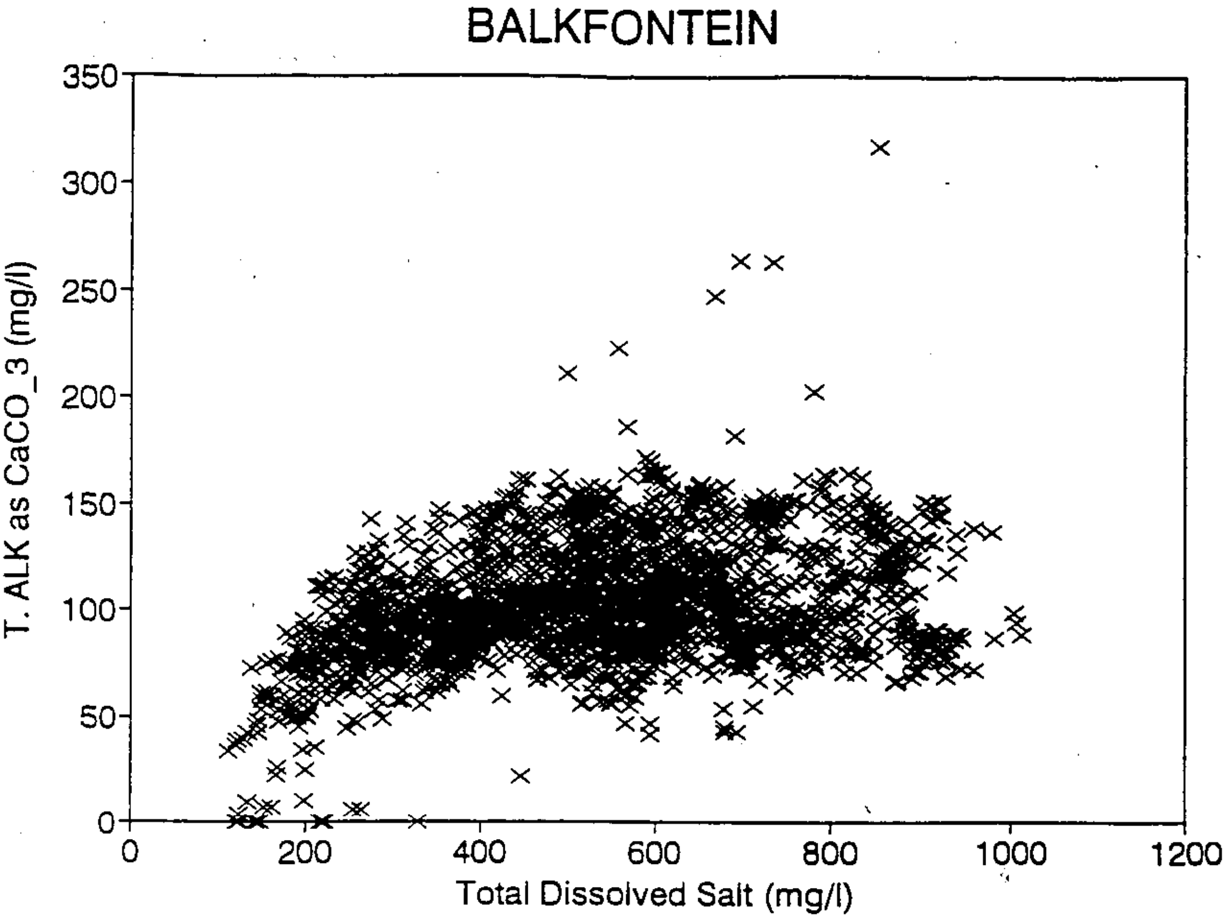

FIGURE 21: Plot of Total Dissolved Salts vs Total Alkalinity as sampled at Balkfontein

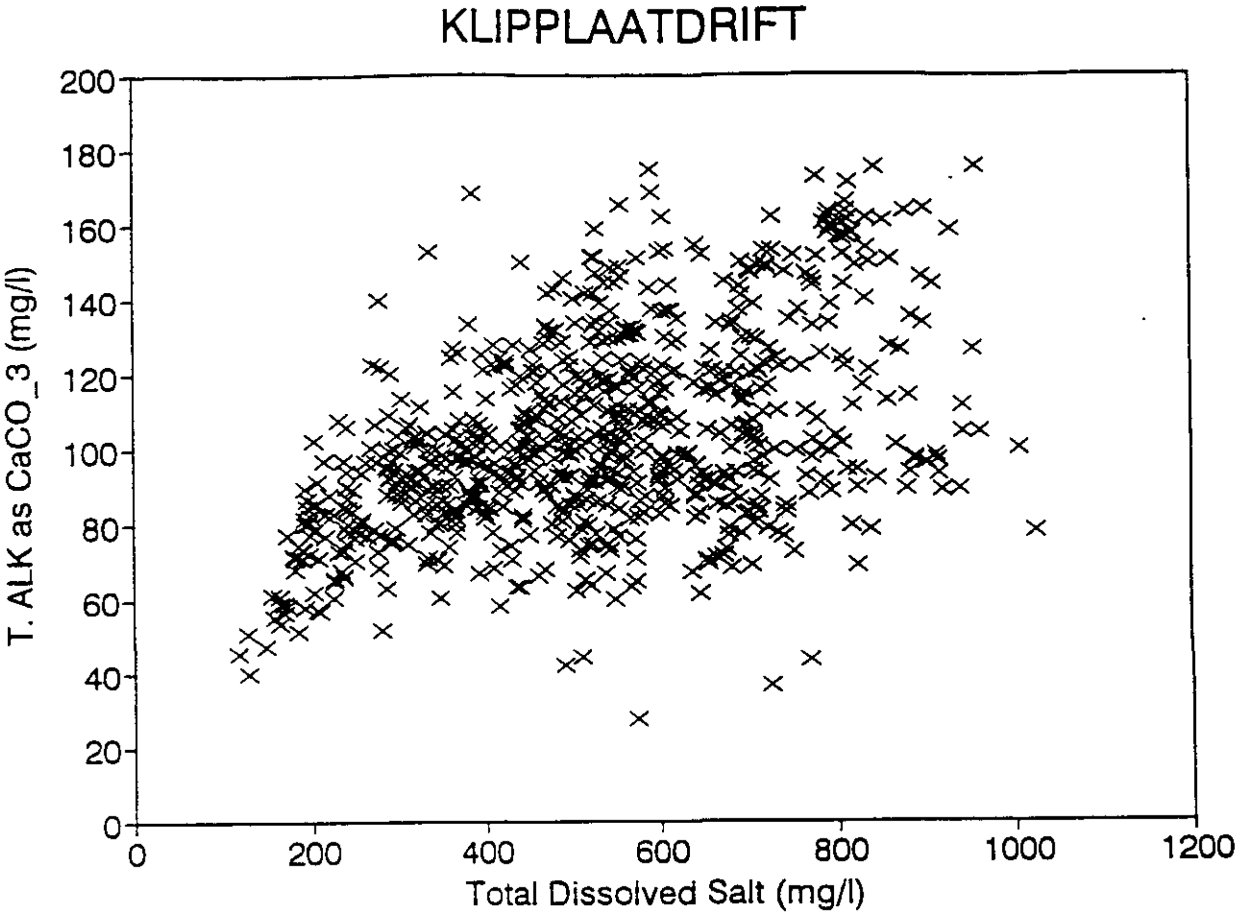

FIGURE 22: Plot of Total Dissolved Salts vs Total Alkalinity as sampled at Klipplaatdrift

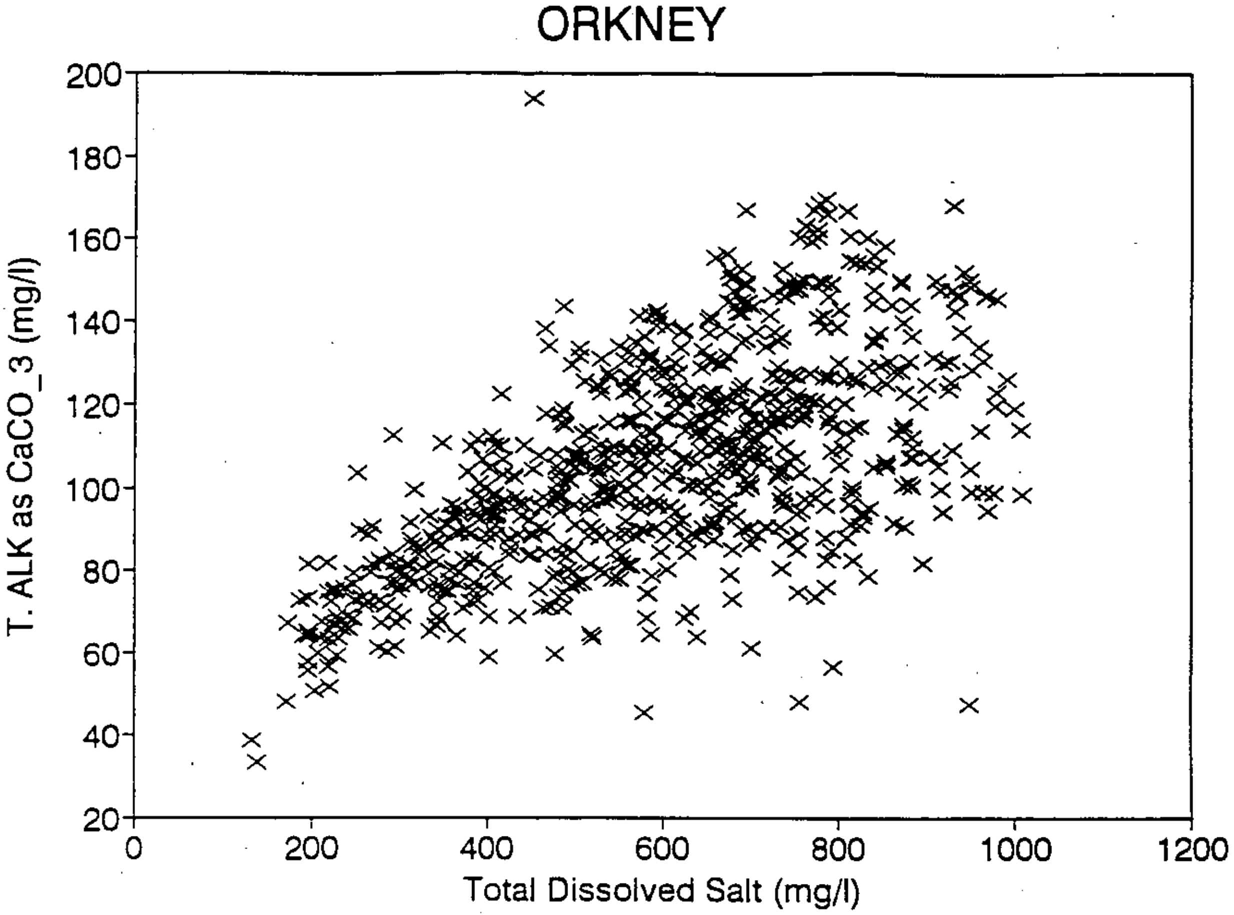

FIGURE 23: Plot of Total Dissolved Salts vs Total Alkalinity as sampled at Orkney

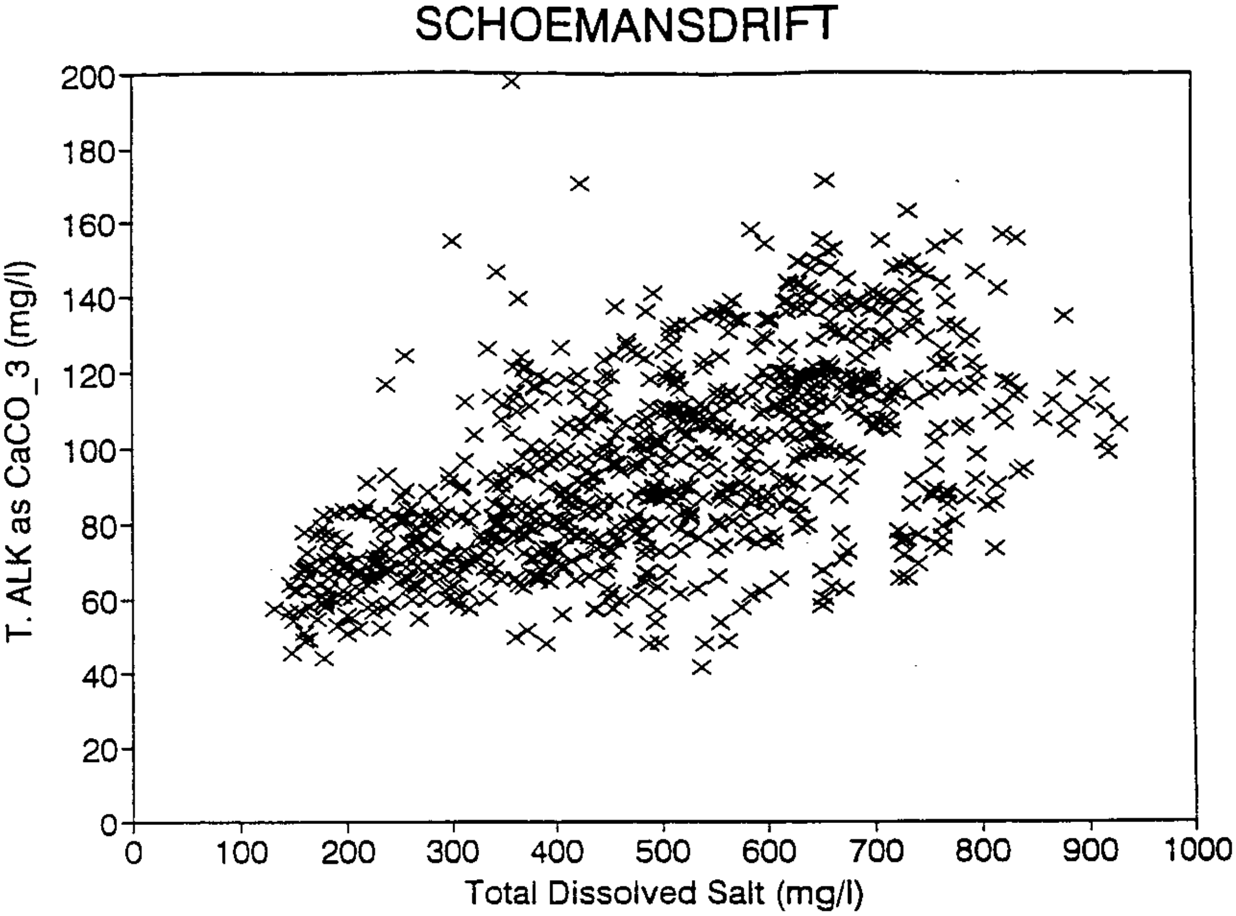

FIGURE 24: Plot of Total Dissolved Salts vs Total Alkalinity as sampled at Schoemansdrift

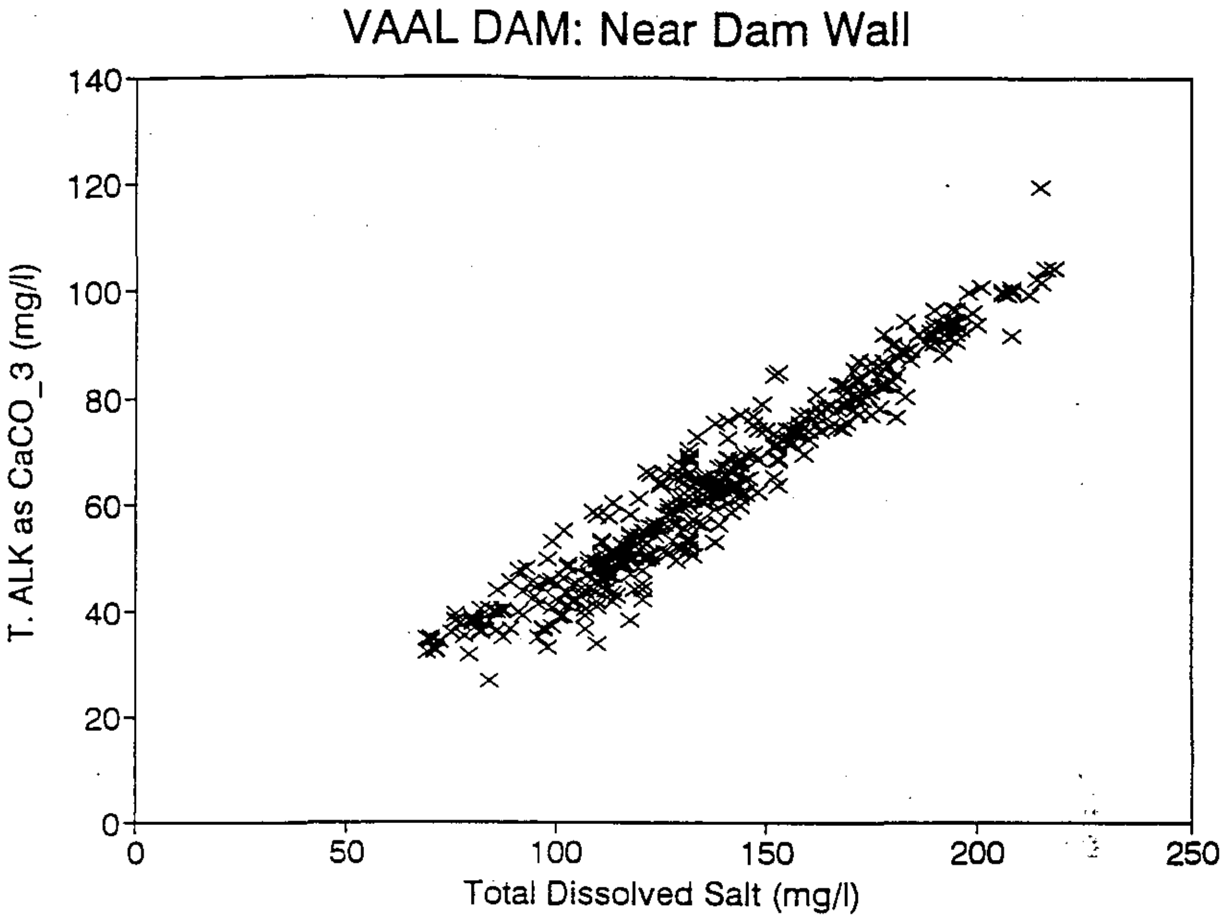

FIGURE 25: Plot of Total Dissolved Salts vs Total Alkalinity as sampled at Vaal Dam (near dam wall)

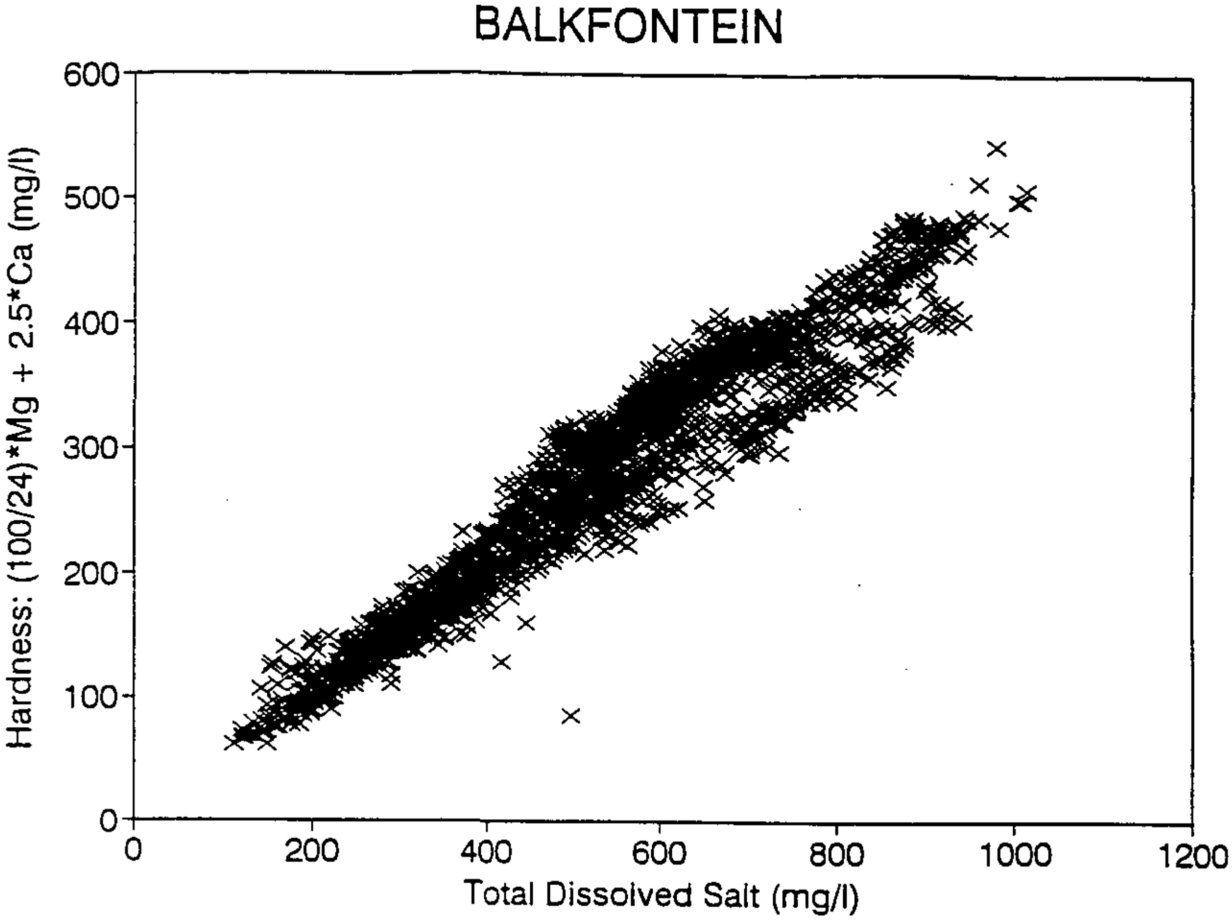

FIGURE 26: Plot of Total Dissolved Salts vs Hardness as sampled at Balkfontein

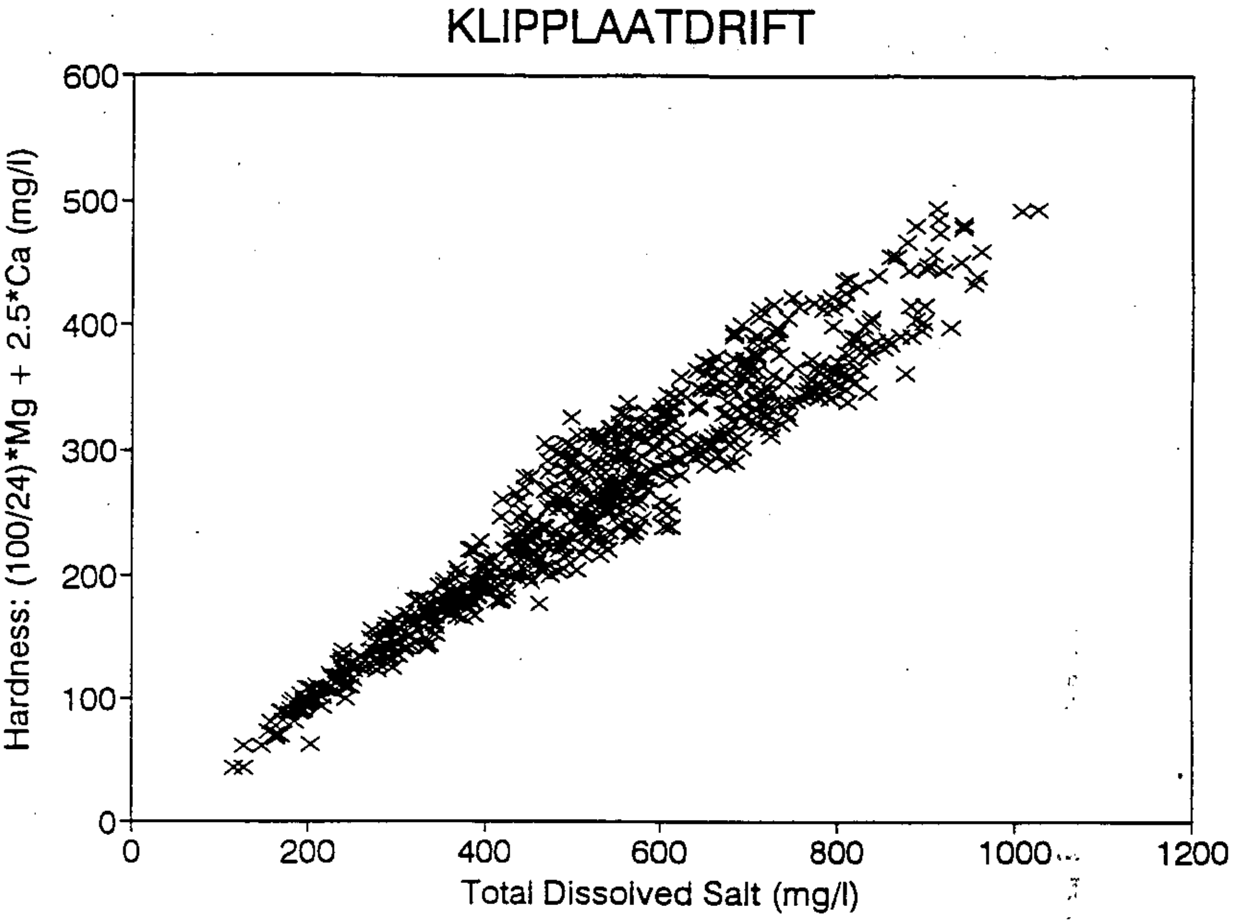

FIGURE 27: Plot of Total Dissolved Salts vs Hardness as sampled at Klipplaatdrift

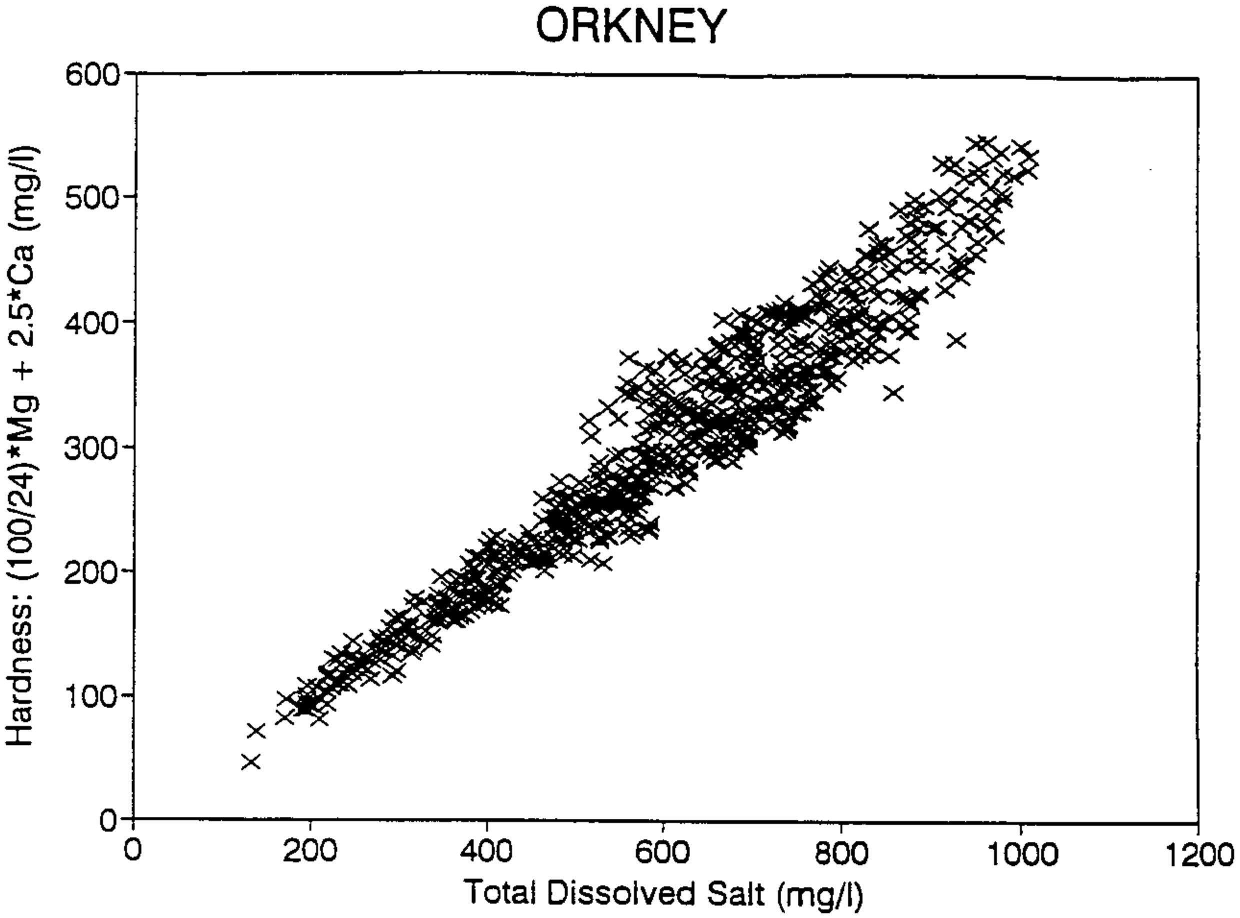

FIGURE 28: Plot of Total Dissolved Salts vs Hardness as sampled at Orkney

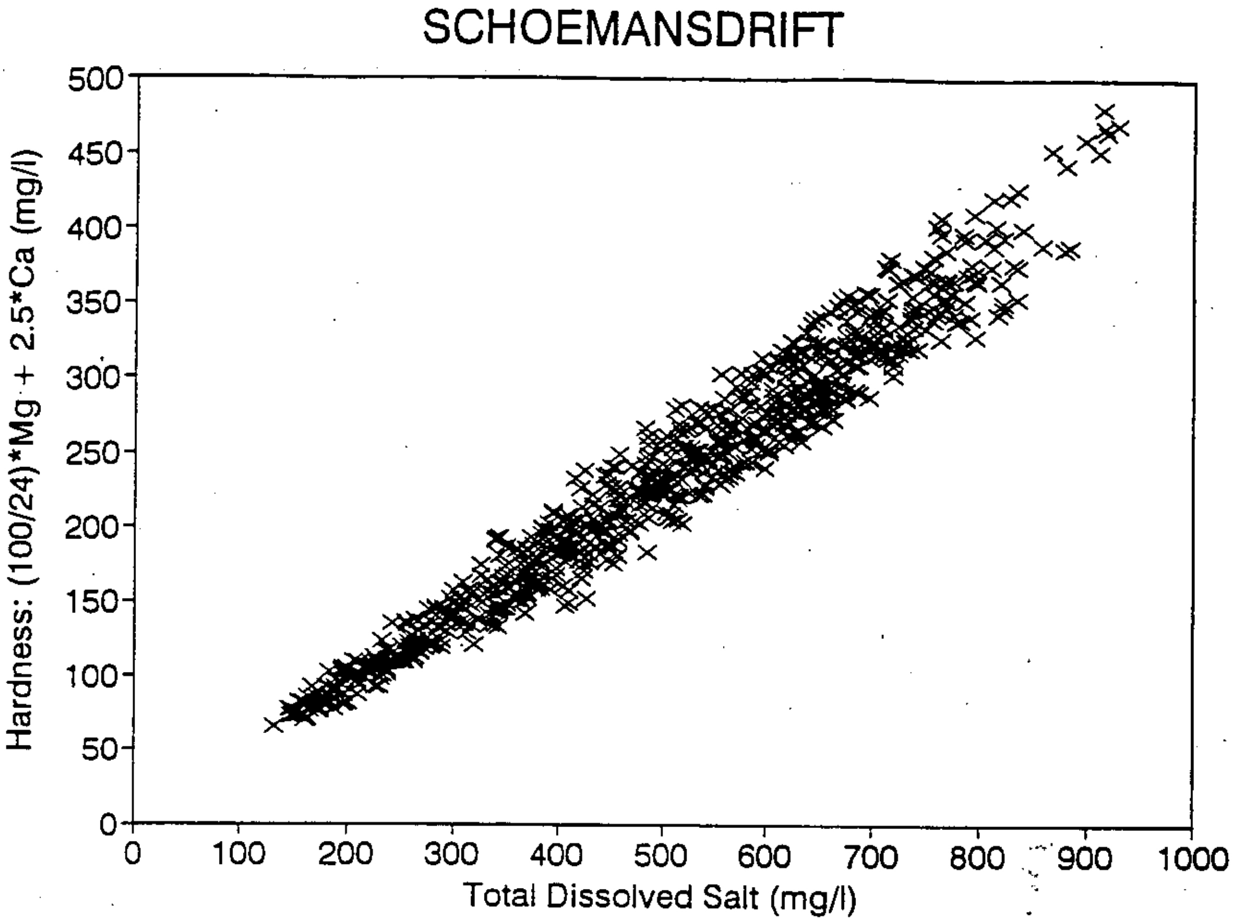

FIGURE 29: Plot of Total Dissolved Salts vs Hardness as sampled at Schoemansdrift

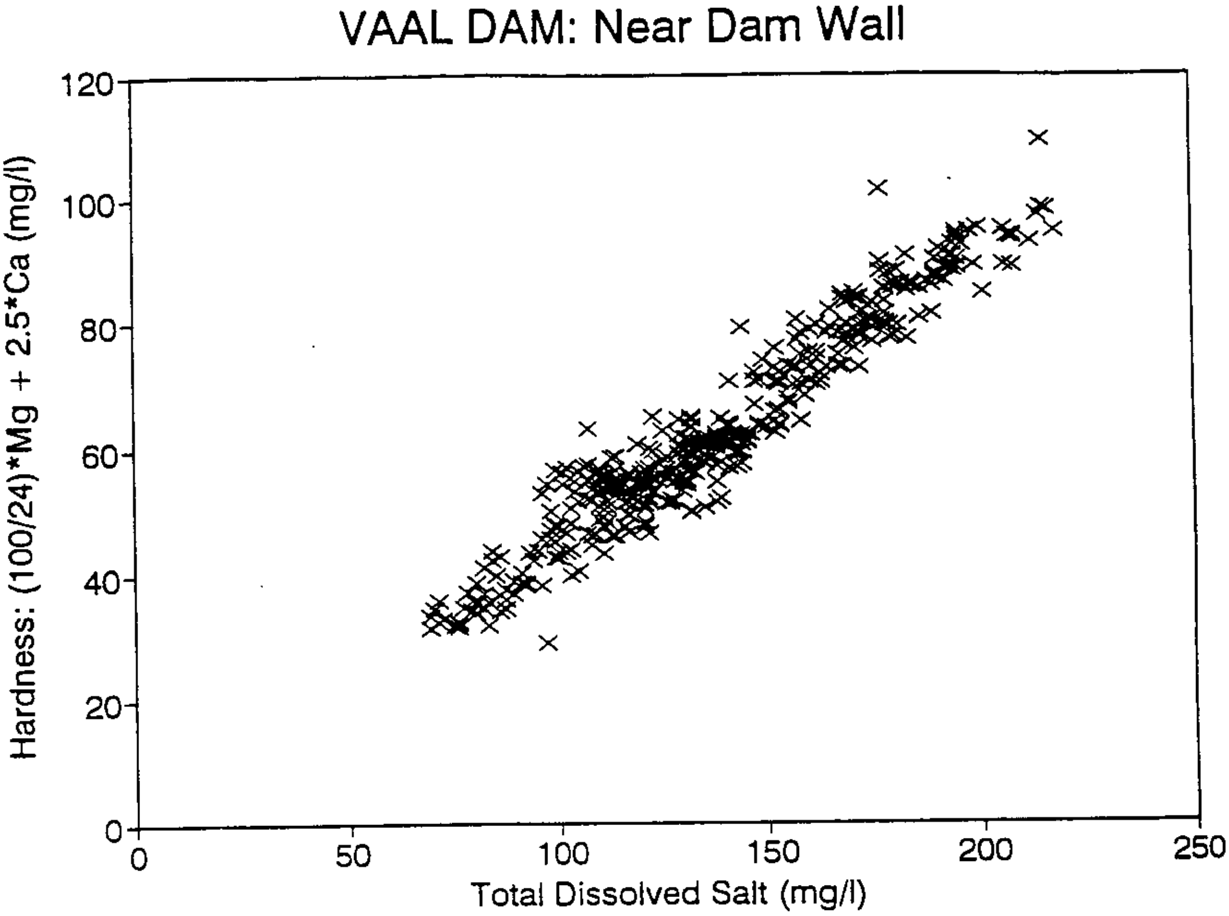

FIGURE 30: Plot of Total Dissolved Salts vs Hardness as sampled at Vaal Dam (near dam wall)

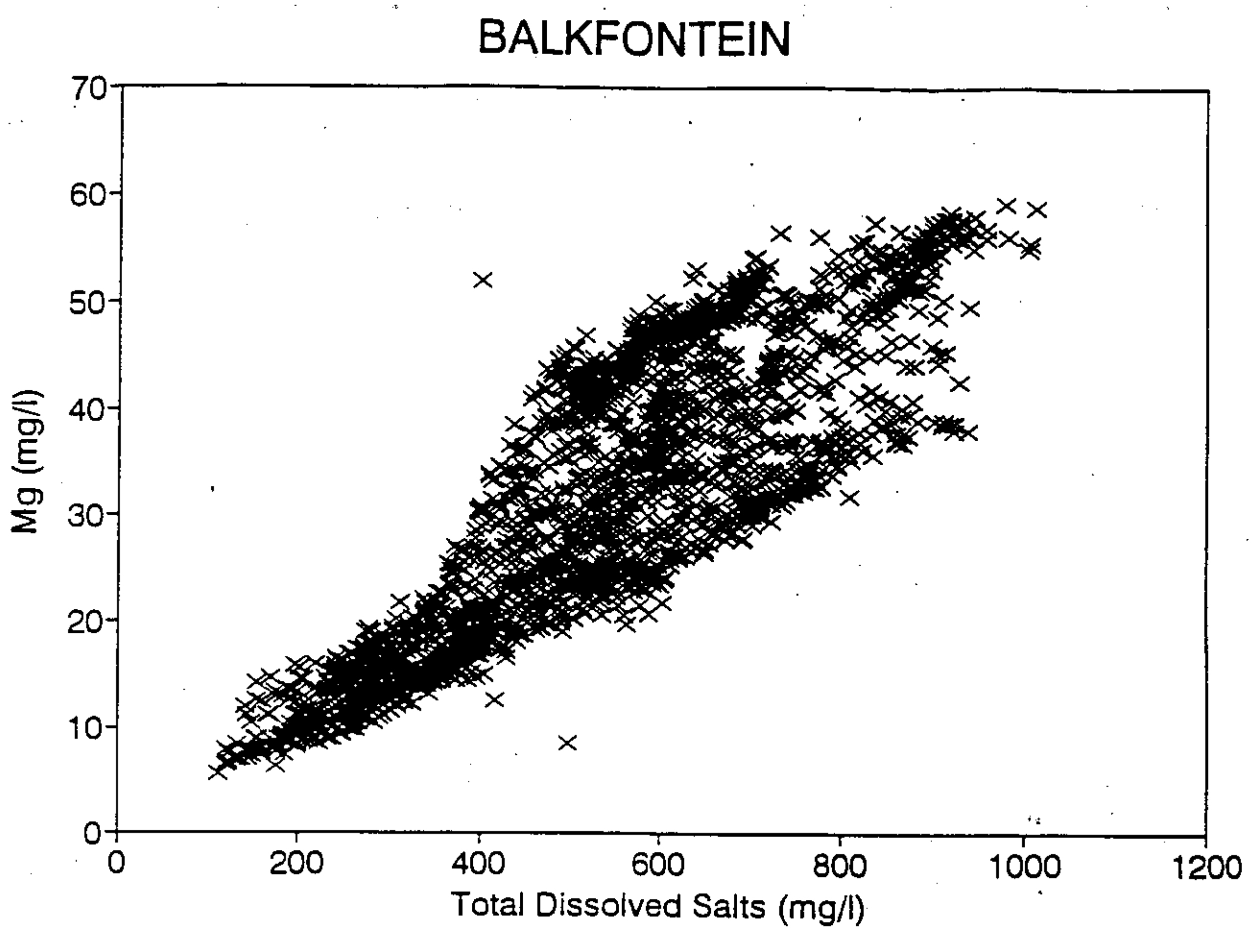

FIGURE 31: Plot of Total Dissolved Salts vs Mg^{2+} as sampled at Balkfontein

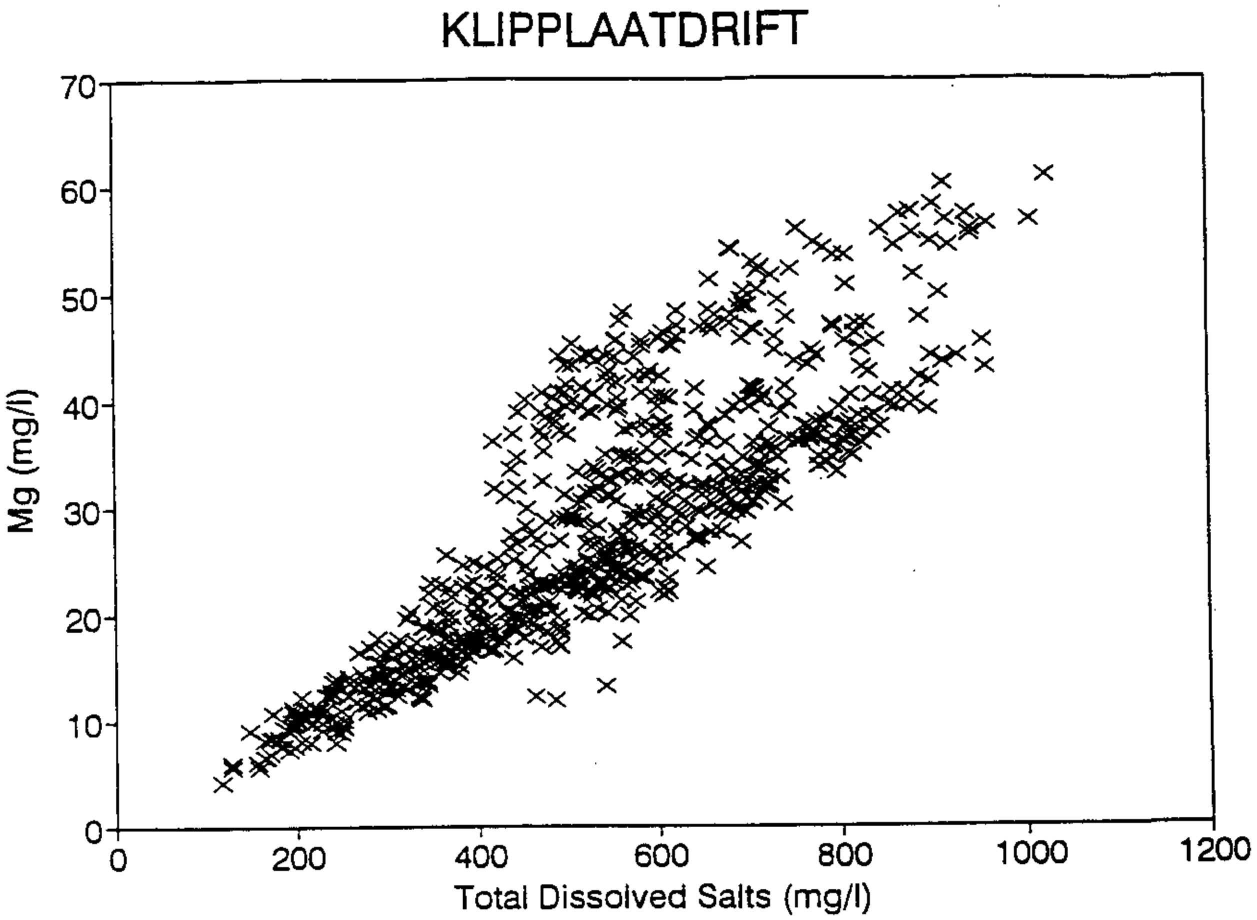

FIGURE 32: Plot of Total Dissolved Salts vs Mg^{2+} as sampled at Klipplaatdrift

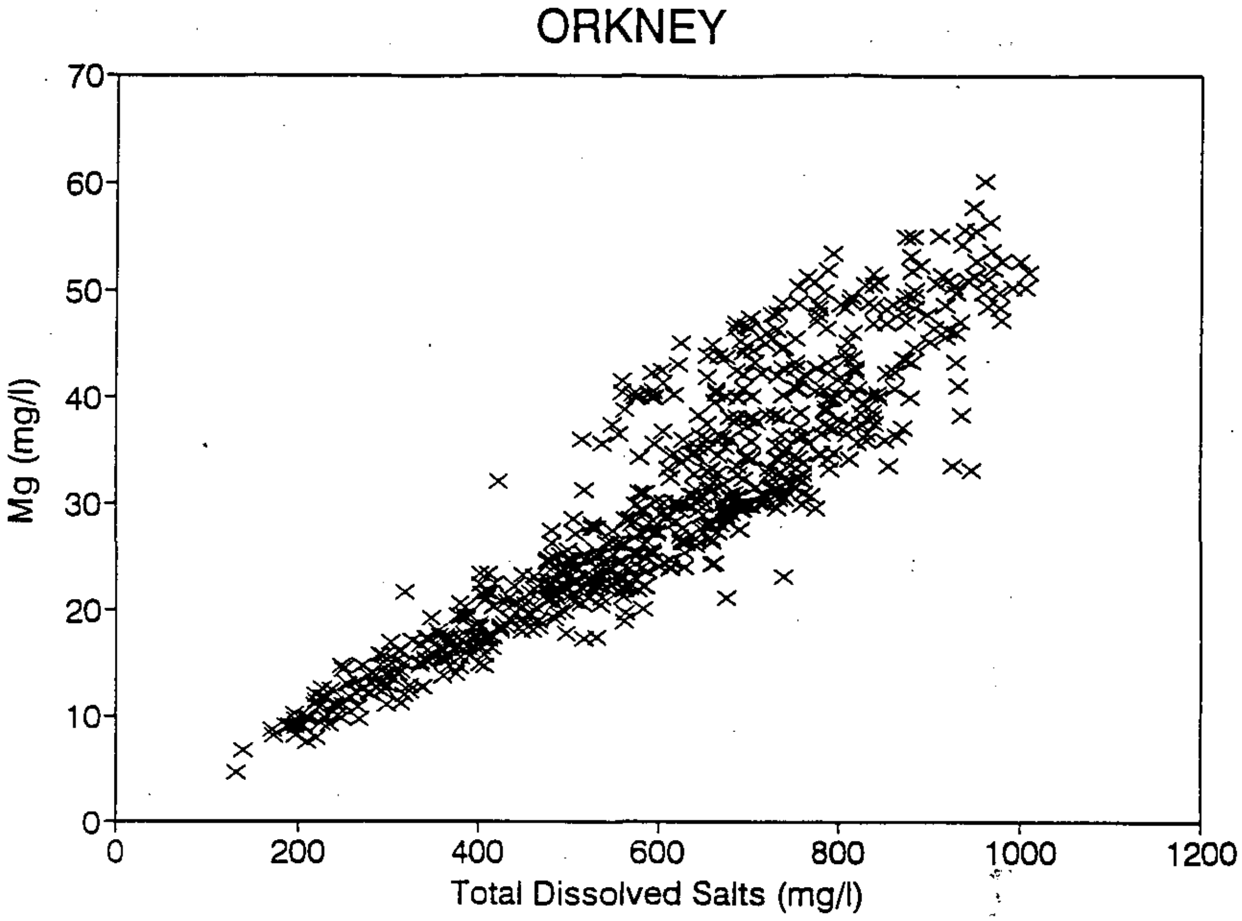

FIGURE 33: Plot of Total Dissolved Salts vs Mg^{2+} as sampled at Orkney

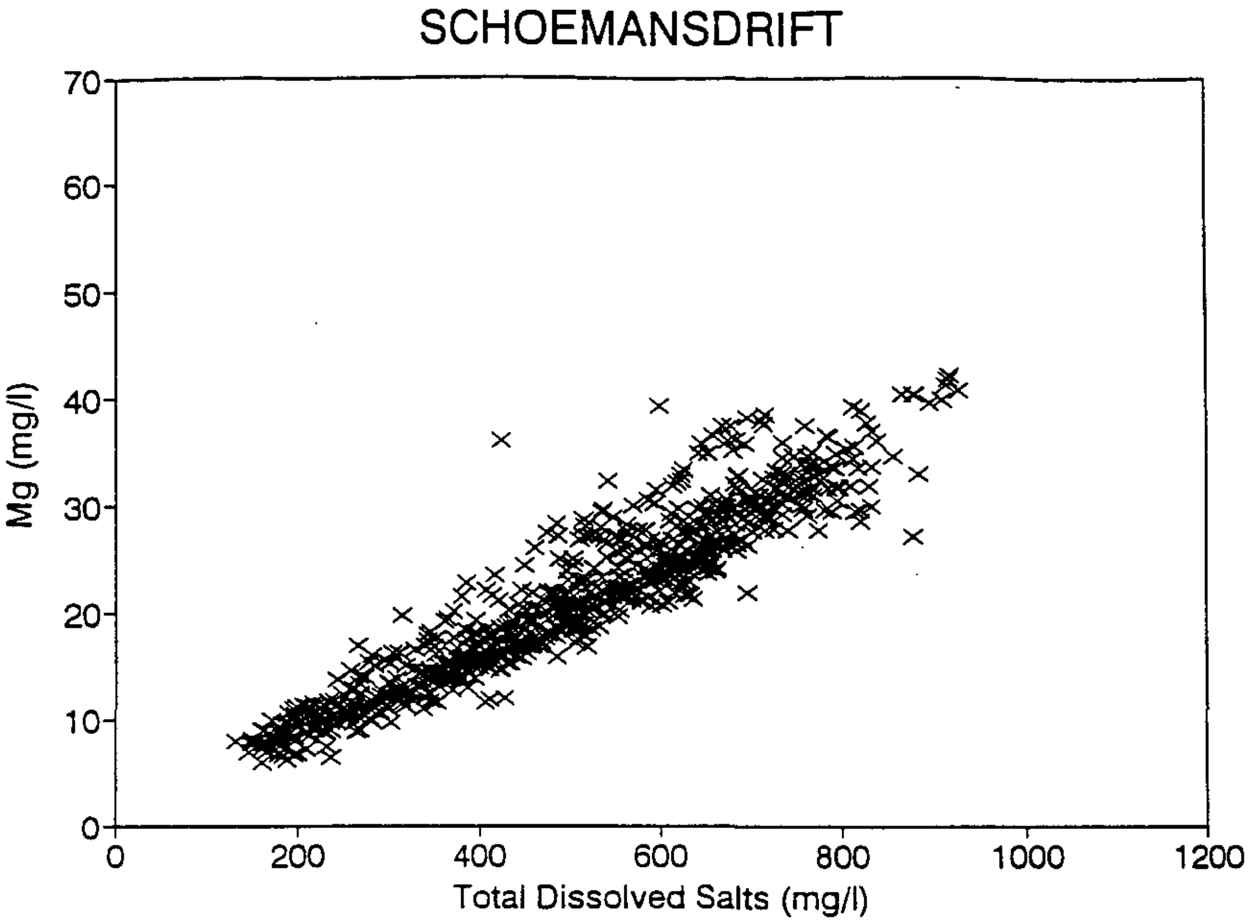

FIGURE 34: Plot of Total Dissolved Salts vs Mg^{2+} as sampled at Schoemansdrift

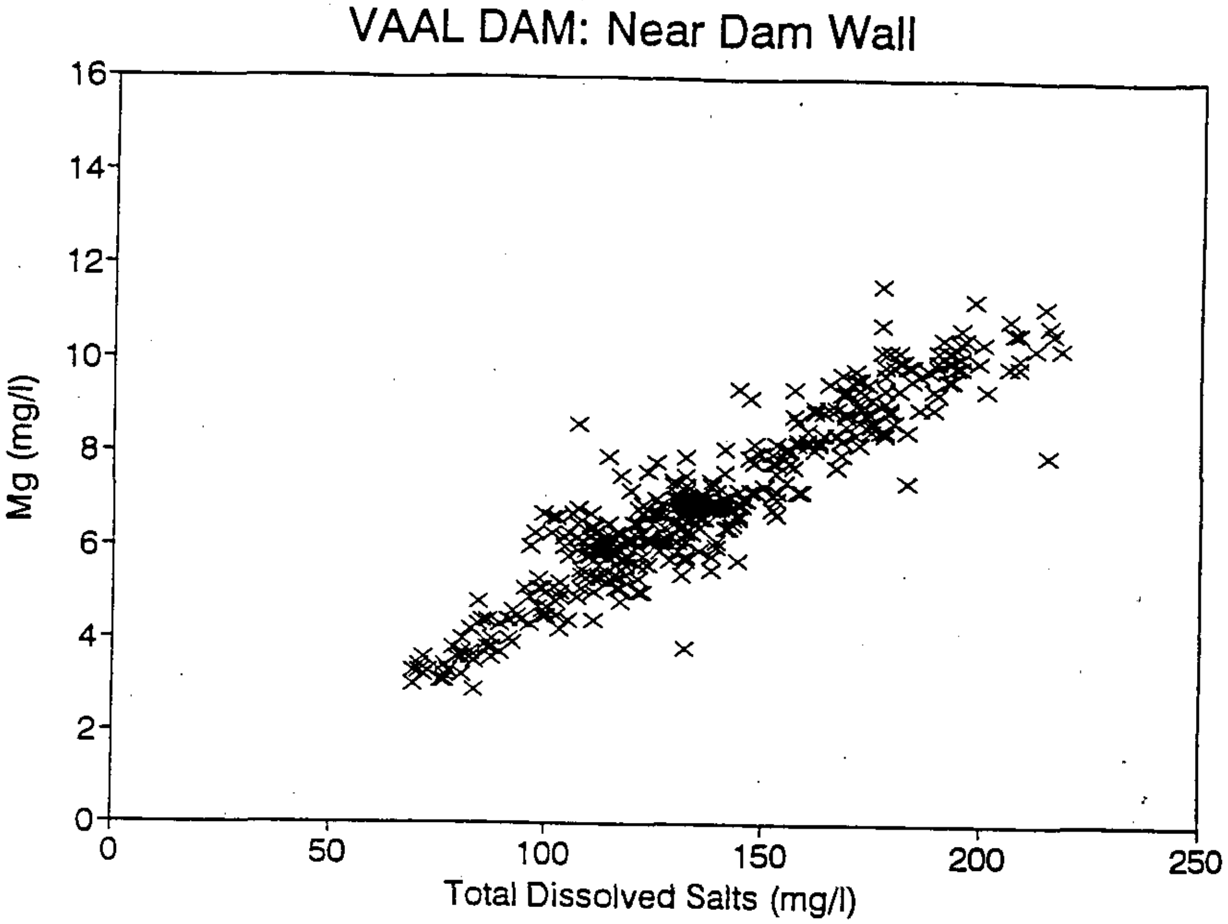

FIGURE 35: Plot of Total Dissolved Salts vs Mg^+ as sampled at Vaal Dam (near dam wall)

THE INFLUENCE OF INCREASED SALINITY ON THE COST OF WATER TREATMENT: CALIBRATION OF PROPOSED MODEL FOR THE MIDDLE VAAL SYSTEM

1. INTRODUCTION

It is generally accepted that increased salinity in water results in increased costs for treatment and conveyance. The increased conveyance cost is normally accepted to be an indirect cost due to corrosion or additional maintenance as the physical properties of the water is not changed sufficiently to impact on the direct conveyance costs. The density, viscosity and other properties remain the same and does not influence pipe or pump sizes.

A comprehensive literature survey has shown that it is impossible to model treatment costs or even corrosion rates with increased salinity based purely on Total Dissolved Solids (TDS) measurements. Both the actual chemical make-up of the TDS (i.e. the concentrations of the different ions) and the physical environment of the raw water play major roles in the cost responses to increased salinity.

The purpose of the total study is to model or determine the incremental costs to the total economical system due to increased salinity. The study covers industrial, domestic and agricultural consumers. It is postulated that a change in salinity will have a possible cost impact on each of the above groups of consumers.

A maximum cost of the impact should be the cost to treat the water to revert back to its original state (i.e. to desalinate the water). It is thus postulated that the cost of desalination should be taken as the upper limit of the costs and if the sum of the individual effects exceed this cost, desalination must be implemented. It will be shown later in this review that the cost responses will at best be in a series of steps and not as a smooth linear function.

In Phase I of the study it was established that the corrosivity of the water for the whole range of TDS (200 mg/ℓ -1200 mg/ℓ) based on the hypothetical ionic makeup will require a suitable corrosion protection system such as Copon coating for all steel pipes. It is not required to install a more sophisticated system for the upper limits of the TDS range. It was thus postulated that incremental costs to feeder systems due to incremental TDS can be accepted as zero and this cost factor can be removed from the cost model.

2. SALINITY STANDARDS FOR DRINKING WATER

SABS 241 sets certain limits or standards for the salinity of drinking water. The standards are measured as Electrical Conductivity (EC) readings for ease of measurement as a well established relationship exists between EC and Total Dissolved Solids (TDS).

Recommended level : 70 mS/m
Maximum level : 300 mS/m

The relationship between conductivity and total dissolved solids (TDS) is:

TDS (mg/ℓ) = EC (mS/m) x 6,5

The factor of 6,5 is approximate and can be influenced by the actual ionic make-up of the dissolved solids. It can be as high as 9,0 or as low as 4,0. The 6,5 is however fairly accurate for the average mixed ionic make-up of natural water.

Based on the above, the preferred or recommended TDS level is 455 mg/ℓ and the maximum allowable level is 1950 mg/ℓ. These standards correspond well with similar standards set by the Environmental Protection Agency (EPA) of the United States of America (500 mg/ℓ TDS recommended level) and the World Health Organization (WHO) with a value of 1500 mg/ℓ TDS as an "excessive limit".

The South African Water Quality Guidelines for Domestic Use (2nd Exdition) classifies a TDS of 1000 mg/ℓ-2450 mg/ℓ as fit for interim use (Class II water). The limits set for Class I waters is 450 mg/ℓ-1000 mg/ℓ. This Class I water has unrestricted use. With the 1200 mg/ℓ upper limit for this study close to the 1000 mg/ℓ limit, it is postulated that it should also be considered fit for longterm use.

The conclusion from the above is that the study range of TDS values between 200 mg/ℓ and 1200 mg/ℓ falls mostly outside the recommended standards (± 450 mg/ℓ TDS) but in all cases below the maximum permissible standards for the various agencies (1500 mg/ℓ to 2450 mg/ℓ).

This statement is only true for a mixed ionic make-up and care must still be taken to ensure that problem ions such as Cl^-, SO_4^{2-}, etc are within the recommended limits.

There is thus no legal obligation to desalinate the water to make it safe for human consumption within the specified study range. If it is however done, it will add significantly to the cost of treatment as desalination involves very sophisticated technologies such as reverse osmosis (RO), electro-dialysis (ED) or any of a large variety of evaporation systems. All of these are high capital, maintenance and energy processes. It may be pointed out that RO or ED (both membrane processes) will be the best options in the 1200 mg/ℓ TDS range.

If a policy decision is made to desalinate water, the cost of the selected process is an add-on to the process of purifying the water as desalination normally only follows full purification.

For the purpose of this study it is accepted that desalination will not be done, but will be used as the upper limit of incremental cost due to increased TDS for the domestic and industrial sectors. As soon as their incremental costs exceed the cost of desalination, the latter must be implemented. The same approach can not be followed for agriculture as the large volumes involved and high cost of desalination makes it unlikely that the incremental costs to this sector will ever exceed that of desalination of the massive volumes of water involved.

It will be necessary to bear the cost of desalination in mind to ensure that the sum of incremental costs do not exceed this.

3. UPPER LIMIT OF COSTS

Increased salinization may result in increased costs to consumers due to, inter alia, increased corrosion costs, increased cost to households due to higher soap consumption, failure of electrical appliances, increased treatment costs, increased process and also treatment costs for industries utilizing very high quality water as part of their process, or making the water totally unacceptable for the specific use.

As with all costs, the model will require periodic updating as technology changes and improvements will tend to lower costs, while normal inflation will increase costs.

The current costs to desalinate water using RO is ± R0,90 /m^3 for a TDS up to 800 mg/ℓ and R1,50/m^3 for a TDS up to 1200 mg/ℓ. Optimized blending can reduce these costs further. The cost of brine disposal will influence costs and a figure as high as R3-50/m^3 (total costs) is not unrealistic. The only purpose of this figure is to alert the team to the likely cost limits and to avoid wasting effort to refine the determination of individual cost increments for domestic or industrial consumers, if it exceeds the above value.

These costs are the upper limits of costs for the commercial, industrial and residential sectors. The agricultural sector will never be able to pay for desalination and it does not apply to them, The CSIR Watertek (Dr Japie Schoeman - tel no 841-2270) provided the above costs in a special report compiled for this study (Economics of desalination Processes - A Literature Study by J J Schoeman, Project no WD021/6000/6840).

4. COST ELEMENTS FOR TREATMENT OF WATER

The treatment cost of water is made up of the following cost elements:

(i) Capital and redemption costs: A
(ii) Chemical costs: B
(iii) Labour costs: C
(iv) Maintenance costs: D
(v) Energy costs: E
(vi) Disposal costs of residue such as sludge, brine etc: F

These elements make up the total cost of water treatment and the potential influence of increased salinity on each will be identified and discussed in the following paragraphs.

4.1 INFLUENCE OF PHYSICAL CHARACTERISTICS OF RAW WATER (TROPHIC STATE) ON COST OF TREATMENT

An important factor regarding the cost of treatment of water is the physical characteristics of the raw water, specifically with reference to its trophic state. If the water has a high turbidity due to inorganic particles in suspension, it will normally have a low particulate organic solids concentration and will be fairly easy to treat by means of conventional treatment systems.

This statement used to be generally accepted, but the Western Transvaal Water Corporation reported high inorganic turbidities together with high chlorophil concentrations during the latter half of 1995. This is a new phenomena.

If the water has a high organic solids content (i.e. the water is eutrophic) it treats less easy and requires more expensive processes such as dissolved air flotation, normally in addition to conventional treatment.

Salinity has an indirect as well as direct role on the trophic state of water. Increased salinity tends to destabilize the stable inorganic colloidal suspension which causes turbidity in raw water. (Personal communication with Mr S. van der Merwe, Rand Water). This can generally happen at TDS concentrations of ± 400 mg/ℓ. If the flow regime of the water is not sufficiently turbulent to keep the colloids in suspension (like in a rapidly flowing river), the water will tend to clarify as the colloids flocculate and settle out. This will reduce turbidity, increase light penetration and the expected result will be increased algae growth resulting in eutrophication. This is based on the assumption that sufficient micro nutrients exist to sustain algal growth. The high organic content is more difficult to treat, requiring the use of the above-mentioned additional treatment processes. The direct effect of increased TDS, if the latter contains significant concentrations of nutrients, will be to enhance the process of eutrophication.

A further complication in the treatment of highly eutrophic water is the fact that taste and odour problems can occur. This will require further sophisticated processes such as ozonation and activated carbon absorption to control.

A complicating factor in the use of the model is the fact that the cost of water treatment is not only influenced by the concentration of phytoplankton in the raw water, but also by the dominant species. If *it is toxic blue-green algae, activated carbon absorption will be required*, while other species, even at higher concentrations, will not be as problematic. In the case of green species or diatoms, activated carbon may perhaps be omitted.

The actual concentration of the phytoplankton will also have an influence on the cost of treatment.

The Umgeni Water (contact person Mr Mark Graham) is currently busy with a WRC study "Modelling the causes of Algal Blooms in Impoundments of the Umgeni Catchment and the Consequences for Portable Water Treatment" which should be consulted when predictions for algae species and concentrations are required.

It must be pointed out that these purification steps are required even if it is decided to ultimately desalinate the water. Only purified water can be desalinated.

While it is thus impossible to predict, by means of a mathematical model, when increased salinity will cause eutrophication and hence increased treatment costs, it can be assumed that at a TDS level of 1200 mg/ℓ it is likely that eutrophication will have occurred, needing more expensive treatment processes. The increased cost will also be in a step (by adding dissolved air flotation) and a further step if activated carbon adsorption is added.

The likely TDS levels where this will occur is at ± 400 mg/ℓ. This figure applies only to a fast flowing river and Umgeni Water experiences algal blooms at much lower TDS values. It must therefore be accepted that there is no direct relationship between increased TDS and treatment costs. The response to increased TDS is very river or water source specific.

4.2 POSTULATED INFLUENCE OF INCREASED TDS ON THE COST OF WATER TREATMENT COST ELEMENTS

The following model is proposed as an effect to quantify treatment costs due to changes in the trophic state of the water.

4.2.1 Capital Cost

Increased TDS can change the trophic state of the water as described in paragraph 3.1 above requiring more sophisticated treatment processes thus increasing capital cost.

Capital Cost : A = $A_1 + / TDS - TDS_{TH_1} / A_2$ 1
$+ /TDS - TDS_{TH_2} / A_3 + /TDS - TDS_{TH_3} / A_4$ 2

A = Total capital cost
A_1 = Capital Cost of a conventional plant
A_2 = Capital Cost for dissolved air flotation
A_3 = Capital Cost for activated carbon
A_4 = Capital Cost for ozone
TDS = Actual TDS
TDS_{TH_1} 3 = threshold TDS for the specific system where algae occur due to eutrophication and where dissolved air flotation is required.
TDS_{TH_2} 4 = threshold TDS for the specific system where odour and taste occurs
TDS_{TH_3} 5 = threshold TDS for the specific system where ozone is required

A, A_1, A_2, A_3 and A_4 depend on size and other site specific factors.
$/TDS - TDS_{TH} /$ = Modulal Function 6 It is 0 if <1 and 1 if ≥ 0

TDS_{TH1} }
TDS_{TH2} } Must be determined for site specific situations.
TDS_{TH3} }

4.2.2 Chemical costs

Chemical costs for coagulation and flocculation can tend to decrease with increased TDS as the latter also contributes to the coagulation process. There may thus be a positive effect on costs, albeit marginal. At a certain point, when eutrophication occurs, more expensive chemicals such as poly-electrolyte may be required for effective flocculation. As the eutrophication increases, the running cost of activated carbon for odour control will also increase.

Chemical Costs : $B = B_1 - K_1 (TDS) B_1 + K_2 (TDS) / (TDS) - TDS_{TH_1} / B_2$ 7
$+ K_3 (TDS) / TDS - TDS_{TH_2} / B_3$ 8

B = Total chemical costs
K_1, K_2, K_3 = constants
TDS = actual TDS concentration
B_1 = normal chemical costs for a conventional treatment process
B_2 = cost of poly-electrolyte or other flocculant aids TDS_{TH_1}, TDS_{TH_2} 9 as before
B_3 = cost of activated carbon

It is proposed that the reduction in chemical costs due to increased salinity be ignored as it will be marginal. Retain the element as part of the model but use $K_1 = 0$. If it is suspected at a later stage that it can contribute significantly to savings, it can be established experimentally.

K_2 and K_3 can only be determined experimentally for specific waters. It is expected that they will be equal to one. The value of B_2 and B_3 will have to determine experimentally for specific water in a specific trophic state.

4.2.3 Labour costs

Labour costs will be independent of actual TDS concentration but will increase in steps as extra processes are added..

This can be linked to the capital cost formula

Labour Costs : $C = f(A_1, A_2, A_3, A_4)$

4.2.4 Maintenance costs

Maintenance costs are normally determined as a percentage of capital costs. As such it will increase in the same stepped fashion as the capital cost. There is a possibility of increased maintenance due to factors such as increased corrosivity. This will be difficult to model and it will in any case tend to decrease first as very low TDS can be more corrosive than water in the 400 mg/ℓ-800 mg/ℓ range. The higher TDS can be more or less corrosive, depending on the actual make-up of the ions.

Maintenance costs can be linked to the capital cost model.

Maintenance costs : $D = f(A)$

4.2.5 Electricity (Energy) costs

Direct energy costs are independent of TDS but will increase in steps as further processes are added. If Reverse Osmosis (for desalination) is added, it will increase dramatically as the latter is an energy intensive step.

Energy costs: $E = f(A_1 \text{ kW installed})$

4.2.6 Cost of disposal of sludge

It will generally be more expensive to dispose of an organic sludge than an inorganic sludge.
Sludge formation is normally a function of chemical consumption.

Sludge disposal costs: $F = f(B)$

4.3 Incremental labour, maintenance, energy and sludge handling costs due to increased TDS

Factors such as labour, maintenance and energy are all linked to the operational and capital costs, but will be relatively small in comparison with the chemical and capital components. It is proposed to ignore these as a first approximation in the model. Only the incremental capital and chemical costs will thus be determined.

5. DISCUSSION OF PROPOSED COST MODEL WITH RELEVANT AUTHORITIES IN ORDER TO CALIBRATE IT

5.1 INTRODUCTION

Interviews were conducted with the following institutions to test the general validity of the model and to quantify costs:

(a) Umgeni Water (Mr D Kerdachi)
(b) Rand Water (Mr S van der Merwe)
(c) Western Transvaal Water Corporation (Mrs M Krüger)
(d) Goldfield Water (Mrs Basson)

There was consensus amongst all the parties consulted that the basic assumption of the model, namely that TDS on its own does not significantly effect treatment costs, is true. Increased treatment costs are as a result of the changes in the trophic state of the water. This renders the water more difficult to treat and either more sophisticated processes or more expensive chemicals or a combination of both are required for the eutrophic waters. There was also consensus regarding the fact that algae concentration alone is not sufficient to predict treatment costs. The type of algae has a major influence with toxic blue-green species much more problematic than diatom species.

None of the water authorities interviews could quantify a threshold TDS for their specific water source. The feeling was that it is impossible to develop a relationship between TDS and chlorophil levels in water. The makeup of the TDS is of paramount importance and a 2 mg/ℓ increase of Ca^{2+} will exhibit no influence whereas 2 mg/ℓ of P will have a major influence on the trophic state of the water. It was further pointed out that problem ions such as Fe^{2+} or Mn^{2+} can have a major influence on treatment costs in spite of it being present in very low concentrations.

5.2 COST OF TREATMENT OF EUTROPHIC WATERS

All the water authorities consulted have experienced treatment problems due to eutrophic conditions and have gone some way towards quantifying these costs. Rand Water and Goldfields Water favour more sophisticated chemical treatment to allow purification of eutrophic raw waters on their conventional treatment plants.

Umgeni Water and Western Transvaal Water Corporation tend to move towards more sophisticated treatment processes such as ozonation and dissolved air flotation to achieve the desired degree of purification.

The following costs have been predicted by the various authorities:

5.2.1 Rand Water

Rand Water predicted the following incremental costs for eutrophic water:

-	Pre-oxidation at 5 mg/ℓ for 6 months per year	0,27 c/kℓ
-	DAFF	3,04 c/kℓ
-	Activated carbon	9,09 c/kℓ
	TOTAL	13,57 c/kℓ

5.2.2 Western Transvaal Water Corporation

This authority has elected to install a dissolved air flotation unit as pre-treatment step. The cost of the unit is R25,0 m to treat 250 Mℓ/d. The predicted capital redemption costs due to this is ± 5 c/kℓ which corresponds favourably with the 3,4 c/kℓ was previously reported by Africon during the Phase I stage of the study for a 10 Mℓ/d dissolved air flotation unit.

It can thus be accepted that dissolved air flotation adds 5c/kℓ to treatment costs. To this the cost of activated carbon must be added. This can be up to 10c/kl depending on dosage.

Once again the total incremental cost for treatment of eutrophic water is in the order of 15c/kℓ. This coincides very well with the previously reported figure of 14,2c/kℓ as developed by Africon for Phase I.

5.2.3 Goldfields Water

This authority can still successfully treat eutrophic water in conventional facilities by modification of the chemical coagulation processes. The basic requirement is for pre-chlorination followed by high lime. The incremental cost is 3c/kℓ which is in the same order of costs as a flotation unit. This still excludes the cost of activated carbon which will be of a similar order of magnitude than for the other authorities, bringing the total incremental costs also to ± 10c/kℓ/15c/kℓ.

5.3 FINANCING OF INCREMENTAL TREATMENT STEPS

It was confirmed by all the water authorities consulted that the incremental costs for more sophisticated treatment technologies is not carried by the specific user alone, but all consumers pay an "average" tariff. A sliding scale of rates may be used, but this is also equally applicable to all consumers.

5.4 CONCLUSIONS FROM THE INTERVIEWS

The following conclusions could be drawn from the interviews:

- The water in the Middle Vaal River is not expected to have any further incremental costs for treatment due to further increases in TDS. It is over the threshold TDS of the model.

- The incremental cost due to the change in trophic state is in the order of 15c/kℓ of which 10c/kℓ is for activated carbon. This value can vary depending on seasonal or other variations in taste and odour problems.

- While the basic hypothesis of the model is supported, the problematic influence of micro elements such as Fe^{2+} and Mn^{2+} is not adequately addressed.

- None of the water authorities consulted could accurately define the threshold TDS for their source of raw water. Factors such as temperature and turbidity influences the actual values.

6. INCREASED COST IN WATER MAIN DISTRIBUTION SYSTEMS DUE TO INCREASED SALINITY

6.1 INTRODUCTION

Increased TDS can increase the corrosivity of water or have the opposite effect in that scale formation can occur.

Corrosion in pipes can be due to purely chemical reactions, but also due to bacteriological actions particularly those of sulphur reducing bacteria (SRB). The latter type of corrosion dominates in the presence of high sulphate concentrations.

TDS concentration alone is thus not a valid parameter to determine the corrosivity of water.

6.2 COSTING THE INFLUENCE OF WATER CORROSIVITY

6.2.1 Chemical corrosivity

If water is known to be chemically corrosive, it can be stabilized in the treatment process to avoid further damage to the downstream pipe system. The cost of this can be deemed to form part of the chemical costs for water treatment. It is thus a treatment cost and normally consist of lime addition, softening or a number of other chemical steps.

Chemical Corrosion : $G = f(A,B)$

G = cost of corrosion protection by incremental treatment costs.

6.2.2 Bacteriological corrosion

There are two main alternatives for avoiding or minimizing the impact of corrosion on pipes. They are:

- use piping materials not subject to corrosion
- use a protection system to protect the pipe against corrosion

This is an incremental capital cost.

6.2.2.1 Costing

The above alternatives can be used as a basis to cost the influence of corrosivity on new or pipes. It can be shown as an incremental cost to use more expensive materials or to protect pipes that can be subject to attack. Once again the cost will be in a step, or a number of steps, if further corrosion protection is required with increased corrosivity. Generally the design life of a pipe is 20 years and corrosion protection systems can at best achieve this. After this the cost of corrosion will be reflected in increased maintenance costs to fix leaks and replace or refurbish pipe sections.

In the case of existing pipes that were installed before the corrosion aspects became a problem, the cost can be defined as the replacement cost minus some credit for the remaining service value of the system. The determination of this service life is very site and circumstance specific and will have to be done using the actual water and materials to obtain corrosion rates. Based on this predictions of remaining service periods can be made.

6.2.2.2 Proposed Cost Model

New pipes:

$H = H_1 + /TDS - TDS_{TH} / H_2$ 10

H = total cost
H_1 = cost of conventional pipe without corrosion protection
H_2 = cost of corrosion protection or incremental cost for alternative materials
TDS_{TH} = TDS concentration which causes need for corrosion protection.

Old pipes:

The best approach to model the incremental cost due to TDS increases in an old reticulation is to accept that an incremental maintenance and refurbishment cost will occur.

I = $f(J, TDS)$
I = Incremental maintenance costs
J = Actual maintenance costs from the local authority's records.

7. COST OF PIPE SYSTEMS

The following are costs (1995 prices) for pipes with various types of sophisticated corrosion protection:

PIPE DIA	UNPRO-TECTED	COPON EP2300	EPOXY POWDER COAT	POLY-URETHANE
400 mm	R130	R230	R223	R450
800 mm	R330	R530	R516	R970
1200 mm	R660	R1350	R1312	R1800

Copon EP2300 can be regarded as the minimum protection required for a water main. Poly-urethane protection will enable the pipe to withstand any combination of ions (ie any concentration of sulphides) in the 1200 mg/ℓ TDS range.

This is thus an upper limit of costs. Depending on size the cost ratio for the standard protection versus special protection is between 1:2,02 (for 400 mm) to 1:1,37 (for 1200 mm).

The sophisticated poly-urethane is only required for exceptionally aggressive waters and in general it should be expected that Copon EP 2300 is sufficient to protect the pipe. There is thus no incremental cost for protection.

In some cases with larger diameter steel pipes, it is possible to use a cement mortar lining.

In feeder systems, using cement-lined mortar lining, the two components most likely to enhance the corrosion of the pipes are the chloride and the sulphate ion concentrations.

From the analyses performed for this study by Dr Loewenthal, the water is already corrosive to bare steel piping and these should be lined regardless of actual final TDS.

Regarding cement lined pipes, the two ion concentrations were examined. From the analytical results, the current chloride ion concentration (Cl^-) is less than 140 mg/ℓ. Even if this was to double, it would still be less than the 500 mg/ℓ threshold level for affecting cement linings. Thus, it can be ignored.

The current sulphate ion concentration ($[SO_4^{2-}]$) is in the region of 500 mg/ℓ which already exceeds the threshold value of 300 mg/ℓ. Thus, the cement lining is already affected and this will possibly worsen. The life expectancy of cement mortar lining (and cost) is a function of the sulphate concentration. However, provided there is pH control (between 6,8 - 7,2) the sulphate problems will not be too greatly enhanced.

Although in theory, cement mortar lining is deleteriously affected under the conditions of this study, in practise little effect is noticed and the corrosive effect can be ignored. As detailed previously, there will be a once off cost due to lining the pipes, whereafter the maintenance and replacement conditions should be no different to those currently experienced.

The simplified formula is:

Cost = cost of lining.

The incremental cost per unit water (m^3) will depend on pipe length, size, etc. For the purpose of the model it is stated that no incremental costs are expected in the study range of TDS once the primary corrosion protection has been installed.

It must be taken into account that the stability of water tends to decrease as the distance from the source increases. This may mean that a municipality furthest away from the source may experience more severe corrosion problems.

There must also be a distinction between corrosion in bulk supply services and domestic plumbing. In the latter case the traditional galvanized piping will be influenced by increased TDS. This particular aspect is covered in another section of the model.

8. CONCLUSIONS

Modelling increased treatment and conveyance costs of water due to increased salinity is difficult. The cost responses depend on physical as well as chemical reactions due to the increased salinity.

It must be accepted that cost increases will occur and that these will be in the form of steps. In some cases, such as chemical corrosivity, the cost can be approximated by the incremental treatment cost to stabilize the water.

For the Middle Vaal River it is predicted that no incremental treatment or corrosion costs will be experienced should TDS levels increase from its current values to 1200 mg/ℓ. The one possible variance is the cost of activated carbon with higher dosages required should the toxic blue-green algae become the dominant species. This can result in further cost increases. The actual incremental cost due to eutrophication is ± 15c/kl.

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1. INTRODUCTION

In many arid and semi-arid areas throughout the world, river ecosystems are becoming increasingly saline (Williams, 1987). It is estimated that world-wide an area of 950 million hectares (slightly larger than Australia) is now affected by salinisation (Hart *et al.* 1990). Salinisation is a naturally occurring phenomenon, through leaching of salts from soils and rocks. However, the extent of salinisation has increased alarmingly with increased human industrial and mining activities, and the resultant pollution. The problem is most noticeable in the semi-arid regions of the western USA, Australia and South Africa. In South Africa concern has been expressed by various sectors that make use of the water (e.g. municipalities, industry, agriculture and mining) due to the rising costs incurred by the high salt content of the water. This marked increase in salinity in river systems is occurring predominantly in developed regions of the country (such as on the Vaal River). With increased economic pressure on South Africa, it is likely that more industrialisation will take place with a possible effect on the salt levels in rivers throughout South Africa.

The economic effects of salinisation have been investigated previously by Heynike (1987) in the Vaal River catchment. However, little is known about the economic costs associated with salinisation in other South African rivers. Information from the Middle Vaal River from the Barrage to Bloemhof Dam has been used to assist in the development of model formulae and as input into the model. Included in the model are the costs associated with changes in the natural environment. It is this sector that is dealt with in this document.

2. METHODOLOGY

The natural environment includes so many interacting components that it would be impossible for all of them to be included in the model without simplification. Thus, most of the sub-sectors were defined in terms of the use of the natural environment and the role of aquatic organisms in relation to man. Most of the sub-sectors are economically important to a greater or lesser extent. The exception is the sub-sector described as "loss of species" which is difficult to quantify in monetary terms, but should not be excluded as a sector for consideration. The eight sub-sectors include:

1. **Terrestrial wildlife** - this includes the non-aquatic animal life which relies on the river for drinking water. Water with a high salt content may be detrimental to the productivity of terrestrial wildlife, or become so impotable that animals will move out of the area. This may have economic implications for the tourism in the area, as well as stock loss costs.
2. **Riparian vegetation** - this is the vegetation which requires river water for its presence. Riparian vegetation is important as a buffer zone between the terrestrial and aquatic environment. The plants also increase the stability of river banks. If the riparian vegetation dies off, severe bank erosion will take place, with associated costs.

3. **Algae** - the algae considered are the phytoplankton and the filamentous green algae. Water for domestic and industrial use are affected by the phytoplankton abundance in the water column. It is believed that changes in salinity levels (represented by total dissolved solids - TDS) will change the abundance of phytoplankton, and thus affect treatment and purification costs. Filamentous algae may clog water transportation systems, thus reducing irrigation flows, with resultant costs.
4. **Fish** - only the fish of angling importance are considered in this sub-sector. Salinity increases may have positive or negative effects on fish populations, their catchability, and the revenue brought in by angling activities.
5. **Aquatic wildlife** - this sub-sector includes vertebrate fauna such as hippopotami, crocodiles and aquatic birds whose primary habitat is water-related. If salinity increases have a negative effect on these animals tourism potential to the area is likely to be affected.
6. **Water weeds** - these are defined as aquatic plants which cause serious nuisance in river ecosystems. Water weeds can interfere with recreational activities and cause blockage problems in intake pipes. Both these have economic effects.
7. **Invertebrate disease vectors** - included in this category are: those blackfly (Simuliidae), mosquito (Culicidae) and other biting midge (Ceratopogonidae) species that require a vertebrate bloodmeal before maturing their eggs and are sometimes disease vectors for certain veterinary (orbiviruses) and human (malaria) diseases, and snails (Molluscs) causing bilharzia and veterinary diseases (liver flukes).
8. **Species loss** - this covers negative effects that the increased levels of TDS may have on the whole ecosystem. However, it is very difficult to attach accurate costs to any species lost.

The chemical requirements of the water for the model are the following:

- the TDS level to be modelled fell in the range of 200 mgℓ^{-1} to 1 200 mgℓ^{-1}, and
- the TDS is for waters of pH between 6.5 and 9.0.

The development of formulae which represent the effect of changing TDS levels in the sub-sectors of each primary sector, forms the basis for this report for the natural environment sector. The approach taken included:

1. Identification of the direct and indirect impacts of changing total dissolved solids (TDS) on the physical and biological environment through a literature review.
2. Identification and preliminary quantification of the costs associated with those impacts through a literature review and consultation with experts in the field.
3. Development of the assumptions and formulae relevant to the model frOm the information compiled in 1. and 2. above.

The content of this report follows these three steps.

3. DIRECT EFFECTS OF TOTAL DISSOLVED SOLIDS

In order to understand the possible influence that total dissolved solids have on the natural environment, it is essential to define what is meant by the term environment. The environment is "the whole sum of the surrounding conditions ... within which an organism, community or object exists" (Walmsley & Walmsley 1993). The natural environment is that portion of the environment which is not man-made. It includes chemical, physical and biological components. These three components interact with each other continuously, so that changes in the one will automatically affect the other two. In this study, we examine how changes in the chemical environment (TDS) in a river can affect the physical and biotic components.

3.1 CHANGES IN THE PHYSICAL ENVIRONMENT

The major change to the physical environment due to a change in the levels of total dissolved solids in the water, is a lowering of the turbidity levels, with a resultant increase in underwater light levels as colloids precipitate out (Chutter 1963; Grobler *et al.* 1983; Pieterse 1986; Grobler *et al.* 1987; Akhurst & Breen 1988). This has several implications for the biota, and may have indirect effects on organisms such as algae and fish.

In many studies it is assumed that turbidity is due to suspended sediment only (Grobler *et al.* 1981, 1983, 1987), and does not include the algal component. Lind (1986) indicates that chlorophyll *a* levels have very little influence on the Secchi depth in mesotrophic waters. However, in a eutrophic situation where algal blooms occur, the average Secchi depth and the average chlorophyll concentrations have a significant inverse correlation to each other ($r = -0.75$; OECD 1982). For the sake of simplicity in the model it may be assumed that turbidity in most South African rivers is due to suspended sediments.

Suspended sediments consist mainly of clay particles of less that 0.2mm (Rooseboom 1978; Grobler *et al.* 1981, 1987) originating from catchment silts. Because of the small size of the particles the settling rate is slow. Clay particles are negatively charged particles which, when suspended in water attract positive cations to them (Grobler *et al.* 1981, 1983). These cations are adsorbed onto the surface of the clays creating a cation layer known as the double layer. The negative charge of the clay particles, the zeta potential, will decrease as more cations are adsorbed into the double layer. Clay particles with a high zeta potential will repel each other and the clay will remain suspended in the water column. If the zeta potential is lowered, individual clay particles will coalesce, forming flocs, which being much larger, will tend to sediment out. As salinity increases in a river more di- and monovalent cations become available for adsorption, suspended clays become more unstable and sedimentation of particles increases.

In Lake Alexandrina on the River Murray, Australia, the turbidity dropped from 93 NTU to 9 NTU with an increase in salinity (Geddes 1988). Although the turbidity levels were not correlated to salinity, the highest level of salinity measured was about 1 000 mgℓ^{-1} (Geddes 1988). Chutter (1963) noted that, in the Vaal River below Vaal Dam, precipitation of colloidal material appeared to occur at levels above 200 mgℓ^{-1}. Grobler *et al.* (1983, 1987) indicate that this may even occur at TDS levels below 200 mgℓ^{-1} (about 150 mgℓ^{-1}; Figure 1). In this case the change is outside the model range of 200 mgℓ^{-1} to 1 200 mgℓ^{-1} TDS. However, unpublished data (Prof AJH Pieterse and J Roos, Botany Department, University of the Orange Free State, pers. comm.) indicate that at Balkfontein the precipitation of colloids occurs at about 230 mgℓ^{-1} TDS (Figure 2) and at Stilfontein this occurs at about 350 mgℓ^{-1} TDS. For the purpose of this study it will be assumed that precipitation occurs between 200 mgℓ^{-1} and 400 mgℓ^{-1} TDS, and that costs are related to this.

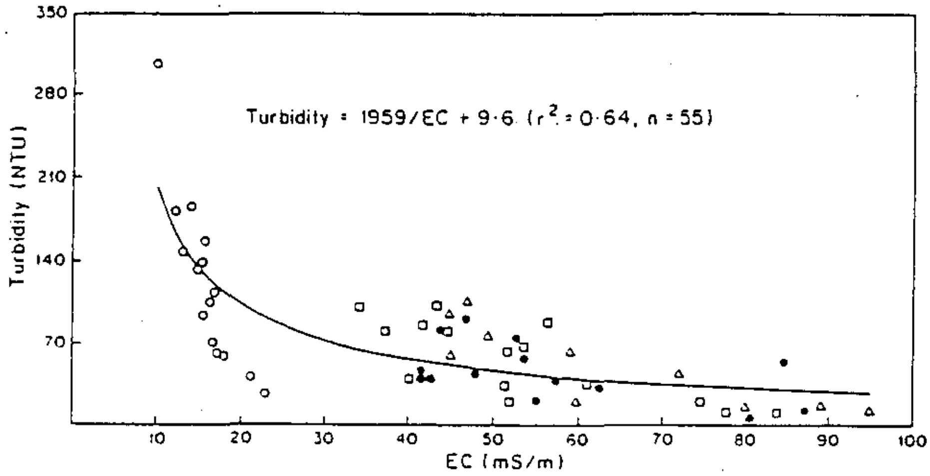

FIGURE 1: Relationship between turbidity and electrical conductivity (EC) in the lower Vaal River (from Grobler *et al.* 1983).

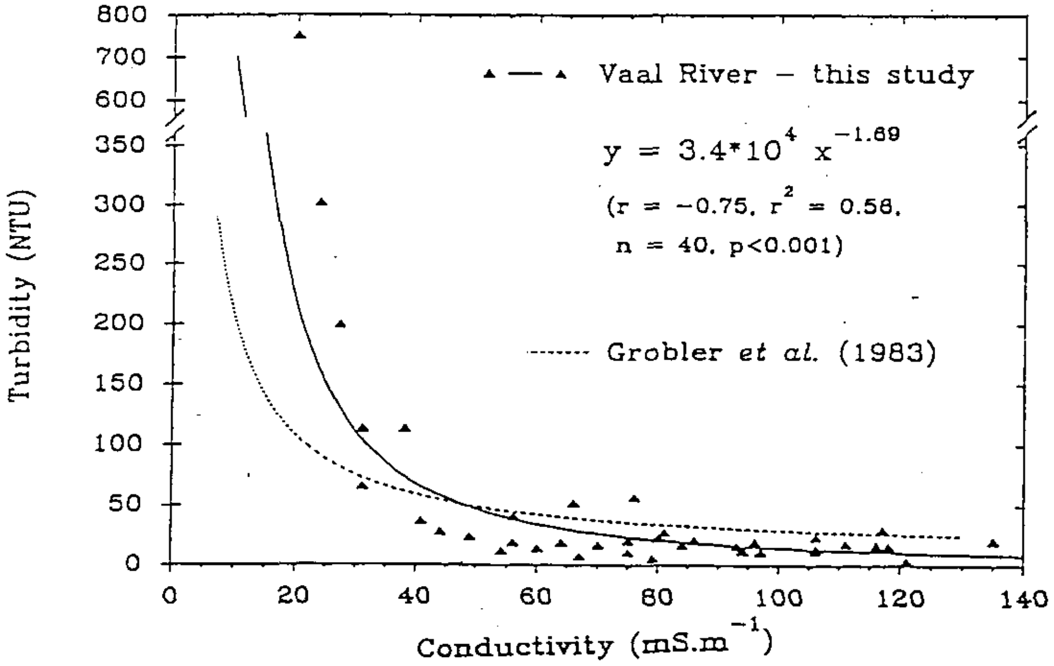

FIGURE 2: Relationship between turbidity and electrical conductivity (EC) in the Vaal River at Balkfontein (from Pieterse and Roos, unpublished data, Botany Department, University of the Orange Free State).

According to Grobler (1983, 1987) turbidity has the following relationship to electrical conductivity and TDS in the Vaal River:

$$\text{Turbidity} = 1959/EC + 9.6 \ (r^2 = 0.64, n = 55) \tag{1}$$

or

$$\text{Turbidity} = 12733.5/TDS + 9.6 \qquad (\text{for } TDS = 6.5 * EC \ mSm^{-1}) \tag{2}$$

where EC = electrical conductivity (mSm^{-1})
TDS = total dissolved solids ($mg\ell^{-1}$).

This relationship was determined from a graph of yearly salinity and turbidity averages from four sites on the Vaal River (Figure 1) - below the Vaal Dam, below the confluence of the Klip and Vaal Rivers, at the Vaal Barrage and below the Vaal Barrage. From the graph it is evident that the Vaal Dam site is physically and chemically different from those further downstream. It may be that the Vaal Dam site would give a better indication of the direct relationship between conductivity and turbidity than all the sites combined. The other sites may be influenced by increased algal concentrations and the data become less reliable. However, a similar relationship was obtained by Pieterse and Roos (unpublished data, Botany Department, University of the Orange Free State) (Figure 2), with the relationship of turbidity to electrical conductivity and TDS in the Vaal River at Balkfontein expressed as:

$$\text{Turbidity} = 34\,000 * EC^{-1.69} \tag{3}$$

or

$$\text{Turbidity} = 34\,000 * (TDS/6.5)^{-1.69} \tag{4}$$

where EC = electrical conductivity (mSm^{-1})
TDS = total dissolved solids ($mg\ell^{-1}$).

A decrease in turbidity increases the availability of light in the water. A number of studies have been undertaken in South Africa to understand the relationship between light penetration and turbidity (Walmsley 1980; Walmsley & Bruwer 1980; Walmsley *et al.* 1980; Stegmann 1982). Most of these studies have been undertaken in lakes, although this relationship will also be relevant to river ecosystems. The relationship between turbidity (NTU) and the extinction coefficient from various studies in South Africa is given in Table 1. The extinction coefficient may be used to define the depth of the photic zone, or the depth to which 1% of the incident light penetrates into a water body (Grobler *et al.* 1983). This has been done for the Bloemhof Dam on the Vaal River (Figure 3). The photic zone depth has a direct effect on the growth of primary producers such as algae and submerged macrophytes.

TABLE 1: Relationship between extinction coefficients (k, m^{-1}) and turbidity (T, NTU) for a number of South African water bodies (from Grobler *et al.* 1983).

WATERBODY	RELATIONSHIP	r^2	n	REFERENCE
Bloemhof Dam	$k = 0.127T + 0.55$	1.00	5	Grobler *et al.* 1983
Wuras Dam	$k = 0.073T + 1.43$	0.94	78	Stegmann 1982
Lindleypoort	$k = 0.082T + 0.84$	0.90	74	Walmsley 1980
Buffelspoort	$k = 0.149T + 0.44$	0.59	76	Walmsley 1980
Rust der Winter Dam	$k = 0.1T + 0.44$	0.70	43	Walmsley & Bruwer 1980

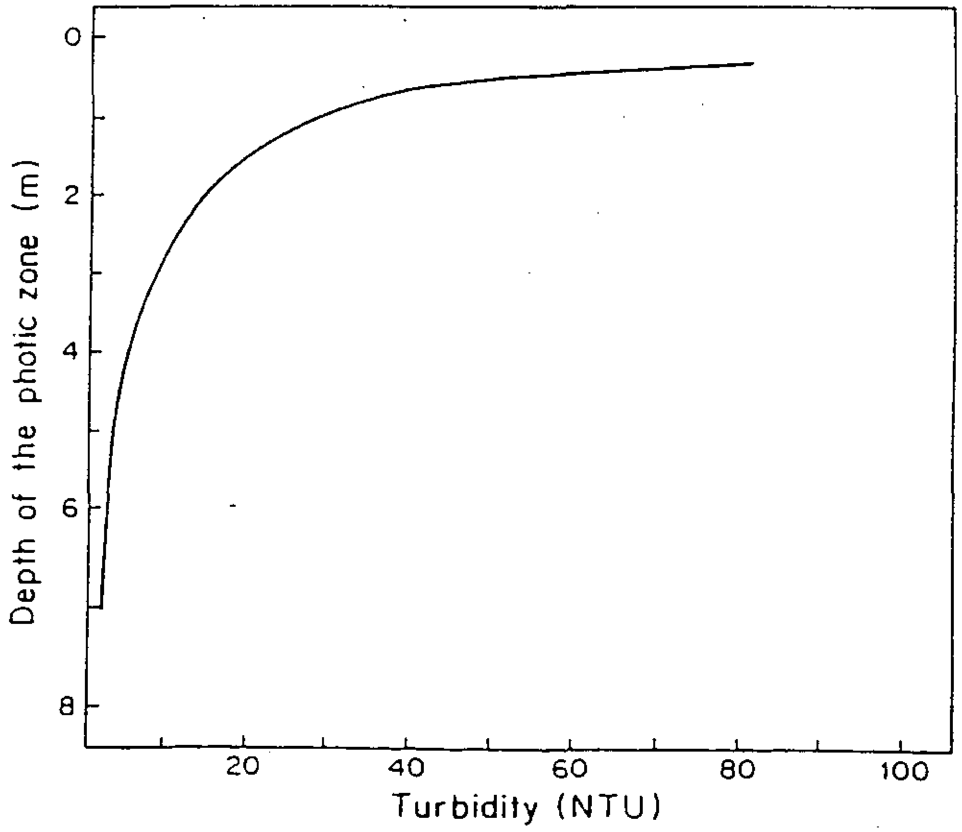

FIGURE 3: The probable relationship between photic zone and the turbidity of the lower Vaal River (from Grobler *et al.* 1983).

3.2 CHANGES IN THE BIOLOGICAL ENVIRONMENT

The environment determines the communities that will be found under various conditions and their adaptation to them. One of the major adaptations is that of organisms to the osmotic pressure of various waters. Organisms in a wide range of aquatic environments have developed physiological mechanisms and adaptations to maintain the balance of water and dissolved ions in their bodies (Hart *et al.* 1991; Schmidt-Nielsen 1983). Freshwater plants and animals are hyper-osmotic regulators, with the concentration of ions in their body fluids above that of the surrounding medium. Thus, water tends to be gained by osmosis and ions lost by diffusion. Animals react to this by excreting dilute urine and possessing a range of mechanisms to take up ions (Hart *et al.* 1991). Plants react by storing excess water in vacuoles. If the salt balance is changed, and the ionic content of the external medium rises, normal osmoregulatory adaptations may not be sufficient to cope with the change. In animals the populations will become stressed and productivity will decrease, while in plant death may occur either from excess ion toxicity or water deficiency due to difficulties in extracting water from the external medium (Hart *et al.* 1991). The direct effects of changing TDS on the biota are outlined below. From these findings it is evident that the organisms that may be affected are fish, water weeds and snails.

Wildlife

Originally it was thought that salinities may have an effect on the productivity and reproductive capacity of wildlife. At higher salinities (>3000 mgℓ^{-1}) this may be the case for the more sensitive animals, but at salinities below 1 200 mgℓ^{-1} there will be no real effect on the animals and birds (Van Heerden, 1994).

Drinking water criteria for livestock and poultry are acceptable as criteria for wildlife (Van Hoven, 1994). Table 2 gives the guidelines for salinity for drinking water for poultry. The values given in this table verify the opinion that at levels of $<1\ 200$ mgℓ^{-1} wildlife will not be affected by increasing salinity. Thus, at levels of TDS $<1\ 200$ mgℓ^{-1}, it is assumed that this sector may be left out of the model.

Riparian Vegetation

High salinity levels may cause reduced plant growth or plant death from toxic effects caused by an excess of ions, or a water deficiency due to difficulties in extracting water from the external medium (Munns & Termaat 1986; Hart *et al.* 1991). The problems that this may cause in the case of riparian vegetation are a decrease in bank stability as the plants die off, a decrease in the aesthetic value of the riparian zone and a decrease in the effectiveness of the riparian strip as a buffer zone. Because riparian zones link the river with the terrestrial environment they can modify temperatures, reduce sediment inputs, provide important sources of organic matter to the river, as well as incorporating, diluting or concentrating substances before they enter the lotic system (Osborne & Kovacic 1993). This function will be altered as the community structure alters and the abundance of plants decreases.

Not much information is available on the effect of increasing salinity on South African riparian species. Most Australian riparian plants can tolerate salinities greater than 5 000 mgℓ^{-1}, and sensitive species will only be adversely affected by salinities above 2 000 mgℓ^{-1} (Hart *et al.* 1991)

TABLE 2: Guidelines for salinity in livestock drinking water (from DWA&F 1993).

Salinity range		**Effects**
Total dissolved solids (TDS) (mg/ℓ)	**Electrical conductivity (mS/m)**	**Suitability for livestock**
Target guideline range *0 - 1 000*	*0 - 154*	*Should not pose any problems for livestock watering.*
1 000 - 3 000	154 - 460	Satisfactory for all classes of livestock and poultry, but some loss in productivity should be anticipated; may cause temporary and mild diarrhoea in livestock not accustomed to the water, or watery droppings in poultry
3 000 - 5 000	460 - 770	Satisfactory for livestock but may cause temporary diarrhoea or be refused at first by animals not accustomed to the water; poor water for poultry often causing watery faeces, increased mortality and decreased growth, especially in turkeys.
5 000 - 7 000	771 - 1 077	Can be used with reasonable safety for beef cattle, sheep, swine and horses; avoid use for pregnant or lactating animals, and dairy cattle; not acceptable for poultry.
7 000 - 10 000	1 078 - 1 540	Unfit for poultry and probably for swine; considerable risk in using for pregnant or lactating cows, horses or sheep, or for the young of these species; in general, use should be avoided, although older ruminants, horses, poultry or swine may subsist on such water under certain conditions.
>10 000	>1 540	Risks with these highly saline waters are so great that they cannot be recommended under any conditions.

Supplementary notes:

- Based on the Canadian Guidelines (1987).

Obviously reaction to increased salinity levels is species specific. However, in all species it is apparent that higher salinities are tolerated if the salinities are increased slowly. In the United States it has been proposed that, for the protection of wildlife habitats (such as the riparian vegetation), salinity should not vary by more than 1 000 mgℓ^{-1} in water where the natural salinity is below 3 500 mgℓ^{-1} (Williams, 1987). This is well with in the variation level proposed for the model.

Long term accumulation of salts in the soils may affect the vegetation over long periods (Rogers, 1994). However, this has not yet been investigated and will be ignored in this model. Thus, from available information, it may be assumed that the effect of increased salinity on riparian vegetation below 1 200 mgℓ^{-1} will be negligible.

Algae

Phytoplankton blooms have several negative economic consequences because of their ability to give rise to large quantities of organic matter in water. Algal blooms in open waters may decrease the aesthetic value of the water body, leading to a decrease in recreational activity; increase fish mortality by depleting the oxygen content of the water when they decompose; and cause interference with water treatment and purification processes, and problems in the distribution systems (Pieterse, 1986).

Phytoplankton communities may change slightly with an increase in salinities, with less salt tolerant species giving way to more salt tolerant species (Forbes & Allanson, 1970). Prinsloo & Pieterse (1994) show that some South African species are more salt sensitive than others. They suggest that under salinity conditions above 250 mgℓ^{-1}, *Cyclotella meneghiniana* and *Microcystis aeruginosa* may be excluded from the water. However, there was no statistical evidence for this, and for the model it is assumed that at levels of TDS between 200 mgℓ^{-1} and 1 200 mgℓ^{-1}, while the species composition of the phytoplankton may change, the algal biomass will not be affected *directly* by the TDS levels.

Similarly, there is little evidence that salinity in the range relevant to the model affects the abundance of filamentous algae, although the more tolerant species, such as *Enteromorpha* spp. and *Compsogon* spp., will become more dominant (Joska & Bolton, 1994). The most problematic alga in South Africa at present is *Cladophora glomerata*. It can occur at TDS levels of up to about 1 400 mgℓ^{-1}, and forms the bulk of large algal growths in the Kalkfontein and Crocodile irrigation systems (Joska & Bolton, 1994). It is evident, therefore, that the filamentous algal abundance will not be affected directly by the TDS levels.

Fish

Little information is available in South Africa regarding the sensitivity of South African angling fish species to salinity increases. Research on the Jukskei River, PWV Region, has indicated that fish populations are unaffected by salinities of about 750 mgℓ^{-1} (Du Preez, 1994). In the Selati and Olifants rivers, fish diversity begins to decrease at TDS levels of 1 600 mgℓ^{-1} as the most sensitive fish (some *Barbus* spp.) disappear out (Engelbrecht, 1994). It has been suggested that, along with the flow régime and a change in silt loads, increasing TDS has played a role in the decline of *Barbus kimberleyensis*, the large yellowfish endemic to the Vaal/Orange River system (Van Zyl, 1994). However, this observation is to some extent refuted by the fact that this population decline is apparent at places where there have not been large increases in salinity.

Australian research has shown that most adult Australian fish can tolerate salinities up to and greater than 10 000 $mg\ell^{-1}$ with no detrimental effects (Hart *et al.* 1991). Willams & Williams (1991) have confirmed this by examining the salinity tolerances of four native Australian fish species from the Murray River. They found that salinity tolerances of the chosen species ranged from an LD_{50} (the point at which 50% of the population die) of 20 800 $mg\ell^{-1}$ to 43 700 $mg\ell^{-1}$. In general, if salinity increases gradually in a system, then fish have a high tolerance to TDS levels (Heath, 1994).

Little is known of the salinity tolerances of fish larvae and eggs except that they are generally less tolerant than the adults (Hart *et al.* 1991). This is because, although they are subject to the same osmotic stresses, they do not have the adult osmoregulatory abilities. It seems that in most cases in Australia the negative effects of salinity increases are only apparent above 10 000 $mg\ell^{-1}$ (Hart *et al.* 1991).

Originally it was thought that direct impacts of salinity on fish populations in South.Africa would be small. However, although species may be able to tolerate higher TDS levels, the additional osmotic stress could lead to additional energy being used in osmo-regulation and a drop in the physical condition of the fish. This would make them more disease-prone and productivity would decline. However, there is little evidence in South African river ecosystems to support this theory.

Aquatic Wildlife

Little information regarding the effect of salinity on the large riverine biota is available. Hart *et al.* (1989) suggest that high salinities will have little effect on the aquatic wildlife in Australia. It is assumed that at the levels under discussion there will be no effect on aquatic wildlife in the Vaal River.

Water Weeds

Bruwer (1986) confirms that the most serious threat on many South African rivers in certain areas is the water hyacinth (*Eichornia crassipes*). Other aquatic macrophytes, such as *Myriophyllum aquaticum, Salvinia molesta* and *Azolla filiculoides* are also of particular concern in the South African Waters (Ashton *et al.* 1986), although not to the same extent as water hyacinth. Of these four species only *M. aquaticum* is rooted in soil; the others are free-floating. The rapid production of photosynthetic material gives these plants a competitive advantage over the indigenous aquatic plants. The primary ecological impacts brought about by these plants is the reduction in water movements, decreased oxygenation and a loss of light penetration into the water. From an economic point of view, impacts include river blockages, aesthetic problems, evapotranspiration losses, water treatment problems and increased breeding sites for disease vectors (Ashton *et al.* 1986).

On the whole it seems that aquatic macrophytes are far less tolerant of high salinities than riparian vegetation. A number of species do not seem to tolerate salinities of greater than 1 000-2 000 mgℓ$^{-1}$, and at 4 000 mgℓ$^{-1}$ most freshwater macrophytes have disappeared (Hart *et al.* 1991). For rooted plants such as *Typha* spp. growth only begins to be affected between 2 900 mgℓ$^{-1}$ and 5 900 mgℓ$^{-1}$ (Hocking 1981). Most of the floating aquatic plants are severely affected at levels greater than 1 700 mgℓ$^{-1}$ (Table 3). *Eichornia crassipes* and *Pistia stratiotes* both die off once salinity levels are over 2 500 mgℓ$^{-1}$, and growth is severely affected from 900 mgℓ$^{-1}$ upward (Haller *et al.* 1974). The proportional reduction in growth at different TDS level can be calculated from the values in Table 3 (Figure 4).

Disease Vectors

Snails

Due to the nature of their biology, snails are fairly sensitive to TDS and changes in salinity. The snail populations are affected by both the level of salinity and the ionic composition of the TDS (De Kock, 1994). In terms of the natural environment, this is the only sub-sector where there is evidence that the ionic composition should be taken into account. If the magnesium content of the water is higher than the calcium content, uptake of calcium is reduced and the structure of the snail shell deteriorates ($CaCO_3$ being the main component of a snail's shell), (De Kock, 1994). Additionally there should be at least 6 mgℓ$^{-1}$ of calcium in the water for snails to thrive (De Kock, 1994).

TABLE 3 : Growth of some aquatic macrophytes exposed for four weeks to various salinity levels. (adapted from Haller *et al.* 1974). Growth experiments were carried out in dilute sea water (dominant ions Na^+ and Cl^-).

SALINITY (mgℓ$^{-1}$)	GROWTH (g dry wt / 28 days)				
	Eichornia	*Pistia*	*Myriophyllum*	*Azolla*	*Salvinia*
170	11.64	7.51	0.6314	7.01	5.79
830	12.48	3.59	no data	no data	no data
1 660	9.84	1.95	no data	no data	no data
2 500	7.19	-	no data	no data	no data
3 330	-	-	0.5232	4.14	6.03
4 160	-	-	no data	no data	no data
5 000	-	-	no data	no data	no data
6 660	-	-	0.4892	2.96	3.36
10 000	-	-	0.4024	2.72	2.92
13 320	-	-	0.2339	1.69	2.50
16 650	-	-	-	1.46	2.25

Australian freshwater molluscs occur at salinities between 400 mgℓ^{-1} and 2 240 mgℓ^{-1}, although the abundance of species decreases above 1 000 mgℓ^{-1} (Hart *et al.* 1991). Vaidya & Nagabhushanam (1979) have studied the survival rate of the freshwater snail *Indoplanorbis exustus* in water of varying salt (NaCl) concentrations. Deaths occur after 200 mgℓ^{-1} and there is 100% mortality at 800 mgℓ^{-1}. In South Africa a study on four species of snail, *Biomphalaria pfeifferi, Bulinus (Physopsis) globosus, Lymnaea natalensis* and *Bulinus (Bulinus) tropicus*, showed that breeding success was severely reduced at levels of TDS below 122 mgℓ^{-1} for all species. Breeding success declined again at salinity levels higher than 510 mgℓ^{-1} for *B. pfeifferi* and *B.(P.) globosus*, 340 mgℓ^{-1} for *L. natalensis* and 595 mgℓ^{-1} for *B.(B.) tropicus* (Jennings 1976).

Snails are the vectors for a number of diseases, amongst them the human bilharzia parasite, *Schistosoma*. The cercaria (larval forms) of the parasite develop in the snail host, and are emitted into the water when they are fully developed. Once released, the cercaria bore their way through the skin into the primary host, generally a mammalian host, and then grow to maturity. It is unlikely that salinities below 1 200 mgℓ^{-1} will affect the cercaria (Hartwig, 1994). It is assumed that this is the case for this model.

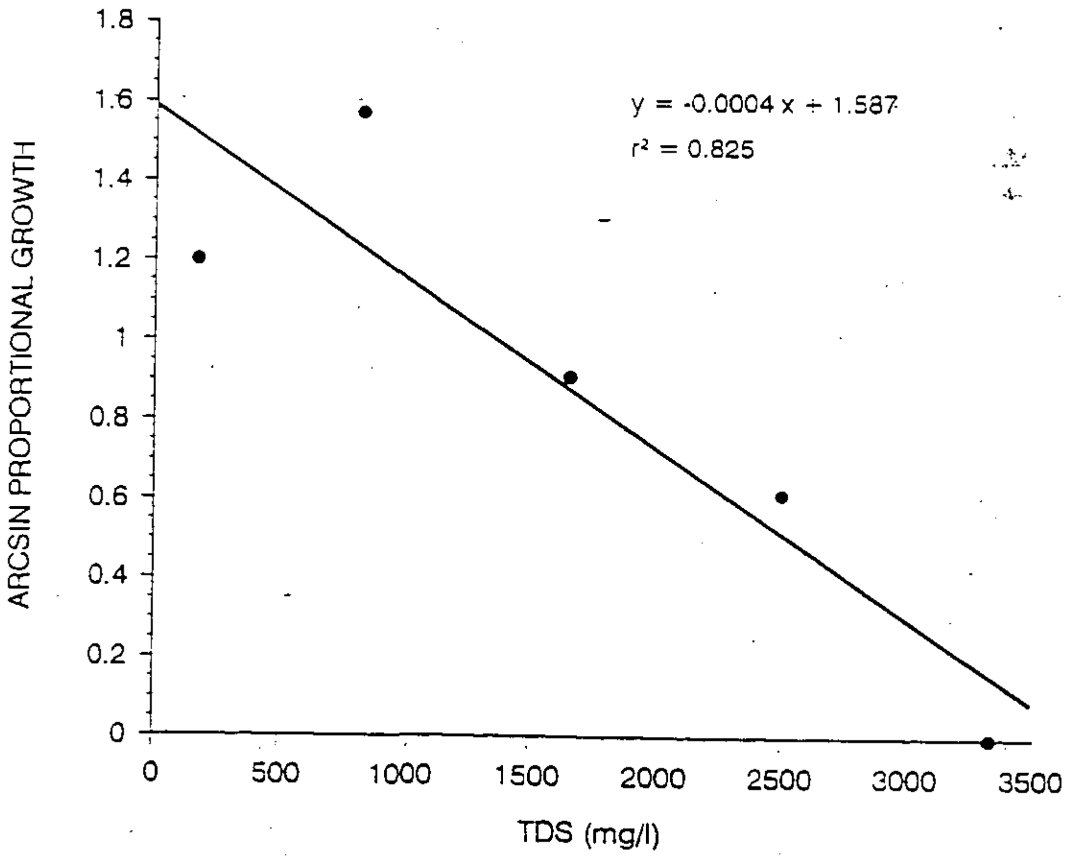

FIGURE 4: Percentage reduction in growth of water hyacinth at different TDS levels.

Ceratopogonidae and Culicidae

The Ceratopogonidae and the Culicidae, although terrestrial as adults, rely on freshwater for their breeding habitats. Most prefer stagnant waters to breed in and are often not associated with rivers (Nevill, 1994). Their abundance in the Vaal River is low (Chutter 1986).

Ceratopoginid larvae have been found at salinity levels of between 500 mgℓ^{-1} and 3 400 mgℓ^{-1} in Australia (Hart *et al.* 1991). In South Africa it has been found that the Culicidae are not affected by levels below about 5 000 mgℓ^{-1} (Leseure, 1994). Thus, at a level of between 200 mgℓ^{-1} and 1 200 mgℓ^{-1} TDS, it seems that the larval populations of the Culicidae and Ceratopogonidae will not be affected.

Simuliidae

The Simuliidae, *Simulium chutteri*, *S. nigritarse*, *S. adersi* and *S. damnosum*, have on occasion reached pest proportions in some South African catchment areas. With increased eutrophication and the building of dams there has been a steady increase in the blackfly populations, and outbreaks have been frequently recorded since 1956 (de Moor 1986). Although they are not disease vectors *per se*, they attack and bite cattle, sheep, poultry and even man in such numbers that animal production may be severely affected. As with the biting midges, *Simulium* spp. also breed in freshwater. All larval stages are completed in flowing freshwater, and larvae may be locally very abundant where suitable rapids for larval growth and pupation occur. Little is known of the salinity tolerances of the larvae, although it is recognised that various species have different tolerance levels (Palmer, 1994).

Hart *et al.* (1991) suggest that the Simuliidae are restricted to waters of salinity <1 000 mgℓ^{-1}. However, a study in Australia on the Blackwood River (Western Australia) and the Glenelg River (Victoria) indicated that blackfly larvae were found at salinity levels greater than 4 000 mgℓ^{-1} (William *et al.* 1991).

It seems that the abundance of blackfly larvae will not be influenced directly by an increase in salinity below 1 200 mg l-1, although community structure may change. A greater influence may be the indirect affect of salinity by an increase in the algal abundances in the river. The relationship between *Simulium* populations and algal blooms requires investigation.

Other Invertebrates

Information on the salt tolerance of freshwater invertebrates is sparse. It is clear, however, that factors such as the degree of acclimation, life stage, condition and temperature can influence sensitivity to salt concentrations (Hart *et al.* 1991). Hart *et al.* (1991) suggest that macroinvertebrates are amongst the most sensitive to increases in salinity in comparison to other Australian biota, with adverse effects apparent in some species at salinities below 1 000 mgℓ^{-1}. Thus changes in community structure and species diversity may occur. However, evidence to the contrary does exist. The Blackwood River in western Victoria, Australia, has an unusual reverse salinity gradient, with the headwaters being more saline than the lower reaches. Comparisons of this with the Glenelg River, which has the opposite salinity gradient, indicated that macrofaunal composition and salinity may not be closely correlated in most Australian rivers (Williams *et al.* 1991). Macroinvertebrates may, in fact, be far more tolerant to salinity effects than first thought (Williams *et al.* 1991). The situation in South Africa is uncertain and little information is available. It is suggested that any change in invertebrate communities will not have measurable direct economic effects and can be excluded from the model.

Loss of Species

Many of the beneficial uses of river and stream ecosystems depend on the natural conditions in these systems. Any disturbance to the ecosystem balance will cause a change in the structure and functioning of the aquatic communities involved (Williams 1987). Although some information is available regarding salinity tolerance of certain individual stream-dwelling organisms, there is a dearth of information on the synecological effects of salinity changes. For instance, there is little documented on the effects of changing salinity on behavioural responses, on communities, and on ecosystem processes such as microbial decomposition and nutrient cycling (Hart *et al.* 1990).

One of the results of these changes is a loss of species, or a change in the species composition in a river. In some cases more tolerant organisms will succeed loss tolerant organisms, as was the case with diatom communities in the Sundays River, Eastern Cape (Forbes & Allanson 1970). In Australia high salt levels in rivers often lead to depauperate fauna, although the levels experienced there are far in excess of the levels experienced in South African rivers.

Whether the change is a change in the species composition or a decrease in the diversity of a system, the loss of species to that system is almost impossible to put a value to. Some research is being undertaken in America on the valuation of large organisms (American Fisheries Society 1992), and there are economic valuation methodologies which estimate society's value for a resource. However, for smaller organism which society is not aware of, this is an impossible task. The cost to the system in terms of species loss will thus not be included in the model.

4. INDIRECT EFFECTS OF TOTAL DISSOLVED SOLIDS

4.1 PHYTOPLANKTON BLOOMS AND RELATED EFFECTS

Primary production (the amount of organic material formed by photosynthetic processes) in a water body is primarily determined by the level of eutrophication of a system, where eutrophication may be defined as the enrichment of a water body with plant nutrients, particularly phosphorus and nitrogen, often as a result of human activity. It is, however, modified by other factors such as light, temperature, oxygen levels and other organisms. (Edwards & Garrod 1972; Barnes & Mann 1980; Goldman & Horne 1983).

Of particular interest is the role of light. Light changes as it enters the water of a river or lake. First it is refracted and then either absorbed, reflected, scattered or transmitted (Goldman & Horne 1983). All these lead to an attenuation of the light or irradiance, which determines the rate of photosynthesis of primary producers at various depths, as well as the growth rates of organisms. Photosynthesis does not occur until a threshold irradiance has been reached. Thereafter the rate is proportional to irradiance until the light saturation level is reached (at about 1/20th of sunlight), at which stage photosynthesis is no longer related to irradiance (Barnes & Mann 1980). At higher levels (half of full sunlight) photosynthesis becomes inhibited. The primary productivity and growth of algae and plants is directly proportional to photosynthesis.

In general South African rivers and lakes are naturally turbid in the rainy season due to high silt loads. Without salinisation of a river the natural turbidity lessens the chances of algal blooms occurring. However, as the concentration of dissolved salts increases in the river, so turbidity decreases and light availability increases. With relatively warm temperatures prevailing, and more than adequate amounts of nutrients, it was originally thought that this change in the light régime would be sufficient to trigger algal blooms. Several studies have related light to algal abundance and growth (e.g. Walmsley 1980; Stegmann 1982; Pieterse 1982). Walmsley (1980), however, suggests that there is no relationship between algal abundance and turbidity in South African waters (Table 4), and that the trophic status of the waters will determine whether blooms occur or not. These relationships were verified by Stegmann (1982) on the Wuras Dam.

TABLE 4: Correlation coefficients between variables associated with light penetration and surface water transparency in Lindleyspoort Dam. Asterisks indicate significance at p=0.001 level (from Walmsley 1980).

	1/SECCHI DEPTH	TURBIDITY(JTU)	SUSPENDED SOLIDS	k
1/SECCHI DEPTH				
TURBIDITY(JTU)	0.90*			
SUSPENDED SOLIDS	0.83*	0.81*		
k	0.92*	0.94*	0.74*	
CHLOROPHYLL a	-0.02	-0.04	-0.04	-0.09

It is apparent that the relationship between algal abundance and turbidity is confused by the fact that algal abundance contributes to the turbidity of a water body. Although the theory discussed above remains the same, it is more exact for modelling purposes to relate chlorophyll *a*, as a measure of algal abundance, to directly to TDS. Information from 28 freshwater sites (slow-flowing rivers or dams - Butty & Walmsley 1979; Walmsley 1980; Bruwer *et al.* 1985) was used to plot mean chlorophyll *a* concentrations against TDS (Figure 5). The relationship was significant at $p < 0.001$ ($r = 0.67$) and may be expressed as:

$$\mathrm{CHLA} = 0.458 * \mathrm{TDS}^{-0.316} \qquad 5$$

where CHLA = chlorophyll *a* concentration ($\mu g \ell^{-1}$)

It must be kept in mind that, although this relationship is significant, TDS only accounts for 56% of the variation in chlorophyll *a*.

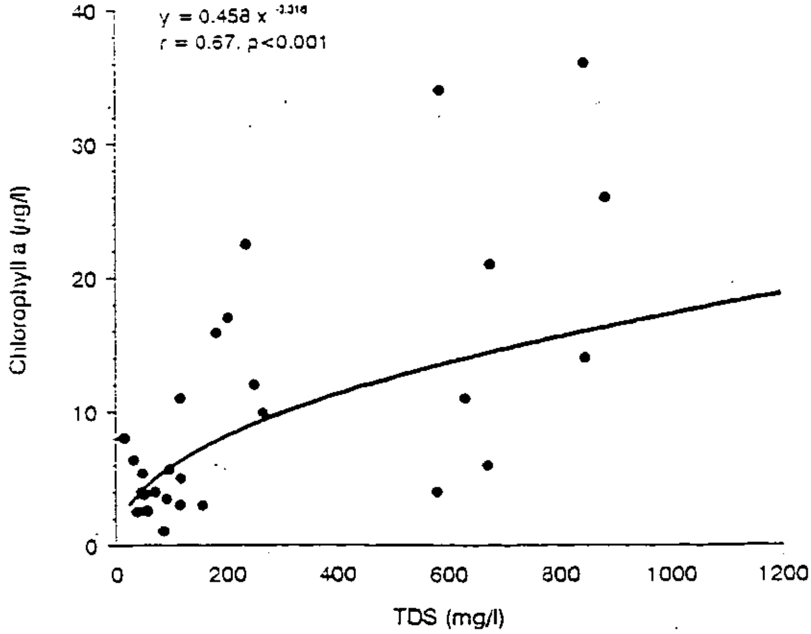

FIGURE 5: Relationship between mean chlorophyll *a* and mean total dissolved solids at 26 freshwater sites around South Africa.

Another factor to be taken into account is that algal blooms will only occur if the rivers are sufficiently slow- flowing, so that phytoplankton populations have opportunity to grow before being reduced by passage through rapids. For instance, the Tugela River is a fast- flowing river with a steep gradient (from Bergville to Colenso the mean gradient is 1:645, and from Colenso to Mandini it is 1:327), and has no recorded algal problems. The Vaal River, on the other hand is slow flowing with a gradient of 1:1992 between the Vaal Barrage and Bloemhof Dam, and experiences frequent algal blooms. Thus, the economic model of the cost of salinity increases will not always include the phytoplankton sub-sector.

The effect of possible algal blooms on the natural environment, is made up of a multitude of complex interactions. For instance, water hyacinth population size affects algal growth and abundance (Pearce 1987), and algae affect the sedimentation of colloids (Avnimelech *et al.* 1982). We have thus simplified the model to include only those impacts that we feel have economic implications both on the natural environment and man:

1. The effect of algal abundance on water treatment (to be dealt with by Corrie Marx).
2. The effect of algal abundance on the blackfly larval population.
3. Toxic effects of blue-green algae.

Two types of algal bloom may occur in the Vaal River: toxic blue-green algae (cyanophyta) or dinoflagellate blooms, and non-toxic green algae or diatom blooms. The presence or absence of blue-green algal blooms depends on the total nitrogen to total phosphorus ratio in the system. They will not occur if the ratio of nitrogen to phosphorus is high (>5). If the ratio drops, then blue-green blooms will occur.

If a non-toxic bloom is present, the overall productivity of the system will increase as productivity increases on every trophic level. In the case of blackfly larvae, food availability will increase and the abundance of flies will increase accordingly. The relationship between the algal abundance and blackfly abundance in South Africa has not been recorded.

If toxic blooms occur (generally blue-green blooms dominated by *Microcystis* spp. and *Oscillatoria* spp.) the effects may be diffuse and many. Some of them include fish kills, drastic declines in the simuliid populations (Palmer, 1994) and stock loss due to poisoning. Large phytoplankton blooms may also decrease the levels of oxygen in the water once the organisms die and start to decompose. This creates anoxic conditions, which may be stressful for the biota (especially fish). Although an attempt can be made to model toxic effects, it is very difficult to put a value on anoxic effect, which will be excluded from the model.

4.2 FILAMENTOUS ALGAE

As with the phytoplankton, filamentous algae are also affected by light levels in the water column. These algae, however, are reliant on light reaching the bottom of the water column, as they are attached to the substrate. Green filamentous algal species such as *Cladophora* spp. appear to require high light intensities to grow (Joska & Bolton 1994). The growth of *Cladophora* spp. in canals, where they cause the most nuisance, depends on the depth of the canal and the turbidity of the water. It is unclear as to how filamentous algal abundance is affected by increases in TDS. With the decrease in inorganic turbidity, an increase in filamentous algal growth is expected. However, phytoplankton blooms decrease the light reaching the filamentous algae, and growth of the filamentous algae will be diminished (Joska & Bolton 1994). For the purposes of this model it is assumed that the increased TDS levels will have little effect on the growth of filamentous algae.

4.3 FISH

Fish productivity can become highly complex in any system, and may be affected by factors such as suspensoids (Bruton 1985; Hecht & Van der Lingen 1992), alkalinity, algal biomass, air temperature, area, benthos standing crop, nutrient levels and water weed abundance (Downing *et al.* 1990). Ryder (1965) makes use of a simple model, the morpho-edaphic index (MEI), to determine fish productivity in North-temperate lakes. He suggests that fish productivity is affected by three principal influences -the morphometric, edaphic (derived from the geology and soils of a catchment) and climatic factors (Ryder 1965). If climatic factors are assumed to be relatively constant, then fish production may be related to the morphometric and edaphic factors only. In the case of the MEI these factors are TDS and the mean depth of the water body, and:

$$\text{MEI} = \text{TDS}/\text{Mean depth} \qquad 6$$

Ryder (1965) found that fish production could be related to MEI (British units) as follows:

$$P = 2.094\ \text{MEI}^{0.44610} \qquad 7$$

where MEI = TDS(ppm)/Mean depth(feet)
P = Production (lb $\text{acre}^{-1}\text{a}^{-1}$)

The value of the MEI lies in its simplicity. However much debate has centred around its validity (Ryder *et al.* 1973; Oglesby 1977; Downing *et al.* 1990; Jackson *et al.* 1990; Downing & Plante 1993). Ryder *et al.* (1973) suggest that, although the model is simple, TDS and mean depth adequately represent morphometric and edaphic conditions in a water body. TDS is considered important as it represents and average edaphic condition for any watershed. It bears a relationship to the geology of the watershed, as well as the nutrient loading in a system. It is often considered a better reflection of the ionic content of a water body than any of its single components, and may reflect environmental features such as physiography, shore development, terrestrial vegetation and aerial nutrient fallout (Ryder *et al.* 1973). Mean depth represents morphometric features other than depth alone. To a degree it correlates inversely with photic zone, as well as correlating with factors such as area and shore development.

Downing *et al.* (1990) suggest that fish production is correlated to primary productivity and not the MEI. However, the data used were from a wide range of climatic regions, whereas the MEI is only valid under similar climatic conditions. Oglesby (1977) found that the relationship between fish production and MEI and fish production and primary productivity were both significant if climatic conditions are fairly constant. The MEI has been found to be valid for several climatic regions, including tropical and sub-tropical rivers in Africa (Henderson & Welcomme 1974; Lesack 1986; Bishai & Khalil 1990). The MEI is considered valid for use in this model, although it must be kept in mind that the relationship between MEI and fish production units varies with changing climatic region.

Henderson & Welcomme (1974) related the catch per fisherman to the MEI in 31 African lakes (Figure 6). They found that there is a significant positive correlation between the two ($r = 0.739$, $p < 0.001$), and the relationship can be described by the following:

$$CPF = 0.7779*MEI^{0.3775} \qquad 8$$

where CPF = catch per fisherman (t)
MEI = TDS($mg\ell^{-1}$)/Mean depth (m)

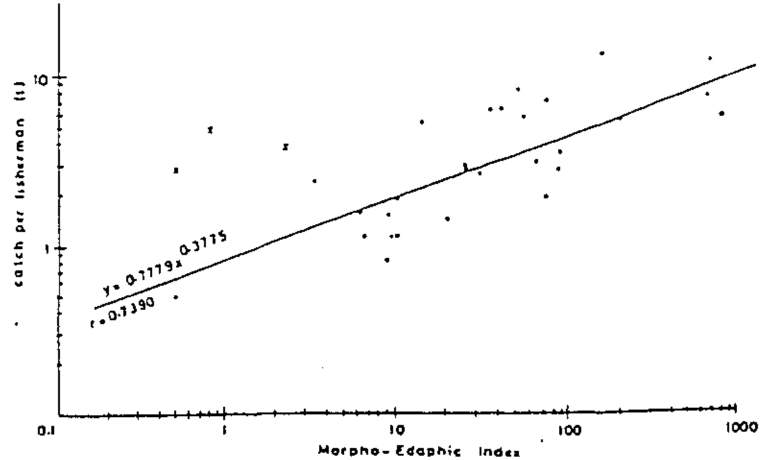

FIGURE 6: The relationship between morpho-edaphic index and catch per fisherman for 31 African lakes. Lakes Victoria, Malawi and Tanganyika are shown as crosses and all the others by dots (from Henderson & Welcomme 1974).

5. COSTS AND BENEFITS OF SALINISATION

5.1 HUMAN HEALTH CARE

Snails

The snail *Biomphalaria pfeifferi* is the secondary host for the human bilharzia parasite. Although TDS may have a slight negative effect on snail abundance, it is unlikely that it will have much economic impact. The number of cercaria that are released into the water by each snail is so large that a small reduction in the snail populations will not significantly affect the number of human hosts infected. This is thus excluded from the model.

5.2 VETERINARY CARE

The two most problematic types of organisms as far as veterinary costs are concerned are the Simuliidae and the disease vectors, snails.

Snails

The snails *Bulinus tropicus* and *Lymnaea* spp. are disease vectors for the liver flukes *Fasciola* spp. and *Paramphistoma* spp. (found in livestock in the Transvaal Highveld and the Orange Free State), as well as transmitters of bovine bilharzia found in the Transvaal Lowveld during high rainfall years. The snails are found primarily in slow flowing or stagnant waters such as vleis, pans, earth dams, pools and slow-flowing rivers. Rivers are important as reservoirs of snails, especially when they flood and form lateral pans.

Increased TDS levels in rivers would not have any economic implications as far as veterinary diseases carried by snails are concerned. Essential roundworm treatment incidently controls *Fasciola* spp., while essential tape worm treatment controls *Paramphistoma* spp (Van Wyk, 1994). Thus, changes in the snail populations will not have extra significant veterinary costs attached to them.

Simuliidae

The cost of *S. chutteri* to sheep farming on the Orange River between Hopetown and Augrabies is estimated at R30 million in direct costs and R7.5 million in lost state revenue in 1990 (Nevill, 1994). The relationship between increased algal abundance, blackfly larval populations and the resultant adult populations is unknown. The blackfly larvae are, however, restricted to certain fast-flowing riffle and rapid areas in rivers. The increase in abundance in these restricted areas will not affect the cost of treatment which is determined per unit area.

Blue-green algal blooms, on the other hand, may wipe out simuliid populations. In economic terms this would be a benefit as there would be a saving in terms of the treatment used (presently estimated at R1 million per annum in the Orange River between PK Le Roux Dam to Augrabies. The most successful treatments are the application of an emulsified bacterial pesticide *Bacillus thuringiensis var israelensis*, known as B.t.i. (Car & de Moor 1984; de Moor 1986) and a chemical agent, temephos. The application of these will be unnecessary should a blue-green bloom deplete the population. The benefit can be calculated from the cost of treatment per unit area saved. At present on the Orange River, the cost is estimated at about R1 600 per rapid or R40 $m^{-3}s$.

5.3 ALGAL BLOOMS

The two types of algal bloom have different cost implications.

Non-Toxic Blooms (Green Algae or Diatoms)

The major implication in terms of non-toxic blooms is the increased cost of water purification.

Toxic Blooms (Cyanophyta and Dinoflagellates)

The excessive growth of blue-green algae in eutrophic systems has several economic impacts. They are:

1. Toxic taste and odour problems which lead to increased water treatment costs (activated carbon treatment is expensive);
2. Anaerobic layers in lakes and slow-flowing waters, with adverse effects on oxygen dependant organisms such as fish;
3. Aesthetic problems, interference with recreational activities and a depreciation of property values.
4. Toxic effects causing skin-irritation in swimmers, gastric problems, and stock and fish poisoning.

Because of the range of problems caused by blue-green blooms, costing of the economic effects is extremely difficult, and falls under other sectors such as recreation and purification as well as the natural environment. However, one way of determining the cost of these blooms is by costing the treatment used to lessen the impact of the blooms. One of the main treatments is spraying of copper sulphate onto the bloom. This may have other ecological implications, which are not taken into account in the model.

Microcystis blooms on the Vaal Dam in 1978 were treated by spraying $CuSO_4$ from a helicopter at a cost of R12 per application per hectare (Bruwer 1979). This is the equivalent of about R88 a^{-1} ha^{-1} in 1993. The cost depends on the area to be sprayed.

5.4 WATER HYACINTH

A high abundance of water hyacinth on a water body has several negative economic implications (Ashton *et al.* 1986), including:

- river blockages and interrupted water flows;
- excessive evapotranspiration losses;
- difficulties in water treatment to obtain potable standards;
- increased breeding sites for disease vectors, and
- inhibition of recreational use of water bodies.

During the hyacinth infestation on Hartebeestpoort Dam, water losses due to evapotranspiration alone were estimated to be 18 million cubic metres over a three year period (Bosman, 1994). At present prices this is equal to about R47,7 ha^{-1} of water hyacinth. In 1975 at Hartebeestpoort Dam, recreational revenue lost due to the hyacinth infestation was estimated at R13 million (R130 million in 1993 real terms).

The cost of removal of water hyacinth gives an indication of the cost of the infestation. This is difficult to estimate due to different methodologies used for removal. Additionally the small effect that TDS changes have on the hyacinth populations may not affect the cost of removal significantly, depending on the method of removal. For the sake of the model it is assumed that chemical spraying is the most appropriate methodology. The cost of hyacinth removal using glyphosate treatment is currently estimated at R230 000 per annum per 30 ha, or R7 700 $ha^{-1}a^{-1}$ (Davidson, 1994). These costs can be related to the change in hyacinth growth with increased TDS.

5.5 ANGLING REVENUE

The strongest attraction of the public to visit inland freshwater sites is that of angling (Cadieux 1979). Table 5 gives the statistics regarding the angling activity in the Transvaal in the 1977-78 season. In 1993 rands the total cost of angling in the Transvaal was R374. million, and on the Vaal River it was R191.1 million. If the change in fish production is assumed to lead to a proportional change in angling revenue, the MEI can be directly related to the fishing expenditure. This assumption may have to be empirically tested at a later stage. For modelling purposes it is suggested that there is a straight line relationship between MEI and angling revenue with an intercept at 0, and the following relationship applies:

$$R_A = MEI/k$$

where R_A = angling revenue
MEI = morpho-edaphic index
k = constant = 1978 MEI/1978 R_A.

TABLE 5: Statistics for the 1977/78 angling season in the Transvaal (from Cadieux 1978).

Total time expended	3 143 673 days
Total number of fish caught	9 497 524 fish
Total mass caught	9 650 430 kg
Average catch per angler per year	83.690 fish
Average mass per angler per year	85.029 kg
Average size of fish landed	1.016 kg
Total expenditure throughout the season	R 32 140 368 (a)
Total economic impact for the season	R 51 484 406 (b)
Cost per fish landed	R 3.38 (a) R 5.42 (b)
Cost per kilogram of fish landed	R 3.33 (a) R 5.33 (b)

6. MODELLING THE SYSTEM

An initial conceptual model included all the parameters that may have had an economic influence. The result was a highly complex interaction between various components. Many of these components were dismissed on the grounds that in the range of TDS of the model there were no apparent impacts. Thus, such components as the terrestrial and aquatic wildlife, riparian vegetation and some disease vectors were withdrawn from the model.

The model thus deals with the direct effects of TDS on snails and hyacinth, the indirect effects of TDS on algal abundance, and the resultant impacts. Further simplification and the use of existing techniques has enabled some of the complex interactions to be expressed as a single equation (e.g. the morpho-edaphic index). The resultant model is given in Figure 7. The total cost of the effect of changing TDS levels on the environment may be expressed as:

$$C_{TOTAL} = C_{FISH} + C_{HYACINTH} + C_{ALGAE} + C_{SNAILS} + C_{SIMULIIDAE} \qquad 10$$

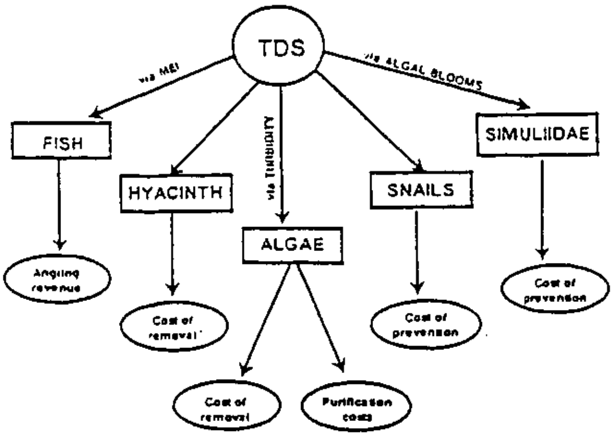

FIGURE 7: Conceptual model of the effect of increased TDS on the natural environment.

6.1 VETERINARY AND HUMAN HEALTH CARE

Snails

No costs are associated with a change in the TDS levels as far as liver flukes, bilharzia and their vectors are concerned. Therefore they have been excluded from the model.

Simuliidae

Assumptions

1. The increase in blackfly larval populations due to increases in non-toxic algal abundance do not involve any extra larval control costs.
2. Blackfly larval populations are seriously depleted if a toxic algal bloom occurs.
3. All problem species are equally expensive to control.

Formulae

If $N/P < 5$, then BLUE GREEN = 1 (present)
If $N/P \geq 5$, then BLUE GREEN = 0 (absent)

If BLUE GREEN = 1,
then $B = -C$ 11
$C = C_T * A$ 12
else $B = 0$

where B = benefit (Rands)
C = cost of treatment (Rands)
C_T = cost of treatment per unit area (Rands m^{-2})
A = area where treatment is occurring.

6.2 ALGAL BLOOMS

Assumptions

1. As salinity increases more light becomes available to phytoplankton through the precipitation of colloids. Because there are sufficient available nutrients, algal blooms will occur if the light conditions are favourable.
2. Algal blooms will only occur if the river is sufficiently slow-flowing. This is related to the gradient of the river where salinity increases occur.

3. As temperature and salinity conditions change the phytoplankton will come to be dominated by the species best adapted to the prevailing conditions, but the algal biomass will be unaffected by this change.
4. For economic modelling purposes the most important consideration is whether or not the dominant phytoplankton are toxic or non-toxic algae. When $N/P < 5$ cyanophyta (toxic) dominate.
5. All phytoplankton species are equally expensive to remove from water.
6. The cost of water treatment is related to the concentration of algae in the water.

Formulae

If GRADIENT > 1:1600, then ALGAL BLOOM POSSIBLE = 0
If ALGAL BLOOM POSSIBLE = 0, then $C_P = 0$ and $C_R = 0$

If GRADIENT < 1:1600, then ALGAL BLOOM POSSIBLE = 1
If ALGAL BLOOM POSSIBLE = 1 and BLUE-GREEN = 1 or 0,

then $$\text{CHLA} = 0.458*\text{TDS}^{-0.316} \qquad 5$$

C_P = COST OF PURIFICATION

If ALGAL BLOOM POSSIBLE = 1 and BLUE GREEN = 1,

then $$C_R = C_A * A \qquad 13$$

where CHLA = chlorophyll *a* concentrations (μg l^{-1})
C_P = cost of purification
C_R = cost of removal treatment (Rands)
C_A = cost of removal per unit area (Rands m^{-2})
A = area treated (m^{-2}).

6.3 HYACINTH REMOVAL

Assumptions

1. Hyacinth growth rates are greatest at intermediate TDS concentrations (about 750 mgℓ^{-1}).
2. The role of nutrients, which stimulate hyacinth growth, may be ignored for the purposes of this model.
3. Hyacinth control costs increase with hyacinth growth rate.

Formulae

$\Delta A = \bar{A} * \%\Delta$ 14

$\%\Delta = \sin(-0.0004*TDS + 1.587) * 100$ 15

$C = C_A * \Delta A$ 16

$B = -C$ 11

where $\bar{A}$ = mean hyacinth population size per annum ($m^{-2}a^{-1}$)
$\%\Delta$ = % change in hyacinth population size (m^{-2}) due to TDS
C = cost of additional removal treatment (Rands)
C_A = cost of removal per unit area (Rands m^{-2})
ΔA = change in area treated (m^{-2}).

6.4 ANGLING REVENUE

Assumptions

1. The value of angling fish is measured as the revenue generated through the angling activity (license fees, equipment, travel, etc).
2. Angling success stimulates revenue generated by angling.
3. "Anglers" who are prepared *not* to catch fish make an insignificant contribution to angling revenue.
4. Angling revenue is proportional to fish production, which is proportional to the MEI.

Formulae

$MEI = TDS/MEAN\ DEPTH$ 6

$R_A = MEI/k$ 9

where R_A = angling revenue (Rands)
MEI = morpho-edaphic index
k = constant = 1978 MEI/1978 R_A.

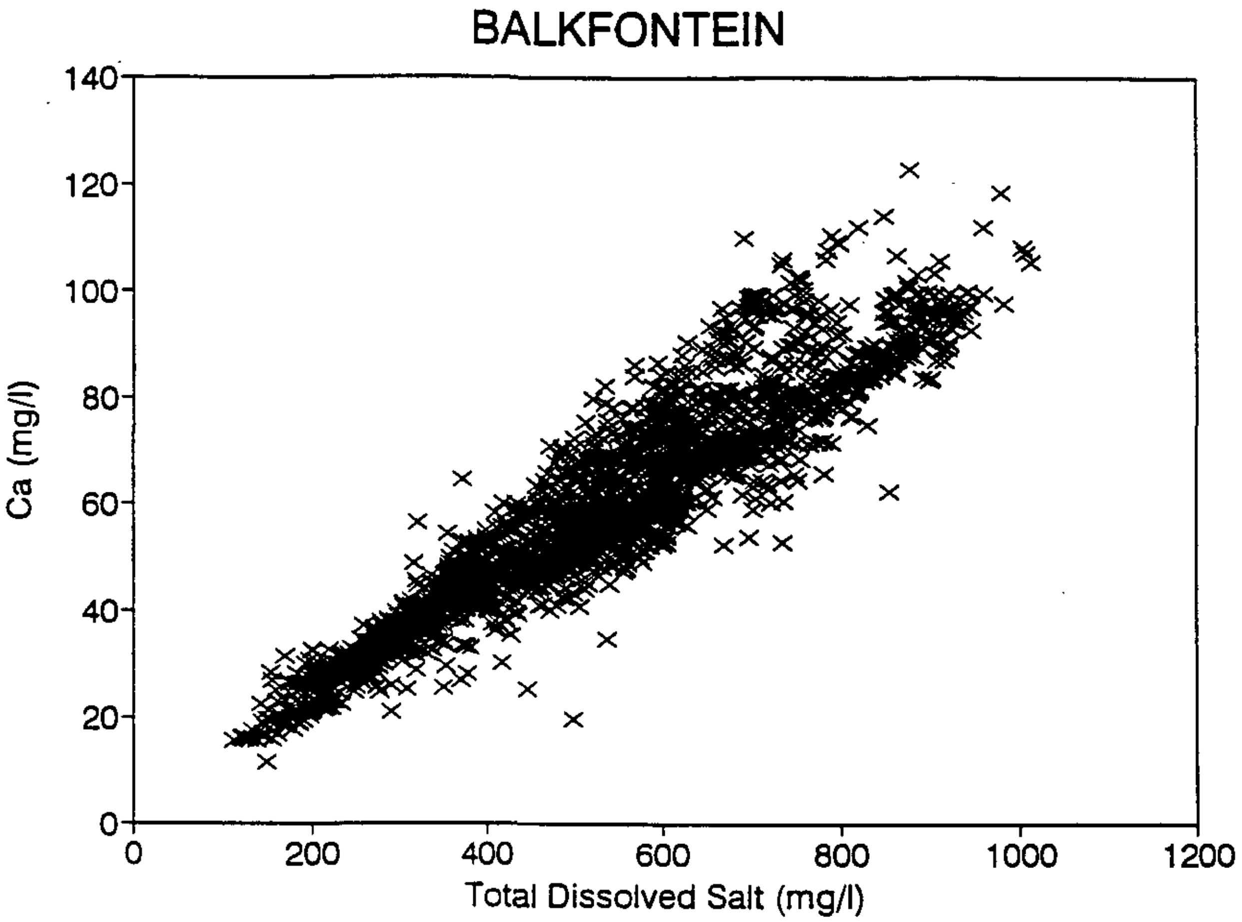

FIGURE 6: Plot of Total Dissolved Salts vs Ca^{2+} as sampled at Balkfontein

6.5 SUMMARY

In economic terms the natural environment has not traditionally been considered as one of the primary sectors, and is often not taken into consideration. Although it is treated separately in this report, the costs or benefits associated with changes in the environment should be incorporated into the traditionally accepted sectors as follows:

1. fish - recreation
2. hyacinth - recreation
3. algae - water treatment
4. disease vectors - agriculture

The total cost of both indirect and direct effects of increased TDS levels are unknown. The possibility exists that the total cost associated with changes in the natural environment are not significant in comparison with the other sectors. When the first run of the overall model of the economic impact of salinity has been made, a first estimate of the costs of impacts on the natural environment relative to the other sectors will be available. This will indicate whether it is worthwhile to continue with the refinements of the costs of impacts on the natural environment or not.

7. MODEL APPLICATION

The purpose of this section is to apply the formulae that were developed for the natural environment sector. The following approach was used:

1. The previous document was refereed by several specialists in each of the sub-sectors of the model. Their comments were taken into account, and any required major changes in the model were made.
2. The model formulae were used to assess the "worst" case (highest cost) and "best" case (highest benefit) scenarios for each sub-sector, using data from the Middle Vaal River. Where assumptions were required, they have been specified.
3. The practical applicability of the formulae was assessed, and the economic implications of each sub-sector evaluated.
4. An abridged version of the model for the natural environment sector is proposed.

7.1 REFEREES COMMENTS

The majority of comments from the selected referees were editorial, and clarification was required throughout the document. Changes to the model were suggested by the referees for the sub-sectors of "water hyacinth" and "algal blooms".

7.1.1 Water hyacinth

Water hyacinth will only expand to fill the available area; once the water surface has been covered, the growth rate will decline to zero, no matter what the TDS levels. If hyacinth is removed, the niche created is available for colonisation by the remaining plants. The area covered by the hyacinth is also affected by the wind and current, and the compaction of the mats. Although it will be used in the following "run" of the model, the relationship used in the model is highly simplified.

7.1.2 Algal blooms

The cost of water treatment is not uniform for all phytoplankton species. It was initially assumed that the cost of water treatment did not vary according to the dominant phytoplankton species. This was considered to be an over-simplification. To take cognisance of all individual species would not be practical. However, the major cost difference is between the toxic blue-green algae on the one hand, and the green algae and diatoms on the other. Treatment of water containing blue-green algae requires an activated carbon filtration stage to remove algal toxins. This stage is not required in waters with large green algal and diatom populations. The model should thus be adjusted as follows:

```
If ALGAL BLOOM POSSIBLE = 1 and BLUE-GREEN = 1,
    then   $CHLA = 0.458*TDS^{-0.316}$
           $C_P$ = COST OF PURIFICATION = f(CHLA, CARBON FILTRATION)
If ALGAL BLOOM POSSIBLE = 1 and BLUE-GREEN = 0,
    then   $CHLA = 0.458*TDS^{-0.316}$
           $C_P$ = COST OF PURIFICATION = f(CHLA)
```

The cost of purification is to be taken further by the water treatment section of the model.

7.2 INITIAL "RUN" OF THE MODEL

The model was "run" for each of the sub-sectors in the natural environment sector, using data from the Vaal River between Bloemhof Dam and the Barrage. The approach adopted was to calculate the "worst" case scenario, that is the highest cost or least benefit, and the "best" case scenario (least cost or most benefit) for each of the sub-sectors. These scenarios were then examined to determine whether inclusion of each sub-sector in the model or further investigation were warranted. This approach assumes that the model formulae are adequate to describe the effects of increasing TDS levels on the natural environment. Estimated values for all the parameters are required, and are calculated from the values discussed in the previous document. Due to the nature of the formulae and estimated parameter values, the resultant costs or benefits should be taken as approximations.

7.2.1 Model Parameters for the Middle Vaal River

The mean depth and width of the Middle Vaal River were estimated using the QUAL2E model, which indicated that for 50% of the time the Middle Vaal had a flow rate of only 7.7 ms^{-1}, a mean depth of 0.3 m and a mean width of 150 m (Rossouw, 1995).

Thus, the following are assumed to be the dimensions of the river between Bloemhof Dam and the Vaal Barrage:

Gradient $<$ 1:1600
Length = 480 km
Width = 150 m
Depth = 0.3 m

and, therefore

Surface area = 24 km^2 - 94 km^2 ≡ 240 ha - 960 ha
Volume = 7.2 million m^3 - 28.8 million m^3

7.2.2 Algae

The cost of water purification is not included in this run of the model, as it is to be treated as part of the water treatment sector. The only algae-related cost that could, therefore, be applicable to the natural environment sector, is the cost of controlling the toxic blue-green algae with copper sulphate. It is assumed for this exercise that $N/P < 5$, and that blue-green blooms are present.

It is estimated from 1978 figures that the cost of controlling the algae would be about R110.00 per hectare (R0.011 m^{-2}) in 1995. This figure was considered low by the referees of the previous document, although at a helicopter hire rate of R1 600 hr^{-1} and a spray rate of 20 ha hr^{-1}, the helicopter cost would only be R80 ha^{-1}.

The area to be treated would depend on the amount of blue-green algae. It is likely that only the dams would be treated. For this exercise, it is assumed that the whole length of the Middle Vaal will be treated. The estimated cost of algal control under these assumptions would be R792 000 for all TDS levels. Obviously if only the dams were treated, this estimate would be far lower.

7.2.3 Simuliidae

The cost of the treatment of the blackfly larvae may be expressed in two forms: as cost per kilometre of river per application, and as cost per rapid. The pros and cons of each are listed in Table 6. From this it was decided that the cost per km river per application would give the closer value to the true cost. It is also assumed that Simuliidae are present in the river in numbers warranting treatment and that a one-off biocide treatment would control blackfly for a whole year.

The following parameters are applicable:

Cost per km river per application = R234.53
No. of applications per annum = 1

Under these assumptions, the benefit gained from the blue-green algal blooms depleting the blackfly larval populations remains constant at R112 574. Even if biocide treatment were to be applied up to three times per annum, the total benefit would only be R337 722. There are, in fact, very few rapids in the Middle Vaal River. Thus the cost per rapid is likely to be far the less expensive, and the cost per kilometre is the greatest cost estimate.

TABLE 6: Characteristics of the two forms of expressing the cost of blackfly treatment on the Vaal River

COST PER KM PER APPLICATION		COST PER RAPID	
1.	Incorporates costs such as travel costs, manpower etc. which will be dependant on the length of the river.	1.	Although travel cost etc. have been incorporated, they relate directly to the Orange River, and are not necessarily valid for the Middle Vaal River.
2.	Does not take the number of rapids into account.	2.	Does not take the length of the river into account.
3.	Does not take behavioural aspects into account (e.g. what determines whether spraying takes place or not).	3.	Does not take behavioural aspects into account.
4.	Easy application.	4.	Difficult to apply.
5.	Assumes that blackfly are found on the stretch of river.	5.	Have to know the number of rapids in which blackfly occur in large numbers

7.2.4 Water Hyacinth

As reported in the previous document, the average cost of hyacinth control on the Middle Vaal River is R230 000. It is assumed that a decrease in the growth rate will decrease the cost of hyacinth control.

Results indicate that the benefit of increased salinity on water hyacinth removal is very small (R45 011 per annum maximum). Additionally, it is apparent from the comments of the referees that the formulae presented may not be valid. Even if salinity has a detrimental effect on the growth of hyacinth, because of the nature of the plant, it is unlikely to affect the cost of removal, and the formulae may be removed from the model.

7.2.5 Angling Revenue

The constant (k) can be calculated using 1978 MEI and angling revenue values. For this the TDS and mean depth for 1978 need to be estimated. It is assumed that the mean depth has not changed much since 1978, and that 0.3m is an adequate measurement of the 1978 mean depth. The mean annual TDS for 1980 at Schoemansdrift and Orkney weir was 567 $mg\ell^{-1}$ and 609 $mg\ell^{-1}$ respectively, and it is assumed that the 1978 mean values would not be much different from these. For the model run, the following estimates were used:

1978 Mean depth = 0.3 m
1978 TDS = 600 $mg\ell^{-1}$

1978 MEI = 600/0.3 = 2000

1978 Angling on Middle Vaal River = 16.23% of Transvaal angling revenue
= R8 355 919 (1978 Rands)

k = 1978 MEI/1978 angling revenue
= 2.393×10^{-4}

Under these assumptions, the lowest resultant value for angling revenue would be R2.79 million (TDS = 200 $mg\ell^{-1}$), and the highest value would be R16.71 million (TDS = 1 200 $mg\ell^{-1}$) in 1978 Rands. These figures are equivalent to R21.71 million and R130.05 million respectively at the 1994 Rand value.

7.3 DISCUSSION

The results for the "worst" case and the "best" case scenarios for each sub-sector are summarised in Table 7.

TABLE 7: Summary of extreme scenarios for each sub-sector for the natural environment sector. Figures in brackets indicate benefits (negative costs)

SUB-SECTOR	- HIGHEST COST - - LEAST BENEFIT -	- LEAST COST - - HIGHEST BENEFIT -
TOXIC ALGAL BLOOMS	R792 000	*
SIMULIIDAE	(R113 000)	*
WATER HYACINTH	(R4 853)	(R45 011)
ANGLING REVENUE	(R21 674 325)	(R130 045 960)

* Least cost not calculated.

The only component that has financial costs attached is the control of toxic algal blooms. These costs are associated with spraying the whole length of the Middle Vaal River and are, thus, an over-estimation of the true situation. However, even the highest cost of algal control is only R792 000, which is insignificant in comparison with, say, the provision of activated carbon filtration in water treatment works. It is also unlikely that copper sulphate would be used for the removal of blue-green algae from the system due to the toxic effect on other components of the ecosystem. Because of the major uncertainties, and the small cost to remove these algae, it is suggested that this formula be removed from the model.

Financial benefits are small in all the other sub-sectors except angling. Thus, the sub-sectors "Simuliidae" and "water hyacinth" may also be excluded from the model for practical purposes. "Angling revenue" is the largest of the sub-sectors, and must be retained in the model. Because of the large benefits to angling, and the number of estimated variables associated with angling revenue, it is important that this sub-sector be investigated further.

Although the MEI has been used on numerous occasions by fisheries managers as a predictive model, its statistical validity has been questioned by Jackson *et al.* (1990). The MEI was criticised on the basis of inherent statistical errors related to the use of ratios. For instance, mean depth is often determined as the ratio of volume to surface area, which leads to an unknown distribution of errors. Additionally Jackson *et al.* (1990) suggested that ratios may lead to spurious correlations and inappropriate null models. An alternative model, the morpho-edaphic model (MEM) was proposed by Rempel & Colby (1991). The MEM is statistically valid, but makes use of area and volume parameters, which, for our purpose, need to be estimated for the Middle Vaal River. This doubles the possible inherent error in estimating morphometric parameters, and use of the MEM would not be valid for South African rivers. Additionally, a comparison of MEM and MEI results indicated that the two methods are very close in their predictions of fish yield (Rempel & Colby 1991).

A re-evaluation of the MEI by Chow-Fraser (1991), determined that the basic assumptions of the model (i.e. that the parameters adequately reflect the morphological and edaphic features of a catchment) can be supported empirically. The predictive value of the MEI may weaken with increasing trophic level, but the MEI is still a valuable predictive tool. It is obvious that the case against the MEI on a statistical basis is not as strong as it seems. The MEI is still acceptable for use in the estimation of angling revenue for this model.

8. CONCLUSIONS

The financial costs and benefits associated with the "angling revenue" and "algae" (water treatment) sub-sectors are high enough to be included in the model. However, non-financial aspects should also be taken into account. In the natural environment sector, "loss of species" is one of the major costs associated with an increase in salinity. Due to a lack of markets for biological species, financial costs cannot be allocated to them. This sub-sector must, however, be acknowledged in the model. The model can, thus, be expressed as follows:

TOTAL COST = FINANCIAL COSTS + NON-FINANCIAL COSTS

where:

FINANCIAL COSTS = ALGAE + ANGLING REVENUE
NON-FINANCIAL COSTS = LOSS OF SPECIES

Practical formulae for the financial costs are outlined below.

ALGAE

If GRADIENT > 1:1600, then ALGAL BLOOM POSSIBLE = 0
If ALGAL BLOOM POSSIBLE = 0, then $C_P = 0$ and $C_R = 0$

If GRADIENT $< 1:1600$, then ALGAL BLOOM POSSIBLE = 1
If ALGAL BLOOM POSSIBLE = 1, and $N/P < 5$,
then BLUE GREEN = 1 (present)

If ALGAL BLOOM POSSIBLE = 1, and $N/P \geq 5$,
then BLUE GREEN = 0 (absent)

If ALGAL BLOOM POSSIBLE = 1 and BLUE-GREEN = 1,
then $CHLA = 0.458*TDS^{-0.316}$
C_P = COST OF PURIFICATION = f(CHLA, CARBON FILTRATION)
If ALGAL BLOOM POSSIBLE = 1 and BLUE-GREEN = 0,
then $CHLA = 0.458*TDS^{-0.316}$
C_P = COST OF PURIFICATION = f(CHLA)

ANGLING REVENUE

MEI = TDS/MEAN DEPTH

$R_A = MEI/k$

where R_A = angling revenue (Rands)
MEI = morpho-edaphic index
k = constant = 1978 MEI/1978 R_A.

APPENDIX A

MODEL RUN FOR THE NATURAL ENVIRONMENT SECTOR

DATA TO INPUT:

Middle Vaal length	480000
Middle Vaal width	150
Middle Vaal depth	0.3
Volume of river	21600000
Hyacinth removal cost	230000
Blackfly removal /km	234.53
Algal removal /m2	0.011
k MEI	0.0002393

MODEL RUN

			TDS values (mg/l)			
SUB-SECTOR	**200**	**400**	**600**	**800**	**1000**	**1200**
Blue-green bloom removal	792000.00	792000.00	792000.00	792000.00	792000.00	792000.00
Blackfly removal	-112574.40	-112574.40	-112574.40	-112574.40	-112574.40	-112574.40
Hyacinth removal	0.00	0.00	-4853.00	-18239.00	-31625.00	-45011.00
Angling revenue	-2785903.33	-5571806.66	-8357709.99	-11143613.32	-13929516.65	-16715419.97
TOTAL (Rands)	-2106477.73	-4892381.06	-7683137.39	-10482426.72	-13281716.05	-16081005.37
COST /m3 (Rands/m3)	-0.10	-0.23	-0.36	-0.49	-0.61	-0.74

ACKNOWLEDGEMENTS

The authors would like to express their sincere thanks and appreciation to the following people for their input into this study:

Dr Pete Ashton, Watertek, CSIR, Pretoria
Dr Garth Batchelor, TPA, Pretoria
Mr HH Bosman, DWA&F, Pretoria
Mr Charel Bruwer, DWA&F, Pretoria
Ms Maureen Coetzee, MRC, Johannesburg
Mr Dudley Davidson, DWA&F, Pretoria
Dr Kenny de Kock, Potchefstroom University for CHE, Potchefstroom
Dr Ferdie de Moor, Albany Museum, Grahamstown
Dr Pierre de Villiers, OFS Provincial Administration, Bloemfontein
Mr Chris Dickens, Umgeni Water, Pietermaritzburg
Prof Hein du Preez, RAU, Johannesburg
Dr Johan Engelbrecht, TPA, Lydenburg
Mr Ralph Heath, Watertek, CSIR, Pretoria
Prof Tom Hecht, Rhodes University, Grahamstown
DR Margaret Joska, University of Cape Town, Cape Town
Dr Dave Lesuere, MRC, Durban
Dr Errol Nevill, Veterinary Research Institute, Onderstepoort
Dr Rob Palmer, Veterinary Research Institute, Onderstepoort
Prof Braam Pieterse, Potchefstroom University for CHE, Potchefstroom
Mr Gavin Quibell, HRI, Pretoria
Prof Kevin Rogers, Wits University, Johannesburg
Mr Nico Rossouw, Watertek, CSIR, Pretoria
Dr Jan van Wyk, Veterinary Research Institute, Onderstepoort
Dr Dan Walmsley, FRD, Pretoria

The authors would like to express their sincere thanks and appreciation to the following people for refereeing the document on formulae development and their resultant input into this document:

Prof Chris Appleton, University of Natal, Pietermaritzburg
Dr Pete Ashton, Watertek, CSIR, Pretoria
Dr Garth Batchelor, TPA, Pretoria
Mr HH Bosman, DWA&F, Pretoria
Prof Tom Hecht, Rhodes University, Grahamstown
Dr Rob Palmer, Veterinary Research Institute, Onderstepoort
Prof Braam Pieterse, Potchefstroom University for CHE, Potchefstroom
Dr Dan Walmsley, FRD, Pretoria
We would also like to thank Dr Chris Elphinstone and Dr Nico Roussow for additional information provided by them.

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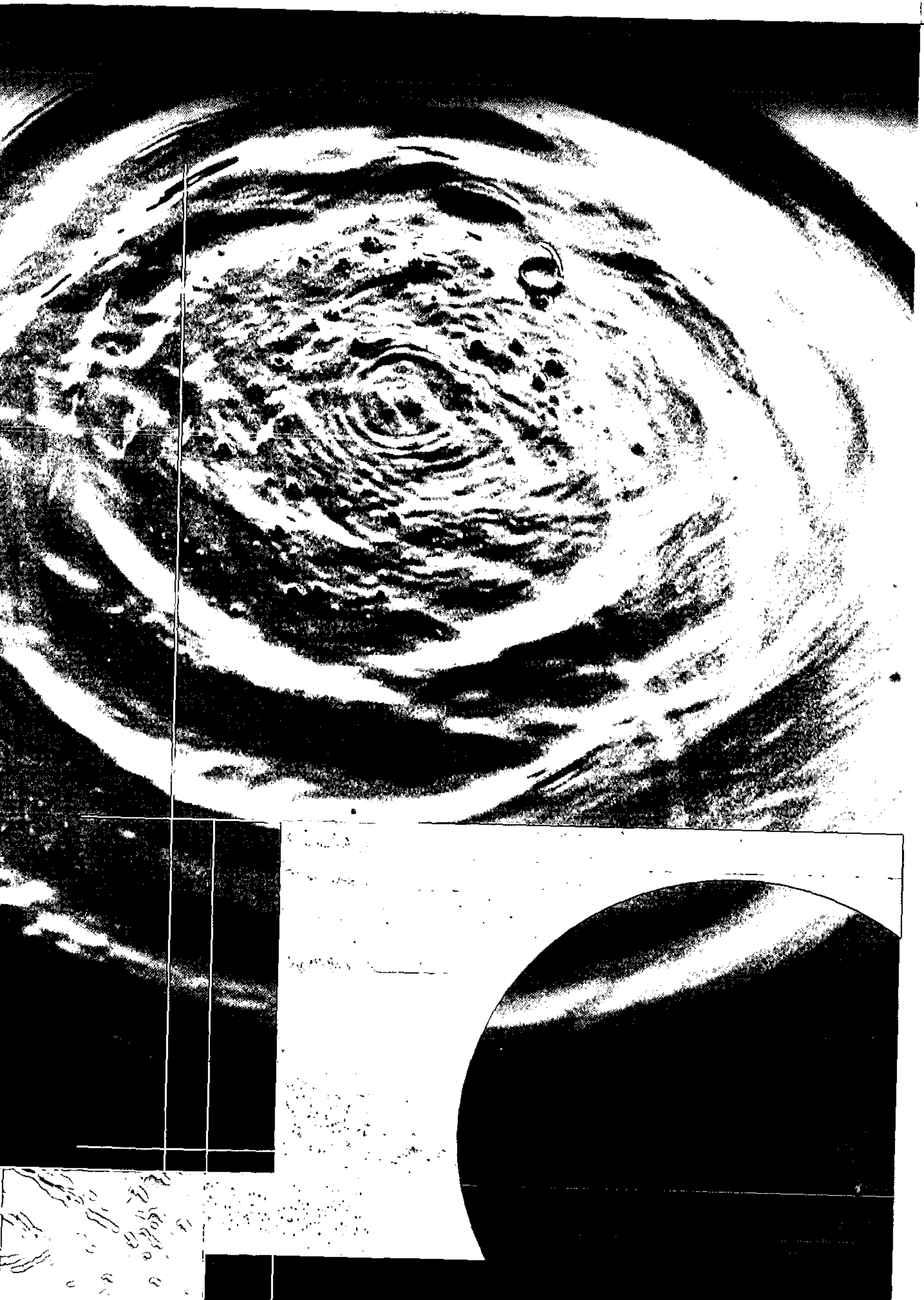

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