TMG AQUIFER ALLIANCE

WITH CONTRIBUTION FROM:

City of Cape Town

Table Mountain Group Aquifer Feasibility Study and Pilot Project

Task 8 – Wellfield Operation

Preliminary Phase

Final Draft

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EXECUTIVE SUMMARY

The Task 8 - Wellfield Operation of the TMG Aquifer Feasibility Study and Pilot Project investigates the requirements for the operation of the wellfield at the selected sites. This includes:

- *Aspects of water quality*
- *Requirements for water treatment*
- *Optimizing the pumping regime*
- *Estimating sustainable yield*
- *Monitoring protocols regarding optimal pumping rates*

During the Preliminary Phase the focus was on the first two items, which was undertaken by means of a literature and desk-top study. The following tentative conclusions are reached based on the limited available data:

i) The Peninsula Aquifer is generally of good water quality. The data indicate that the water may be highly corrosive. In some instances there might be elevated concentrations of sodium, chloride, iron and or manganese.

ii) Separate treatment facilities either remote at the wellfields or at the existing treatment works will probably be required for the removal of iron and manganese.

iii) Stabilisation of the water to reduce its corrosive character seems to be required.

iv) The treated water should be diluted with suitable water to reduce the sodium, chloride and possibly the sulphate concentrations to currently accepted levels.

v) The treatment with respect to both stabilisation and removal of iron and manganese would be ideally situated at the source (i.e. each borehole) or at least at the wellfield, to avoid corrosion and deposition of iron and manganese in the downstream conveyance systems.

The literature review on iron biofouling revealed that the causes and processes of iron biofouling and precipitation in parts of the TMG Aquifers have to be taken into account for further planning and assessments during the study. However, the factors determining the risk of clogging in the boreholes and conveyance systems are known.

For most of these factors (i.e. salinity, geology of the host rock, pH, dissolved oxygen, dissolved gases, organic carbon) the preliminary assessment and the experience from other studies indicate a less probability of iron biofouling occurring in the Peninsula Aquifer than in the Nardouw Aquifer, which was mainly utilised in the water supply schemes with high maintenance due to borehole clogging.

Detailed planning and investigation during the Exploratory Phase can significantly further reduce the risk of failure due to iron biofouling. However, clogging of boreholes and conveyance system due to iron biofouling and precipitation can be managed, if the problem is assessed in detail and continuously monitored.

A more extensive water quality sampling program is required to allow for thorough investigation into the treatment requirements of the underground water and its effect on the water system of Cape Town. The following recommendations were drawn based on the desktop review of currently available data:

i) Better groundwater chemistry data – there appeared to be uncertainty about how representative the groundwater chemistry data was of water that may be transferred in future. The typical concentrations in the TMG Aquifer water are much lower than the concentrations in the data sets supplied for the assessment. It is recommended that the estimates of the TMG Aquifer water chemistry be refined through monitoring once the exploration begins.

ii) More detailed assessment - A number of assumptions were made about average water chemistry in the inflows to the reservoirs, the in-lake water quality and the chemistry of the source water. It is recommended that the potential impacts on water chemistry be assessed in more detail when the system yield analysis is undertaken during a later phase of the project. That assessment should then use more realistic inflow rates and specifically, more site specific TMG Aquifer water chemistry data.

iii) Location of discharge and abstraction points – an implicit assumption in this assessment was that the pumped TMG water is instantly mixed with the whole reservoir. In reality this is not the case and in a more detailed assessment during a later phase of this project, the location of the discharge and abstraction points in the reservoir need to be considered to examine aspects such as the mixing zone, impacts of in-lake currents and potential short-circuiting to the abstraction point.

iv) Information on the volume of inflow from the TMG Aquifer to the in-lake reservoirs must be confirmed during the Exploratory Phase. A detailed assessment might be required for the Berg River Dam and Steenbras Dams, depending on the location of the production boreholes.

v) Since the source of the water and the host rock are determining factors for the risk of iron biofouling, the exploration phase should focus on the Peninsula Aquifer. Furthermore, hydraulic connection of the targeted aquifer to the overlying aquitard (i.e. Cedarberg Shale) and aquifer (i.e. Nardouw) should be avoided. During the borehole siting and the drilling program this factor has to be taken into account.

vi) The influence of the borehole construction material should be investigated further, as there appears to be a controversery. This will inform the wellfield costing for the Pilot Phase.

vii) Detailed and site specific assessment of the water quality (see below) is required to determine the risk of and management options for iron biofouling.

viii) If the assessment in the Exploratory Phase verifies the potential risk, a pilot trial of different treatment and management options is required prior to or as first step in the Pilot Phase of the study.

TABLE OF CONTENTS

APPENDICES

List of Tables

List of Figures

List of Abbreviations

atm	atmospheric pressure
AWSS	Atlantis Water Supply Scheme
$BART^{TM}$	Biological Activity Reaction Test
$BCHT^{TM}$	Blended Chemical Heat Treatment
CAGE	Citrusdal Artesian Groundwater Exploration
Ca	Calcium
CCTV	Closed Circuit Television
CH_4	Methane
Cl	Chloride
CO_2	Carbon dioxide
CO_3	Carbonate
DBI	Droycon Bioconcepts Incorporated
deg C	Degree Celcius
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DWAF	Department for Water Affairs and Forestry
EC	Electrical Conductivity
Fe	Iron
H_2O	Hydrogen oxide (Water)
H_2S	Hydrogen sulphide
K	Kalium
KKRWSS	Klein Karoo Rural Water Supply Scheme
M	mole
Mg	Magnesium
mg/l	milligram per liter
min	minute
Mm^3	million cubic meters
Mn	Manganese
mS/m	milli Siemens per meter
mV	milli Volt
Na	Sodium
NH_4	Ammonia
NO_3	Nitrate
O_2	Oxygen
pH	Potential hydrogen
pH_2	Pressure of hydrogen gas in water
pO_2	Pressure of oxygen in water
ppm	parts per million
PVC	Polyvinyl chloride
TDS	Total dissolved solids
TMG	Table Mountain Group
TMGA	Table Mountain Group Aquifer
TMGAA	Table Mountain Group Aquifer Alliance
S	Sulphur
SABS	South African Bureau of Standards
SADC	Southern African Development Community
Si	Silica
SO_4	Sulphate
UK	United Kingdom
uPVC	hardened Polyvinyl chloride

1. INTRODUCTION

1.1 BACKGROUND OF THE STUDY

At present the Western Cape is highly reliant on surface water resources. About 6% of the Western Cape's human-related water requirement is currently drawn from groundwater resources, and many of these groundwater resources are poorly managed, over-utilised and/or susceptible to pollution and contamination. There is very little conjunctive planning and management of ground and surface water resources. In that surface and ground water systems are interconnected, "open" systems, this lack of integrated, conjunctive management is a major shortcoming in strategic water resource management.

The TMG Aquifer project is seen against this background. It seems likely that the TMG Aquifer is capable of yielding high and sustainable volumes of good quality water, and can thus be a vital resource in the Western Cape and regional water management strategies. It should not be seen as a stand-alone project, but an integrated part of the provincial and regional strategic water resource network, to be managed accordingly. For this reason, the analysis and management of risks pertaining to the TMG Aquifer project must also be seen in the context of regional water resource management. Many of the risks associated with the TMG Aquifer project are interrelated with and can be managed so as to mitigate risks associated with surface water exploitation and regional water resource management in general.

The Table Mountain Group (TMG) Aquifer Feasibility Study and Pilot Project is staged in four discrete phases, viz.,

- Inception Phase
- Preliminary Phase
- Exploratory Drilling Phase
- Pilot Wellfield Phase

In the project proposal and the consequent report of the Inception Phase the focus of whole Preliminary Phase is summarised as "…selection of the most favourable target areas for wellfields and sites for pilot boreholes after having considered all the relevant factors and ramifications …" (TMGA Alliance, 2002)

The project is divided into eight tasks, focusing on different aspects of the overall scope, viz.:

- Task 1 – Hydrogeology
- Task 2 – Drilling and Engineering
- Task 3 – Infrastructure
- Task 4 – Ecological and Environmental
- Task 5 – Societal and Legal
- Task 6 – Risk Definition
- Task 7 – System Model
- Task 8 – Wellfield Operation

1.2 WELLFIELD OPERATION

The Task 8 - Wellfield Operation investigates the requirements for the operation of the wellfield at the selected sites. This includes:

- Aspects of water quality
- Requirements for water treatment
- Optimizing the pumping regime
- Estimating sustainable yield
- Monitoring protocols regarding optimal pumping rates

1.3 TERMS OF REFERENCE

In the preliminary phase of the study this task focuses on a desktop study on water quality aspects. According to the technical proposal the following subtasks have to be addressed:

a) Comprehensive review of biofouling and its avoidance/management strategies

b) Brief assessment of wellfield management requirements of different target zones.

1.4 WATER QUALITY IN THE TMG AQUIFER

Table 1-1 provides a breakdown of the general water chemistry characteristics expected for water abstracted from the Peninsula formation of the TMG aquifer. The water tends to be oligotrophic (low in nutrients), acidic and low in salinity, which is characteristic of water flowing through or over TMG formations.

Table 1-1 Typical water chemistry characteristics expected for water abstracted from the confined TMG aquifer. All concentrations in mg/l unless otherwise indicated (Smith et al. 2002).

	EC (mS/m)	pH	Na	Mg	Ca	Cl	SO_4	Alkalinity as $CaCO_3$	Si	K	Fe	δD (‰)	$\delta^{18}O$ (‰)
Boreholes from Nardouw Subgroup													
Mean	30.0	6.0	30.8	5.8	10.2	56.2	27.6	42.8	6.4	5.1	3.3	-45.2	-7.3
Minimum	9.2	3.1	7.2	1.5	1.3	6.1	3.2	1.0	2.1	0.4	<0.1	-53.2	-7.9
Maximum	155.0	8.3	232.8	43.1	73.4	395.2	220.5	147.3	18.1	16.2	15.4	-27.7	-6.3
Boreholes from Peninsula Formation													
Mean	10.4	6.2	11.1	1.8	3.3	18.0	5.2	14.5	3.8	0.8	0.2	-42.8	-7.3
Minimum	2.6	4.3	2.0	0.9	0.4	4.5	1.0	3.5	1.4	0.2	0.1	-51.1	-7.7
Maximum	26.3	7.6	21.2	3.2	30.4	34.1	14.0	77.9	9.4	2.3	0.2	-35.5	-7.06

Unlike the water in the Nardouw Aquifer in the Western Cape, which tends to be high in iron and salts, the water from the Peninsula Formation can be expected to be relatively pure. In contradiction results from a study in the Hermanus area show high iron content and higher EC values for water samples from the Peninsula Aquifer, probably due to the position of the waterstrikes close to the Cedarberg / Pakhuis contact (Umvoto 2002).

Since no hydrogeological exploration took place in this phase, the assessment of the water quality related requirements is based on existing datasets from other areas of the TMG. The following information was used:

- Macrochemical analysis of borehole water from both the Nardouw and Peninsula aquifer in the surroundings of Citrusdal (CAGE project, Umvoto 2000)
- Trace element analysis of borehole water from both the Nardouw and Peninsula aquifer in the surroundings of Citrusdal (CAGE project, Umvoto 2000)
- Macrochemical analysis of borehole water from both the Nardouw and Peninsula aquifer in the surroundings of Hermanus (Hermanus project, Umvoto 2002)

The data and statistical analysis of the above datasets are given in Appendix A. The main parameters are given in Table 1-2 below.

Table 1-2 Water chemistry characteristics for water abstracted from the TMG aquifer at two different places. All concentrations in mg/l unless otherwise indicated (Umvoto 2000 + 2002).

Hermanus data set

Variable	**n**	**Average**	**Min**	**Max [1]**	**25%tile**	**Median**	**75%tile**
pH	22.00	5.65	4.00	6.78	5.35	5.65	6.10
EC mS/m	26.00	53.32	33.00	108.00	39.25	45.30	58.75
TDS mg/l	17.00	358.29	211.00	691.00	243.00	358.00	378.00
Ca mg/l	17.00	5.74	2.90	11.00	4.40	5.40	5.80
Mg mg/l	17.00	10.47	6.90	23.00	7.80	8.70	12.00
Na mg/l	17.00	76.59	39.00	182.00	49.00	76.00	79.00
Cl mg/l	17.00	135.88	80.00	249.00	93.00	127.00	150.00
SO^4 mg/l	17.00	31.60	8.50	117.00	13.00	16.00	23.00
Fe mg/l	23.00	4.80	n.d.	12.90	3.15	3.90	5.55
Mn mg/l	23.00	1.34	0.04	2.50	0.74	1.40	1.75

Peninsula data set

Variable	**n**	**Average**	**Min**	**Max [2]**	**25%tile**	**Median**	**75%tile**
Na	35	168.27	10.3	1251.1	55.9	103.6	172
Mg	35	30.31	1.5	304.5	7.5	16.5	34.85
Ca	35	20.69	1.1	229.2	3.55	7.1	15.3
Cl	35	360.75	18.8	2972.5	104.1	206.7	366.05
SO^4	35	28.35	1.7	192.4	8.15	14.8	24.9

n – Number of samples
n.d. – not detectable
1 – Maximum values influenced by proximity to sea and industrial sites
2 – Maximum values influenced by connection to other aquifers, such as Bokkeveld

2. REVIEW OF IRON BIOFOULING

2.1 INTRODUCTION

Iron biofouling of boreholes and aquifers presents a threat to the sustainability of many groundwater supply schemes in South Africa and internationally. In the Western Cape, iron fouling has had a severe economic impact on both the Atlantis Water Supply Scheme and the Klein Karoo Rural Water Supply Scheme. The concern that this problem may arise in the new wellfield(s) in Table Mountain Group aquifers, which the City of Cape Town is investigating as an option for bulk supply, must be addressed. However, the above-mentioned wellfields target different aquifers with different lithological settings. An understanding of the causes and controlling factors in iron biofouling under local conditions is an area of research that can guide informed decisions regarding the management and rehabilitation of affected aquifers.

In many aquifer systems worldwide, hydrochemical and biogeochemical reactions involving iron have been a limiting factor in the sustainability of groundwater schemes. The presence of iron in aquifers may become problematic as a result of:

- high concentrations of dissolved iron. This affects the potability of the groundwater and often requires treatment before reticulation.
- “iron biofouling”, i.e. precipitation of iron oxyhydroxides and development of associated microbial biofilms. These clog boreholes and aquifer formations, causing losses in production capacity.

Iron-related problems are often accelerated by poor wellfield or aquifer management. Given the high costs of water treatment, borehole construction and rehabilitation of clogged boreholes, inadequate understanding of iron biogeochemistry may have important financial implications for groundwater schemes.

2.2 HIGH CONCENTRATIONS OF DISSOLVED IRON

2.2.1 Sources of iron

Iron is a major element in the earth’s crust and is found as a major component in almost all rock types. It occurs naturally as native Fe and in a wide range of minerals. Iron in minerals generally occurs in the reduced ferrous form in silicates, carbonates and sulphides and may be slowly released to solution as these minerals weather. Sources of dissolved iron in groundwater include dissolution of Fe(II)-bearing oxide and sulphide minerals, such as magnetite, ilmenite, pyrite, siderite, and Fe(II)-bearing silicates, including amphiboles, pyroxenes, olivine, biotite, glauconite and clay minerals (Appelo and Postma, 1996). Reduction of Fe(III) oxyhydroxides in aquifer sediments is also an important source of dissolved Fe^{2+} in groundwater. A considerable amount of iron is present in aquifers as FeOOH coatings on the surface of quartz and feldspar grains.

2.2.2 Controls on iron dissolution

Most iron-bearing minerals have low solubility in water in natural environments, which prevents the accumulation of high concentrations of dissolved iron. The solubility of iron is controlled by pH and redox geochemistry and iron may exist in solution as stable dissolved Fe^{2+} or Fe^{3+} under certain conditions. Other factors controlling iron solubility include the presence of sulphur and carbonate species in the system. Favourable conditions for iron dissolution include very low pH, reducing E_h and the absence of dissolved sulphide. Dissolved iron as Fe^{2+} is a common constituent of anoxic groundwater, since ferrous iron is the more soluble form in the normal pH range between 5 and 8 in groundwaters. Ferric iron (Fe^{3+}) is only soluble at low pH. Iron is rarely found at high concentrations in surface water.

Relationships between pH, redox potential and iron species were presented by Garrels and Christ (1965) and Eh-pH diagrams such as those in Figures 2-1 and 2-2 have become standard features in most aqueous geochemistry textbooks (e.g. Stumm and Morgan, 1996; Appelo and Postma, 1996; Drever, 1997). These predict the stability of various inorganic iron phases in solution, based on thermodynamic principles. Such diagrams must, however, be interpreted with caution, since they are only valid for the systems and concentrations used for their construction and are seldom representative of the complexities found in natural environments (Nealson, 1983).

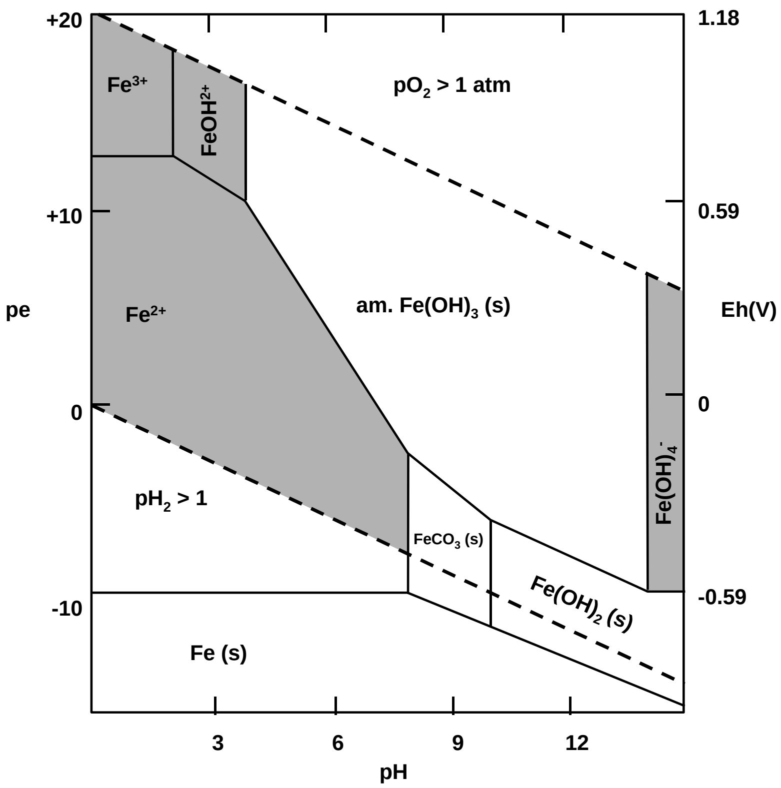

Figure 2-1 Eh-pH diagram for the system Fe-O-H2O-CO2 at 25ºC with solid phases amorphous ferric hydroxide (am. Fe(OH)3), Fe(OH)2, siderite (FeCO3), and native iron (Fe). The diagram was constructed with total carbonate species concentration = 10-3 M and total iron = 10-5 M. Stability fields for dissolved iron species are shaded (After Stumm and Morgan, 1996).

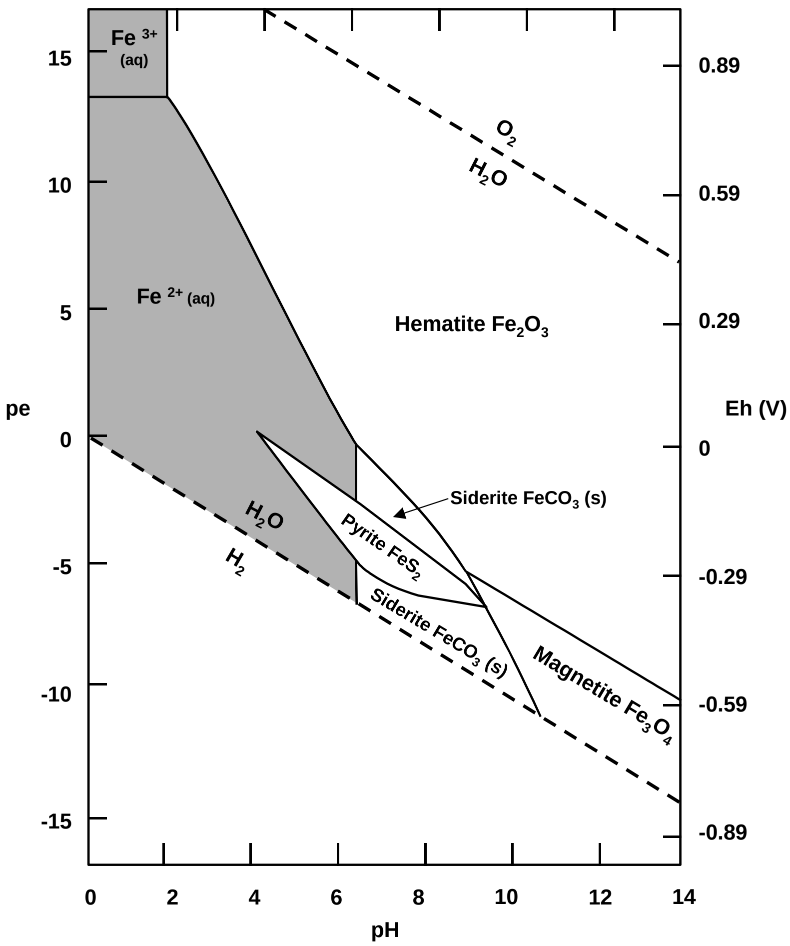

Figure 2-2 Eh-pH diagram for the system Fe-O-H_2O-S-CO_2 at 25ºC with solid phases hematite, magnetite, siderite and pyrite. The diagram was constructed with total carbonate species concentration = 10^0 M, total sulphur = 10^{-6} M and total iron = 10^{-6} M. Stability fields for dissolved iron species are shaded (After Garrels and Christ, 1965).

The diagrams show that iron dissolution is favoured by the reduction of Fe(III) species to Fe(II), since the Fe^{2+} ion is stable under a much greater range of pH. Iron reduction reactions may be abiotic, or they may be mediated by micro-organisms. Iron reducing bacteria use iron (III) hydroxide minerals as electron acceptors in the absence of free oxygen. Where reducing conditions prevail, the oxygen in the iron hydroxides is used by the bacteria to oxidise organic carbon or other reduced species and in the process Fe^{2+} is released into solution. Reductive dissolution of iron hydroxides becomes important when an electron donor such as dissolved organic matter, H_2S or CH_4 moves into the aquifer.

The concentration of Fe^{2+} in groundwaters is most often limited by the reverse of this reaction, i.e. the oxidation to Fe^{3+} and precipitation as oxyhydroxides, but the low solubility of siderite and pyrite are also important controlling factors at high Fe^{2+} concentrations. Pyrite has extremely low solubility and will precipitate virtually all iron from solution, provided enough dissolved sulphide is available.

The iron dissolved by reductive dissolution of Fe(III) hydroxides may be re-precipitated nearby as pyrite if sulphate reduction follows iron reduction in the aquifer. Once the sulphide is depleted, iron may also precipitate as siderite in waters with high alkalinity.

The dissolution of Fe(II)-bearing silicates (e.g. detrital amphiboles and pyroxenes) is considerably faster under anoxic than oxic conditions. Kinetic and mechanistic studies have shown that the rate of dissolution decreases with exposure time in an environment where free oxygen is present, because of the coating of the mineral surface with an inner layer of Fe(III) silicate and an outer layer of FeOOH, a phenomenon that is sometimes referred to as “armouring”. Diffusion control of reactants through the surface coating inhibits the dissolution reaction (Appelo and Postma, 1996).

2.2.3 Effects of high dissolved iron concentrations

Iron seldom occurs in groundwater at levels that would pose a health threat to users, but operational and aesthetic effects of high iron concentrations may be a limiting factor in the use of the water. Dissolved iron concentrations up to several milligrams per litre can exist in groundwater without discolouration or turbidity when first brought to the surface, but over time, the iron will precipitate out, giving the water an undesirable reddish-brown colour. Iron also imparts an astringent taste to the water at a very low taste threshold of 0.3 mg/l. Iron in water causes aesthetic problems such as staining of laundry, walls and plumbing fixtures at levels above 0.2 mg/l, and it is generally desirable to remove dissolved iron from groundwater to a level of 0.1 mg/l immediately after abstraction (Mackintosh and de Villiers, 2002).

At very high levels, exceeding 10 to 20 mg/l dissolved iron, chronic health effects may be observed in young children. Excessive ingestion of iron may result in haemochromatosis, wherein tissue damage occurs as a consequence of iron accumulation. This is generally a result of long-term consumption of acid foodstuffs cooked in iron utensils, rather than from water. Iron poisoning from drinking water is extremely rare because excessively high concentrations of iron do not occur naturally in water. Lethal toxicity arises at levels above 3000 mg/l (DWAF, 1996).

2.2.4 Removal of dissolved iron

The removal of dissolved iron is a relatively simple process that is well understood and documented. All methods oxidise the relatively soluble Fe(II) to insoluble Fe(III), either by simple aeration or chemical dosing with oxidants such as chlorine, potassium permanganate, ozone or chlorine dioxide. This is followed by conventional coagulation/flocculation and sedimentation in a settling tank before final filtration. Soft, acidic groundwaters with low alkalinity require an additional dosing of alkali before oxidation to assist iron removal. Oxidation rates are slow at low pH and Fe(II) requires a pH above 7 to be oxidised at a reasonably rapid rate (Figure 2-3).

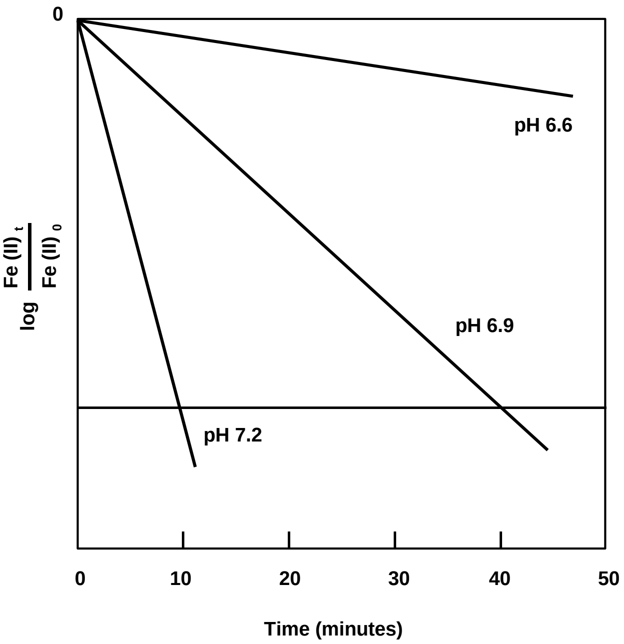

Figure 2-3 Rates of oxidation of iron in contact with atmospheric oxygen at various pH values (Van Duuren, 1997).

Aeration is simpler and less expensive than chemical dosing and has the added benefit of allowing CO_2 to degas, which also increases the pH. The reaction product, Fe(III), catalyses the oxidation reaction and improved efficiency of removal can be obtained by contacting the aerated water with Fe(III) in, for example, filter media (Mackintosh and DeVilliers, 2002).

2.3 IRON BIOFOULING

Iron biofouling refers to bacteriologically assisted precipitation of iron. It incorporates both the formation of mineral precipitates containing ferric iron (scaling) and the growth of extensive microbial mats (fouling) in boreholes and aquifer materials. Iron oxidation is always a chemical process, but biological clogging or biofouling is distinguished from chemical iron oxidation by far more rapid and severe clogging. The deposition of chemical precipitates can be initiated by changing conditions of pH (shifts to alkaline range), the presence of oxygen, or increases in the redox potential. Mineral solids and bacterial biomass accumulate over time, eventually plugging borehole screens and the pore spaces of the aquifer materials.

Micro-organisms attach themselves directly to surfaces in and around the borehole where they make use of groundwater constituents, such as nutrients, that flow into the borehole. In the process, they modify the chemical environment in their immediate vicinity and can cause side-effects such as slimy coatings or corrosion of pumps and casings (Rohde and Keevill, 2000).

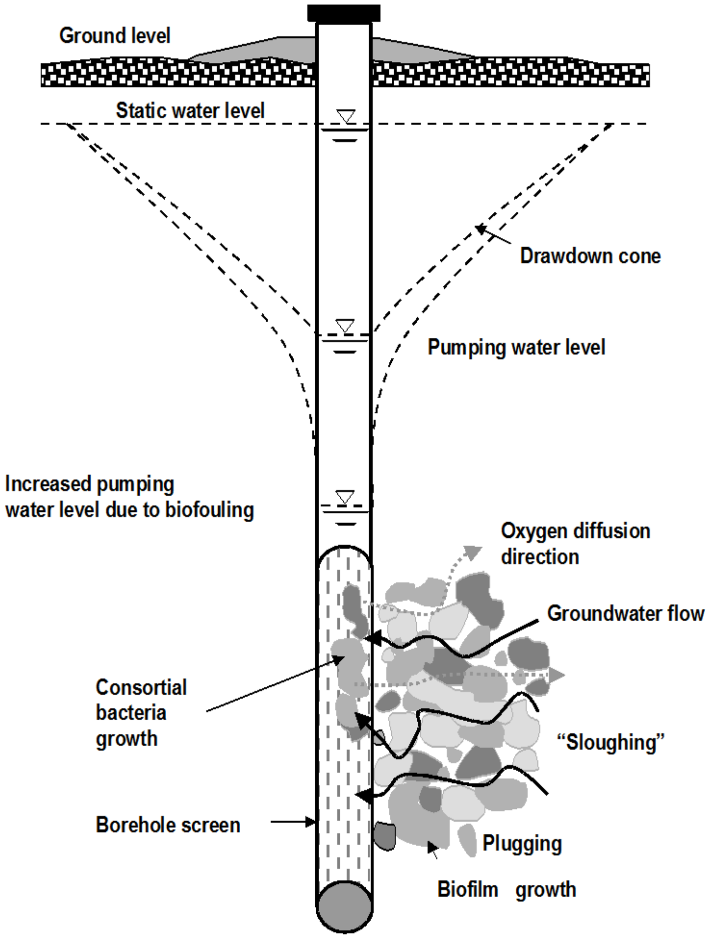

Figure 2-4 Effects of iron biofouling in a production borehole (After Rohde and Keevill, 2000).

Scaling and/or fouling generally occurs at the oxic-anoxic interface (redox front) where the greatest microbial activity occurs, e.g. when anoxic water, rich in Fe(II), flows into an oxic zone. Iron (II) is oxidised rapidly at circumneutral pH when anoxic groundwater becomes aerated. Iron biofouling reactions can be induced by human disturbance of the system during drilling and pumping, or occur by purely natural biogeochemical interactions that take place within the aquifer when there is recharge with oxygenated water, discharge at streams or seasonally fluctuating water tables.

Oxidation causes precipitation of iron (III) oxyhydroxides, which may rapidly clog aquifer pore spaces, borehole screens and pumps and distribution systems. Iron and sulphur bacteria in the aquifer catalyse redox and precipitation reactions and biocatalysed iron precipitation by *Gallionella*, *Sphaerotilus* and *Thiobacillus ferrooxidans* bacteria is well known. The bacteria reduce oxygen and oxidise Fe(II) to Fe(III) by a reaction that generates energy to support growth. The formation of iron (III) oxyhydroxides quickly follows (Perez-Paricio, 1998). Table 2-1 summarises some of the features of iron related bacteria in aquifers. Rapid rise and fall in water level due to an intermittent pumping regime can enhance both chemical iron oxyhydroxide precipitation and biofilm growth, by increasing oxygen diffusion to the aquifer.

Table 2-1 Comparison of features of iron-oxidising and iron-reducing bacteria (After Tyrrel and Howsam, 1997).

	Carbon source	Energy source	Use of iron	Redox environment	Iron problem
Chemoautotrophic iron oxidisers e.g. *Gallionella, T. ferroxidans*	Inorganic	Oxidation of Fe(II)	Energy (electron donor)	Oxic/anoxic interface	Iron biofouling
Chemoheterotrophic iron oxidisers e.g. *Leptothrix, Crenothrix*	Organic	Oxidation of organic carbon	Unclear	Oxic/anoxic interface	Iron biofouling
Chemoheterotrophic iron reducers	Organic	Oxidation of organic carbon	Energy (electron acceptor)	Anoxic	High dissolved iron

2.4 SITUATION ASSESSMENT

2.4.1 Local occurrences of iron-related problems

Iron biofouling is a widespread problem in South Africa, particularly in the Western and Eastern Cape. Iron-related problems seem to be common in the Table Mountain Group (TMG) Aquifers, specifically in boreholes, which target the Nardouw Subgroup in the Gouritz Water Management Area (WMA), and both the Peninsula and Nardouw Groups in the Eastern Cape (Tredoux and Smart, 2002). It is noteworthy that the lithology of these aquifers and the geological settings at the wellfields differ from the situation of the TMG Aquifers in the Breede and Berg WMAs; especially of the selected Target Site Areas. Groundwaters in the Malmesbury Group in the Western Cape and the Natal Group sandstones in KwaZulu Natal are also commonly affected. Boreholes in alluvial deposits may also suffer from iron-related problems.

Jolly and Engelbrecht (2002) cite the following examples of areas in TMG aquifers affected by iron biofouling in South Africa:

- St. Francis Bay (Cohen and Wood, 2001)
- Steytlerville (Jolly and Welman, 1999)
- Plettenberg Bay (Jolly, 1998a)
- Walboomskraal, inland of George (Miller, 2000)
- Calitzdorp (Jolly, 1998b)
- Cape Agulhas (Toens and Associates, 1991)
- Arabella, Kleinmond (Parsons and Associates, 2000)
- Clovelly Country Club, Cape Town (Maclear, 2000)

Most of these wellfields target aquifers from the Nardouw Formation, which have naturally higher iron content in the host rocks (see below). The Peninsula aquifer consists of quartzite and contains far less iron (see below for more detail).

At the Klein Karoo Rural Water Supply Scheme (KKRWSS) in the Gouritz WMA, wellfields near the towns of Dysselsdorp, Calitzdorp and Oudtshoorn have been affected by iron biofouling (Shand and Little, 1999; Kotze *et al*., 2000). At least twelve boreholes have been rehabilitated, some on more than one occasion. Similarly at the Atlantis Water Supply Scheme (AWSS) thirty-seven shallow boreholes have had to be treated for biofouling since 1999.

2.4.2 International examples of iron biofouling

Biofouling and iron-related problems occur globally and extensive work has been conducted on identifying the problem, understanding the causes of biofouling and managing its effects on several continents. Although the list is by no means comprehensive, a few examples of these investigations are mentioned here:

- **Canada:** Extensive biofouling of community water supply and agricultural boreholes has been found, especially in the Canadian Prairies, and many of the remediation methods in current use worldwide were developed in this country (Rohde and Keevill, 2000; Cullimore, 1993; Gehrels and Alford, 1990).

- **United States**: A survey of 154 Water Utility Companies supplying groundwater in the south central and south Atlantic regions found that 39 Utilities (25%) were affected by iron bacteria and 12 (8%) by iron/manganese clogging (Smith, 1990). Several production boreholes in Suffolk County, Long Island , New York were affected by iron encrustation, when high pH municipal water was flushed down boreholes to clean the pumps (Walter, 1997).

- **United Kingdom:** The Triassic sandstone aquifer of the Otter Valley that supplies South West Water's public supply boreholes is reportedly affected by iron biofouling (Bowen, 1990). The Lower Greensand aquifer in Southern England and the Isle of Wight also contains high iron content (Banks, 1990; Packman, 1990).

- **Australia:** Microbially assisted borehole clogging has been reported in Western Australia (Ralph and Stevenson, 1995), New South Wales (McLaughlan et al., 1993) and South Australia (Forward, 1994).

- **Argentina:** Alcade and Gariboglio (1990) report widespread biofouling in boreholes in the Rio Negro province.

- **Peru:** Municipal supply boreholes for the city of Lima are located in an alluvial fan aquifer which has been affected by biofouling (Puri et al., 1989).

- **Eastern Europe:** Barbic and Savic (1990) looked at the effects of chemical regeneration on iron bacteria and ochre deposits in tube-wells in Yugoslavia.

- **Germany:** A survey of 12 000 boreholes used for water supply revealed that 3000 boreholes had been rehabilitated in the previous 5 years, of which 87% were affected by iron or manganese oxide encrustation (Houben, 2003a).

The volume edited by Howsam (1990) also includes case studies from alluvial aquifers in Pakistan (Indus Plain), India (Eastern and Northeastern regions), France (Loire Valley) and Zambia (Zambesi flood plain).

Problems of high iron concentrations in drinking water have also been reported from many sites in developing countries where shallow wells are used for rural water supply such as the Philippines (Fujita et al., 1990), Bangladesh and Uganda (Tyrrel et al., 2001).

2.5 RATES OF IRON BIOFOULING

Biofouling rates are site specific and thus dependant on prevailing environmental conditions, as well as wellfield management and infrastructure. Factors affecting the rate of biofouling include aquifer materials, water composition, flow dynamics, temperature and dissolved oxygen concentrations. Growth rate (doubling) of bacteria is also a determining factor in the rate of clogging.

General consensus among those investigating iron biofouling problems is that the timescales for clogging to occur after borehole installation or rehabilitation may be in the order of a few months. Gehrels and Alford (1990) report that nuisance bacteria caused a community water supply well in Northern Ontario to steadily lose its efficiency during a 15-month period. According to Jolly and Engelbrecht (2002), “clogging (in the Table Mountain Group, South Africa) can be extremely rapid, reducing borehole yields … within months of installation”. They cite an example of a borehole with an initial iron concentration of 11.5 mg/l and pH 5.9. Only 2 years after installation, the pH had dropped to 3.3 and iron concentration increased to 63 mg/l. In some cases the symptoms of borehole deterioration may not be apparent until the yield is severely impaired (Tuhela et al., 1997).

Ferrous iron in groundwater is readily oxidised in the presence of dissolved oxygen. Iron oxidation is usually rapid and is sensitive to pH and oxygen concentration as described by the following equation (Stumm and Morgan, 1996):

$$\frac{dFe(II)}{dt} = kFe(II)(OH^-)^2\, pO_2$$

where k = 8±2.5 x 10^{13} min^{-1} atm^{-1} $mole^{-2}$ at 20ºC. In fully aerated water at pH 7, the half-life of Fe(II) oxidation is 15 minutes, and under these conditions, iron bacteria may not be able to compete with chemical reaction rates (Emerson and Moyer, 1997). With an initial Fe(II) concentration of 10 ppm, pO_2 of 0.05 atm and at a pH of 6, five grams of Fe(III) would take about 2000 days to be produced, provided the system contained sufficient Fe(II) (Ralph and Stevenson, 1995). The rate constant, k, is affected by temperature (rates increase with increasing temperature) and by ionic strength.

Ralph and Stevenson (1995) conducted laboratory experiments to compare abiotic and biotic rates of iron oxidation. They found that rates were significantly enhanced when Fe(II) solutions were inoculated using ochreous borehole drilling sludges from Western Australia over those for sterile solutions. In general, kinetic estimates for simulated groundwater conditions suggest that abiotic rates of Fe(II) oxidation are not sufficient to cause the rapid blocking of screens evident at field study sites (Tuhela et al., 1997). Rates of biological reactions can be 100 000 times greater than rates of chemical clogging alone (Jolly and Engelbrecht, 2002).

2.6 CONTROLLING FACTORS IN IRON BIOFOULING

2.6.1 Temperature

Temperature has an effect on all chemical reactions, with the rates of most reactions increasing as temperature increases (Chapelle, 1993). Abiotic iron oxidation rate constants, for example, increase with increasing temperature and it might be expected that faster clogging will occur in South Africa's relatively warm groundwater than reported for other countries such as Canada, where much of the iron biofouling research has been conducted.

There are also other chemical reactions, which are temperature sensitive, that may indirectly affect the rate of iron precipitation. Temperature affects the solubility and diffusion rate of O_2, for example. Dissolved oxygen concentrations decrease as temperature increases, which could decrease the amount of iron oxidation. Temperature and ionic strength of groundwater affect the CO_2 and $CaCO_3$ solubilities, and hence the buffering of pH of the system (McLaughlan and Knight, 1989). Temperature also affects the activity and behaviour of microbial communities (see below).

2.6.2 Salinity

Salinity affects the ionic strength and hence the rate of iron oxidation. Salinity and pH are environmental factors affecting the cardinal temperature for the growth of a bacterial community. Very high salinity levels are toxic to some bacteria. A study in Yugoslavia found that iron rich waters were the most saline waters for the area investigated by Barbič and Savić (1990). Although salinity is recognised as a major factor in borehole deterioration in terms of corrosion (McLaughlan, 2002), there appears to have been little work conducted on the influence of salinity on iron biofouling. The role of groundwater salinity and salinity changes in cases of iron biofouling may be an area where further research could contribute. However, in the context of this study for possible bulk water supply it is not of relevance, due to the low salinity levels in the Peninsula Formation.

2.6.3 Parent Geology

Parent geology plays an important role in that it provides the source of iron (See Section 2.2.1). As an example, iron problems in the Table Mountain Group aquifers are more commonly reported in boreholes targeting the Nardouw Subgroup than in the Peninsula Formation (Smith et al., 2002). Although both parent geologies are quartzitic, the Nardouw rocks have a higher natural concentration of iron than the Peninsula rocks; especially the members of the Nardouw Formation, which contains layers of siltstone and sandstone, viz. the Goudini and to lesser extent the Rietvlei, have naturally higher iron content in the host rock.

The abundance of iron in the parent geology is not the only determining factor. Aquifers with low total iron content may still be affected by iron biofouling if environmental and or wellfield management conditions favour the mobilisation of iron from the rocks or sediments. Furthermore, if an aquifer with low iron content is hydraulically connected to iron-bearing rocks upstream (e.g. Cedarberg Shale, Nardouw Aquifer), iron mobilisation from these formations can result in increased iron concentrations in the groundwater. The findings from the Hermanus study confirm that water from the Peninsula close to the Cedarberg Shale contact and or the fault structure has higher iron content (Umvoto, 2002).

Iron is dissolved from iron-containing formations under anaerobic or acidic conditions, while a rise in pH or Eh at the source of oxygenated water will initiate chemical precipitation of iron (Tyrrel and Howsam, 1990). A common process that releases dissolved iron is the microbially-catalysed oxidation of accessory sulphide minerals such as pyrite.

In the Table Mountain Group, iron appears to be related to pyrite and illite in the more argillaceous sediments, including the Cedarberg shale formation (Smith et al., 2002). Magnetite may also be a source of iron in the TMG, but its presence has not yet been confirmed (M. Smith, *pers. comm*.). Detailed logging of drill chips and borehole cores can indicate mineralisation of the fracture zones, when signs of iron and manganese deposition are detected. Shales of the Bokkeveld Group and the Nardouw Subgroup (primarily the Goudini Formation) could also contain sources of carbon that support for bacterial growth (Jolly and Engelbrecht, 2002).

2.6.4 Organic Carbon

Environments with oxidized iron minerals and organic debris provide favourable conditions for iron reduction and high concentration of ferrous iron in solution (Jolly and Engelbrecht, 2002). Organic carbon also provides a substrate for microorganisms and an electron acceptor for microbially-mediated redox reactions. When organic carbon is present in excess, dissolved oxygen in the aquifer is rapidly depleted and reduction of ferric hydroxides may become the electron donating half reaction in the decay of organic matter. The iron reduction reaction may be catalysed by bacteria that utilise organic carbon as an energy source (Loveley and Phillips, 1988).

Dissolved organic carbon compounds, such as soluble humic and fulvic acids, which have reducing and complexing properties, may react with metals in groundwater systems (McLaughlan and Knight, 1989). Organometallic complexation has a great effect on the solubility of Fe and Mn compounds, which is often not taken into account in conventional solubility diagrams or modelling calculations. The fact that accretion of Mn-compounds is observed before Fe-compounds, has been attributed to the greater stability of Fe-organo-metallic compounds (McLaughlan and Knight, 1989). The organic part of the organo-Fe compounds is often utilized by microorganisms during biological oxidation of Fe (II), during which the Fe is deposited or released (McLaughlan and Knight, 1989).

The quantity of organic carbon available also influences the composition of microbial communities as reported by McLaughlan *et al.* (1993) from an extensive investigation of Australian biofouling deposits. They found that two main groups of deposits were present: 1) biofilms dominated by the stalked bacteria such as *Gallionella* sp. and 2) biofilms mainly comprising heterotrophic bacteria. Biofilms with dominant stalked bacteria were associated with a lower amount of organic carbon in the aquifer formation. *Gallionella ferruginea* are chemoautotrophs that oxidise iron. These bacteria use Fe(II) as an energy source, and in the process, convert inorganic CO_2 or bicarbonate into bacterial biomass (Tuhela *et al*., 1997). *Thiobacillus ferrooxidans* are also chemoautotrophic bacteria that can oxidise both sulphide and ferrous iron under acidic conditions without an organic carbon source. When organic carbon is available, iron may be oxidised by chemoheterotrophic bacteria such as *Leptothrix* and *Crenothrix* (Tyrrel and Howsam, 1997).

Dissolved organic carbon (DOC) in groundwater serves as an inhibiter of crystallization when iron oxides precipitate. If DOC and silicate ions are present in solution, Fe is precipitated as ferrihydrite rather than lepidocrocite. The presence of DOC also retards the transformation of ferrihydrite into the more thermodynamically stable form, goethite (Tuhela *et al.,* 1997).

2.6.5 Dissolved Oxygen

Oxygen is seen as one of the key factors affecting biofouling, as it is responsible for both the growth of aerobic iron precipitating bacteria and the oxidation of soluble ferrous to insoluble ferric iron. An increase in oxygen concentrations leads to an increase in Eh, and this initiates iron precipitation at the aerobic/anaerobic interface (Tyrrel and Howsam, 1990; Tuhela *et al*., 1997). When a source of circumneutral pH, anoxic water that is rich in ferrous iron, flows into an oxic zone, bacterial growth will develop at the oxic/anoxic interface (Emerson and Revsbech, 1994). This is specifically relevant for the Nardouw Aquifers with pH values around 6 to 7. The pH in the Peninsula Aquifer is often less (between 4 and 6), which results in less bacterial growth (see below).

Many iron-oxidising bacteria, e.g. *Gallionella*, are microaerophillic, i.e. they prefer a narrow band of low dissolved O_2 concentrations (Tyrrel and Howsam, 1997). The lower limit of dissolved oxygen required for bacterial activity is 0.05 mg/l (Engelbrecht and Jolly, 1999). Ralph and Stevenson (1993) view dissolved oxygen (DO) as a limiting factor in biotic oxidation reactions during which oxygen acts as an electron acceptor. Oxygen is, however, required in small amounts for Fe (II) to remain mobile within aquifers. Engelbrecht and Jolly (1999) found that DO was lower for boreholes in which biofouling or biological activity occurs, while areas with limited biological activity had high DO values, as found in their study of the Klein Karoo Rural Water Supply Scheme.

The pumping rate influences the amount of oxygen introduced to a system. When pumped at a high rate, an increase in dissolved oxygen affects the redox reactions and hence the Fe concentrations (Emerson and Revsbech, 1994). The pumping rate also has some effect on the amount of nutrients available in the subsurface at any one point. They can therefore either limit or enhance bacterial growth.

In a study in Exeter, UK, clogging of a well by precipitation of iron oxide was ascribed to pumping of the borehole (Bowen, 1990). The author explains that pumping the borehole induces upward leakage of low pH water, grossly unsaturated with respect to iron, into an area with greater iron content. The iron is then reoxidized as the pH rises to form a colloidal ferric oxyhydroxide precipitate.

Tyrrel and Howsam (1990) and Jolly and Engelbrecht (2002) recommend that boreholes should be operated continuously at lower pumping rates, rather than in a stop-start pumping regime, to minimise biofouling problems. Pumping water levels should not be allowed to fall below the depth of the screen, and pumps should be used in such a way that flow is as low as possible in all parts of the system (i.e. aquifer, borehole, pump, pipeline). This will aid in avoiding oxygenated water from mixing with Fe-containing water in the borehole. Flow velocities are the greatest in the pump, and hence any flow-related enhancement of biofouling would be greatest at the inlet, within the pump bowls and up the riser (Tyrrel and Howsam, 1990). The high flow velocity supplies large amounts of nutrients and oxygen to the already established bacteria in the system, which might cause serious and costly damage to pumps.

The effects of continuous pumping in supplying a steady stream of nutrients and reagents to the microorganisms in biofilms, and the possibility of “starving” the microorganisms by altering pumping regimes, has apparently not been investigated. Emerson and Revsbech (1994) state that sites at which active flow over the microbial “mat” occurred at a high rate were ones that showed slow accretion rates and associated unicellular microbes were predominant, while slower flow rates favoured accretion. Intermittent bursts of very fast pumping could possibly detach biofilms from screens and casing.

2.6.6 pH

Fe (II) is oxidized at 100-200mV at near neutral pH i.e. pH 7. In fully aerated water at this pH, chemical oxidation rates are extremely rapid without biological catalysis (Emmerson and Moyer, 1997). The higher the pH, the more rapid the abiotic oxidation of Fe (II). When the pH is lower, abiotic oxidation occurs at lower rates and biological oxidation becomes more important (Tuhela et al., 1997).

The solubility of ferric species is below 0.01 mg/l at a pH above 4.8. Fe (III) rapidly hydrolyses and is precipitated at pH 5 to 7, which is typical of most groundwater systems. Both ferrous and ferric iron are soluble at low pH values, such as those found under acid mine drainage conditions, and high concentrations of dissolved iron may develop at pH 2 and below. Decreasing pH and increasing dissolution of pyrite causes increased dissolved Fe concentrations in groundwaters in the Table Mountain Group, South Africa (Tredoux and Smart, 2002).

In natural groundwater systems, pH does not present a significant problem to the development of bacteria, but polluted water where the pH can often be as low as 3.0 or as high as 11 presents a great deal of stress on bacterial growth (Chapelle, 1993). Some iron bacteria thrive in conditions of neutral pH, while others are capable of oxidizing Fe under acid conditions (pH 0.5 – 4). In the case of acid mine drainage, low pH and high sulphate environments occur, which are very selective for acidophilic S and Fe-oxidizers (Tuhela *et al.*, 1997). At a pH of less than 5.5, biofouling is retarded, since micro-organisms do not grow freely under these conditions (Engelbrecht and Jolly, 1999). The pH in the Peninsula Aquifer varies normally between 4 and 6.5, with minimum values found of pH 3. The pH in the Nardouw Aquifers are normally around 6 to 7 and, hence, the aquifer is more vulnerable to iron biofouling.

Bacterial growth may cause modification in Eh-pH conditions, which results in oxidation of iron to its insoluble state. Low Eh and pH conditions lead to reduction of iron, but the bacterial iron reduction process is inhibited by the presence of nitrate, which is the preferred electron acceptor in such cases (Nealson, 1983).

2.6.7 Light (photochemical reactions)

Both iron and manganese have been noted to participate in photochemical reactions, but no references to these processes were found in the iron biofouling literature. These reactions are not relevant in groundwater systems and would only play a role once the water was brought to surface.

2.6.8 Pressure

Pressure in the groundwater system is a variable which has been given little consideration in the iron biofouling literature. Ralph and Stevenson (1993) give one account of the effects of gas partial pressure in a laboratory experiment. They observed that at pH 5.8 their experiment was reacting as expected, however at higher pH (6.0 and 6.3), the experiments were proceeding slower than anticipated. They found that partial pressure differed in different parts of the system and attributed slower reaction rates to lack of control over the partial pressure of oxygen at very low levels. Pressure changes during pumping may result in the release of CO_2 gas (McLaughlan and Knight, 1989) and might affect partial pressures of oxygen, which is an important factor in the iron oxidation rate equation (See Section 2.6.5).

2.6.9 Other dissolved gases e.g. CO_2, H_2S, and CH_4

Carbon dioxide acts as a source of carbon to chemoautotrophic bacteria in groundwater systems where organic material is absent (Tuhela *et al*., 1997) and increased availability of CO_2 or HCO_3^- could promote iron oxidation by these micro-organisms. The CO_2 equilibrium system also acts as an important pH buffer in groundwater systems, which may have indirect effects on iron biofouling. Groundwater rich in iron and CO_2 is suitable for precipitating Fe^{3+} (as ferric hydroxide) without bacterial intervention (Engelbrecht and Jolly, 1999). Dissolved CO_2 also plays a role in controlling the solubility of the iron carbonate mineral, siderite.

Reduced gases such as H_2S and CH_4 moving into an aquifer can trigger the reduction of iron hydroxide minerals and increase concentrations of dissolved Fe (Appelo and Postma, 1996).

2.6.10 Borehole construction materials

Design problems and poor performance are not uncommon in boreholes used for water supply or for monitoring and contaminant plume control. Borehole design presents problems if the borehole is not correctly matched to the chemical and physical properties of the aquifer. The shape of slots as well as the material that the screen is composed of can affect the rate of clogging (McLaughlan and Knight, 1989).

The problem of corrosion is aggravated by poor material selection for installations (Tuhela *et al*., 1997). Tyrrel and Howsam (1990) recommend that corrodible materials should be avoided and replaced by materials such as glass-reinforced plastics and epoxy. Clogging rates appear to differ between PVC, iron and stainless steel screens. Jolly and Engelbrecht (2002) claim that deposition across PVC screens is more even, while more tuberculation tends to occur on stainless steel and steel screens. In general stainless steel wells plug faster than PVC screens, with plugging of stainless steel materials being observed to occur up to 63% faster under the same environmental conditions (Jolly and Engelbrecht, 2002).

Contradictory to this finding are the results from the Atlantis Water Supply Scheme, where the problem of iron biofouling has diminished since the uPVC casings were replaced with Johnson stainless steel screens. One of the reasons might be that the angled slots in uPVC create a funnel with a high degree of roughness, while the cuts in the Johnson screens are perpendicular to the edge and possibly smoothed.

3. WATER TREATMENT REQUIREMENTS

The expected water treatment requirements are reviewed from different perspectives:

a) Water quality and water treatment requirements

b) Treatment in existing Water Treatment Works

c) Effect on water quality in existing reservoirs (dams)

3.1 WATER QUALITY

3.1.1 Introduction.

The available data, as mentioned in Section 1.4 and Appendix A, are reviewed against the SABS 241 Class 1 standard. The classes are defined as Class 0 being ideal, Class 1 being acceptable and Class 2 being the maximum allowable with a qualification to the maximum period over which this quality could be consumed.

3.1.2 Hermanus Data

Below is a brief description of the water quality, based on the data from Hermanus:

- The water has a fairly stable pH generally in the range 5,5 to 6,5 with outliers of 4 and 7.
- Calcium is in a tight range from 4 to 11 mg/l and alkalinity from 2 to 10 mg/l as $CaCO^3$.
- Magnesium is in the range 8 to 16 mg/l with an outlier of 23 mg/l.
- Sodium is relatively high from 50 to 100 mg/l with a high value of 182 mg/l recorded.
- The chloride concentration is high being in excess of 100 mg/l, values up to 180 mg/l have been recorded.
- Sulphate concentrations are generally below 30 mg/l, however, a high of 117 mg/l has been recorded.
- The only metal concentrations measured are iron and manganese, both are high. A minimum iron concentration of 3 mg/l and a maximum of 12,9 mg/l have been recorded. Manganese concentrations are generally in the range 0,6 to 1,5 mg/l.

The water, in its raw untreated state will be corrosive to metals and aggressive to concrete. It is likely that the water will have to be treated. The classic treatment would be the addition of lime and carbon dioxide.

Iron and manganese removal, as needed, will require the oxidation of the divalent metal and removal of the precipitate.

3.1.3 CAGE data

Below is a brief description of the water quality, based on the data from the CAGE project:

- The water has a higher dissolved solids concentration than the Hermanus data.
- Sodium concentrations show a greater variation with many samples having concentrations in the range for Class 1 water and a few exceeding the Class 2 water criteria.
- Calcium concentrations and alkalinity is similar to the Hermanus data, however higher outliers were recorded.
- The average chloride concentration was 360 mg/l and few samples had concentrations below 100 mg/l.
- Sulphate concentrations were similar to the Hermanus data.
- Regrettably no measurements of metals were done. It is, however, possible that this water will have high concentrations of iron and manganese, possibly higher than the Hermanus data.
- No data on pH is given.
- Due to the high chloride ion concentration, the water, even after stabilisation, will be corrosive to metals.

Assuming that the concentrations of iron and manganese are similar to the Hermanus data, treatment of this water will be the same as for the Hermanus water.

However, if the treated water is to meet the SABS Class 0 requirements it will have to be diluted with a greater concentration of low TDS water than the Hermanus water.

Dilution will also be required to reduce the corrosive nature of the treated water to currently accepted standards within the City of Cape Town, (chloride concentration of less than 50 mg/l).

3.1.4 Location of treatment

Especially with respect to the corrosive character of the water and the probable high iron and mangenese concentration the location of the treatment relative to the borehole or wellfield is of concern. The options include:

- Do not remove iron and manganese;
- Removal and stabilisation at each borehole;
- Treatment at a combined location, within the wellfield;
- Treatment at a combined location in a more accessible area, possibly at the inlet to an existing impounding reservoir(s);
- Treatment at the existing water treatment works, which draw water from the impounding reservoir(s).

Treatment close to each well or at least at a central location as close as possible to a group of wells may prove necessary. Considerations include:

- Effect of iron and manganese on conveyance systems, whether pipes, open channels, or natural watercourse(s);
- Remoteness of treatment works site;
- Disposal of iron and manganese sludge;
- Corrosiveness and aggressiveness of the borehole water

Without addition of alkalinity, which provides a buffer against pH change, the pH will change easily, for example by the production of CO_2 from biological activity, as may occur in pipelines downstream of the boreholes.

If there is no removal of iron at the source it is likely to precipitate out in the conveyance pipeline, and at the discharge point at the receiving reservoir.

Some manganese deposits can be expected in the conveyance pipelines, but much of it is likely to stay in solution, to be discharged into the receiving reservoir, where it may eventually cause problems. If the reservoirs already contain significant manganese concentrations then additional manganese from the boreholes may not be significant. The long term risk is that manganese will accumulate on the bed of the reservoir, to be released in relatively high and troublesome concentrations in the years to come.

Removal of iron and manganese at the source, together with stabilisation, would be the ideal choice, to avoid corrosion and deposition of iron and manganese in the downstream conveyance systems, and in the receiving reservoir.

However, it is recognised that there are many factors to be taken into account, and it may be found that some partial treatment at source will be sufficient. Stabilisation and removal of iron would seem highly desirable.

3.2 TREATMENT IN EXISTING WATER TREATMENT WORKS

3.2.1 Introduction.

The available data, as mentioned in Section 1.4, were reviewed from the aspect of the treatment of the water in the existing water treatment works of the City of Cape Town for potable use.

The water can be treated to comply with the requirements of SABS 241 (2001) Class 0 water, except for sodium and chloride. Sodium concentrations will be within the requirements of a Class 1 water. Chloride will normally meet the Class 1 criteria, but will occasional only meet the Class 2 criteria.

If the treated water is required to meet the Class 0 criteria at all times it must be mixed with another water with lower sodium and chloride concentrations.

The main treatment objective will be the removal of iron and manganese, disinfection and stabilisation of the treated water. Iron and manganese removal will require the oxidation of the divalent metal and removal of the precipitate.

3.2.2 Iron and Mangenese Removal

The options for iron and manganese removal may include:

- Removal by biological treatment;
- Removal of iron and manganese by conventional oxidation processes;

For each of these processes the water may need to be stabilised upstream of the existing water treatment works (by increasing alkalinity and the calcium concentration).

The removal of iron and manganese will be more complicated if they are chemically complexed with organic material, e.g. colour. This is unlikely, judging from the preliminary water quality data, but needs to be confirmed by future water quality analyses.

<u>Biological Treatment</u>

Pilot trials would be essential to determine the feasibility of biological treatment to remove iron and manganese. The process can be more economic than conventional oxidation but it requires reasonably well controlled conditions, including:

- controlled concentration of dissolved oxygen;
- pH control;
- redox potential within an acceptable range;
- absence of agents poisonous to the bacteria, such as hydrogen sulphide, heavy metals (e.g. zinc), and chlorine.
- temperature, above 10 deg C, and ideally in the range 10 to 15 deg C.

The sludge from biological treatment is reported to be well suited to thickening and dewatering, which will facilitate disposal.

<u>Treatment by Conventional Oxidation</u>

A conventional oxidation process is likely to include aeration, pH adjustment, dosing of a chemical oxidant, clarification in solids-contact type clarifiers, and filtration. This could be similar to a conventional treatment works centred on sludge blanket clarifiers and rapid gravity filters. The filtration stages could be contained in open tanks or in pressure vessels.

3.2.3 General Treatment

For the general treatment some process modifications at the existing treatment works will be required to:

- Adjust pH.
- Dose an oxidising agent.
- Provide oxidation contact tank.
- Depending on the current coagulation pH and filtration pH (i.e the point of lime addition for stabilisation) at the existing treatment works, a separate unit will probably be required for the removal of the precipitate.

- Depending on the configuration of the existing works and order of disinfection and stabilisation, separate disinfection and stabilisation facilities may also be required.

With regard to the latter two points existing disinfection, stabilisation and filters will probably not be required at Faure, where the pH is raised prior to filtration. At the other works separate units will probably be required.

However, it might be preferable to install remote treatments for stabilisation and iron removal close to the boreholes or wellfields. The blended water can then be mixed with surface water either in the treatment works or in the dams (see below).

3.3 MIXING WITH SURFACE WATER

3.3.1 Introduction

One of the options being considered for the use of TMG Aquifer water is to transfer water to any of the three main bulk water supply reservoirs from where it is then treated for domestic use and distributed to the Cape Town metropolitan area. The terms of reference for this task was to assess the potential impacts, at a scoping level, of transferring water from the TMG Aquifer to Voëlvlei, Wemmershoek or Theewaterskloof dams at a rate of 15 Mm^3/year, 15 Mm^3/year and 40 Mm^3/year respectively. The potential impacts, if water is transferred all year round or if water is transferred for six months only during the summer months, were also assessed.

3.3.2 Methodology

A quantitative and qualitative assessment of the potential impacts on in-lake water chemistry of blending water from the TMG Aquifer with reservoir water was done.

The quantitative assessment involved setting up a spreadsheet that calculates the water chemistry changes if TMG Aquifer water is mixed with reservoir water. Typical monthly inflows and demands for each reservoir were obtained from the Western Cape Systems Analysis and this was used to set up a water balance for each reservoir and to estimate the change in quality over an eight-year period. The in-lake and inflow water qualities were obtained from statistical analysis of historical data collected at the three dams and the TMG Aquifer data sets for Hermanus and Peninsula CAGE that were provided for the assessment. The summary statistics of the data sets used are presented in Appendices A and B. In all cases the median concentrations were used because the average concentration is often distorted by one or two high outlier values. Since the TMG data sets are probably distorted by different water quality (seawater, different aquifer) and the median values are higher than expected, the 25% quartile values were used additionally to compare with the median values.

A visual assessment of the seasonal change in water chemistry in the reservoirs indicated that there was not a strong seasonal trend in the conservative substances (e.g. salinity, sodium, chloride) and the median concentration of these substances changed very little between summer and winter seasons. The resultant time series graphs of the estimated in-lake concentrations were examined for the time it took to reach a new equilibrium once pumping commenced, and what the new equilibrium concentration would be. The new equilibrium concentrations were then compared to the guidelines for domestic water supply (Water Research Commission, 1998) to assess the fitness for use.

The qualitative assessment involved identifying additional considerations or impacts that cannot be quantified at a scoping level but that should be investigated at a later stage of the project.

3.3.3 Results

The impact on the in-lake water chemistry of blending TMG Aquifer water with reservoir water is reported below for each reservoir. The impact on salinity (electrical conductivity) was assessed as well as the other variables of concern, sodium (Na), chloride (Cl) and sulphate (SO_4). No EC data was available for the Peninsula CAGE data set and only Na, Cl and SO_4 impacts were therefore assessed.

The results are summarised in tables that show the present chemistry in the reservoir, the chemistry of the transfer water and the resultant median concentration after equilibrium has been reached. The percentage change and the variability in the resultant chemistry are also reported. The WRC guideline for the ideal water quality can be used to assess the fitness for use of the estimated condition in the reservoir after implementation.

Voëlvlei Dam

Table 3-1 Estimated in-lake concentrations after pumping of Peninsula CAGE water to Voëlvlei Dam has commenced

Variable	Sodium (Na in mg/l)		Chloride (Cl in mg/l)		Sulphate (SO_4 in mg/l)	
Pumping	All year	Summer	All year	Summer	All year	Summer
Transfer volume (Mm^3/month)	1.3	2.5	1.3	2.5	1.3	2.5
Initial concentration[1]	11.8		20.5		6.27	
TMG concentration[2]	103.6		206.7		14.8	
Final concentration[1]	20.2	20.3	37.1	37.4	7.0	7.1
% increase	71%	72%	84%	86%	12%	13%
Variability (10-90% range)	16 - 23	17 - 24	29 - 42	31-45	6.7-7.3	6.8-7.4
Guideline for ideal quality	<100		<100		<100	

Variable	Sodium (Na in mg/l)		Chloride (Cl in mg/l)		Sulphate (SO_4 in mg/l)	
Pumping	All year	Summer	All year	Summer	All year	Summer
Transfer volume (Mm^3/month)	1.3	2.5	1.3	2.5	1.3	2.5
Initial concentration[1]	11.8		20.5		6.27	
TMG concentration[3]	55.9		104.1		8.15	
Final concentration[1]	15.8	15.9	28.1	28.2	6.4	6.4
% increase	34%	35%	37%	38%	3%	3%
Variability (10-90% range)	14 - 17	14 - 18	25 - 30	25 - 32	6.4-6.5	6.4-6.5
Guideline for ideal quality	<100		<100		<100	

1 – median concentration
2 – source water is Peninsula CAGE, median concentration
3 – source water is Peninsula CAGE, 25th percentile concentration

The sodium and chloride concentrations in the transfer water are significantly higher in the transfer water than in Voëlvlei Dam and the sulphate concentrations are somewhat higher in the transfer water. Although the transfer results in an overall increase in in-lake concentrations, the resultant concentrations in both scenarios are still well within the guidelines value for ideal water quality for domestic water supplies.

Wemmershoek Dam

Table 3-2 Estimated in-lake concentrations after pumping of Hermanus water to Wemmershoek Dam commenced

Variable	Electrical conductivity (mS/m)		Sodium (Na in mg/l)		Chloride (Cl in mg/l)		Sulphate (SO_4 in mg/l)	
Pumping	All year	Summer	All year	Summer	All year	Summer	All year	Summer
Transfer volume (Mm^3/month)	1.3	2.5	1.3	2.5	1.3	2.5	1.3	2.5
Initial concentration[1]	4		3.8		7.05		3.35	
TMG concentration[2]	45.3		76		127		16	
Final concentration[1]	11.9	11.6	17.6	17.1	29.9	29.2	5.8	5.7
% increase	197%	191%	362%	351%	324%	314%	72%	70%
Variability (10-90% range)	9-14	9-15	12-21	13-24	21-35	22-40	4.8-6.3	4.9-6.8
Guideline for ideal quality	<70		<100		<100		<100	

Variable	Electrical conductivity (mS/m)		Sodium (Na in mg/l)		Chloride (Cl in mg/l)		Sulphate (SO_4 in mg/l)	
Pumping	All year	Summer	All year	Summer	All year	Summer	All year	Summer
Transfer volume (Mm^3/month)	1.3	2.5	1.3	2.5	1.3	2.5	1.3	2.5
Initial concentration[1]	4		3.8		7.05		3.35	
TMG concentration[3]	39.25		49		93		13	
Final concentration[1]	10.7	10.5	12.4	12.1	23.4	22.9	5.2	5.1
% increase	168%	163%	227%	220%	233%	225%	55%	53%
Variability (10-90% range)	8-12	8-14	9-14	9-16	18-31	17-27	4.5-5.6	4.5-6.0
Guideline for ideal quality	<70		<100		<100		<100	

1 – median concentration
2 – source water is Hermanus data set, median concentration
3 – source water is Hermanus data set, 25th percentile concentration

The water quality in Wemmershoek Dam is very good. The electrical conductivity, sodium and chloride concentrations in the transfer water are significantly higher in the transfer water than in Wemmershoek Dam. Overall, this results in increases in in-lake concentrations of greater than 200%. The large increase is a function of the smaller volume of Wemmershoek Dam and volume transferred. However, the resultant concentrations in both scenarios are still well within the guidelines value for ideal water quality for domestic water supplies.

Theewaterskloof Dam

Table 3-3 Estimated in-lake concentrations after pumping of Hermanus water to Theewaterskloof Dam commenced

Variable	Electrical conductivity (mS/m)		Sodium (Na in mg/l)		Chloride (Cl in mg/l)		Sulphate (SO_4 in mg/l)	
Pumping	All year	Summer	All year	Summer	All year	Summer	All year	Summer
Transfer volume (Mm^3/month)	3.33	6.66	3.33	6.66	3.33	6.66	3.33	6.66
Initial concentration[1]	7.8		7.0		13.0		5.5	
TMG concentration[2]	45.3		76		127		16	
Final concentration[1]	12.7	12.5	16.0	15.6	27.9	27.2	6.9	6.8
% increase	63%	60%	105%	100%	114%	109%	25%	24%
Variability (10-90% range)	10-14	10-14	11-18	11-19	19-30	20-32	6.1-7.1	6.2-7.2
Guideline for ideal quality	<70		<100		<100		<100	

Variable	Electrical conductivity (mS/m)		Sodium (Na in mg/l)		Chloride (Cl in mg/l)		Sulphate (SO_4 in mg/l)	
Pumping	All year	Summer	All year	Summer	All year	Summer	All year	Summer
Transfer volume (Mm^3/month)	3.33	6.66	3.33	6.66	3.33	6.66	3.33	6.66
Initial concentration[1]	7.8		7.0		13.0		5.5	
TMG concentration[2]	39.25		49		93		13	
Final concentration[1]	11.9	11.7	12.5	12.2	23.4	23.0	6.5	6.4
% increase	53%	50%	78%	75%	80%	77%	18%	17%
Variability (10-90% range)	9-13	10-13	9-13	10-14	17-25	18-24	5.9-6.6	6.0-6.7
Guideline for ideal quality	<70		<100		<100		<100	

1 – median concentration
2 – source water is Hermanus data set, median concentration
3 – source water is Hermanus data set, 25th percentile concentration

The electrical conductivity, sodium and chloride concentrations in the transfer water are significantly higher in the transfer water than in Theewaterskloof Dam and the sulphate concentrations are somewhat higher in the transfer water. Although the transfer results in an overall increase in in-lake concentrations, the resultant concentrations in both scenarios are still well within the guidelines value for ideal water quality for domestic water supplies.

Using the 25th percentile values will still result in elevated concentrations in the reservoir although it is not as high as when using the median concentrations. However, in all cases the resultant waters were well within the guideline values for ideal water quality. This exercise underlines the issue of uncertainty in the quality of the groundwater and that an effort should be made to address this during the further phases of the project.

3.3.4 Discussion

Should water from the proposed boreholes discharge to one of the existing impounding reservoirs the effect will depend on whether the borehole water receives treatment prior to discharging into the reservoir, and upon the type of treatment.

Parameters of main interest appear to be:

- chlorides
- nitrates
- nutrients e.g. phosphates
- softness
- iron deposition
- manganese deposition

Without Treatment

The effect of chlorides has been assessed, and judged to be tolerable. After an initial increase over the first year or two it is presumed that chloride concentrations in the reservoir water will not continue to increase, assuming there is sufficient natural inflows of low-chloride water each year. Further studies should be carried out to verify this. Computer modelling, using dispersion techniques may help to assess localised water quality effects, e.g. streaming.

Water quality analyses on the borehole water are needed to assess the concentrations of nitrates and phosphates, but it appears that these are likely to be of low significance, with little or no impact on water quality in the receiving reservoir.

The reservoir waters are also very soft, so no material effect on reservoir water quality is envisaged.

With Treatment

If the borehole water is stabilised and iron and manganese removed before reaching the reservoir it appears that the effects on reservoir water quality would be minimal.

3.3.5 Other potential impacts and considerations

Aesthetic impact of iron and manganese

The iron and manganese concentrations reported for the Hermanus data set is quite high. It is possible that it would form a precipitate when the transfer water is oxygenated at the discharge point in the dam. This would probably result in a dark rust coloured deposit in the mixing area close to the discharge point, which may be aesthetically objectionable and may have an impact on the bottom dwelling organisms in the affected area. However, this effect is expected to be limited in time and space due to dilution with rain water and establishing an equilibrium.

Nutrients

The nutrient concentration in groundwater is generally quite low and not regarded as a concern when mixed with surface water. However, the nitrogen concentrations in the Peninsula CAGE data set appears to be high for groundwater and may cause enrichment at the point of discharge. This potential impact should be addressed in the Exploratory Phase with water chemistry data from the boreholes drilled for this project.

Downstream impacts

There are already concerns about the impacts of Berg River (Skuifraam) Dam on water quality in the lower Berg River. Releases from Voëlvlei Dam are sometimes used to control elevated salinities in the lower Berg River and any change in the overall salinity of Voëlvlei Dam can potentially have a cumulative effect on the lower Berg River. This should be investigated during further phases of the project.

4. MANAGEMENT CONSIDERATIONS

4.1 ADAPTIVE MANAGEMENT

In order to manage the current uncertainties for the wellfield operation the implementation of an adaptive management is recommended. Adaptive Management is an incremental approach, which increases the confidence level of the system during operation and keeps the flexibility to react appropriate to different events. Therefore it helps to minimise risks and associated costs.

The adaptive management starts with the wellfield implementation during the exploratory phase and will be fully implemented in the pilot phase. A comprehensive monitoring network for both water level fluctuations and water quality changes enables fast and immediate reaction for necessary changes in the operation rules. The assessment procedures can be automated, e.g. using datalogger and a telemetry system to control the drawdown in the borehole, resulting in immediate decrease of discharge rates or switch to another borehole, if a specified drawdown is reached.

Regular water quality measurements, as also mentioned below, can indicate changes in composition, which could require additional treatment and or changes in operation set-up, e.g. increase in salinity could result in a switch between boreholes as well as in changing the mixture with surface water.

Further detailed assessments of the water quality and of the possible yield at the actual target site are needed to develop and optimise the monitoring network, which is the core of the adaptive management strategy. A regular cycle of operation, monitoring, re-assessment and, if required, change of operation ensures the flexibility and security in the wellfield operation.

In the light of uncertainty, adaptive management is the responsible and sustainable approach to groundwater development.

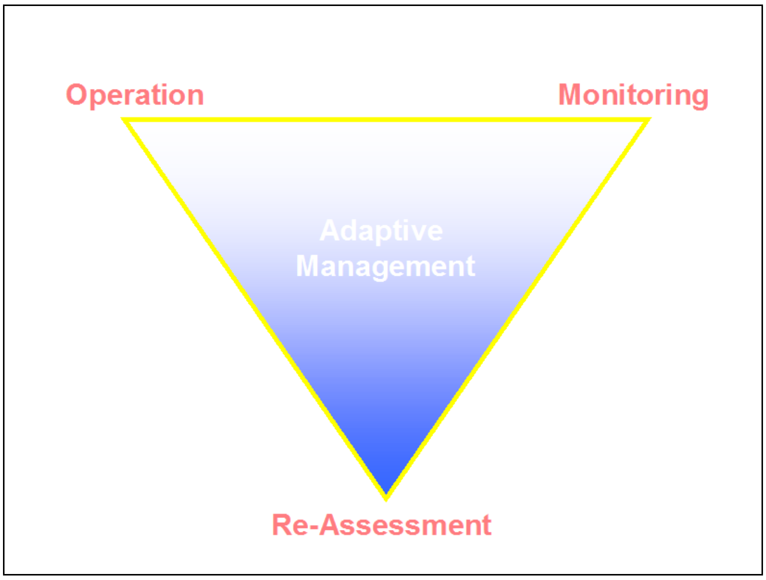

Figure 4-1 Principle of Adaptive Management

4.2 MONITORING AND PREVENTATIVE MAINTENANCE

Boreholes affected by iron biofouling are usually identified when there are dramatic decreases in yield or increases in dissolved iron concentrations. However, by the time changes are noted in these parameters, clogging is already substantial and rehabilitation becomes expensive. Rohde and Keevill (2000) recommend that boreholes should be treated before a 40% reduction in specific capacity or two orders of magnitude increase in biological aggressivity has occurred.

Many scientists have stressed the need for ongoing monitoring and informed management systems to control biofouling (e.g. Alcade and Gariboglio, 1990; Tyrrel and Howsam, 1990; Jolly and Engelbrecht, 2002; McLaughlan, 2002). Early detection and preventative maintenance may save some of the costs associated with cleaning or replacing boreholes, since the restoration of a severely biofouled borehole is extremely difficult. Rohde and Keevill (2000) provide guidelines for borehole design and construction to facilitate monitoring and maintenance.

Monitoring programmes should include regular collection of pumping and rest water level data and periodic pump testing to measure changes in specific yield. Samples should also be collected for chemical and microbiological analysis. Closed Circuit Television (CCTV) cameras are also being widely used for surveys of biofouled boreholes. Biological Activity Reaction Tests (BART™) have been developed by DBI to monitor changes in biological activity. These test kits are used to evaluate the effectiveness of borehole treatment and as a warning system for when further treatment is required (Rohde and Keevill, 2000). Howsam and Tyrrel (1989) also describe a monitoring cell (moncell), which is basically a sand-filled clear acrylic tube connected to the pump discharge pipework, that allows for visual inspection of biofilm growth without the need to dismantle the pumping infrastructure.

Preventative maintenance involves activities designed to interrupt the deterioration process at an early stage, based on systematic, scheduled programmes of providing early detection (McLaughlan, 2002). Shock chlorination and/or physical disruption by air-lift pumping, jetting or brushing have been recommended as a means of disrupting biofilms in their early stages of development. The application of an electric current is also being investigated as a prevention technique to prevent biofilm growth (Rohde and Keevill, 2000).

4.3 TREATMENT OPTIONS FOR IRON BIOFOULING

Various methods have been developed to treat iron biofouling in boreholes. Traditional treatments include acid treatment and shock chlorination to both remove mineral precipitates and disinfect the borehole of bacteria. In some cases, chemical treatment is accompanied by mechanical disruption of the biofilm using jetting or wire brushes. Two treatment processes, which have been widely used in North America, are the Blended Chemical Heat Treatment (BCHT™) and the Ultra Acid Base Treatment (UAB™) developed by Droycon Bioconcepts Incorporated (DBI). The BCHT™ method has also been used with some success at Atlantis and the Klein Karoo, South Africa.

Examples of the variety of chemical agents that are employed for cleaning fouled boreholes include: mineral acids (hydrochloric acid, sulphamic acid), organic acids (oxalic acid, acetic acid, citric acid, ascorbic acid), caustic alkalis (calcium hydroxide, potassium hydroxide, sodium hydroxide), chlorine compounds (calcium hypochlorite, sodium hypochlorite), oxidants (hydrogen peroxide), reductants (sodium dithionate) and a variety of proprietary commercial products including acids, oxidants, dispersants and surfactants (Houben, 2003b; Stewart, 2000). Some of these treatments use harsh chemicals that may ultimately damage the aquifer, particularly if applied in high concentrations or in unstable geological environments. Stewart (2000) presents results of laboratory testing of the effects of some of these chemicals on aquifer materials from Canada. Laboratory experiments conducted by Houben (2003a) found that sodium dithionate, ascorbic acid, malonic acid, oxalic acid and sulphuric acid were effective for dissolving synthetic ferrihydrite. Dithionite and oxalic acid were also effective in dissolving the more stable iron hydroxide, goethite. Citric acid, sulphamic acid and hydrochloric acid only achieved partial dissolution of ferrihydrite after 7 hours. Ferrihydrite and goethite were insoluble in sodium hydroxide.

The chemical and mechanical treatment options usually improve borehole performance over the short term, but clogging problems tend to recur frequently in the same boreholes, often within the space of a few months. Regular borehole cleaning thus becomes part of the maintenance programme in wellfields that are prone to iron biofouling. In cases of severe clogging, the most economical option may even be to abandon the borehole and drill a new one in its place.

5. CONCLUSIONS AND RECOMMENDATIONS

5.1 CONCLUSIONS

5.1.1 Iron biofouling

The literature review on iron biofouling revealed that the causes and processes of iron biofouling and precipitation in parts of the TMG aquifers have to be taken into account for further planning and assessments during the study. However, the factors determining the risk of clogging in the boreholes and conveyance systems are known:

- Temperature
- Salinity
- Geology of the host rock
- Organic carbon
- Dissolved oxygen
- Dissolved gases
- pH
- Light
- Pressure
- Borehole construction material

For most of these factors (i.e. salinity, geology of the host rock, pH, dissolved oxygen, dissolved gases, organic carbon) the preliminary assessment and the experience from other studies indicate a less probability of iron biofouling occurring in the Peninsula Aquifer than in the Nardouw Aquifer, which was mainly utilised in the water supply schemes with high maintenance due to borehole clogging. A comparison between the two aquifers is shown in Table 5-1 below.

Factor	Peninsula	Nardouw	Comment
Geology of host rock	Quartzite, very low iron content	Sandstone, siltstone, contains iron	Less iron at source, less risk of biofouling
pH	4 - 6.5	6 - 7	Lower pH reduce risk of iron-biofouling
Salinity	Water is very pure, low salinity	Higher salinity due to more solvents from host rock	Higher salinity seems to increase risk of biofouling
Organic carbon		Layers with higher contents of organic carbon	Presence of organic carbon increases risk and effect of biofouling

Detailed planning and investigation during the Exploratory Phase can significantly further reduce the risk of failure due to iron biofouling.

Clogging of boreholes and conveyance system due to iron biofouling and precipitation can be managed, if the problem is assessed in detail and continuously monitored.

5.1.2 Water quality and treatment

The following tentative conclusions are reached based on the limited available data:

a) The Peninsula Aquifer is generally of good water quality. The data indicate that the water may be highly corrosive. In some instances there might be elevated concentrations of sodium, chloride, iron and or manganese.

b) Separate treatment facilities either remote at the wellfields or at the existing treatment works will probably be required for the removal of iron and manganese.

c) Stabilisation of the water to reduce its corrosive character seems to be required.

d) The treated water should be diluted with suitable water to reduce the sodium, chloride and possibly the sulphate concentrations to currently accepted levels.

e) The treatment with respect to both stabilisation and removal of iron and manganese would be ideally situated at the source (i.e. each borehole) or at least at the wellfield, to avoid corrosion and deposition of iron and manganese in the downstream conveyance systems.

5.1.3 Water mixing in water supply reservoirs

The preliminary assessment was aimed at assessing the potential impacts under average conditions in the system to determine whether there are any fatal flaws if TMG Aquifer water is first transferred into the Cape Town water supply reservoirs before treatment and distribution. As such the assessment does not give an indication of what might happen during extreme conditions such as during a severe drought with reduced, poor quality inflows.

A number of conclusions can be drawn from the preliminary assessment:

a) The groundwater chemistry data sets available for the assessment show that there is a difference between in-lake and TMG aquifer chemistry. Salinity, and its individual constituents, is expected to be higher in the TMG water than in the reservoirs. This results in an increase in in-lake concentrations when the two waters are blended.

b) The resultant chemistry after blending the two waters is, however, still well within the guidelines for domestic water supply. It is not expected that the resultant water quality has negative impacts on the aquatic ecology in the reservoirs.

c) There appeared to be little impact between year-round pumping and summer pumping on the average water chemistry. However, the rate of change in chemistry when pumping only during the summer months is greater due to a change in mixing conditions (i.e. higher rate of transfer, less surface water in dam).

It was concluded that, based on the preliminary assessment of the impacts on reservoir chemistry, there were no fatal flaws that would stop the project from going to a further phase.

5.2 RECOMMENDATIONS

5.2.1 Biofouling

Based on a literature review the following are recommended for consideration in the next phases of the study:

a) Since the source of the water and the host rock are determining factors for the risk of iron biofouling, the exploration phase should focus on the Peninsula Aquifer. Furthermore, hydraulic connection of the targeted aquifer to the overlying aquitard (i.e. Cedarberg Shale) and aquifer (i.e. Nardouw) should be avoided. During the borehole siting and the drilling program this factor has to be taken into account.

b) The influence of the borehole construction material should be investigated further, as there appears to be a controversery. This will inform the wellfield costing for the Pilot Phase.

c) Detailed and site specific assessment of the water quality (see below) is required to determine the risk of and management options for iron biofouling.

d) If the assessment in the Exploratory Phase verifies the potential risk, a pilot trial of different treatment and management options is required prior to or as first step in the Pilot Phase of the study.

5.2.2 Water Quality

A more extensive water quality sampling program (see Appendix C) is required to allow for thorough investigation into the treatment requirements of the underground water and its effect on the water system of Cape Town. The following recommendations were drawn based on the desktop review of currently available data:

a) Better groundwater chemistry data – there appeared to be uncertainty about how representative the groundwater chemistry data was of water that may be transferred in future. The typical concentrations in the TMG Aquifer water are much lower than the concentrations in the data sets supplied for the assessment. It is recommended that the estimates of the TMG Aquifer water chemistry be refined through monitoring once the exploration begins.

b) More detailed assessment - A number of assumptions were made about average water chemistry in the inflows to the reservoirs, the in-lake water quality and the chemistry of the source water. It is recommended that the potential impacts on water chemistry be assessed in more detail when the system yield analysis is undertaken during a later phase of the project. That assessment should then use more realistic inflow rates and specifically, more site specific TMG Aquifer water chemistry data.

c) Location of discharge and abstraction points – an implicit assumption in this assessment was that the pumped TMG water is instantly mixed with the whole reservoir. In reality this is not the case and in a more detailed assessment during a later phase of this project, the location of the discharge and abstraction points in the reservoir need to be considered to examine aspects such as the mixing zone, impacts of in-lake currents and potential short-circuiting to the abstraction point.

d) Information on the volume of inflow from the TMG Aquifer to the in-lake reservoirs must be confirmed during the Exploratory Phase. A detailed assessment might be required for the Berg River Dam and Steenbras Dams, depending on the location of the production boreholes.

5.2.3 Wellfield Management

To establish and maintain a wellfield management programme that takes the current uncertainties into account, a comprehensive monitoring system is required comprising of:

i) Abstraction monitoring

- Abstraction rate, continuously
- Drawdown, continously
- Water quality of abstracted water, regularly (daily - weekly)

ii) Near field monitoring

- Water level, continuously
- Water quality of groundwater, regularly (weekly - monthly)

iii) Far field monitoring

- Water level, regularly (weekly - monthly)
- Water quality of groundwater, regularly (monthly - quarterly)

The monitoring system should be developed during the Exploratory Phase and implemented latest in the Pilot Phase of the study. While the TMG Aquifer Alliance will develop the monitoring system, it becomes the responsibility of the City of Cape Town to implement the system and to undertake the monitoring according to the specified programme. Based upon the results the operation rules will be determined during the Pilot Phase.

5.2.4 Terms of Reference for Exploratory Phase

The recommendations given above are in line with the original proposed terms of reference and specify in more detail the required scope of work for the Exploratory Phase of the study.

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APPENDICES

Appendix A Summary Statistics of TMG Aquifer data sets

Appendix B Summary Statistics of in-lake data

Appendix C Future Data Collection

Appendix A
Summary Statistics of TMG Aquifer data sets

Hermanus data set

Variable	n	Average	Min	Max	25%tile	Median	75%tile
pH	22	5.65	4.00	6.78	5.35	5.65	6.10
EC mS/m	26	53.32	33.00	108.00	39.25	45.30	58.75
TDS mg/l	17	358.29	211.00	691.00	243.00	358.00	378.00
Ca mg/l	17	5.74	2.90	11.00	4.40	5.40	5.80
Mg mg/l	17	10.47	6.90	23.00	7.80	8.70	12.00
Na mg/l	17	76.59	39.00	182.00	49.00	76.00	79.00
K mg/l	17	4.04	1.90	13.00	2.20	2.50	3.30
PALK mg/l	17	6.04	n.d.	32.00	3.00	5.00	6.50
MALK mg/l	17	6.04	n.d.	32.00	3.00	5.00	6.50
Cl mg/l	17	135.88	80.00	249.00	93.00	127.00	150.00
SO4 mg/l	17	31.60	8.50	117.00	13.00	16.00	23.00
Cations	17	4.61	2.55	9.54	3.05	4.46	4.88
Anions	17	4.62	2.53	9.62	3.04	4.34	4.83
Balance	17	1.52	0.07	4.32	0.80	1.15	2.47
N NO3mg/l	16	0.10	n.d.	0.87	0.00	0.00	0.10
Fe mg/l	23	4.80	n.d.	12.90	3.15	3.90	5.55
Mn mg/l	23	1.34	0.04	2.50	0.74	1.40	1.75
N_Amonia mg/l	2	0.465	n.d.	0.93	0.2325	0.465	0.6975

Peninsula data set

Variable	n	Average	Min	Max	25%tile	Median	75%tile
Na	35	168.2657	10.3	1251.1	55.9	103.6	172
Mg	35	30.30857	1.5	304.5	7.5	16.5	34.85
Ca	35	20.68571	1.1	229.2	3.55	7.1	15.3
F	35	0.138571	0.01	0.35	0.09	0.12	0.16
Cl	35	360.7457	18.8	2972.5	104.1	206.7	366.05
NO_3+NO_2-N	35	3.464514	0.013	12.767	1.033	2.726	4.6615
SO4	35	28.35143	1.7	192.4	8.15	14.8	24.9
PO_4-P ortho	35	0.023571	0.006	0.311	0.009	0.013	0.0195
TAL as $CaCO_3$	35	18.1	n.d.	118.3	1.6	5.8	15.2
HCO_3	37	20.88838	n.d.	144.326	0.854	6.832	18.056
Si	35	4.890571	0.16	11.27	4.305	4.62	5.18
K	35	2.659714	0.17	14.45	0.95	1.68	3.025
NH_4-N	35	0.016143	n.d.	0.174	0.007	0.012	0.0165

n.d. not detectable

Appendix B
Summary Statistics of in-lake data

Voëlvlei Dam Data period: 15/3/69 – 18/9/01

Variable	Valid N	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Variance	Std.Dev.
PH	1115	7.193625	7.35	4.24	9.26	6.81	7.515	0.356301	0.59691
KJEL_N_T	908	0.363163	0.3405	0.02	4.721	0.268	0.426	0.061401	0.247791
NO3_NO2_	1113	0.065466	0.04	0.02	0.954	0.02	0.082	0.006682	0.081746
NH4	1090	0.042417	0.02	0.02	2.246	0.02	0.055	0.00541	0.073553
F	1082	0.128218	0.12	0.05	0.53	0.05	0.16	0.0061	0.078101
TAL	1095	15.19439	14.2	4.3	90.1	11.7	17.3	42.55521	6.523435
NA	1084	11.90304	11.8	2	60	10.6	12.8	7.242436	2.691178
MG	1084	3.027623	3	0.5	13	2.7	3.2	0.58796	0.766786
SI	1090	0.660028	0.59	0.2	9.895	0.2	0.84	0.327282	0.572086
P_TOT	906	0.027672	0.025	0.0025	0.23	0.017	0.033	0.000307	0.017512
PO4_P	1099	0.014901	0.011	0.0025	0.649	0.005	0.018	0.001152	0.033939
SO4	1084	6.469904	6.2695	2	29.3	4.6855	8.1	9.983673	3.159695
CL	1084	20.51326	20.15	0.5	105	17.3	23.25	28.8155	5.368007
K	1078	1.01688	0.82	0.15	3.99	0.71	1.04	0.282157	0.531185
CA	1084	3.745017	3.5	0.5	21.9	3.1	4	2.353826	1.534218
EC	1116	11.92598	11.625	2.4	52	10.8	12.6	6.315575	2.513081
DMS	1070	65.14832	64	33.8	199	59	70	141.1005	11.87857

Wemmershoek Dam Data period: 11/5/68 – 13/1/86

Variable	Valid N	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Variance	Std.Dev.
EC	209	4.703349	4	1	88	3.7	4.6	35.99859	5.999883
TDS	189	28.24868	24	13	595	21	27	1849.358	43.00416
PH	209	5.640096	5.8	4	9	4.7	6.42	0.962471	0.981056
NA	208	4.186538	3.8	0	69.4	3.3	4.1	23.40542	4.837915
MG	208	0.957212	0.7	0.1	25.7	0.6	1	3.582943	1.892866
CA	208	1.514423	1	0	66	0.7	1.4	21.81081	4.670204
F	199	0.076281	0.06	0	0.56	0.03	0.1	0.006032	0.077663
CL	208	7.642308	7.05	1	73.5	6.2	8.25	26.4524	5.14319
NO2_NO3	209	0.0419	0.03	0	0.58	0.02	0.05	0.003199	0.056563
SO4	208	5.023077	3.35	0	266.7	2.3	5	339.1856	18.41699
PO4	199	0.006698	0.005	0	0.065	0	0.01	6.55E-05	0.008095
TAL	208	6.259615	5	0	73.6	3.3	7.4	52.89131	7.272641
SI	190	1.715053	1.675	0.42	3.73	1.45	1.92	0.193361	0.439728
K	198	0.461616	0.33	0	11.56	0.24	0.43	0.798568	0.893626
NH4	190	0.046679	0.047	0	0.13	0.023	0.062	0.00079	0.02811
KN	173	0.239376	0.207	0.017	1.069	0.138	0.29	0.023576	0.153546
TP	173	0.024162	0.01	0	1.57	0	0.01	0.020502	0.143186

Theewaterskloof Dam Data period: 28/1/80 – 17/8/89

Variable	Valid N	Mean	Median	Minimum	Maximum	Lower Quartile	Upper Quartile	Variance	Std.Dev.
EC	644	8.750311	7.8	1.5	49.3	6.4	8.8	24.76888	4.976834
TDS	598	46.52676	42	16	349	33	50	653.841	25.57031
PH	644	5.982873	6.095	2.12	8.48	4.72	7.12	1.512851	1.22998
NA	599	8.316027	7	1.7	53.9	5.8	8.1	37.22118	6.100916
MG	599	1.981135	1.8	0.6	16.5	1.3	2.2	1.351098	1.162368
CA	599	2.462771	2.3	0.2	34.1	1.6	2.9	3.307592	1.818679
F	599	0.112237	0.09	0	0.56	0.04	0.16	0.009742	0.098702
CL	599	15.66461	13	5.1	79.1	10.7	15.7	126.9941	11.26916
NO2_NO3	602	0.128831	0.11	0	0.94	0.054	0.189	0.009832	0.099159
SO4	599	5.914691	5.5	0	64.3	3	8.1	17.89326	4.230043
PO4	599	0.013684	0.012	0	0.195	0.007	0.017	0.000149	0.012208
TAL	602	8.503488	7.8	0	170.5	5.4	10.6	64.01831	8.001144
SI	599	0.935843	0.71	0	7.12	0.42	1.13	0.615892	0.784788
K	599	1.006611	0.81	0.02	10.3	0.54	1.37	0.529197	0.72746
NH4	599	0.057895	0.053	0	0.62	0.034	0.073	0.001463	0.038254
KN	6	0.296833	0.2965	0.237	0.363	0.249	0.339	0.002777	0.052697
TP	6	0.031667	0.02	0.01	0.1	0.01	0.03	0.001177	0.034303

Appendix C
Future Data Collection

The following is a summary of the key data collection that seems necessary. The data requirements will need to be updated regularly, as information becomes available.

Borehole water

a) *Very accurate and reliable measurement of pH, accurate to 0.05pH unit;*
b) *pH measurements to record changes in pH as water is drawn to the surface, and thereafter along the discharge pipework;*
c) *temperature of the water, corresponding to all pH measurement points;*
d) *total dissolved solids (TDS);*
e) *alkalinity;*
f) *calcium.*
g) *Redox values of water within the aquifer, to determine the 3 Redox zones, when boreholes are in production and idle;*
h) *Continuous recording of the abstraction rate from a borehole and aquifer draw-down water level adjacent to the borehole, to relate to the times when water quality samples were taken;*
i) *Sulphate and hydrogen sulphide;*
j) *Iron and manganese, both in solution and total;*
k) *Natural organic compounds (relevant to chemical oxidation);*
l) *Dissolved oxygen (also for chemical oxidation).*
m) *heavy metals (e.g. zinc),*
n) *ammonia,*
o) *hydrocarbons.*
p) *Chlorides*
q) *Nitrates*
r) *Nutrients, e.g. phosphates*

A full suite of analyses to determine physical properties and chemical composition properties should also be carried out at regular intervals, to build up a data base of water quality data. Data from the next 3 years would provide a reasonable base for future process design.

Reservoir water

s) *manganese, sampled over the full depth;*
t) *phosphorous, sampled over the full depth.*