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editors

GROUNDWATER POLLUTION IN AFRICA

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Groundwater Pollution in Africa

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Preface by UNEP

At the Millennium Summit in 2000 and during the 2002 World Summit on Sustainable Development in Johannesburg, world leaders recognized the vital importance of surface and subterranean fresh water to human development, and committed themselves to a precise, time-bound agenda for addressing the world's current and future water resource and sanitation needs. This commitment formed a strong basis for discussion at CSD-13 on developing strategies to achieve the goal of universal safe drinking water supply, as well as providing a policy framework to leverage national and international support in pursuit of this goal.

Groundwater is of particular importance in Africa, where numerous countries rely on a combination of hand-dug wells and public boreholes for their drinking water. Although groundwater use is generally less visible than surface supplies, it increasingly provides the main source of agricultural irrigation in rural areas, as well as a vital safety net for dry-season food security. In urban centres, groundwater is an important source of affordable municipal and private freshwater supplies.

In the past two or three decades, rapid urbanization across Africa has led to the growth of large areas of unplanned sub-standard housing in most cities. Residents of such areas usually resort to groundwater as a source of inexpensive, high-quality domestic water supply. However, the uncontrolled expansion of this kind of housing, together with increasing sewage and effluent leakage, indiscriminate waste disposal, and uncontrolled industrial and commercial activities all lead to the increasing pollution and deteriorating quality of groundwater – and to mounting public health problems.

In 2000, the United Nations Environment Programme (UNEP) and the United Nations Education Science and Cultural Organization (UNESCO), with the support of the United Nations Economic Commission for Africa (UN-ECA) and the United Nations Programme on Human Settlements (UN-Habitat), launched a collaborative effort to assess the vulnerability of groundwater in several African cities. The assessments had one primary goal: to assist policymakers to make informed and appropriate decisions for protecting their groundwater. Under the title *Urban Pollution of Surface and Groundwater Aquifers in Africa*, the project's initial two-year phase addressed the issue of aquifer vulnerability and the need to protect the quality of key groundwater resources. This was followed in 2003–05 by a second phase, *Assessment of the Pollution Status and Vulnerability of Water Supply Aquifers of African Cities*, which focused on monitoring and assessing the pollution threats arising from unplanned developments, open sewers, leaking septic tanks and latrines, and uncontrolled industrial and commercial activities. This phase also involved capacity building and networking between governments, local authorities and university researchers in the participating countries.

All of these activities have been aimed at alleviating the detrimental effects of human activity on Africa's groundwater resources, by providing accessible and reliable academic and public information. Aquifer stress now constitutes a genuine crisis in some African countries, where towns and cities are often wholly dependent upon groundwater resources. It is clear that a proper scientific diagnosis and prioritization of the problems affecting each aquifer are critical in order to design an effective urban groundwater protection strategy. This in turn is dependent upon the methodical collection and reliable interpretation of accurate groundwater quality and exploitation data. The establishment of a continent-wide monitoring network of aquifer surveillance will contribute to effective groundwater protection by enabling the consistent assessment of the extent, rate and progress of aquifer degradation across the region.

The UNEP/UNESCO project initially covered six French-speaking countries and one English-speaking country in West Africa (Benin, Côte d'Ivoire, Niger, Ghana, Mali, Burkina Faso and Senegal), before expanding to encompass an additional four English-speaking countries in East and Southern Africa (Ethiopia, Kenya, Zambia and South Africa) in 2004.

This publication provides an assessment of the current status of both shallow and deep-seated aquifers in some of the most groundwater dependent cities in these 11 countries. It also provides vital information on the methodologies developed for the monitoring, management and sustainable use of Africa's urban aquifers. In the first two years, this involved a uniform methodology for sample collection and analysis, together with basic measures for pollution reduction, while in the second phase more sophisticated techniques were developed for water sampling, vulnerability mapping, the interpretation of results, and their dissemination to governments, policymakers and the general public.

It is becoming increasingly clear that the availability of groundwater resources will have a critical impact on the success of future efforts to alleviate poverty across Africa. The Water Agenda of the New Partnership for Africa's Development (NEPAD) and the constitution of the Africa Ministerial Council on Water (AMCOW) exemplify positive recent developments in the pursuit of this objective. At the 2003 Pan-African Implementation and Partnership Conference on Water, African ministers pledged their commitment to the coordinated, multi-sectoral development of the continent's water resources in order to provide sustainable supplies for urban development, agricultural and industrial uses, low and flat lands management, and other activities covered by new Integrated Water Resource Management (IWRM) policies. In November 2005, this commitment was further advanced at South Africa's University of the Western Cape, where an international groundwater workshop brought together some of the continent's leading scientists, water authorities and NGOs with key international supporters to develop strategies to manage and protect Africa's groundwater resources.

The information and recommendations contained in this publication build upon the pollution and risk assessments carried out by the UNEP/UNESCO project, as well as groundwater protection experiences in a number of developing and developed countries, and attempt to develop a groundwater protection strategy specifically designed for Africa's physical and socio-economic environment. It is our hope that this information will go some way towards assisting policymakers, institutions and other stakeholders in addressing what remains one of the greatest challenges facing the future of this continent.

Dr. Achim Steiner,
Executive Director
United Nations Environment Programme

Message from UNESCO

Dr. A. Szöllösi-Nagy
Director of Water Science, UNESCO, Paris

This book is largely based on the investigation or review of the situation of aquifer's water supply in sub-Saharan African cities, from the UNEP/UNESCO project on 'Assessment of Pollution Status and Vulnerability of Water Supply Aquifers of African Cities'. Strategically this project resulted in the setting up of groundwater monitoring networks in the eleven participating countries and provided groundwater quality information through various aquifer vulnerability maps and bulletins. National water managers and planners were thereby provided with information pertaining to water quality trends and status in both space and time for resource planning, management purposes and policy formulation for groundwater protection and contamination prevention.

Degradation of groundwater is one of the most serious water resources problems in Africa. Groundwater is vulnerable because of its relation to surface water and land use activities where pollution often occurs. Groundwater is particularly problematic in areas where the aquifers provide a large part or are the sole source of water supply. Because groundwater movement is usually slow, polluted water may go undetected for a long time. In fact, most contamination is discovered only after drinking water has been affected and people become sick. By this time, the volume of polluted water may be very large, and even if the source of contamination is removed immediately, the problem is not solved. Although the sources of groundwater contamination are numerous, the solutions are relatively few, and the focus must be on prevention of the problem.

The UNEP/UNESCO project is aligned with the focus of the Sixth International Hydrological Programme (IHP) of UNESCO (2002–2007) on 'Water interactions: systems at risk and social challenges'. Water is an integral part of the environment and its availability is indispensable to the efficient functioning of the biosphere. Water is of vital importance to all socio-economic sectors – human and economic development simply is not possible without a safe, stable water supply. IHP VI seeks to contribute to our knowledge of the state of global freshwater systems, stresses the importance of a holistic, integrated groundwater management approach, and emphasizes the social, economic, cultural and environmental value of water.

Therefore, it is a great pleasure for UNESCO/IHP to be part of this important endeavor and we are very pleased to present this book, as a concrete outcome of the fruitful cooperation between UNESCO, UNEP and the UNESCO Chair on Groundwater based at the University of the Western Cape over the course of the project. We believe that in order to achieve and maintain an acceptable standard of living, access to a safe, reliable source of clean drinking water is essential. The content of this book has highlighted a range of groundwater pollution problems and clean drinking water will become a viable objective only by keeping our valuable water resources as pollution – free as possible. The challenge now is to develop an improved understanding of groundwater water quality and respond to the growing need for improved water services.

Acknowledgements

This book contains a selection of papers that initially originated from an eleven-country UNEP & UNESCO project, entitled 'Assessment of pollution status and vulnerability of water supply aquifers of African cities'. A call for papers resulted in submissions from across the African continent and beyond. Several organizations and project management teams are thanked for their support and input over a number years, during and after the project. The support of the following organisations is gratefully acknowledged and has facilitated the realisation of this book:

- Belgium Development Cooperation
- Development Cooperation of Ireland
- UN Development Account
- UNEP – United Nations Environment Programme
- UNESCO – United Nations Education Educational, Scientific and Cultural Organization
- UN HABITAT – United Nations Human Settlements Programme

The project was led by Prof Dr. Salif Diop from the Water Unit and Ecosystems Section of the Division of Early Warning and Assessment of UNEP Nairobi, and implemented by Dr. Emmanuel Naah from UNESCO Nairobi. Dr. Loic Giorgi was involved in the initial coordination and consulting services. The coordination and training functions were largely provided by the Centre of the UNESCO Chair on Groundwater, based at the University of the Western Cape (UWC), South Africa. The effort of Mr. Patrick Mmayi of UNEP Nairobi with the coordination of much of the logistics deserves special mention.

During the project, many short courses for capacity building were organized by the UNESCO Chair, and provided by various experts, as follows:

- GIS application for African hydrogeologists by UWC (Dr. R. Knight) and GEOSS (Mr. J. Conrad)
- Introduction to groundwater protection in Mombasa, Kenya by UWC (Prof. Y. Xu) and the University of the Free State (Dr. B. Usher)
- Groundwater interaction with surface water by UWC (Prof. Y. Xu, Prof. L. Raitt, Dr. A. Thomas, Mr. A. Scheepers) and CSIR (Mr. J. Zhang)
- Modelling and its calibration at UWC by USGS (Dr. M. Hill and Ms. C. Tedman)
- Assessment of fractured rock aquifer at UWC by USGS (Dr. A. Shapiro) and staff of the UNESCO Chair at UWC
- International workshop on groundwater protection for Africa at UWC

The editors would like to thank the following members of the review board, who made valuable contributions to the quality of the work and its improvement through their willingness to help.

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Our gratitude goes to all the contributors, regardless of whether they are a part of the project or not, who made a proud strike to report their work from a variety of African aquifers under different circumstances.

The publication of this book was made possible through financial support from the UN Development Account, UNEP and the UNESCO Nairobi Office. While many organisations have increasingly focused on global water issues, we gratefully acknowledge the UNEP Nairobi office for maintaining its interest in, and concern about, groundwater pollution issues in western, eastern and southern Africa.

All the authors and their affiliated organisations are gratefully acknowledged for their technical input and kind support.

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Introduction

1

Issues of groundwater pollution in Africa

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ABSTRACT: Groundwater quality issues in African countries are very complicated. Human activities result in the groundwater resources being polluted, aquifers being incorrectly exploited and utilised and abstraction facilities (boreholes or wells) being vandalized.

Based on the contribution that follows in this book, the groundwater quality problems are discussed under the main headings (1) resource value versus access, (2) contamination source identification and prioritization, (3) vulnerability and protection and (4) borehole performance and monitoring.

1 INTRODUCTION

The African continent is believed to be the cradle of humanity. From satellite imagery it looks innocent and peaceful, but this landmass suffers from poverty and hardship resultant of natural and man-induced disasters. General features of water in Africa are associated with highly variable rainfall, extreme flooding events and frequent droughts. Water supplies tend to be vulnerable to contamination from various sources, and can be the vector for water-borne diseases. However, much effort has been made in recent decades to improve water supply and provide adequate sanitation. Groundwater resources have been increasingly used to rectify the supply backlogs and many urban areas rely on groundwater as a fundamental part of the drinking water supply.

Based largely on a recent UNEP-UNESCO project entitled ‘Assessment of pollution status and vulnerability of water supply aquifers of African cities’, this book is aimed at disseminating the knowledge gained during the project and highlighting the African capacity for groundwater science and technology. With groundwater as the sole source or major economic supply in urban areas of Africa, contamination and pollution problems should be dealt with in a scientific manner. The book, consisting of a series of peer-reviewed papers, will provide a window through which current issues of groundwater quality in Africa can be viewed, and thoroughly discussed with the initiation of solutions. As the papers presented demonstrate, there are several challenges in terms of the current regulation, protection and technical proficiency on the continent that require attention.

2 GROUNDWATER – THE HIDDEN TREASURE IN AFRICA

Unlike surface water, groundwater accumulates and flows underground in both unconsolidated sediments and hard rock formations. It is reported that the major aquifers occur in the African platform and in the folded zones (United Nations, 1988; United Nations, 1989; Zektser and Everett, 2004). They are characterized by dominantly localized aquifers with a few transboundary systems significant

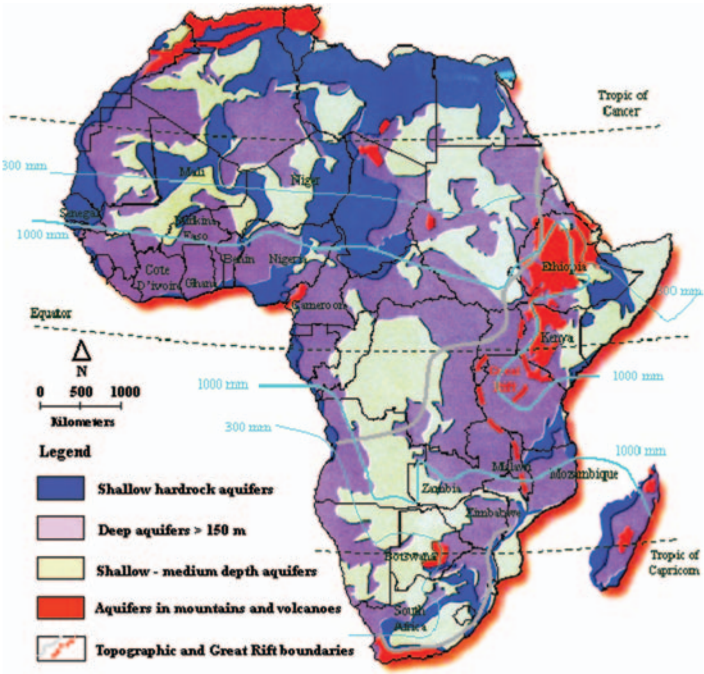

Figure 1.1. Simplified hydrogeological map of African countries involved in the studies.

at regional scale (Fig. 1.1). The regional groundwater flow basins in Africa can be divided into two zones by the equator (Zektser and Everett, 2004): (1) the arid and semi-arid zone north of the equator consisting of the Nubian, Sahara, Chad, Niger, and Taoudeni basins, and (2) the zone south of the equator, including the equator zone, composed of the Congo, Kalahari and Karoo basins. Due to socio-economic development, several regional aquifer systems have been recognised along coastal Africa. Among them is the TMG (the Table Mountain Group Sandstone) aquifer in South Africa, which has received a great attention from local water resource planners (Pietersen and Parson, 2002).

For Sub-Saharan Africa, MacDonald and Davies (2000) report that the unconsolidated sediments occupy only 22% of the land area, whereas the remaining 78% are found in basement (40%), sedimentary (32%) and volcanic (6%) rocks. This means that the people in vast rural areas have to rely on hard rock aquifers locally available to them.

The socio-economic and ecological values of groundwater can be easily appreciated from its role played in the history. The first human settlements may have been established around lakes and springs along the Great Rift Valley in eastern African a couple of million years ago, but since then water availability has driven many of the settlement patterns and socio-political changes across the continent. The history of the water use by humankind evolves in three phases.

- Epoch 1 (Dawn of civilization to last century): Water is nature's gift to mankind. During this period the spring, for instance, is essential for life. Cape Town was originally named Camissa by the Khoisan tribe, meaning 'the place of sweet water' and their name for Table Mountain was Hoerikwaggo, meaning 'the mountain in the sea' (Yeld, 2004).
- Epoch 2 (Late 1800s to present): Borehole water sustained the rural livelihoods and urban development. Construction of wells and boreholes to meet various water supplies for village, town and industrial development may come relatively late in the African continent, but the availability

Table 1.1. Groundwater resources use compiled by African countries involved in the UNEP & UNESCO project.

Country	Population (Million)	Surface water (10^9 m^3)	Groundwater (10^9 m^3)	Groundwater (%)
Botswana	1.6	1.7	1.7	50.00
Burkina Faso	12.6	13	9.5	42.22
Cameroon	16.2	265	57	17.70
Cote d'Ivoire	16.8	74	37.7	33.75
Ethiopia	67.7	202	45	22.28
Ghana	20.3	29	26.3	47.56
Kenya	31.2	17.2	3	14.85
Mali	11.4	50	20	28.57
Niger	10.7	1	2.5	71.43
Nigeria	130.0	214	87	28.90
Senegal	10.6	23.8	7.6	24.20
South Africa	43.7	40	4.8	10.71
Zimbabwe	11.4	13.1	5	27.62

of borehole water is vital for community's prosperity. In South Africa farm names often ended with 'fountain', 'oog', etc. as many European settlers stocked or cultivated farmlands in the vicinity of streams or springs.

- Epoch 3 (1990s to the present): Groundwater is critical for maintenance of healthy ecosystems. Emphasis is placed on the integration of groundwater with surface water for sustainability of human endeavours and ecosystems. The introduction of the human dimension also implies that anthropological impact on water quality must be assessed.

As pointed out by Weyman (1975), surface water is usually found in one of three capacities: either too much, too little, or too dirty. Unlike surface water bodies, groundwater as the hidden treasure is often out of sight until the aquifer is pierced through by a borehole or well. Access to such a water point is complicated by land ownership policies within various countries. Thanks to both local and international efforts for sustainable development, the importance of groundwater as the hidden treasure is increasingly recognized and appreciated through the adaptation of various legal frameworks in Africa (Table 1.1).

3 KEY ISSUES OF GROUNDWATER CONTAMINATION

In addition to the lack of appropriate management of groundwater resources in Africa, the following 4 key issues are identified as causes of water quality deterioration, as illustrated in Figure 1.2:

1. Resource value versus access
2. Contamination source identification and prioritization
3. Vulnerability and protection
4. Water supply and monitoring

A brief discussion of these key issues is based on an understanding of papers that follow in this book.

Resource value versus access. Water resources need to be classified and valued for management purpose. More often than not, small villages are unable to afford piped water from outside their village boundaries. This information can be used by the decision maker to allocate and manage the resources. In ideal circumstances, water is considered in a framework of social justice, economic affordability and technically feasibility.

Contamination source identification and prioritization. The pollution sources vary from place to place. Accurate identification of these sources would help the authorities and communities to

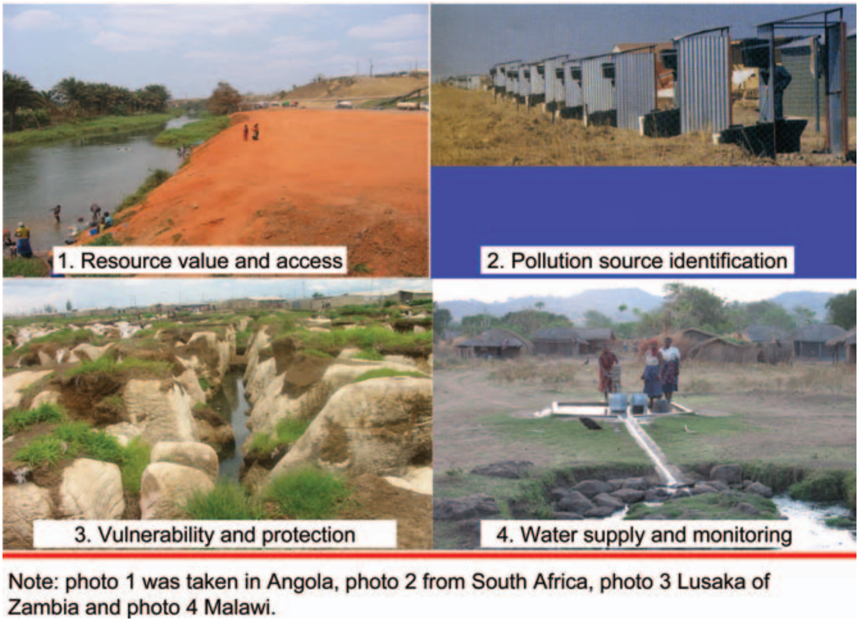

Figure 1.2. Key issues of groundwater contamination in Africa.

focus on prioritized problems with the limited resources available, in order to minimize the negative impact of pollution.

Vulnerability and protection. Groundwater vulnerability assessment proves useful in identifying 'hot spots' at regional scales. The concept of vulnerability assessment should also be applied to the resource point or wellfields. Groundwater vulnerability tends to address probability of the vertical movement of pollutants through the unsaturated zone, whereas wellhead protection is often based on horizontal flow dynamics in the saturated zone. The wellfield vulnerability assessment requires the combination of the two in order to attain the objective of sufficient, appropriate quality and water supply.

Borehole performance and monitoring. Boreholes and hand dug wells are reliable sources of water supplies, especially in rural Africa. Adequate protection measures such as borehole sanitary seals should be in place to safeguard basic water sources. This should be supported by regular water quality monitoring and these assessments should be calibrated through a dedicated monitoring programme.

In general, there is insufficient strategic management in Africa and the water crisis has been managed without satisfactory strategic planning. Ownership of boreholes or access to water points is often one of the causes of conflict. The vandalism, destruction or contamination of groundwater abstraction points may be used as a weapon during conflict.

Due to the lack of good governance for groundwater development and utilisation the degradation of groundwater resources is taking place. The authors' recent visits to eight African countries confirmed such a trend. In the case of Lusaka, the country is heavily dependent on the karstic groundwater resource underlying the city. The karstic aquifer (marble cavities) is interconnected through a network of conduits through which groundwater flows. The sanitation facilities, mostly in form of toilets (often unimproved pit latrines) with limited sewer facilities e.g. in the John Laing area, are identified as a major source of contamination. As a result, water-borne diseases such as cholera and dysentery occur from time to time. This scientific information needs be translated into language that decision makers would understand and would inspire them to do something about it.

What legal provision and policies must be in place to improve the situation? It is reported that good legal frameworks are in place in many African countries. The laws are there, but are often toothless! The challenge now lies in the satisfactory implementation of these policies throughout the African continent. In South Africa, for example, the government has introduced the Water Service Act of 1997 and the Water Act of 1998, which have had tremendous impact in the water industry and on potential impactors on the water resources.

4 THREATS TO GROUNDWATER

Understanding the flow mechanisms and transport of contaminants in fractured rock aquifers prevalent in Africa proves to be very challenging, especially in most African countries where resources are limited.

The chemistry of groundwater is generally conditioned by the nature of the rock formations through which it flows, as illustrated in the case of Cameroon. Despite the fact that the dissolved content in groundwater is normally higher than that of surface water, it is naturally protected from surface pollution and therefore often potable as the subsurface provides natural attenuation processes for common contaminants such as bacteria. There are tremendous backlogs in provision of water and sanitation in the continent. One estimate suggests that there are more than 300 million people who have inadequate access to safe water in Africa. Groundwater can play an important role in eradicating these backlogs. This prospect could be severely jeopardized by inadequate control or management of groundwater qualities.

Based on near 30 papers received from 16 countries (7 from Northern Africa, 1 Central Africa, 2 Eastern Africa, 4 Southern Africa and 2 Europe), the major sources of contamination reflected in this book are as follows:

- on-site sanitation
- solid waste dumpsites, including household waste pits
- surface water influences
- agricultural chemicals (fertilisers, herbicides and pesticides)
- petrol service stations (underground storage tanks)
- mismanagement of wellfields.

A list of contaminants one would expect from the above sources is as follows:

- microbiological contaminants, typically viruses, bacteria, and protozoa
- chemical contaminants, consisting of both inorganic and organic components. The inorganic components of primary concern are nitrogen species. The organic components of primary concern are toxins and those that decay rapidly, depleting the oxygen in the carrier water or forming odorous by-products.

5 DISCUSSION

The above list may be expanded by adding the quality variations caused by other processes e.g. fluoride, biofouling and sea water encroachment. Even so, such a list may still be incomplete because the lack of monitoring programmes and laboratory facilities in many African countries results in no detection of certain problem constituents in groundwater.

On a global scale, major problems of water quality can be characterised by pathogenic agents, organic pollution, salinisation, nitrate pollution, heavy metals, industrial organics, acid mine drainage and hydrological modification. In Africa, the major issues can be listed in order of importance as follows: (1) Nitrate pollution, (2) pathogenic agents, (3) organic pollution, (4) salinisation, (5) acid mine drainage.

From the above discussion it is evident that groundwater supplies in Africa are increasingly threatened by human activities, which result in the groundwater resources being polluted, aquifers being incorrectly exploited and utilised and abstraction facilities (boreholes or wells) being vandalised.

Once groundwater is polluted, it is very difficult, if not impossible, to remediate. With increasing incidences of water pollution reported in the continent, the availability of the fresh water resources, including groundwater for future generations, is getting more and more problematic. It is possible that the groundwater industry may in the future no longer be able to exploit the treasure hidden underground, due to both technical and economical unfeasibility. Unfortunately this scenario is not fiction. It will be a reality, unless we take immediate action in adopting and implementing sustainable development strategies across the continent.

Groundwater has been recognised as playing a very important role in the development of the continent. As such, significant resources have been made available in different African countries for the location and establishment of groundwater supply schemes. The overwhelming majority of groundwater-related work is focused on the successful location, drilling and sustainable supply from aquifers for the purposes of water supply for domestic use, stock watering and irrigation. Due to the extent of the continent and the need for supply for basic human needs and poverty alleviation, it is natural that the focus is on supply from groundwater. The political pressure is also on delivery and the consensus is that communities will only concern themselves with the quality of the water when there is enough to meet these needs.

As a result of this, the focus of funding, research, education and technical proficiency is largely related to groundwater supply. However, the supply of poor quality groundwater will have far-reaching impacts, and this book showcases the continental effort into aspects of groundwater protection.

The 29 peer-reviewed papers included in this book indicate an emerging network of African specialists dealing with the problems. Ideally Africa would have enough expertise to tackle these issues, and through expanded education and training programs, this is possible in the longer term.

The current level of expertise in the continent is growing, but budgetary constraints and the historical research focus has meant that contaminant hydrogeology is still in its infancy. The papers presented are therefore the start of a long road towards including water quality protection of groundwater in the strategies of governments, training institutions and funding agencies. Despite these limitations, the efforts presented here will add to the understanding of the problems, and begin the process of implementing the necessary measures to ensure true ‘sustainable development’.

6 WAY FORWARD

The range of papers presented here illustrates the pollution problem endemic in Africa, but they also seek solutions to mitigate and improve the situation. The problems associated with rapid urbanisation, inadequate sanitation, lack of alternative water supplies and lack of public awareness and implementation require urgent addressing.

Groundwater protection is not optional; there MUST be groundwater protection strategies in place in developing countries. There are many factors that influence the design and implementation, including availability of scientific information, socio-economics, institutional arrangements and legal frameworks. The most important issues to be addressed are:

- Political will: Groundwater quality protection is closely related to the government policy towards economic development and the political will for sustainable development and utilisation of resources. The correct regulations are a good start but implementation of such policies is the key to successful protection, i.e. implementation of the resolutions taken at the Pan Africa Conference on Water (Dec 2003 in Addis Ababa) organized by AMCOW (Africa Ministerial Council on Water).
- Knowledge dissemination: The small number of technical people involved in groundwater in the continent are well aware of the problems and approaches to be followed. This needs to be fed

through to decision makers and the broader public in ways that effectively communicate the issues, their consequences and solutions. Only through the technical aspects being communicated and accepted by the stakeholders (the public and regulators/decision-makers) will implementation be appropriate and successful.

- Capacity building: By comparison, Africa has little research capacity. It is for that reason that the review process in which the board of reviewers participated, consolidated and enhanced the network of active water scientists for Africa. Such capacity can be increased by
  - Establishing more formal networks of African universities working on water and sanitation
  - Improving communication by increasing access to internet facilities.

In Africa this will mean a change from the mode of 'buying service' to a combination of 'institutional development' with 'technology transfer', which will lead to local expertise being created and utilised.

Groundwater contamination and pollution problems are a growing threat to African development, as highlighted by the papers in this book. These aspects deserve far more attention, and insufficient research has been done to date on these topics. This has resulted in uninformed decisions by authorities, planners and regulators.

Attention should be paid to the following areas.

- Development of water resource protection strategies, including the socio-economic and institutional aspects thereof
- Establishment of early warning monitoring system
- Incorporation of freshwater indicators into resource management
- Knowledge transfer at various levels, from grass roots awareness to ministerial consideration of groundwater pollution protection as part of planning and implementation.

Africa has survived natural disasters in various guises; we cannot allow ourselves to be plagued by man-induced pollution.

REFERENCES

DWAF, 1997. A protocol to manage the potential of groundwater contamination from on site sanitation (Edition 1), The National Sanitation Coordination Office & Directorate of Geohydrology at the Department of Water Affairs and Forestry, Pretoria, 1997.

DWAF, 2003. A protocol to manage the potential of groundwater contamination from on site sanitation (Edition 2), The National Sanitation Programme, the Department of Water Affairs and Forestry, Pretoria, March 2003.

Fourie AB and Van Ryneveld MB, 1995. The fate in the subsurface of contaminants associated with on-site sanitation: A Review, Water SA, vol.21, no.2, April, pp.101–112.

MacDonald and Davies, 2000. A brief review of groundwater for rural water supply in sub-Saharan Africa, Technical Report WC/00/33, Overseas Geology Series, British Geological Survey.

Pietersen K and Parsons R, 2002. A synthesis of the hydrogeology of the Table Mountain Group – formation of a research strategy, WRC Report No TT 158/01, Pretoria, January 2002.

United Nations, 1988. Ground water in North and West Africa, Natural Resources/Water Series No. 18, United Nations, New York.

United Nations, 1989. Ground water in eastern, central and southern Africa, Natural Resources/Water Series No. 19, United Nations, New York.

Weyman DR, 1975. Runoff processes and streamflow: theory and practice in geography. Oxford University Press, Oxford.

Xu Y and Braune E, 1995. A guideline for groundwater protection for community water supply and sanitation programme, Pulished by the Department of Water Affairs and Forestry, Pretoria, 1995.

Yeld J, 2004. Cape Town News paper 'Weekend Argus', South Africa, March 13, 2004.

Zekster and Everett (eds), 2004. Groundwater resources of the world and their use, UNESCO IHP-VI, Series on groundwater No.6, ISBN 92-9220-007-0.

Overview

Editorial Comment: Overview

The collection of papers in this book is significant, not for the scientific content necessarily, but especially for the knowledge dissemination of ‘hidden’ data on African groundwater quality. Many of the papers contained would not seem particularly noteworthy in isolation, but the cumulative worth of this collection in increasing our understanding of the continent’s groundwater pollution issues provides a clear insight into the problems and the technical level of proficiency available to deal with them.

The first section of this book provides a set of papers that provide a broad overview of the situation in Africa. These papers highlight topics of special significance to the continent, and some broad country perspectives, whereas the subsequent section provides specific case studies to substantiate these general perspectives.

Nitrate, as will be emphasized by the majority of the papers, is a contaminant of great significance on this continent. The first paper is a detailed discussion on its occurrence, the hydrochemical interactions, its effects and some specific cases to illustrate these. It is noted in Tredoux & Talma’s paper that nitrate values in excess of 500 mg/l have been measured in southern African groundwater. The causes are numerous but their conclusions that in most cases, the occurrence of high levels of nitrate in groundwater is due to pollution related to anthropogenic activities is of great importance. Of particular significance is the finding that inappropriate on-site sanitation at rural villages and towns frequently leads to groundwater pollution by nitrate and the abandoning of wellfields, as this is mirrored by many of the cases studies which follow.

This is followed up by the situation in Nigeria, which builds on these concepts and shows the validity in a different area of Africa, and indicates that over a third of the shallow wells have qualities that make them unsuitable for long term use. Three country perspectives, from Cameroon, Ghana and Zimbabwe, are provided in this section. These papers give national perspectives rather than the explicit cases in the next section, and give the reader a feel for the type of issues to be addressed at national level. The shallow groundwater in Cameroon is highly polluted with very high microbial contamination, while in Ghana it is of interest that more than two thirds of the population is rural but may still be effected by aspects such as elevated metals, acid mine drainage, and naturally occurring flourosis. The recurrence of themes is highlighted in these national reviews, with many of the sources and effects from Love *et al.*’s paper for example consistent with Vogel and Adelana in later papers, and the topics covered in the later sections reiterate these general findings with site-specific values.

The continental issue of pesticide pollution is dealt with in this section, and highlights the potential time bomb lying in wait, with for example over 200 metric tones of redundant pesticides stockpiled throughout Zambia. A later paper refers to elevated DDT levels leading to warning being issued in Mali, showing this to be a very real issue beyond the agricultural areas.

The final paper in this section emphasizes the socio-economic and technical challenges of dealing with the groundwater pollution problems. The inadequate urban sanitation infrastructure with an estimated 150 million urban Africans without access to adequate provision for water and up to 180 million with inadequate sanitation provision, poor wellhead construction, lack of groundwater protection zones and rising urban populations discussed in this paper, are articulated in all the case studies presented in this book. The social aspects and the economic challenges (and the move toward demand management of water) in dealing with urban development are valid considerations, and the importance in this section is that these issues have a direct and negative impact on groundwater quality.

Africa has particular challenges, but these are not unique to this continent. The groundwater issues raised here apply equally to any other developing city where the rate of settlement and growth outstrips the city’s ability to provide the necessary services. It is hoped that these broad issues place the continent in context, laying the foundation for the subsequent sections. The implications should facilitate enhanced actions to protect the groundwater resources of Africa.

2

Nitrate pollution of groundwater in southern Africa

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ABSTRACT: Groundwater nitrate exceeds drinking water specifications in many parts of the southern African region. High nitrate levels in drinking water lead to infant methaemoglobinaemia and, at higher levels, livestock poisoning. Both are potentially fatal. A simplified map of nitrate occurrences in Botswana, Namibia and South Africa indicates regionally significant areas of high nitrate levels. Case studies in some of these areas show the extent of nitrate pollution observed during the past two decades. Isotopic methods indicated that the nitrate in groundwater can be derived from natural or anthropogenic sources. Nitrate originating from anthropogenic sources is the major problem and needs to be approached as an environmental management issue. Cattle husbandry, in the form of feedlots and concentrations around watering points, has great pollution potential and has caused fatalities. On-site sanitation systems also caused fatalities and have led to the abandoning of well fields. The impact of industry and sewage sludge disposal seems to be locality limited, while fertilizer use by agriculture was low and did not have any significant effect on groundwater. The lack of recorded information on the occurrence of infant methaemoglobinaemia and livestock losses tends to obscure the seriousness of the situation among developing nations. There is a need for a better understanding of the processes leading to nitrate pollution, natural enrichment, and impact on the health of the population.

1 INTRODUCTION

The single most important reason for groundwater sources in South Africa to be declared unfit for drinking is nitrate levels exceeding 10 mg/L (Marais, 1999). Nitrate in groundwater can be derived from natural or anthropogenic sources. The presence of nitrate from anthropogenic sources is largely an environmental management issue and needs to be approached as such. High nitrate levels in drinking water cause infant methaemoglobinaemia and, at higher levels, livestock poisoning. Both are potentially fatal. Risks associated with infant methaemoglobinaemia are multiplied by bacterial pollution, malnutrition, and weak health, e.g. immune compromised individuals. Although no statistics are available on morbidity and mortality, it is known from the literature that infant methaemoglobinaemia occurs in southern Africa (Super *et al.*, 1981, Hesseling *et al.*, 1991, Tredoux *et al.*, 2005). Similarly, no statistics are available on the loss of livestock due to nitrate poisoning, but considerable losses have been incurred over the last three decades (Anonymous, 1974, Marais, 1991, Tredoux *et al.*, 2005). The overall lack of recorded information on the occurrence of infant methaemoglobinaemia and livestock losses tends to obscure the seriousness of the situation among developing nations.

South Africa follows the lead of Europe and the USA to consider a nitrate concentration of less than 6 mg/L (as N) as *ideal* for drinking water. In Western Europe, drinking water is generally denitrified to a level of 5.5 mg/L. However, the South African guidelines are somewhat more lenient,

Table 2.1. South African water quality guidelines for nitrate and nitrite for potable use and livestock watering.

Drinking water class	as N*	as NO_3	Comments
Nitrate plus Nitrite (mg/L)			*(DWAF, 1998)*
Ideal	**<6**	<26	Negligible health effects
Acceptable	**6–10**	26–44	Insignificant risk
Marginal	**10–20**	44–89	Slight chronic risk to some babies
Poor	**20–40**	89–177	Possible chronic risk to some babies
Unacceptable	**>40**	>177	Increasing acute health risk to babies
Livestock watering	as N	as NO_3	
Nitrate (mg/l)	0–90.3	**0–400**	*(DWAF, 1996)*
Livestock watering	as N	as NO_2	
Nitrite (mg/L)	0–12.2	**0–40**	*(DWAF, 1996)*

* Numbers in bold indicate units specified in the guidelines.

and denitrification is generally considered only when acceptable or marginal limits are exceeded (Table 2.1).

In many parts of the Kalahari nitrate is common in groundwater. Occasionally concentrations are well below the 'acceptable' drinking water specification of 10 mg/L (as N), but in certain instances nitrate levels far exceed the guideline values reaching levels of several hundred mg/L. This has particularly been observed after high seasonal rainfall. A simplified map of the groundwater nitrate distribution (Tredoux *et al.*, 2001) shows the areas where nitrate (as N) exceeds 20 mg/L, i.e. of 'poor' quality, at twice the recommended limit of 10 mg/L set by the World Health Organization (WHO, 1998).

The map shows that high nitrate levels occur extensively in the northern parts of South Africa, the south-eastern parts of Botswana, and also in the south-eastern parts of Namibia (Fig. 2.1). It is also evident that the area with high nitrate in south-eastern Namibia, at the western edge of the Kalahari Desert, links to an adjoining area in northern South Africa with high nitrate. Similarly, the occurrence of high nitrate in central Namibia extends eastwards to western Botswana.

Whereas the south-western Kalahari is sparsely populated, the northern parts of South Africa and the south-eastern part of Botswana have a relatively higher population density. This indicates that diverse processes or combinations of processes may be operative in the different areas.

The extensive occurrence of nitrate in groundwater along the western edge of the Kalahari is primarily ascribed to natural causes, and this was indeed confirmed by ^{15}N isotope determinations (Heaton, 1984). On the other hand, the high nitrate levels causing stock losses in the area extending from Ghanzi in Botswana to Gobabis in Namibia are mainly pollution related (Tredoux *et al.*, 2005).

The distribution and sources of nitrate in groundwater have been studied in some detail in Botswana (Lewis *et al.*, 1978, Lewis *et al.*, 1980, Lagerstedt *et al.*, 1994, Jacks *et al.*, 1999, Staudt, 2003, Vogel *et al.*, 2004, Schwiede *et al.*, 2005, Stadler, 2005), Namibia (Kirchner & Tredoux, 1975, Heaton, 1984, Tredoux & Kirchner, 1985, Wrabel, 2005), and South Africa (Heaton, 1985, Connelly & Taussig, 1991, Tredoux, 1993, Tredoux, 2004). These studies have shown that pollution by anthropogenic activities is the main source of high and variable nitrate levels. Such activities include inappropriate on-site sanitation and wastewater treatment, improper sewage sludge drying and disposal, and livestock concentration at watering points near boreholes. Other activities, such as the tilling of soil, will contribute through mineralisation of the natural soil nitrogen. The case studies identify a number of the main sources of nitrogen to be managed for groundwater protection. The interplay between natural and anthropogenic groundwater nitrate enrichment processes can complicate source identification in certain instances. This paper discusses a number of cases of anthropogenic groundwater pollution, but this cannot be viewed in isolation from natural enrichment processes or cases where natural soil nitrogen is mobilised due to anthropogenic

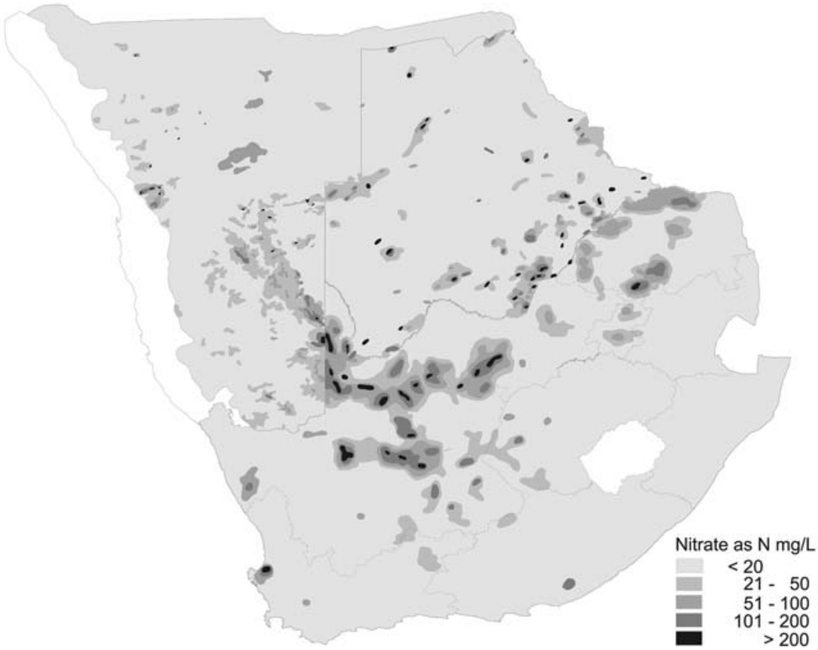

Figure 2.1. Simplified map showing the distribution of nitrate in groundwater in Namibia, Botswana and South Africa (*Tredoux et al., 2001*). White area along Namibian coast denotes sparse data.

activities. In urban industrial pollution, the source may be clearly identifiable due to substantial nitrogen isotope fractionation (Talma & Meyer, 2005).

The management approach for dealing with high levels of nitrate in groundwater depends on the levels observed and the purpose for which the water is required. Solutions may range from finding an alternative source to distributing nitrate free bottled water to people at risk. These are short-term solutions. The only sustainable longer-term solution is to prevent anthropogenic nitrogen from reaching groundwater. This requires both legislative control and public awareness campaigns, as water users need to realise that harmful practices cause them to pollute their own water sources.

2 ACCURACY OF ANALYTICAL RESULTS FOR NITRATE

Since nitrogen compounds serve as nutrients for most organisms, the concentrations of nitrogen compounds in water samples change with relative ease when samples are stored without precautionary measures for any length of time. Bacterially polluted water samples are particularly unstable. The accuracy of groundwater nitrate data is of major importance when attempting to compare results for identifying concentration trends over time. Analytical inaccuracies may be present in some of the older data, as nitrate analysis was less reliable several decades ago.

Nitrate is usually determined by reduction to nitrite, and many laboratories are therefore content to report the total concentration of nitrate and nitrite as only one result (called 'NO_x-N'). When nitrate concentrations are high, they do not create a problem, as nitrite concentrations are generally much lower. However, when nitrate concentrations are low and the water is polluted, the error may become significant.

In this paper nitrate concentrations are expressed as an equivalent quantity of nitrogen. This allows the direct comparison of molar concentrations of the various nitrogen compounds, e.g. ammonium, nitrite, and nitrate, when all are expressed in this form. The conversion factor for converting Nitrate-N to Nitrate-NO_3 is 4.43.

3 ISOTOPE DETERMINATION FOR SOURCE IDENTIFICATION

The nitrate ion (NO_3^-) consists of both nitrogen and oxygen atoms. Nitrogen and oxygen have several isotopes (^{14}N and ^{15}N; ^{16}O, ^{17}O and ^{18}O respectively). In this paper the ratios $^{15}N/^{14}N$ and $^{18}O/^{16}O$ are discussed. For ease of reporting, the ratios are expressed as differences (δ, in per mil = parts per thousand) from a standard that is universally used: atmospheric air (AIR) for nitrogen and Standard Mean Ocean Water (SMOW) for oxygen. The two isotope ratios of NO_3 are, therefore, reported as $\delta^{15}N$ (‰ AIR) and $\delta^{18}O$ (‰ SMOW).

The characteristic $^{15}N/^{14}N$ ratios of industrially produced nitrate (fertiliser, explosive, chemicals), naturally occurring soil derived nitrogen and faecally produced nitrate, are sufficiently different to use the characteristic ratios to identify nitrate sources in groundwater (Heaton, 1986; Kendall, 1998). Industrial processes frequently involve phase changes of gaseous, dissolved, and liquid forms of nitrogen, usually as ammonia/ammonium (Talma & Meyer, 2002, 2005). Such phase changes can generate significant changes in the isotope ratios of the nitrogen products and effluents produced. Such ratios serve a useful purpose for tracing pollution underground.

Kreitler (1975) was the first to show that different sources of nitrate in groundwater can be distinguished by different $^{15}N/^{14}N$ ratios. This established the basis for the use of this approach by many studies for the source apportionment of nitrate (see reviews by Heaton, 1986 and Kendall, 1998). Distinct isotopic differences can be seen when the nitrate sources are classified as rain, animal and human waste, natural soils and fertilisers. While there is an overlap of ranges between the different sources, a practical distinction for most samples can be made (Table 2.2) and is frequently used as such.

^{18}O in nitrate serves as an additional tracer for identifying sources and processes. Its application is based on the source of the oxygen in the nitrate ion. $\delta^{18}O$ in most natural water ranges from −5 to 0‰ SMOW, while that in atmospheric oxygen is much higher at +23‰. This significant isotopic distinction is geochemically transferred, so that the nitrate formed in soils and water bodies can be distinguished from nitrate fertilisers, which are industrially produced with air as oxygen source. Nitrate in rainfall derives at least one of its oxygen atoms from atmospheric oxygen. In addition, there are large ozonation effects, which react with high isotope fractionation. The result is that the ^{18}O contents of rainfall nitrates can vary in the range of +23 to +70‰ SMOW. Nitrates in desert deposits are solely derived from rainfall and carry the high rainfall ^{18}O isotope signature. Plots of ^{15}N and ^{18}O clearly show the influence of different sources and have become quite versatile tools for source identification (Kendall, 1998).

The process of denitrification can clearly be identified in ^{15}N – ^{18}O plots along a slope of 1:2. This is a consequence of the bacterial processes responsible. A set of samples of similar source and different stages of denitrification will plot along such a typical straight line in ^{15}N and ^{18}O space (Böttcher *et al.*, 1990).

Table 2.2. Distribution of $\delta^{15}N$ in groundwater nitrate for different nitrogen sources. *(Data from Kreitler, 1975, Heaton, 1986, Kendall, 1998).*

Source	$\delta^{15}N$ ‰ (AIR)
Rain	−12 to +3
Fertiliser	−5 to +5
Soil nitrogen and natural groundwater nitrate	0 to +9
Nitrate from animal or human waste	+8 to +20

4 POLLUTION SOURCES

In highly developed countries, with intensive agriculture, fertiliser and manure application to land were identified as the main source of anthropogenic nitrate entering the groundwater resources (Addiscott *et al.*, 1991). Accordingly, much research has been conducted into the impacts of agriculture on groundwater quality. Particularly in Europe, the land application of surplus nitrogenous wastes from intensive animal husbandry and dairy farms is currently closely managed. On the other hand, most of the methaemoglobinaemia mortalities in the USA and Europe have been due to local (private) drinking water supplies that were contaminated with inadequate on-site sanitation systems (Walton, 1951, O'Riordan & Bentham, 1993). In southern Africa, excess sludge application to land and inappropriate on-site sanitation are considered the main anthropogenic sources of nitrate. These sources are discussed in more detail below.

4.1 *Agriculture*

Agricultural farming consists of a variety of activities, ranging from extensive dry land grazing to intensive animal feeding units, and from dry land cultivation to large irrigation schemes. The nitrate pollution potential of these activities varies widely. The area covered by the most important arable crops, such as maize and wheat in South Africa, constitutes approximately 9 per cent of the total farming area of 82.9×10^6 ha. Most of the remaining farming area in the country is used for grazing. The largest individual impact is expected to be the result of maize. Most maize is produced under dry land cultivation. To maintain and even increase the production per unit area, fertilization is essential. Tilling the soil causes oxidation of organic soil nitrogen, followed by leaching (Kreitler, 1975). This was also confirmed in the case of the Springbok Flats (Heaton, 1985). In addition, nitrification is enhanced due to the low organic C/N ratio of South African soils, high ambient temperatures, and other factors. This is possibly also the reason for the presence of nitrate in groundwater in the maize growing areas of the northern and north-western Free State Province (Henning & Stoffberg, 1990, Cogho, 1991).

In 1955 nitrogen fertiliser consumption in South Africa was only about 20,000 tons. This increased slowly until 1966. During the fifteen years from 1966 to 1981 (when the highest annual consumption of 510,000 tons was recorded), fertiliser consumption increased at a high rate, but the increase of crop production occurred at a slower rate (Van der Merwe, 1991). This would mean that an increased surplus quantity of nitrogen was available in the soil for oxidation to nitrate and subsequent leaching to groundwater. Considering the experience in countries abroad, a considerable quantity of this nitrate could still be present in the unsaturated zone in areas where the groundwater level is deeper. In areas with shallow water tables ($<$10 m below surface), it is expected that the nitrate derived from the nitrogen applied during the previous two decades has already reached the groundwater.

A further important aspect with respect to soil nitrogen inputs is highlighted by Power & Schepers (1989). Fertiliser nitrogen has largely replaced legume nitrogen as the primary source of nitrogen for American agriculture. According to Power & Schepers (1989) there was a steady increase in fertiliser nitrogen usage in the USA from about 1950 to 1980, while there was a corresponding decrease in legume production. The nitrogen provided by the legumes is in organic form, and has to be mineralised and oxidised before leaching to the subsurface can occur. In South Africa, nitrogen fixed by legumes (including grain legumes, lucerne and legume-based pastures) amounted to 95,000 tons in 1981 (Strijdom & Wassermann, 1984). Should all agronomic soil and natural pasture suited for legume pastures be exploited for this purpose, the total contribution by nitrogen fixation could rise to over 400,000 tons.

Biesenbach (1984) compiled a nitrogen balance sheet for the agricultural soils in South Africa. This balance sheet took into account additions such as fertilisers, atmospheric contributions, nitrogen from irrigation water and natural nitrogen fixation. Nitrogen derived from manure and compost was excluded in the compilation, as no statistics were available. It was also stated that the quantities were relatively small and would not play a significant role in the balance sheet. The nitrogen

Table 2.3. An agricultural nitrogen balance sheet *(after Biesenbach, 1984).*

Year	1960/61		1980/81	
Fertilised area (10^6 ha)	12.3		14.4	
Irrigated area (10^6 ha)	0.7		1.1	
	kg/ha	Total (t)	kg/ha	Total (t)
Additions				
fertilisers	3.8	46,046	32.3	466,385
irrigation	2.4	1,680	2.4	2,678
atmosphere	(10)	(122,660)	(10)	(144,250)
N-fixation	?	unknown	?	unknown
Total		47,726		469,063
Removals				
crops & plant materials	14	181,715	26.7	414,397
erosion	3	36,798	3	43,274
losses (leaching, denitrification, volatilization)	?	(122,660 + unknown)	?	(144,250 + unknown)
Total		218,513		457,672
Net gain (+)/loss (−)		−170,787		+11,391

removals included in the balance sheet were the following: crops and plant residues, soil erosion by wind and water, as well as losses through leaching, denitrification, and volatilization. The data were compiled for a number of years over the period from 1960/61 to 1980/81. For the purpose of this report, only the years 1960/61 and 1980/81 are of importance, as they show the relative weights of the various components, as well as the changes over the 20-year period (Table 2.3).

It is noteworthy that there was a considerable net loss of nitrogen from soil in 1960/61, while in 1980/81, there was a slight gain according to these balance sheets. The loss in 1960/61 was due to the small fertiliser input, which could not cover the nitrogen losses, due to removal of crops and plant materials. Two decades later, the total fertiliser application had increased ten-fold (8.5-fold for the application rate per hectare), while the removal by crops and plant materials had only doubled.

Biesenbach (1984) had no estimate for nitrogen fixation in the soil and assumed that, together with the nitrogen derived from the atmosphere, it balanced the losses due to leaching, denitrification, and volatilization. Strictly speaking, this might not be true. In the case of Danish agriculture (Schroder *et al.*, 1985), it was found that biological nitrogen fixation decreased as other nitrogen inputs increased. Thus, even though the nitrogen balance sheet for agricultural soils in South Africa has certain shortcomings, evident trends point towards a nitrogen surplus developing in the soil zone, possibly creating a greater potential for leaching nitrates to the subsurface.

When comparing the fertiliser application rate of 192 kg N/ha in Denmark in 1980 with 32 kg N/ha in South Africa, it is evident that the nitrogen application rates in Denmark are approximately six times higher. This explains at least part of the very high losses of 187 kg N/ha calculated for Denmark. The balance sheet of Biesenbach (1984), although incomplete, gives an approximation of the situation in South Africa. This would imply that, by 1960/61, there was a considerable over-exploitation of the soil, while in 1980/81 fertiliser application rates were slightly too high.

There are clear indications that nitrogen recovery by crops vary considerably under South African conditions (Dijkhuis *et al.*, 1984). Nitrogen recovery was found to range from 31 to 122 per cent, with exceptional cases as high as 258 per cent. Dijkhuis *et al.* concluded that nitrogen recovery by crops may be governed by weather conditions, soil form, and management. It was also concluded that more mineralisable nitrogen than is generally accepted may be stored in the soil. Much more local research is needed on the nitrogen delivery potential of soils.

Conrad *et al.* (1999) carried out various studies regarding the impacts of agriculture on groundwater quality, of which three case studies investigated the impact of fertiliser usage on nitrate in groundwater. The first of these studies was carried out in the Hex River Valley, approximately 150 km north-east of Cape Town. The area is underlain by alluvium, mainly derived from the Table Mountain Group (largely the Nardouw Sandstone) and to a lesser extent from the Bokkeveld Group (Gydo Shale). This area is known for its export quality table grapes. Nitrogenous fertilisers were applied at various stages before and during the growth season. Fertiliser application rates were as high as 100 kg/ha, considerably higher than the average values presented in Table 2.3 (Biesenbach, 1984). Nitrate concentrations in groundwater showed a high degree of spatial variability ranging from less than 0.1 to 13.2 mg/L (Conrad *et al.*, 1999). However, $\delta^{15}N$ values exceeded the range of −2 to +4‰ of commercial fertilisers and were mainly in the range of human and animal waste, i.e. +10 to +20‰. Although organic fertilisers are also used, it may be that the nitrogen isotopes rather show the effects of the on-site sanitation systems.

The second study was carried out in the so-called Sandveld, near Elands Bay, along the West Coast approximately 200 km north of Cape Town (Conrad *et al.*, 1999). The area is covered by Tertiary to Quaternary alluvial sand deposits of estuarine-lagoonal facies, overlying predominantly horizontally bedded rocks of the Table Mountain Group and Klipheuwel formation. An unconfined, primary aquifer with an average saturated thickness of 15 m exists in the sand deposits. The main land use in the area is potato farming. Over the entire growing season, a total of 240 kg N is applied per hectare (Conrad *et al.*, 1999). The nitrate-N concentration in the groundwater varied from 0.1 to 24.5 mg/L (average 5.7 mg/L). The highest concentrations occurred near dwellings and are ascribed to pollution related to on-site sanitation. In a number of cases the isotope analyses indicated that the nitrate was at least partly derived from commercial fertilisers (Conrad *et al.*, 1999). It was concluded that low levels of nitrate derived both from fertiliser and soil organic nitrogen were found in this area, which indicated that at least some of the nitrate fertiliser applied at a high rate reached the aquifer.

The third study site was located in a wheat farming area at Hertzogville in the Free State Province, that has been used for this purpose for many decades. This area receives summer rain. In 1996 the annual rainfall exceeded 800 mm. The study area is underlain by the Volksrust Shale Formation, which forms part of the Ecca Group (Karoo). The formation is 150 to 250 m thick and mainly comprises shale. Alluvium, sand, and calcrete, in total approximately 3 to 4 m thick, cover most of the area. Water was derived from the shale. The boreholes in the area are generally 40 to 50 m deep, with the main water strike at approximately 35 m below surface. This site differs from the first two in that the aquifer is fractured, while the first two are primary aquifers. Analytical results show that the highest nitrate concentrations occur in the immediate vicinity of the agricultural area. The lowest nitrate level was observed in the Hertzogville town borehole, located some distance from the nearest agricultural land. In the agricultural area, the nitrate varied between 17 and 22 mg/L. $\delta^{15}N$ values for these boreholes ranged from +6.3 to +7.4‰, while the town borehole with a low nitrate-N of only 4.7 mg/L, had a $\delta^{15}N$ value of +7.8‰, i.e. slightly higher than those in the agricultural area (Conrad *et al.*, 1999). Too little data are available for a detailed analysis, but it would seem that nitrogen is mainly derived from the natural soil organic nitrogen pool in this case, but possibly with a limited contribution by nitrogen from inorganic fertiliser. It is important to note that these fields are ploughed twice per annum: before planting and after harvesting. This maximises the nitrification of soil organic nitrogen and the bare soil will promote the leaching of nitrate to the subsurface. It is possible that denitrification may play a role in increasing the $\delta^{15}N$ values.

These three case studies confirm the earlier described observations that inorganic nitrogen fertiliser application rates in South Africa are generally too low to cause significant leaching of nitrate derived from the fertiliser to the subsurface. However, tilling the soil probably plays a key role in oxidising soil organic nitrogen and mobilising nitrate for leaching to the subsurface.

4.2 *Intensive animal husbandry ('feedlots')*

Dry land grazing of stock is expected to have little direct influence on nitrate accumulation, because of the low animal load on the land. However, the gathering of livestock in 'kraals' (enclosures) or around feeding/watering points significantly increases the risk of groundwater pollution. Dairy farming presents a particular problem, as additional feeds are provided and the concentration of animals around milking facilities is usually higher. In their discussion of pollution sources in (then) Bophuthatswana, Xu *et al.* (1991) refer to agricultural practices such as stock-farming, which tends to increase the nitrate load in the underlying aquifer. They state that farmers often construct kraals for their livestock close to the boreholes. This is also the situation in most of the region, including the western Kalahari (Heaton, 1984). At such sites, manure is concentrated, and leachate seeping into the subsurface contains inorganic and organic compounds as well as coliform organisms. Xu *et al.* (1991) link the loss of cattle due to very high nitrate groundwater in the district of Kudumane to pollution under such conditions.

Intensive animal feeding units (feedlots) represent the most significant threat to groundwater quality. Information from the United States indicates that the risk of groundwater pollution by nitrate is relatively low while the feedlot is in full operation (Mielke & Ellis, 1976). Under such conditions, nitrogen remains in reduced form and is largely lost as ammonia. However, when feedlots are only used partially or intermittently, or eventually abandoned, nitrification and transport of nitrates to the groundwater take place. Botha (1984) estimated the total quantity of nitrogen contained in manure produced annually in intensive feeding units in South Africa to be at 69,313 t/a. This was based on data for 1981. No statistics could be found with regard to the present number and distribution of feedlots in South Africa. From an analysis of livestock, it appears that the total number has remained relatively constant over the last two to three decades. However, meat production has increased by approximately 50 per cent over the past thirty years. Based on these numbers, the pollution potential of intensive animal feeding units is expected to have increased considerably. Cogho (1991) found nitrate concentrations in groundwater exceeding 30 mg/L near feedlots in the northern Free State Province.

One major environmental impact of intensive animal husbandry is the large quantity of animal waste that is produced within a limited area. Waste management at such locations is, therefore, of concern. Conrad *et al.* (1999) selected three study sites with this in mind: a dairy and piggery farm, a dairy, and a feedlot. The first two sites are located in the winter rainfall area, and the feedlot in the summer rainfall area. The site with the dairy and piggery was located on unconsolidated alluvium. Although these had relatively well-run management systems, groundwater nitrate reached a maximum of 156 mg/L due to pollution. Several pollution sources were noted, e.g. the manure-covered area at a feeding trough, effluent irrigation areas, and pit latrines (Conrad *et al.*, 1999). Soil nitrogen profiles were also determined and it was concluded that the area adjacent to the feeding trough was possibly the main pollution source. The shallow water table at this site ($<$1.5 m below surface) increased the vulnerability of the aquifer.

At the dairy farm, groundwater nitrate pollution was also noted at a maximum of 27 mg/L. This area is underlain by granite. Soil nitrogen profiles were determined in the weathered material. Some of these had significant nitrate-N concentrations of up to 400 mg/kg of soil. However, it was not possible to link the analytical data from the boreholes to a specific source, as the groundwater intake areas could not be identified (Conrad *et al.*, 1999).

The feedlot site was located on a fractured aquifer in the Gauteng Province, which is a summer rainfall area. In 1996, when the study was carried out, the feedlot had been in operation for 23 years. Despite the relatively shallow piezometric water level (approximately 3 to 7 m below ground surface) and high nitrogen levels in the upper 0.5 m of the soil column, nitrate levels in the groundwater were very low. It was assumed that recharge to the underlying dolomite aquifer (the top of which varies from 10 to 70 m below ground level) was taking place at another location. Pollution from the feedlot could therefore not reach the aquifer mainly due to the low permeability of a thick unsaturated zone ($>$20 m at the feedlot) (Conrad *et al.*, 1999).

The general conclusion is that intensive animal husbandry and dairy farming have a high potential of groundwater pollution. Unless proper waste management practices are in place and aquifers are naturally protected by impermeable zones, nitrate pollution of groundwater will occur.

4.3 *Irrigation*

Irrigation is generally practised in areas where the rainfall is relatively low and crop production has to be stimulated with irrigation water. In many cases, over-application of water for the purpose of salinity leaching forms part of the management of the irrigation system. It is therefore to be expected that nitrate occurrences ascribed to percolate from irrigated lands will occur. In the case of the Hex River Valley, Weaver (1991) found nitrate levels of up to 33 mg/L in some of the wells monitoring shallow groundwater ($<$3 m below the surface). The higher nitrate values were accompanied by potassium values that were also considerably above the natural background levels. This suggests pollution by fertiliser or on-site sanitation.

Nitrate concentrations exceeding 20 mg/L are quite common in boreholes within the irrigated areas extending over 80 km along the Crocodile River in the Limpopo Province (Tredoux, 1993). The associated high electrical conductivities of the water from these boreholes implied irrigation percolate as the nitrate source.

Although high nitrate levels in groundwater can be a result of irrigation practices, an increase in groundwater salinity is generally the main impact noted in South Africa (Conrad *et al.*, 1999).

4.4 *On-site sanitation*

On-site sanitation is economically attractive, but often entails a groundwater pollution risk. In most cases this risk can only be evaluated by carrying out a detailed hydrogeological field study before implementation of such a scheme. Generally, such studies are few in number. Pollution may therefore be expected from many of the settlements located in vulnerable areas in the future. The intricacies of on-site sanitation and its link to nitrate in groundwater warrant a detailed expose in a separate chapter. Several studies have been undertaken in southern Africa, notably also in Botswana, e.g. Lewis *et al.* (1978), Palmer (1981), Jacks *et al.* (1999) and Staudt (2003), and in Moçambique (Muller, 1989). These can serve as useful introduction to such groundwater impacts.

In rural settlements near Makhado (formerly Louis Trichardt) in Limpopo Province, Connelly & Taussig (1991) found that nitrate concentrations in many boreholes exceeded 20 mg/L (up to 54 mg/L). The nitrate was ascribed to two possible sources: mobilisation of natural nitrate by cultivation of the soil, or on-site sanitation. Xu *et al.* (1991) ascribed the high nitrate concentrations in the Northwest Province to the placement of kraals close to water sources, as well as the effect of sewage effluent and sewage sludge disposal to land overlying aquifers. Hesseling *et al.* (1991) studied the occurrence of nitrates at Rietfontein in the arid western part of Northern Cape Province. They found unacceptably high levels of nitrate in the centre of the village, considered to likely be due to human and animal pollution, while water of a better quality was present on the periphery of the village. Despite the widespread occurrence of on-site sanitation in South Africa, the information is largely anecdotal. Even in relatively densely populated areas, pit latrines are used. Depending on local geohydrological conditions, such sanitation can cause serious pollution if precautionary measures are not taken (Lewis *et al.*, 1980).

Septic tanks may be a localised or regional source of groundwater pollution (Canter & Knox, 1985). This depends on the density of septic tanks and the population in a given area. A density of 40 systems per 2.6 km^2 is considered as having significant contamination potential at discharge rates of 170 L/person per day (Canter & Knox, 1985). The design, construction, and maintenance are important. For example, the systems should be cleaned every three years. The effluent quality and the efficiency of constituent removal depend on the underlying soil and the thickness of the unsaturated aerobic soil below the leach field. Nitrification generally occurs at a depth of 60 to 150 cm (Andreoli *et al.*, 1979). The nitrogen originates from several sources, such as urine (up to 80 per cent of the nitrogen), faeces, organisms and household wastewater, and may be in the forms of urea, faecal proteins, and ammonium. The nitrogen loads have been estimated between 11 and 22 g N/capita per day (Kaplan, 1991). Other studies have given an average for the seepage of 62 $\pm$ 21 mg N/L (Bauman & Schafer, 1985) of which 75 per cent is in ammonium (NH_4^+) form. Palmer (1981) stressed the risks involved in introducing water-based on-site sanitation systems such as septic tanks. For a low nitrate impact dry system, a modified pit latrine is recommended, and particularly a urine separating toilet.

The implementation of on-site sanitation remains an important issue that is sometimes decided on a non-technical basis. This complicates the protection of groundwater against the impacts of such practices. However, a general policy for sufficient separation of on-site sanitation systems from water supplies needs to be adopted. Muller (1989) proposed a type of flow chart approach consisting of sets of questions relating to nine parameters for assessing the on-site sanitation risk to any local groundwater supply. In addition, a system of protection zones around production wells and wellfields is needed, such as that proposed by Vrba *et al.* (1985), Van Beek (1985) and locally in South Africa by Xu & Braune (1995). These are generally based on subsurface travel times of pollutants, which may be complex in particular circumstances and in fractured aquifers but can serve as a basic, workable approach to groundwater protection.

5 NITRATE POLLUTION CASE STUDIES

5.1 *Ghanzi (Botswana)*

In October 2000, approximately 200 heads of cattle died on two neighbouring farms near Ghanzi. This prompted an investigation into the cause of death and eventually veterinarians identified the cause as acute nitrate poisoning by contaminated groundwater. Analysis of the groundwater chemistry revealed the potential source of nitrate.

The Ghanzi Ridge forms a prominent topographic feature extending over several hundreds of kilometres and reaches an elevation of 1300 m in the west, approximately 300 m higher than the surrounding sediments of the Kalahari Group.

The main aquifer of importance in the area is in the dominantly siliclastic Ghanzi Group. The rocks consist of ‘grain-supported textures, slightly metamorphosed, and with a fine cementing material made of calcite, chlorite and clay minerals’ (Modie, 1996). Groundwater is found in secondary fracture systems. The degree of folding, shearing and jointing is such that arkose/mudstone interfaces are unlikely to influence the water table. Depths to first water strikes range from 22 to more than 200 m (Modie, 1996).

Groundwater samples of some 13 boreholes collected along the Ghanzi Ridge in October 2000 had nitrate-N concentrations from 14 to 508 mg/L. Further samples from the same and other sources were collected on four occasions during the next four years and showed variable or constant chemical compositions (Fig. 2.2). In October 2000, for instance, one of the boreholes where the cattle died (A-post WP) showed a nitrate level of 508 mg/L. From that time until November 2004, the nitrate level decreased steadily with a linear relationship between the nitrate and chloride (Fig. 2.2). At another borehole, virtually no change could be observed in the nitrate concentration and only a very small variation in the chloride concentration (Fig. 2.2). In both cases the water levels were relatively shallow in 2000, but whereas the level at the high nitrate borehole was at 6.12 m below surface, the water level at the low nitrate borehole was initially only 3.07 m below surface. Based on this information, the low nitrate borehole with the shallower water table should theoretically be more vulnerable to pollution.

The experience of Kreitler (1975) in Texas and the observations in Namibia in 1974 linked the increase in nitrate with exceptionally high rainfall. Analysis of rainfall data for the farm Oakdene near Ghanzi, extending over more than 40 years, confirmed that the 1999/2000 rainy season with a rainfall of 813 mm, was nearly double the long-term mean value of 420 mm (Fig. 2.3). Precipitation in this semi-arid area varies considerably over a short distance, and on one of the farms (Farm No 72-NK) where the cattle died, the rainfall for 1999/2000 was 1121 mm. Most of this high rainfall occurred in February (425 mm) and March (356 mm), which caused flooding in large parts of the area. Such exceptional conditions modify the general recharge patterns in the area. The conceptual hypothesis is that salts that had collected in specific parts of the unsaturated zone during drier seasons were leached into the subsurface during this exceptionally high rainfall event. These salts also contained nitrate.

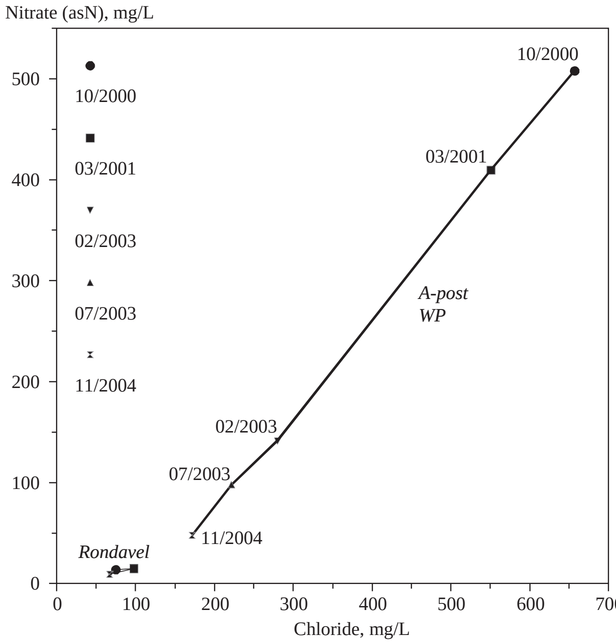

Figure 2.2. Nitrate vs chloride in groundwater at Ghanzi, Botswana, for two boreholes with different responses over time.

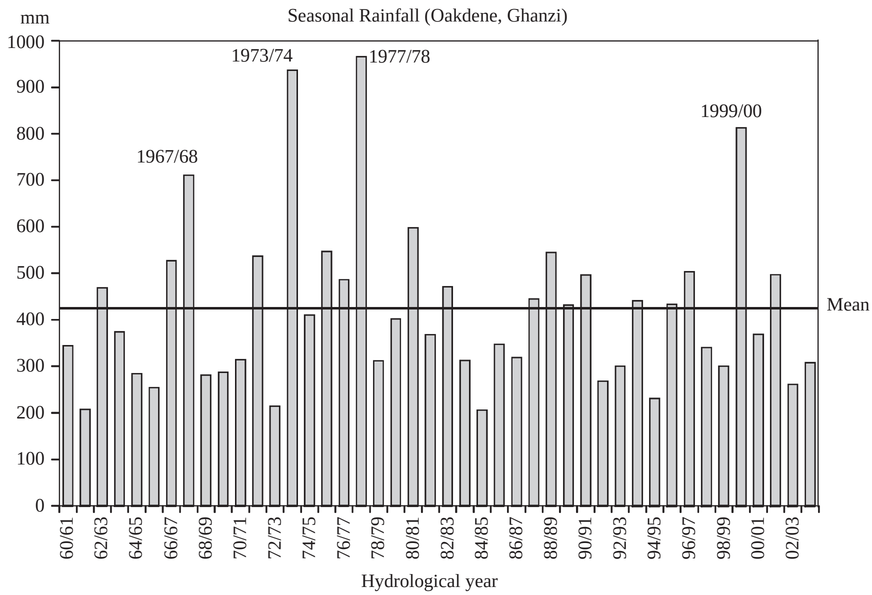

Figure 2.3. Seasonal rainfall record at the Farm Oakdene, near Ghanzi (*supplied by RC Eaton*). The labelled seasons are those with exceptionally high rainfall.

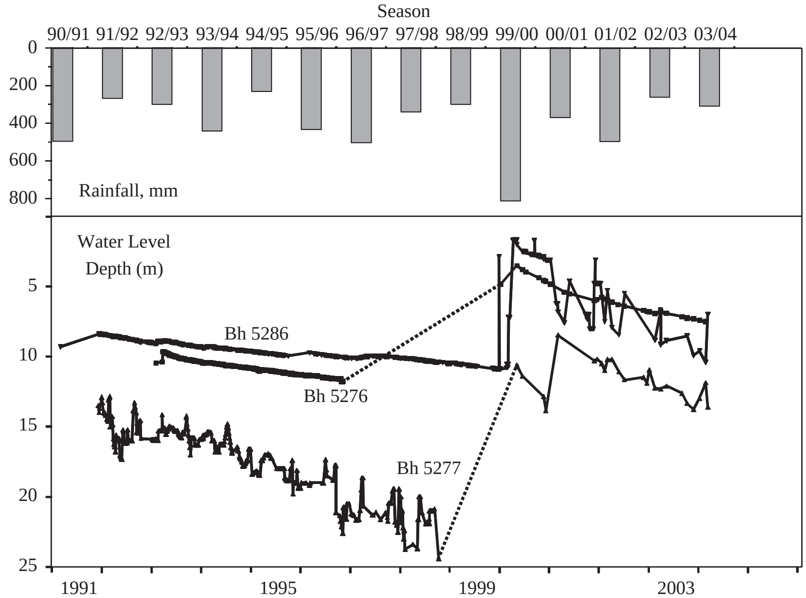

Figure 2.4. Water level response in observation boreholes at Ghanzi, compared to the Oakdene rainfall.

Water-level data from observation boreholes at Ghanzi (obtained from the Department of Water Affairs) show the water-level response during the 1999/2000 rainy season (Fig. 2.4). At Boreholes 5286 and 5276, the water level rose by approximately 8 m within 2 months in response to the major 1999/2000 rainfall, while at Borehole 5277, the extrapolated rise exceeded 10 m. This confirms that exceptional recharge conditions prevailed during that period and led to the leaching of certain parts of the unsaturated zone, which is not regularly affected by infiltrating rainwater.

Previously, Morosini (1996) concluded that contamination of groundwater by nitrate occurs in the recharge area on the ridge, but stated that the source of the nitrate was still uncertain. All of the sources with nitrate-N concentrations exceeding 100 mg/L during the present study had $\delta^{15}N$ values exceeding +8‰, thereby confirming that the nitrate in these particular boreholes was derived from pollution sources (cattle manure from adjacent enclosures), while the low nitrate water has lower $\delta^{15}N$ values (less than +8‰), indicating natural nitrate (derived from vegetation) (Fig. 2.5a). The ^{18}O content of nitrate in the analysed samples shows a negative correlation with ^{15}N (Fig. 2.5b). The end-members of this mixing line show that the low-^{15}N samples derive their oxygen from atmospheric air, while the high-^{15}N samples used water as oxygen source. This is consistent with the concept that soil nitrate is formed in the unsaturated zone, while pollution nitrate is formed where organic matter, urea and ammonia are present in moist environments.

Following a high recharge event, it appears that most of the high salinity water was removed either through pumping or diluted by natural migration. As a result, groundwater salinity and nitrate concentrations at nearly all the affected boreholes returned to lower levels within three to four years. This lower salinity groundwater has to represent the larger quantity of groundwater held in storage and it is assumed to represent recharge that regularly enters the aquifer during 'average' rainfall seasons. Litherland (1982) states that the Ghanzi Group aquifer is recharged annually by surface run-off and seepage. ^{14}C values of 80 to 90 pmc reported for groundwater of the Ghanzi Group outcrop by Morosini (1996) confirm that frequent recharge occurs. In contrast the thick sand of the Kalahari Group absorbs most of the rainfall, which is then again lost to evaporation.

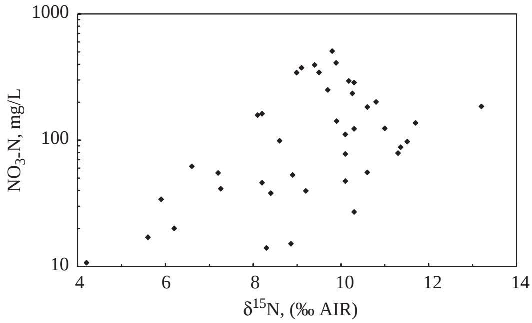

Figure 2.5a. Nitrate vs ^{15}N in sampled boreholes at Ghanzi, Botswana. Even when nitrate levels decreased in the same borehole water, the ^{15}N levels remained constant over time.

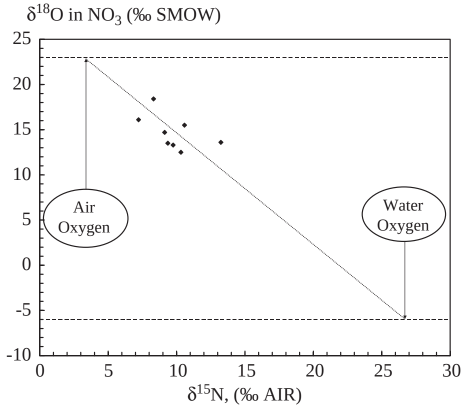

Figure 2.5b. ^{18}O vs ^{15}N in nitrate in groundwater at Ghanzi, Botswana. The relation extrapolates to end-members that indicate the contribution of atmospheric and water values of ^{18}O indicative of the source of oxygen for the unpolluted (low ^{15}N) and polluted (high ^{15}N) samples.

An exceptionally high-rainfall episode in 1973/74 (Fig. 2.3) extended into Namibia and led to the loss of hundreds of heads of livestock, both cattle and sheep (Anonymous, 1974). The Kamtsas Formation in the adjacent part of Namibia is correlated with the Mamuno Formation in Botswana. No information is available on livestock losses in Botswana during that season.

From the above, it is evident that exceptional rainfall events cause the high levels of nitrate to reach the groundwater. This highlights an important component to the variability in salinity and particularly the nitrate concentrations. Groundwater that is generally suitable for stock watering can therefore become poisonous under specific conditions.

As the Ghanzi Group forms a secondary aquifer with a relatively shallow water table and a high permeability, the attenuation of pollutants in the subsurface is very limited. Accordingly, the Botswana National Groundwater Pollution Vulnerability Map (Busch *et al.*, 1996) rates the vulnerability of most of the Ghanzi Ridge as 'moderate' to 'high'. The pollution episode of 2000 proves this statement to be very appropriate.

5.2 *Springbok flats (South Africa)*

In the Springbok Flats, located in the south of the Limpopo Province (the northernmost Province of South Africa, Fig. 2.1), two areas with high nitrate concentrations are clearly delineated. These areas coincide with the distribution of Jurassic basalt rocks of the Letaba Formation (part of the Karoo Sequence). A vertisol, locally known as the 'black turfs of the Springbok Flats', formed on top of the basalt. No fertilisers have been applied, despite more than five decades of agriculture in this area. Verhoef (1973) completed the first extensive investigation of water quality in this area and found that higher nitrate values were largely confined to the areas with black turf soil. Grobler (1976) determined the nitrogen content of the soil and came to the conclusion that the C:N ratio of the soil was low enough to ensure that nitrogen was freely available for autotrophic nitrifying bacteria. Ploughing the soil caused mobilisation and leaching nitrate was followed by its accumulation in the subsurface beyond the root zone. Heaton (1985) confirmed, by means of a combined isotopic ($^{15}N/^{14}N$) and hydrochemical study, that increased cultivation was the only important process leading to the leaching and accumulation of nitrate in the subsurface. This anthropogenic impact due to soil tilling is supported by the findings of Conrad *et al.* (1999) at Hertzogville. However, the black turf soil of the Springbok Flats is much more productive in the generation of nitrate from soil organic nitrogen and, therefore, has higher leaching potential.

The typically high nitrate concentrations in the area are mostly linked to moderate $\delta^{15}N$ values (below +7‰) that indicate nitrification of organic soil nitrogen as the origin of these nitrates (Heaton, 1985). During recent (1999 to 2005) sample runs nitrate-N concentrations of 18 to 46 mg/L were found with similar $\delta^{15}N$ values (Tredoux, 2004). During this recent sampling, pollution occurrences were also identified and could be characterized by higher ^{15}N values (Fig. 2.6). At one borehole (SF-52 on the farm Tuinplaas), situated some 30 m from the residence on the farm, a high nitrate level of 193 mg/L and ^{15}N of +11.8‰ was found (Fig. 2.6). In this case it was concluded that water from the soak-away of the septic tank at the residence was seeping into the borehole through a fissure in the basalt. One could actually discern water dripping down the borehole from a relatively shallow depth. It is likely that more such occurrences will appear as the effects of population pressure become reflected in local groundwater.

The common occurrence of high nitrate pollution in areas where there is already a high 'natural' nitrate background (as is also the case in Ghanzi) raises the question whether nitrate pollution is more severe in an area where natural enrichment of nitrate in groundwater prevails.

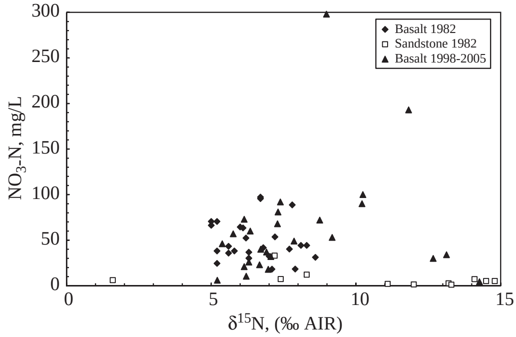

Figure 2.6. Nitrate vs $\delta^{15}N$ for borehole samples from the Springbok Flats. The basalt soils showed a consistent narrow ^{15}N (soil nitrate) signal in 1982. This pattern was repeated during later sampling, but some polluted samples were also encountered. The sandstone samples showed both polluted and unpolluted ^{15}N patterns in 1982. Data from Heaton (1985), Tredoux (2004) and unpublished measurements.

5.3 *Ramotswa (Botswana)*

The town of Ramotswa is located in the Southeast District of Botswana. This is a summer rainfall area with most of the precipitation occurring between November and March. The mean annual rainfall is 475 mm (Staudt, 2003). The geology in the Ramotswa study area consists of three lithological Supergroups, i.e. Ventersdorp, Transvaal, and the Waterberg (Staudt, 2003). The Transvaal Supergroup is of greatest hydrogeological importance: specifically the Ramotswa Dolomite Formation and the Lephala Shales Formation. Active groundwater circulation has additionally brought about local karstification along structural lineaments in the dolomite, producing high transmissivity and storativity. The Ramotswa wellfield extends over an area of 29 km^2, which includes part of the village (Staudt, 2003). As the most productive wellfield in Botswana, it was important as a backup supply to the capital, Gaborone, and also as a local water supply. Hence several studies of this aquifer have been carried out over the previous two decades (Staudt, 2003, Zwikula, 2005).

Of the total 31 boreholes sampled in 2001, eleven had elevated nitrate levels exceeding the Botswana standard of approximately 10 mg/L (or 45 mg/L as NO_3), with the maximum at 99.8 mg/L (as NO_3-N) (Staudt, 2003). Seven of the boreholes sampled in 1983 were re-sampled in 2001 and three of these showed increases in nitrate ranging from approximately 4 times to 55 times (Staudt, 2003). The Ramotswa wellfield was shut down in 1997 and the village is currently supplied with surface water conveyed by pipeline from the Gaborone Dam. Completion of the sewerage system for the village and elimination of the pollution problems were recommended, to be followed by regular monitoring for the potential reintroduction of the wellfield into the water supply system in future (Staudt, 2003).

The main environmental hazards include the village of Ramotswa, the industrial area, and the Ramotswa railway station (Staudt, 2003). It was concluded that approximately 3000 pit latrines constituted a major groundwater pollution hazard. According to Zwikula (2005), the use of pit latrines was ‘successfully promoted in the 1980s in order to deal with the problem of sanitation.’ Several hundreds of private septic tanks were also in use apart from several communal ones in the industrial complex and at the railway station. Considering the shallow water table (in places as little as 3.3 m below surface) pit latrines and septic tanks created a serious hazard. The present (2005) drought has forced the authorities to revisit this wellfield as a standby source of water for the capital.

5.4 *Marydale (South Africa)*

The rural town of Marydale in the Northern Cape Province is located in an arid area, with the low mean annual rainfall of approximately 200 mm/a. The town is situated on the contact between the Kaapvaal Craton and the Namaqua metamorphic belt. The contact passes through the town in a northwest-southeast direction. The dominant rock types in the south-west are metamorphic rocks of the Namaqua metamorphic belt, primarily quartzite, schist, and porphyritic granite. North-east of the contact there are primarily gneissic granites of the Kaapvaal Craton. Quaternary windblown sands of the Kalahari Group occur in the lower-lying areas, between the ridges consisting of more resistant rocks (Hofmann, 1997). A hydrogeological study identified both a primary and secondary aquifer in the area. The primary aquifer occurs in low-lying areas between ridges, and is composed mainly of sedimentary deposits, i.e. sandstone and silt, and calcrete. It reaches a maximum thickness of approximately 12 m and overlies bedrock consisting either of quartzitic gneiss or granite. The sediments are mainly of aeolian origin.

The groundwater flow in secondary aquifers in the quartzitic gneiss and granite is not well understood, but average yields are lower in these aquifers than in the primary aquifer. The primary aquifer also seemed to yield water of a lower salinity than the secondary aquifers. Water levels in the primary aquifer generally vary from 5 to 10 m, but can reach depths up to 25 m in the secondary aquifers. Aquifer transmissivity calculated from pumping tests varies over a wide range from 50 to 610 m^2/d and illustrates the heterogeneity within the aquifers (Hofmann, 1997).

Marydale obtains its water supply both from the shallow, unconfined alluvial aquifer and secondary aquifers scattered around the town. The groundwater is mainly of the sodium-chloride type,

Table 2.4. Hydrochemical and isotope data for town supply boreholes at Marydale (15 February 2000) *(Tredoux, 2004).*

Borehole	Mar-1	Mar-8	Mar-9	Mar-21	Mar-23	Mar-25
Conductivity, mS/m	186	192	177	205	212	204
Potassium, mg/L	31	26	25	31	25	24
Sulphate as SO_4 mg/L	152	172	150	217	250	185
Chloride mg/L	255	258	226	252	273	294
Nitrate-N, mg/L	13	19	25	19	20	25
δ^{15}N, ‰ AIR	+12.4	+10.6	+10.6	+10.8	+10.3	+9.9

it is slightly hard, and has a characteristically high nitrate and potassium concentration. Nitrate-N is generally close to the maximum allowable limit of 20 mg/L. Careful blending of water from different boreholes is required to ensure that the nitrate levels supplied to the town do not exceed 20 mg/L. Whereas the groundwater from the secondary aquifers generally has a lower nitrate, the salinity is higher. The final blended water used for town supply is slightly saline, with the electrical conductivity in the order of 200 mS/m. The hydrochemical data for the sampling run at Marydale in February 2000 are summarized in Table 2.4 (Tredoux, 2004).

A comparison of these results with earlier analytical data indicates that the water quality varies to some extent. This is often the case in such shallow, unconfined aquifers. Overall, the nitrate values have increased. The relatively high δ^{15}N values at Marydale are at levels that are characteristic for faecal pollution. Although sheep are grazing in the river catchment, it was considered unlikely that the groundwater was polluted. However, it is practically impossible to find another explanation and, together with the relatively high potassium, this may be due to pollution from animal waste. Borehole Mar-1, which is located down-gradient of the other boreholes and the closest to the town, has a lower nitrate concentration and a higher δ^{15}N, which would seem to indicate partial denitrification.

5.5 *Industrial pollution (South Africa)*

Certain industrial processes also generate nitrate pollution. Invariably, in older plants some spillage occurs (or has occurred) and this affects the environment. Talma and Meyer (2002, 2005) show examples of these pollution occurrences in various industries, and explain the use of isotopic methods to trace the pollutants around the plants. The uniqueness of the particular industrial process can, in some cases, generate abnormal isotope ratios that are useful for tracing the pollutant in question.

The most comprehensive study of this nature was done at the site of a coking plant of Mittal Steel (previously known as Iscor) at Vanderbijlpark in 1998/2000. The conversion of coke from coal produces up to a few per cent ammonia as a by-product, which is sold off as fertiliser. The study shows that the liquid in wastewater evaporation ponds contained ammonia at levels of 41 to 472 mg/L, nitrate between 1 and 11 mg/L, and δ^{15}N between +25 and +59‰ (Talma & Meyer, 2002). These isotope ratios are way above the values found in the more common situations described in Table 2.1, and are the result of fractionation effects between gas and solvents within the ammonia extraction plant (Talma & Meyer, 2005). It is found that these high ratios of nitrogen isotopes persist throughout the system and this enables a distinction to be made between the industrial pollutant N and other N sources.

i) Isotope tracing in vegetation

δ^{15}N of plants growing on unpolluted nitrogen sources ranges from 0 to +5‰ (Kendall, 1998). This was also found for trees well away from waste pond 4 (Table 2.5). Various plants growing on the edge of the pond, drawing their nutrients from pond water, showed the high ^{15}N signal of this water. For a tree 100 m away from the pond, we could show that about 40 per cent of the nitrogen in the plant would have been pollution derived.

Table 2.5. Isotope analyses of plants growing in or near Mittal Steel pond 4 *(data from Talma & Meyer, 2002).*

Description	Location	$\delta^{15}N$	Nitrogen from pollution (%)
Phragmites, grass and rushes	Pond 4 north wall	+24.7 to +31.9	100
Eucalypt leaves	100 m east of pond 4	+12.5	38
Eucalypt leaves	2000 m west of pond 4, background	+2.9	0

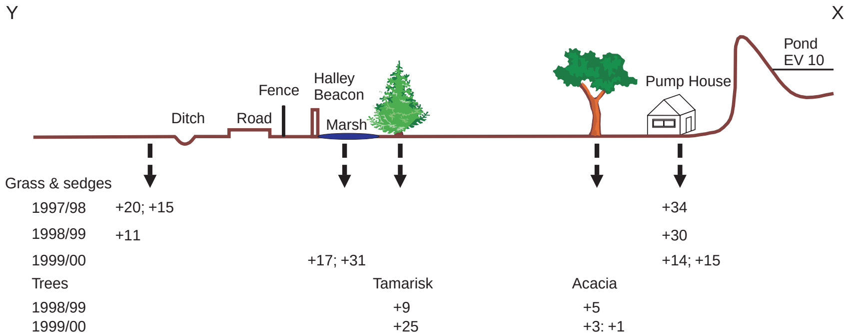

Figure 2.7. Cross-section along a line (X-Y of Figure 2.8) from pond EV10 to the west at Mittal Steel, Vanderbijlpark showing the locations of tree and grass samples that were sampled. $\delta^{15}N$ obtained for each season is shown for the grass and sedges (*modified from Talma & Meyer, 2005*).

A similar situation was shown to be the case for waste water from another pond (EV10) where some leakage had earlier occurred through one of the pond retaining walls. High nitrogen levels were found in the seepage water. The high ^{15}N of seepage water was also traced in vegetation that was growing below the wall (Fig. 2.7). The grassy and shallow shrubs had already attained high ^{15}N values when the samples were taken. The trees with deeper roots showed very little such pollution evidence (Talma & Meyer, 2002, 2005) since the nitrogen pollution had not yet migrated deep enough to reach the roots of the trees. It is therefore important to view vegetation sampling in the context of the nitrogen source available.

ii) Tracing of underground seepage

For a ten-year period in the past, before evaporation ponds were established on this site, waste water effluent was disposed of by spray irrigation in a corner of the coke oven site. Water application rates were in the order of 3 mm/day. This rate is lower than the local evaporation rate (even in winter) and it was reasonable to expect that all of the water, except for the excess during heavy rainfalls, would have evaporated. Chemical and isotopic analysis of water from boreholes in this area, eight years after this practice had ceased, showed that the disposed nitrogen had in fact reached the groundwater (Table 2.6). Apart from one borehole, all the ammonium had already been converted to nitrate. It is not known where (saturated or unsaturated zone) oxidation occurred. The data are consistent with the fact that ammonium oxidises easily and is seldom found in local groundwater. The low flow rate and well-aerated nature of the deep soils are probably factors that ensure complete oxidation of reduced nitrogen compounds to nitrate.

Evidently, in these boreholes, a historical isotope signature exists that has not been flushed for the eight years after spray irrigation was discontinued. With annual recharge rates of only 20–70 mm, it will require many years to flush the nitrate load from the aquifer.

Table 2.6. Isotope data from boreholes in the area that was used for spray irrigation during 1978 to 1988. Sampling was done in 1996 *(Talma & Meyer, 2002).*

	NH_4		NO_3	
Sample	mg N/L	$\delta^{15}N$	mg N/L	$\delta^{15}N$
Borehole IP2 (shallow)	160	+37.3	135	+27.3
Borehole IP3 (shallow)	<0.1	–	160	+37.2
IP3 (deep)	<0.1	–	246	+28.6
MJZ (shallow)	2.1	–	260	+46.7
MJZ (deep)	0.3	–	75	+38.1

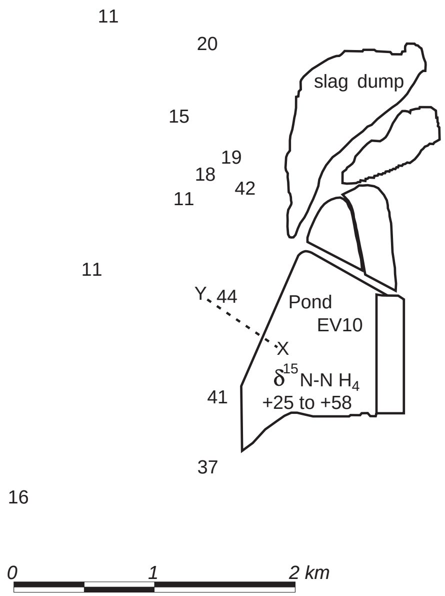

Figure 2.8. Plan of evaporation pond EV10 at Mittal Steel, Vanderbijlpark, and the boreholes to the west. The numbers show ^{15}N of nitrate in boreholes (*modified from Talma & Meyer, 2005*).

iii) Nitrate source apportionment

The nitrate levels in boreholes up to 2 km to the west of Evaporation Pond EV10 were variable, ranging from zero to 24 mg/L. Seepage through the wall of the pond had already been established on the basis of a chloride front that was located a few hundred metres to the west of the dam wall. This raised public concern, since the front was located beyond the borders of the industrial site onto agricultural land. This prompted the question to what extent nitrogen pollution from the evaporation pond could have progressed westwards as well. The ^{15}N content of water from all boreholes with either nitrate or ammonium levels exceeding 5 mgN/L was analysed (Fig. 2.8). Isotope analyses showed that there was, indeed, a front of $\delta^{15}N$ values in excess of +20‰. Beyond that front, $\delta^{15}N$ values of the nitrate ranged between +11 and +20‰, which are generally attributed to human and animal faecal pollution (Table 2.2). There were, at that time, a sufficient number of animals and humans living in the area for a sufficiently long time to make nitrate pollution from that source a distinct possibility. *E coli* were found in the water of one of the boreholes (with $\delta^{15}N = +16‰$) which indicates pollution from a faecal source. Amongst the entire collection of boreholes, there were also a number with low nitrate levels which support the concept that point sources of N pollution exist.

The conclusion from these data was that nitrogen pollution by seepage, leakage or spillage from the evaporation ponds was limited to the area immediately adjacent to the dam wall and that the elevated nitrate levels in the boreholes further away were caused by local pollution from animal or human waste.

6 CONCLUSIONS FROM REGIONAL NITRATE STUDIES

From the southern Africa regional studies it can be concluded that:

- High nitrate levels occur in groundwater in a variety of geological formations and a diversity of environmental conditions. In most cases, the occurrence of high levels of nitrate in groundwater is due to contamination related to anthropogenic activities.
- Geological formations can only serve as a primary source of nitrogen in exceptional cases where ammonium ions are incorporated in rock minerals to be released by weathering and oxidised to nitrate.
- The apparent correlation between the occurrence of high nitrate levels and certain geological formations such as the Ghanzi Group in Botswana and its equivalents (i.e. the Nosib Group in Namibia), the Stormberg Basalt in the Springbok Flats (South Africa) and its equivalents (e.g. the Kalkrand Basalt in Namibia) is due to secondary characteristics of the geological formation and associated factors allowing enrichment with nitrate derived from other sources.
- Nitrogen isotope ratios can serve an extremely useful purpose in identifying nitrogen sources. Anomalous nitrogen isotope ratios remain preserved in the plants that withdraw such water.
- In unconfined aquifers, groundwater nitrate levels can be highly variable over the short term, as they are a function of the recharge processes in the area combined with the features of nitrate generation or polluting activity.
- Feedlots and dairy farming generate large quantities of animal wastes that have a high potential of groundwater pollution. Efficient management of waste materials is required for groundwater protection. Such activities should be restricted to areas where aquifers are naturally protected by impermeable zones such as clay layers of sufficient thickness.
- Inappropriate on-site sanitation at rural villages and towns frequently lead to groundwater pollution by nitrate and the abandoning wellfields.
- In the interior of the subcontinent, soils have a low organic content. Therefore, the possibility of denitrification is very limited, and once nitrate is in solution in the subsurface, it will tend to be persistent.
- Considering the threats listed above, a system of wellfield protection zones, based on groundwater flow and associated travel times to the production wells, needs to be introduced.

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REFERENCES

Addiscott, T. M., Whitmore, A. P. & Powlson, D. S. 1991. *Farming, fertilizers and the nitrate problem.* C.A.B. International, Oxon, U.K., 170 p.

Andreoli, A., Bartilucci, N., Forgione, R. and Reynolds, R. 1979. Nitrogen removal in a subsurface disposal system. *J. Water Poll. Control Fed.*, 51, 841–854.

Anonymous. 1974. Nitrates poison water in the north: Cattle and sheep dying (*in Afrikaans*). *Die Suidwester,* Windhoek, Namibia, 26 June.

Bauman, B. J. and Schafer, W. M. 1985. Estimating groundwater quality impacts from on-site sewage treatment systems. In: *ASAE Pub. 07-85*, pp. 285–295. ASAE, St. Joseph, MI, USA.

Biesenbach, F. W. 1984. An NPK balance sheet for the agricultural land of the Republic of South Africa (*in Afrikaans*). Proc. Nitrogen Symposium, Pretoria, 1982, SIRI, Dept. Agriculture, 60–79.

Böttcher, J., Strebel, O., Voerkelius, S. and Schmidt, H. L. 1990: Using isotope fractionation of nitrate–nitrogen and nitrate–oxygen for evaluation of microbial denitrification in a sandy aquifer. *Journal of Hydrology* 114: 413–424.

Botha, A. D. P. 1984. Increasing importance of organic fertilizer in the Republic of South Africa (*in Afrikaans*). *Proc. Nitrogen Symposium*, Pretoria, 1982, SIRI, Dept. Agriculture, 141–144.

Busch, K., Mabua, I., Mokokwe, K., & Von Hoyer, M. 1996. 1:1,000,000 National Groundwater Pollution Vulnerability Map Explanatory Notes. Geological Survey Department, Lobatse, Botswana, Bulletin 41, 32 p. Map.

Canter, L. W. and Knox, R. C. 1985. *Septic tank system effects on groundwater quality.* Lewis Publishers, Chelsea, MI, USA.

Cogho, V. E., 1991. *Written communication.* Institute for Groundwater Studies, UFS, Bloemfontein.

Connelly, R. J. & Taussig, D. 1991. Investigation into the nitrate contamination of ground water in the Kutama and Sithumule Districts, Venda. *Biennial Ground Water Convention, Midrand*, Paper No 33.

Conrad, J. E., Colvin, C., Sililo, O., Görgens, A., Weaver, J. & Reinhardt, C. 1999. Assessment of the impact of agricultural practices on the quality of groundwater resources in South Africa. *Water Research Commission Report* 641/1/99, Pretoria.

Department of Water Affairs and Forestry (DWAF). 1996. South African Water Quality Guidelines, 2nd edition. Volume 5: Agricultural Use: Livestock Watering. Department of Water Affairs and Forestry, Pretoria.

Department of Water Affairs and Forestry (DWAF). 1998. Guidelines for Domestic Water Supplies. Department of Water Affairs and Forestry, Pretoria.

Dijkhuis, F. J., Jurgens, C. A. and Botha, F. 1984. Nitrogen recovery by maize and wheat. *Proc. Nitrogen Symposium*, Pretoria, 1982, SIRI, Dept. Agriculture, 58–59.

Grobler, D. C. 1976. Report on the investigation into the origin of the high nitrate content of groundwater in certain parts of the Springbok Flats (*in Afrikaans*). *Technical note 74*, Dept. Water Affairs, Pretoria.

Heaton, T. H. E. 1984. Sources of the nitrate in phreatic groundwater in the western Kalahari. *J. Hydrol.*, 67, 249–259.

Heaton, T. H. E. 1985. Isotopic and chemical aspects of nitrate in the groundwater of the Springbok Flats. *Water SA*, 11, 199–208.

Heaton, T. H. E. 1986: Isotopic studies of nitrogen pollution in the hydrosphere and atmosphere: a review. *Chemical Geology*, 59, 87–102.

Henning, J. A. G. & Stoffberg, D. J. 1990. Nitrate concentrations in groundwater in the northwestern Free State (*in Afrikaans*). Poster paper, Soil Science Congress, Pretoria.

Hesseling, P. B., Toens, P. D. and Visser, H. 1991. An epidemiological survey to assess the effect of well-water nitrates on infant health at Rietfontein in the northern Cape Province, South Africa. *S.A. Journal of Science*, 87, 300–304.

Hofmann, R. H. 1997. Geohydrological investigation of the groundwater resources at Marydale, Northern Cape Province (*in Afrikaans*). *Technical Report No.: 04/27/96/11, submitted to Marydale Municipality*, August 1997.

Jacks, G., Sefe, F., Carling, M., Hammar, M. and Letsamao, P. 1999. Tentative nitrogen budget for pit latrines – eastern Botswana. *Environmental Geology*, 38, 199–203.

Kaplan, B. O. 1991. *Septic systems handbook.* Lewis Publishers, Chelsea, MI, USA.

Kendall, C. 1998. Tracing nitrogen sources and cycles in catchments. In: Kendall, C. and McDonnell, J. J. (Eds) *Isotope Tracers in Catchment Hydrology*, 519–576. Elsevier, Amsterdam, 839p.

Kirchner, J. & Tredoux, G. 1975. The Stampriet Artesian Basin, with special reference to the Salt Block. *Proceedings of SARCCUS meeting, Bulawayo.*

Kreitler, C. W. 1975. Determining the source of nitrate in ground water by nitrogen isotope studies. *Univ. Texas, Austin, Texas, Bur. Econ. Geol. Rep. Invest.*, 83, 57 p.

Lagerstedt, E., Jacks, G. & Sefe, F. 1994. Nitrate in groundwater and N circulation in eastern Botswana. *Environmental Geology*, 23, 60–64.

Lewis, W. J., Farr, J. L. & Foster, S. S. D. 1978. A detailed evaluation of the of the pollution hazard to village water supply boreholes in eastern Botswana, *Report GS 10/4,* Dept. Geol. Survey, Botswana.

Lewis, W. J., Farr, J. L. & Foster, S. S. D. 1980. The pollution hazard to village water supplies in eastern Botswana. *Proc. of the Institute of Civil Engineers,* Part 2, 69, 281–293.

Litherland, M. 1982. The geology of the area around Mamuno and Kalkfontein, Ghanzi District, Botswana. *District Memoir* 4, Botswana Geological Survey, Lobatse, Botswana, 145 p.

Marais, S. 1991. *Written communication.* Department of Water Affairs, Mafikeng (Mmabatho), North-West Province (formerly Bophuthatswana).

Marais, S. 1999. Dependency of communities on groundwater for water supply and associated nitrate and fluoride problems. Paper presented at Water Research Commission Workshop on Fluorides and Nitrates in Rural Water Supplies, Mafikeng, South Africa, 9–10 March 1999.

Mielke, L. N. and Ellis, J. R. 1976. Nitrogen in soil cores and groundwater under abandoned cattle feed lots. *J. Environ. Qual.*, 5 (1), 71–75.

Modie, B. N. 1996. The geology of the Ghanzi Ridge. District Memoir 7, Department of Geological Survey, Lobatse, Botswana, 81 p.

Morosini, M. 1996. Chemical-Isotopic Character, Origin and Evolution of Groundwater in Western Ghanzi District, Botswana. Department Geological Survey, Lobatse, Botswana, 56 p, figures & tables.

Muller, A. M. 1989. The evaluation and health impact of on-site sanitation. *M.Sc. Eng thesis, Univ. Witwatersrand*, 121 p.

O'Riordan, T. and Bentham, G. 1993. The politics of nitrate in the UK. In: *Nitrate: Processes, Patterns and Management.* Burt, T. P, Heathwaite A. L. and S. T. Trudgill (Eds), John Wiley & Sons, Chichester, UK, 403–429.

Palmer, I. 1981. Nitrates in the groundwater supply to villages in Botswana, M.Sc. Eng thesis, Pennsylvania State Univ., 122 p.

Power, J. F. and Schepers, J. S. 1989. Nitrate contamination of groundwater in North America. *Agriculture, Ecosystems and Environment,* 26, 165–187.

Schroder, H., Harremoës, P. and Simonsen, J. F., 1985. Water pollution caused by nitrogen from urban wastewater and from agriculture. *Paper presented at Conf. Nitrates in Water, Paris.*

Schwiede, M., Duijnisveld, W. H. M. & Böttcher, J., 2005. Investigation of processes leading to nitrate enrichment in soils in the Kalahari region, Botswana. Proc. International Workshop on Environmental Geology in Semi-arid Environments, Selebi-Phikwe, November 2004, Department of Geological Survey, Lobatse, pp. 103–112.

Stadler, S., 2005. Investigation of natural processes leading to nitrate enrichment in aquifers of semi-arid regions. Ph D Thesis, University of Karlsruhe, Germany, 262 p.

Staudt, M. 2003. *Environmental hydrogeology of Ramotswa*. Environmental Geology Division, Department of Geological Survey, Lobatse, Botswana, 65 p.

Strijdom, B. W. and Wassermann, V. D. 1984. Existing and potential contributions by biological nitrogen fixation systems to the South African agriculture (*in Afrikaans*). *Proc. Nitrogen Symposium*, Pretoria, 1982, SIRI, Dept. Agriculture, 80–86.

Super, M., Heese, H de V., MacKenzie, D., Dempster, W. S., Du Plessis, J. and Ferreira, J. J. 1981. An epidemiological study of well water nitrates in a group of Namibian infants. *Water Res.*, 15, 1265–1270.

Talma, A. S. and Meyer, R. 2002. Origin and tracing techniques of high ^{15}N nitrogen compounds in industrial environments. In: The Application of Isotope techniques to the Assessment of Aquifer Systems in Major Urban areas, IAEA-TECDOC-1298, p135–160. International Atomic Energy Agency, Vienna, 179p.

Talma, A. S. and Meyer, R., 2005. The application of 'anomalous' nitrogen isotopic tracers in industrial groundwater pollution situations. Biennial conference: Ground Water Division of GSSA, Pretoria, 7–9 March 2005 (on CD).

Tredoux, G. & Kirchner, J. 1985. The occurrence of nitrate in groundwater in South West Africa/Namibia. *Proceedings Nitrates in Water, Paris.*

Tredoux, G. 1993. A preliminary investigation of the nitrate content of groundwater and limitation of the nitrate input. *Report to the Water Research Commission*, Report No 368/1/93, ISBN 1 86845 009 0.

Tredoux, G., Engelbrecht, J. F. P. and Talma, A. S. 2001. Nitrate in groundwater in southern Africa. In: *New approaches characterizing groundwater flow*, Seiler & Wohnlich (Eds), Swets & Zeitlinger, Lisse, pp. 663–666. ISBN 90651 848 X.

Tredoux, G. 2004. Nitrate and associated hazard quantification and strategies for protecting rural water supplies. Water Research Commission Report No. 1058/1/04, Pretoria. ISBN No 1-77005-173-2.

Tredoux, G, Engelbrecht, J. F. P., & Talma, A. S., 2005. Groundwater nitrate in arid and semi-arid parts of southern Africa. In: Vogel, H. & Chilume, C. (Eds), *Environmental Geology in Semi-arid Environments*, DGS, Lobatse, Botswana, pp 121–133. ISBN 99912-0-501-2.

Van Beek, CGEM. 1985. Nitrate in groundwater extracted by the waterworks in the Netherlands: Present and Future situation. *Paper presented at Conf. Nitrates in Water*, Paris, 8 p.

Van der Merwe, A. J. 1991. Personal communication. SIRI, Department of Agriculture, Pretoria.

Verhoef, L. H. W. 1973. The chemical quality of groundwater in the Springbok Flats (*in Afrikaans*). *Unpubl. Rep.*, Dept. Water Affairs, Pretoria.

Vogel, H., Mokokwe, K. and Setloboko, T. 2004. Nitrate hotspots and salinity levels in groundwater in the Central District of Botswana. Environmental Geology Division Report, Department of Geological Survey, Lobatse, Botswana, 34 p.

Vrba, J., Pekny, V. and Skorepa, J. 1985. Contents of nitrates in groundwater – Lessons of the Czechoslovak experience. *Paper presented at Conf. Nitrates in Water*, Paris, 25 p.

Walton, G. 1951. Survey of literature relating to infant methaemoglobinaemia due to nitrate-contaminated water. *American Journal of Public Health*, 41, 986–996.

Weaver, J. M. C. 1991. Groundwater pollution by pesticides – a case study from the Hex River valley, South Africa. Biennial Ground Water Convention (Midrand), Paper No 8.

World Health Organization (WHO). 1998. WHO Guidelines for drinking water quality. 2nd edition, addendum to Volume 1: Recommendations. Geneva: WHO, 1998, p-8–10; and addendum to Volume 2: Health Criteria and other Supporting Information.

Wrabel, J. 2005. Investigation of the processes leading to nitrate enrichment in the Kamtsas quartzite aquifer, Namibia. In: Vogel, H. and Chilume, C. (Eds), *Environmental Geology in Semi-arid Environments*, DGS, Lobatse, Botswana, 134–139. ISBN 99912-0-501-2.

Xu, Y., Marais, S. and McCaffrey, L. 1991. A preliminary discussion of aquifer protection strategies for some areas in Bophuthatswana. *Biennial Groundwater Convention*, *Midrand*, Paper No 28.

Xu, Y. and Braune, E. 1995. Minimum distance as an important concept for borehole source protection in South Africa. Groundwater '95, *Biennial Groundwater Convention, Midrand*, Paper No 6.

Zwikula, T. 2005. Evaluation of groundwater quality in Ramotswa, Southeastern Botswana. Proc. International Workshop on Environmental Geology in Semi-arid Environments, Selebi-Phikwe, November 2004, Department of Geological Survey, Lobatse, pp. 24–28.

3

Nitrate pollution of groundwater in Nigeria

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ABSTRACT: The average level of nitrate in groundwater in Nigeria has increased in the last 20–30 years. This is based on the analyses of groundwater samples from over 2,200 wells (1985–2004) and 350 samples (pre-1970). The results of the survey show that 33% of wells produced water with a nitrate concentration that is above the WHO guide limit of 45 mg NO_3/L. It was difficult to estimate the number of people drinking water with nitrate concentrations above the permissible limit, yet a significant percentage of the population is assumed to be at risk of ingesting high doses of nitrate through drinking water and food. The purpose of this paper is to present the occurrence of nitrate in groundwater for Nigeria and discuss the implication and sources of nitrate in groundwater from different aquifers within the various groundwater regions in Nigeria.

1 INTRODUCTION

The problem of nitrate pollution, particularly in groundwater, is widespread in many countries of the world and is reported in the various regions of Nigeria (Adelana & Olasehinde, 2003, Edet, 2000, Mbonu & Ibrahim-Yusuf, 1994, Uma, 1993, Egboka & Ezeonu, 1990, Langenegger, 1981). In an overview of groundwater contamination in Nigeria high concentrations of nitrate, chloride, sulphate and bacterial pollution have been observed in municipal as well as rural water supplies.

Low exposure levels of nitrates already start showing health effects. Studies clearly demonstrate that nitrate concentrations above 10 mg/l, as NO_3-N, adversely affect human health (Shuval & Gruener, 1977, Wolff & Wasserman, 1972, Hartman, 1983, WHO, 1985). Therefore, knowledge of the distribution of the elevated concentrations is necessary for effective management of the nation's water resources and precautionary measures against adverse effects.

The overall objective of the evaluation is three-fold:

1. To describe nitrate distribution in various aquifers in order to bring the nitrate problem to the attention of the groundwater practitioners and stakeholders;
2. To identify the sources of nitrate pollution in groundwater in Nigeria;
3. To evaluate which geochemical environments are associated with occurrences of elevated nitrate concentrations.

2 GEOLOGY AND GROUNDWATER DISTRIBUTION IN NIGERIA

Geologically, Nigeria is made up of two main rock types: the Basement Complex fluvio-volcanics, and the Cretaceous-Tertiary sedimentary rocks; each covering approximately 50% of the land area in Nigeria. In the basement complex terrain (comprising the west, north-central and the south-east areas), rock types are predominantly of migmatitic and granitic gneisses, quartzites, slightly migmatised to

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unmigmatised meta-sedimentary schists and dioritic rocks (Rahaman, 1989). The sedimentary rocks overlying the basement complex (in the south, north-east and north-west) consist of arkosic, gravely, poorly sorted and cross-bedded sandstones (Cretaceous and Tertiary). The full description of the geology of Nigeria is reported in Kogbe (1989).

Aquifer distribution in Nigeria is categorized into two systems: basement fluvio-volcanic aquifers and sedimentary aquifer systems in accordance to the various hydrogeological basins or groundwater regions delineated by Akujieze et al. (2002). The basement fluvio-volcanics consists of four hydrostratigraphic units: Younger granites aquifuges, Fluvio-volcanics aquifers, Older granites aquifuges and Metamorphic aquifuges (1a, b, c, d). The sedimentary aquifer systems are found within the sedimentary hydrogeological basins (2–10) as indicated in Figure 3.1. Nigeria has significant potential in terms of groundwater resources; far exceeding that of surface water.

There are eight significant aquifers in Nigeria. They are the:

1. Ajali Sandstone aquifer with yields of 7–10 l/s,
2. Benin Formation (coastal plain sands) aquifer with yields of 6–9 l/s,
3. Upper aquifer with 2.5–30 l/s,
4. Middle aquifer with yields of 24–32 l/s,
5. Lower aquifer with yields of 10–35 l/s (of the Chad Formation),
6. Gwandu Formation aquifer with yields of 8–15 l/s,
7. Kerrikerri Sandstone aquifer with yields of 1.25–9.5 l/s,
8. Crystalline fluvio-volcanic aquifer with a 15 l/s yield in the Jos Plateau region.

These eight mega-regional aquifers have an effective average thickness of 360 m, with a thickness range of 15–3000 m at a depth range of 0–630 m b.g.l (below ground level), with an average depth of 220 m.

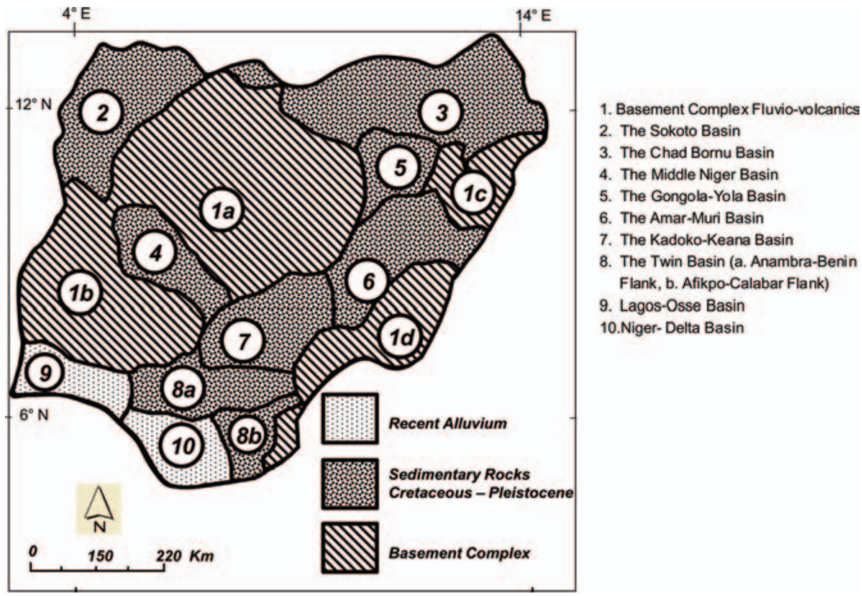

Figure 3.1. Hydrogeological basins of Nigeria (modified from Akujieze, 2002).

3 DATA GATHERING

The dataset in this study was obtained through field investigations by the author, in most cases through collaborative staff-students research projects. This is supplemented by a literature survey and an evaluation of 2,120 NO_3 analyses to determine the distribution of nitrogen in groundwater, either in the form of nitrate or ammonia. In addition, wells supplying drinking water to homes (or for domestic use) were sampled rather than those used for irrigation and livestock watering. This selection criterion were imposed in order to identify the major problems related to nitrate in groundwater due the associated risk with the ingestion of such waters by humans.

4 RESULTS

4.1 *The condition of groundwater in Nigeria*

The majority of the water supply wells did not contain detectable nitrate thirty to forty years ago. More than 200 water wells in the Northern Nigeria were sampled between 1938 and 1960 (du Preez & Barbers, 1965) and only 4.5% found with a nitrate content of above 50 mg/L (WHO standard). Another set of 130 boreholes was sampled for chemistry between 1961 to 1968 by the Geological Survey of Nigeria in north-western Nigeria (Anderson & Ogilbee, 1973). These results showed that 9.6% of the water supply wells had nitrate exceeding the drinking water standard.

Recent surveys show that there is an increasing trend in nitrate concentration and water quality deterioration in the greater part of the country. High nitrate levels in groundwater currently occur in many areas of Nigeria, in places sometimes exceeding the international drinking water standards ten-fold (Adelana & Olasehinde, 2003). Elevated concentrations of NO_3 often extend over several square kilometers, both in sparsely and densely populated areas. The distribution of NO_3 (during the preliminary stages) from more than 1,120 dug wells and boreholes in many of these areas is shown in Figure 3.2.

Figure 3.2. The distribution of nitrate in selected wells in Nigeria (Adelana & Olasehinde, 2003).

These elevated concentrations can partly be explained as by increased urbanisation coupled with indiscriminate waste disposal, industrialization, overcrowding of cities without adequate sanitation facilities, animal husbandry and agricultural use of manure and chemical fertilisers.

4.2 *Distribution of nitrate according to the groundwater regions in Nigeria*

The distribution of nitrate values in groundwater is not uniform, and ranges from <1 to >200 mg/L NO_3. This range reflects heterogeneity in nitrate sources and pathways, resulting in differences in soil type and depth, vegetation types and coverage, as well as the disposal mode of wastes and lithological variations. Some of the lower values reflect preferred infiltration routes, and not necessarily a good sewage system. No geographical bias to NO_3 is observed. Very high values (>1000 mg/l NO_3) occur in the south-eastern part of Nigeria in hand-dug wells and boreholes (especially within sedimentary basins), while low values (<5 mg/l) generally occur in the Basement Complex fluvio-volcanic rocks.

4.2.1 *The basement complex – Fluvio-volcanic basin*

The nitrate concentrations of water in the Basement Complex areas are generally lower than those within sedimentary basins.

Table 3.1. Nitrate distribution in groundwater from various geological formations in Nigeria.

Groundwater region	Hydrostratigraphic units	Lithology	Maximum NO_3 level (mg/L)	No. of samples	% over 45 mg/L
Basement-fluvio-volcanics	Younger granite aquifuge	Weathered basalts, buried alluvium regolith	300	120	13
	Granite, metamorphic aquifuge	Fractured granite, gneiss, schists, metasediments	225.4	288	17
Sokoto Basin	Kalambaina aquifer	Fine-coarse sand 'limestone', clay	97.1	52	33
	Taloka/Wurno aquifer	Coarse sand, limestone	39.1	74	18
Chad Basin	Upper aquifer	Silts, sands, gravels, clays	134.4*	360	29
	Lower aquifer	Sands, fine-coarse sands, clay intercalations	112	12	20
Nupe Basin	Nupe sandstone aquifer	Fine-coarse sands	88	69	<1
Yola Basin	Alluvial aquifer	Grits, sandstone, clay	72	12	16
	Bima sandstone aquifer	Sandstone, micaceous shale-mudstone	150	18	10
Anambra-Benin Basin	Ajali sandstone aquifer	Coarse/medium sands, clay interbeds	135	65	27.5
	Enugu shales		472	14	25
Calabar Flank Basin	Coastal plain sand aquifer	Red sands, thin clay interbeds	1101	67	80
	Eastern Delta	Sands, clay interbeds	3869	120	95
Lagos-Osse Basin	Alluvial, coastal sand aquifer	Sands, gravels, silt, clay	284.7	88	52
	Ilaro/Ewekoro aquifer	Clay sands, sand silt, limestone beds	107.8	112	25

* NO_3-N.

Many boreholes are however sited within residential areas, and as such the high nitrate contents in this aquifer system were attributed to leachates from poorly disposed domestic and agricultural wastes (Mbonu & Ibrahim-Yusuf, 1994).

Organised waste disposals systems are also lacking in many of these areas.

4.2.2 *The sedimentary basins*

The nitrate occurrence in the groundwaters of the sedimentary basins is of concern. The heavily polluted surface and borehole water coincided with areas of intense agricultural activity and fertiliser applications. Even tap water sometimes contains nitrate in the range of 13.5–22.7 mg/L (see Table 3.2). The concentrations of groundwater with nitrogen in the form of NH_4 can be up to 78.6 mg/L in the Lagos mainland.

5 MAIN SOURCES OF NITRATE POLLUTION IN NIGERIA

Groundwater is exposed to active pollution in the major cities of Nigeria due to the increase of urbanization and indiscriminate waste disposal. There are cities without organised waste disposal systems; and where poorly managed municipal landfills exist. Figures 3.3a-b illustrates the negligence and improper managements of domestic wastes, as well as human activities that could encourage nitrate accumulation.

The direct source of nitrate in groundwater originates as NO_3 from wastes or commercial fertilisers applied to the land surface. In some other cases, nitrates are introduced by conversion of organic nitrogen or NH_4^+, which occurs naturally or is introduced into the soil zone by man's activities. In certain areas these intensive agricultural practices and industrial activities have been going on for several decades.

Table 3.2. Contamination levels of nitrate and nitrite in potable water sources in Onitsha.*

Water source	No. of samples	pH range of samples	NO_2^- (mg/l)	NO_3^- (mg/l)
Well water	40	4.8–5.7	2.41–36.03	184–380
Metallic tanks (Probably rusting)	10	4.7–5.1	–	180–306
Tap (with rusty & leaky pipes)	10	5.1–5.5	–	13.5–22.7

* Source: Egboka & Ezeonu (1990).

Figure 3.3a. Roadside refuse dumping in a city.

Figure 3.3b. Wastewater in unlined drainage.

Isolated occurrences of high nitrate are observed in the rural areas of Nigeria and are considered to be due to anthropogenic effects and indiscriminate waste disposal near unprotected wells. Occurrences of nitrate in the coastal plain sands of southern Nigeria are due to identified anthropogenic activities.

6 CONSEQUENCES OF NITRATE POLLUTION

Groundwater contamination in Nigeria is of great importance, as most aquifers are used for water supply purposes and, in some cases, bottled by the mineral water industry. It should be noted that the contamination of groundwater from industry and waste dumps is a serious problem in the big cities of Nigeria just like other developed countries. High concentrations of nitrates are the other main results of groundwater pollution. Concentrations of nitrate above 10 mg/L in drinking water should normally generate concern. Concentrations that approach or exceed 45 mg/L as NO_3, already present health hazards.

The international drinking-water quality standard is set at 10 mg/L for NO_3 as Nitrogen, which is approximately 45 mg/L as NO_3 (WHO, 1985). It is therefore important to consider the health implications of this common pollutant of groundwater with respect to humans, livestock and the environment. The range of nitrate values in Nigerian groundwater is compared with limits and standards from different international organisations (Table 3.3) in order to evaluate the associated health risks.

High nitrate levels in water can result in a number of diseases, such as infant methemoglobinemia, a disease commonly known as 'blue baby' syndrome (Canter, 1996). Clinical effects become obvious when drinking water contains nitrate in the neighborhood of 50 mg/L. Records have shown that infants are typically exposed to unsafe levels of nitrate in drinking water when it is used to mix formula milk or other types of baby food, where children are not breast-fed (Colvin, 1999). Infants under or at about 3 months of age are the primary concern, because they are the most vulnerable. Such infants are much more sensitive to nitrate toxicity than the rest of the population for many reasons. One reason is that the bacteria that live in the digestive tracts of such infants convert nitrate into toxic nitrite.

Although there are no current statistics on infant methemoglobinemia in Nigeria, several cases have been reported. Countries with sufficient statistics on infant methaemoglobinaemia are Hungary (WHO, 1985), South Africa (Hesseling *et al*., 1991, Colvin, 1999), Namibia (Super *et al*., 1981, Tredoux, 1993), the United States of America and Europe. From 1945 until 1970, some 2000 cases of methemoglobinemia have been reported in the world literature, with a case fatality of about 8 per cent (Shuval & Gruener, 1972). This number is likely to have grown as a result of

Table 3.3. Specification limits and standard guideline values for nitrate in drinking water.

			Concentration (mg/l)	
Organization	Year	Limit of specification	as NO_3	as N
WHO (European standard)	1970	Recommended	50	(11.3)[1]
		Acceptable	11.3–22.6	
WHO (International)	1971		45	10.2
WHO	1984	Guide value	(44.3)	10
[2]USEPA	1977		(44.3)	10
European Communities	1980	Maximum admissible	50	(11.3)
	1980	Guide level	25	(5.6)
[4]SABS	1984	Recommended[3]	(26.6)	6
		Maximum allowable[3]	44.3	10
[5]NFEPA	1991	Recommended	45	10

[1] Brackets indicate derived units.
[2] United States Environmental Protection Agency.
[3] Nitrate plus nitrite.
[4] South African Bureau of Standards.
[5] Nigerian Federal Environmental Protection Agency.

the increasing trend of nitrate in groundwater globally. Infants are not the only ones at risk; it is possible that high nitrate concentrations can cause cancer in adults. 'Nitrate itself is not directly carcinogenic. However, there is recognition of the fact that nitrate could be converted to nitrite in the human body that can react with secondary and tertiary amines to form nitrosamines – which have been identified as potent carcinogens' (Fedkew, 1991).

Several epidemiological studies have indicated significant positive correlations between exposure to nitrate and cancer risk. For instance, nitrate in drinking water has been correlated with gastric cancer risk in Colombia and England; and exposure to nitrate-containing fertilisers appeared to be linked to gastric cancer mortality in Chile (Canter, 1996). It should be noted that a high risk for gastric cancer does not only correlate with nitrate, but also with several other dietary or environmental factors.

Other human health effects suspected to be caused or aggravated by nitrate intake are hypertension, the 'hot dog headache', certain cancers, some birth defects (congenital malformations) and spontaneous abortions (Spalding & Exner, 1993). It has also been suggested that chronic exposure to high levels of nitrate in drinking water may have adverse effects on the cardiovascular system. Increased concentrations of nitrate often cause blood disorders (Bowman, 1994). Moreover, a high level of nitrate and phosphates in drinking water encourages the growth of blue-green algae, resulting in deoxygenation (eutrophication) and subsequent reduction in metabolic activities of the organisms that serve as purifiers of faecal polluted water in the human system.

Other possible effects of nitrates relate to the thyroid function. Some animal studies indicate that chronic exposure to high levels of nitrate can reduce the intra-thyroid iodine pool and thus render the gland more sensitive to goitrogens (WHO, 1985). However, whether or not exposure to nitrate is an etiological factor in human goiter is subject to further research. Losses of livestock by nitrate poisoning occur occasionally in the northern parts of Nigeria. Detailed information on the health effects of nitrate on humans and animals, as well as other health-related information on nitrates, is fully discussed in Adelana (2005).

7 CONCLUSION

The occurrence and distribution of nitrate in Nigerian waters have been described and illustrated in this paper. The elevated nitrate contents of the groundwater are generally not attributable to geologic factors. The sources of high concentrations of nitrate are mostly anthropogenic, particularly indiscriminate waste disposal and agricultural practices.

Exposure to high doses of nitrate is generally perceived to be associated with adverse health effects in humans and other species. Exceptionally high NO_3-borehole water must be abandoned for drinking and domestic purposes. Possible bio-denitrification water treatment could play a role as a possible solution.

Even though a detailed health risk assessment of data currently available in Nigeria has not been attempted on a provincial scale, a high percentage of the population is at risk of ingesting increased doses of nitrate through drinking water. Nitrate pollution control programs must therefore be introduced at the national level to protect groundwater against nitrate pollution caused by agricultural activity. This is very important in Nigeria in view of the fact that agriculture still dominates in the occupational and employment status of the country.

Finally, this paper is written with the purpose to characterise the groundwater quality of the country in view of the global nitrate problem; it seems necessary to carry out a complete shallow groundwater sampling across the country. The establishment of a groundwater monitoring network is therefore suggested in this regard.

REFERENCES

Adelana, S.M.A. 2005. Nitrate health effects. In J.H. Lehr & J. Keeley (eds), *Water encyclopedia Vol.4: Municipal and industrial water supply*. New York: John Wiley & Sons.

Adelana, S.M.A. 2004. Water pollution by nitrate in a weathered/fractured basement rock aquifer: The case of Offa area, Nigeria. In R. Xi, W. Gu & K.P. Seiler (eds), *Research basin and hydrological planning*: 93–98. London: Balkema.

Adelana, S.M.A. & Olasehinde, P.I. 2003. High nitrate in water supply in Nigeria: implications for human health. *Water Resources* 14(1): 1–11.

Adelana, S.M.A., Bale, R.B. & Wu, M. 2004. Water quality in a growing urban centre along the coast of southwestern Nigeria. In R. Xi, W. Gu & K.P. Seiler (eds), *Research basin and hydrological planning*: 83–91. London: Balkema.

Adelana, S.M.A., Bale, R.B. & Wu, M. 2003. Quality assessment and pollution vulnerability of groundwater in Lagos metropolis, SW Nigeria. *Proc. Aquifer Vulnerability Risks (AVR03), Salamanca, Mexico, 28–30 May 2003*.

Adelana, S.M.A., Olasehinde, P.I. & Vrbka, P. 2001. Hydrochemical Characteristics of groundwater in the Sokoto Basin, NW Nigeria, West Africa. *Zentralblatt für Geologie und Paläontologie,* Teil I, Heft 3/4: 365–374.

Akiti, T.T. 1982. Nitrate levels in some granitic aquifers from Ghana. *Proc. Intern. Symp. on Impact of Agricultural Activities on Groundwater*, IAH Mem. XVI, Part1, pp. 87–98.

Akujieze, C.N., Coker, S.J.L. & Oteze, G.E. 2002. Groundwater in Nigeria – a millennium experience – distribution, practice, problems and solutions. *Hydrogeol. Journal* 11: 259–274.

Anderson, H.R. & Ogilbee, W. 1973. Aquifers in the Sokoto Basin, Northwestern Nigeria, with a description of the general hydrogeology of the region. *Geol. Survey Water Supply paper* 1757-L, 1–79.

Aremu, D.A., Olawuyi, J.F., Meshitsuka, S., Sridhar, M.K. & Oluwande, P.A. 2002. Heavy metal analysis of groundwater from Warri, Nigeria. *Int. Jour. Env. Health Res.* 12(3): 261–267.

Bassey, N. & Opeloye, S.A. 2000. Groundwater quality data in the Yola arm sedimentary Basin of Adamawa State, NE Nigeria. *Water Resources* 11: 26–30.

Bowman, J. 1994. Water is best: Would Pindar still think so? In B. Cartledge (eds), *Health and the environment: The Linacre lectures 1992–3:* 85–125. Oxford: Oxford University Press.

Canter, L.W. 1996. *Nitrates in groundwater*. New York: Lewis.

Colvin, C. 1999. Increased risk of methemoglobinemia as a result of bottle-feeding by HIV positive mothers in South Africa. Paper presented at IAH Congress, Bratislava, Slovakia.

Du Preez, J.W. & Barber, W. 1965. The distribution and chemical quality of groundwater in Northern Nigeria. *Geol. Surv. Nigeria Bul.* 36: 1–93.

Edet, A.E. 2000. Water pollution by nitrate near some waste disposal sites in Calabar, Nigeria. *Groundwater 2000. Proc. Int. Conf. Groundwater research, Copenhagen, 6–8 June.*

Edmunds, W.M., Fellman, E., Goni, I.B. & Prudhomme, C. 2002. Spatial and temporal distribution of groundwater recharge in northern Nigeria. *Hydrol. J.* 10: 205–215.

Edmunds, W.M., Fellman, E. & Goni, I.B. 1999. Lakes, groundwater and palaeohydrology in the Sahel of northeast Nigeria: evidence from hydrogeochemistry. *J. Geol. Soc. Lond.* 156: 345–355.

Egboka, B.C.E. & Ezeonu, F.C. 1990. Nitrate and Nitrite pollution and contamination in parts of SE Nigeria. *Water Resources* 2(2): 101–110.

Faillat, J.P. 1990. Origin desnitrates dans les nappes de fissures de la zone tropicale humide. Exemple de la Cote d'Ivoire. *J. Hydrol.* 113: 231–264.

Faillat, J.P. & Rambaud, A. 1988. La teneur en nitratea des nappes de fissures de la zone tropicale humide en relation avec les problems de deforestation. *C.R. Acad. Sci. Paris 306*, Series II: 1115–1120.

Fedkiw, J. 1991. Nitrate Occurrence in Waters. *USDA*, Washington D.C.

Goni, I.B., Fellman, E. & Edmunds, W.M. 2001. Rainfall geochemistry in the Sahel region of northern Nigeria. *Atmos. Environ.* 35: 4331–4339.

Kiser, R.T. 1968. Chemical quality of groundwater in northern Nigeria. Open-file report, Geol. Survey Nigeria, Kaduna.

Kogbe, C.A. 1989. *Geology of Nigeria*. Jos: Rock View Publ. Co.

Hartman, P.E. 1983. Review: putative mutagens and carcinogens in foods. *Environmental Mutagenesis*, 5, 111–121.

Hesseling, P.B., Toens, P.D. & Visser, H. 1991. An epidemeological survey to assess the effects of well water nitrates on infants' health at Rietfontein in the northern Cape Province, South Africa. *South Afr. J. Sci.* 87, 300–304.

Langenegger, O. 1981. High nitrate concentrations in shallow aquifers in a rural area of Central Nigeria caused by random deposits of domestic refuse and excrement. In W. van Duijvenbooden, P. Glasbergen & H. van Lelyveld (eds), Quality of Groundwater in The Netherlands. *Studies in Environmental Science*, Vol. 17, pp. 135–140.

Mbonu, M. & Ibrahim-Yusuf, A.I. 1994. Groundwater quality in the Basement Complex region of north-central Nigeria – A preliminary survey of nitrate concentration. *Water Resources* 5(1 & 2): 16–21.

Nigerian Federal Environmental Protection Agency (NFEPA). 1991. Drinking water standards recommendations, Lagos: NFEPA.

Ntekim, E.E. & Wandate, S.D. 2001. Groundwater chemistry of the Basement Rock units of Adamawa area NE Nigeria: A reflection of the aquifer rock types. *Water Resources* 12: 67–73.

Offodile, M.E. 1992. *An approach to ground water study and development in Nigeria*. Jos: Mecon publishers.

Onugba, A. 1990. Contribution a la connaissance des systems aquifers de la Haute Benoue (Nigeria). Etude hydrochinique de zones tenoins du socle et du sedimentaire. Vulnerabilite des forages a la pollution. These de doctorate. Univ. d'Avignon. 203p.

Rahaman, M.A. 1989. Review of the basement geology of southwestern Nigeria. In C.A. Kogbe (eds), *Geology of Nigeria (2nd edition)*: 39–56. Jos: Rock View Publ.

Shuval, H.I. & Gruener, N. 1977. Infant methaemoglobinemia and other health effects of nitrates in drinking water. *Prog. Water Technol.* 8, 183–193.

Shuval, H.I. & Gruener, N. 1972. Epidemiological and toxicological aspects of nitrates and nitrites in the environment. *Ame. J. Publ. Health* 62, 1045–1052.

South African Bureau of Standards (SABS). 1984. Specification for water for domestic supplies,SABS241-1984, Pretoria.

Spalding, R.F. & Exner, M.E. 1993. Occurrence of nitrate in groundwater – A review. *J. Environ. Qual.* 22: 392–402.

Super, M., Heese, H., de, V., MacKenzie, D., Dempster, W.S., Du Plessis, J. & Ferreira, J.J. 1981. An epidemiological study of well water nitrates in a group of Namibian infants. *Water Res.*, 15, 1265–1270.

Tredoux, G. 1993. Preliminary investigation of nitrate content of groundwater and limitation of the nitrate input. WRC Report No. 368/1/93, Pretoria, 76pp.

Uma, K.O. 1993. Nitrates in shallow (regolith) aquifers around Sokoto Town, Nigeria. *Environ. Geol.* 21: 70–76.

United States Environmental Protection Agency (USEPA). 1994. Nitrogen Control, Technomic Publ. Inc. Lancaster, Pennsylvania, pp. 1–22.

Wolff, I.A. & Wasserman A.E. 1972. Nitrates, nitrites and nitrosamines. *Science* 177: 15–19.

World Health Organisation (WHO). 1985. *Health hazards from nitrates in drinking water*: 1–102, Copenhagen: WHO.

4

Groundwater quality in Cameroon and its vulnerability to pollution

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ABSTRACT: A synthesis of the work carried out on groundwater quality in Cameroon was done in an attempt to identify the geologic (natural) and anthropogenic (artificial) controls on the quality of the resource. The results show that the mineral content of groundwater is generally highest in the volcanic (500 mg/l TDS), intermediate in the sedimentary (300–500 mg/l) and lowest in the crystalline (300 mg/l) formations. Exceptions to this general trend are the salt springs of the Mamfe basin which, although found in sediments, have the highest TDS values (36000 mg/l) in the country. Anthropogenic input has not yet had any significant impact on the quality of the resource. Indicators however suggest a very high pollution potential, especially from a biological viewpoint. Consequently, if mitigative schemes are not established early, anthropogenic influences will have a very severe impact on groundwater quality in Cameroon.

Key words: groundwater, quality, pollution, vulnerability, Cameroon

1 INTRODUCTON

Cameroon is endowed with a large amount of water resources. It is the country with the second highest volume of available water in Africa (after the Democratic Republic of Congo) estimated at 322 billion m^3. Groundwater constitutes 21.5% (57 billion m^3) of this resource and plays a very important role in the socio-economic life of the country. Common uses of groundwater include drinking, domestic, industrial and agricultural. Understanding the quality of groundwater is therefore fundamental for the sustainable use of the resource. Unfortunately, little has been done to understand the quality of groundwater in Cameroon. That notwithstanding, the available limited results and findings need to be assessed and evaluated before further detailed work is commissioned.

This paper is a synthesis of the results of the few studies that have been done to investigate the quality of groundwater in Cameroon. Because groundwater quality is invariably influenced by geology, the discussion is constrained within the geologic framework of Cameroon. The discussions on the vulnerability of groundwater to pollution are mostly speculative. The approach was to first infer the types and causes of pollution from circumstantial evidence and then upgrade the findings with hard evidence from chemical analyses. The chemical analyses support the speculations and emphasize the need for the implementation of best management practices. These practices will help to minimize the incidence of groundwater pollution and even prevent it in unspoiled areas.

2 GROUNDWATER OCCURRENCE

Groundwater in Cameroon occurs in crystalline (metamorphic/plutonic), volcanic and sedimentary terrains. Figure 4.1 shows the main saturated geological formations in the country.

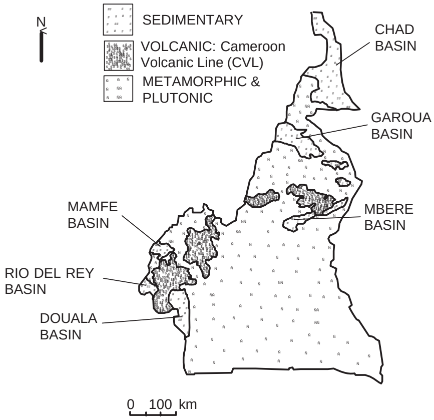

Figure 4.1. Main saturated geological formations in Cameroon.

3 GROUNDWATER QUALITY

3.1 *Sedimentary formations*

The most studied groundwater resource in Cameroon is that in sedimentary formations. These aquifers are easily accessible (due to a generally shallow water table) and the water is generally of good quality. For these reasons, groundwater in the sedimentary formations is widely used and a very significant resource.

3.1.1 *Douala basin*

The most exploited aquifers in Cameroon are those of the Douala sedimentary basin. The main aquifer in this basin is the Plio-Pleistocene Wouri alluvial aquifer, typified by a multi-layer aquifer system composed of an alternating sequence of marine sands and estuarine mud and silt (Regnoult, 1986). The groundwater composition varies with the configuration of the aquifer.

The upper aquifer in this system is an unconfined sandy horizon. It is hydraulically connected with the brackish waters of the Wouri estuary and the saline waters of the coastal wetlands. From time to time, seawater intrusion occurs in this aquifer. This occurs in the dry season when low recharge causes a reversal in the hydraulic gradient (Mafany, 1999). Calculated Revelle's coefficients are less than one for most of the samples, while trilinear plots (Fig. 4.2) show most of the samples plotting towards the seawater end. The Na-Cl water type dominates the chemical facies in this shallow water table aquifer.

In the deep confined aquifers, the water is dominated by $Na\text{-}HCO_3$ and $Ca\text{-}HCO_3$ water-types. The confining clays provide a buffering effect and this prevents seawater intrusion. The good quality of the water makes it a resource that is widely used by industries and households in Douala.

Another important aquifer in the Douala basin is the sandstone aquifer that is separated from the Plio-Pleistocene aquifer by the shales of the Nkappa formation. This aquifer is not susceptible to seawater intrusion due to the overlying impermeable shales. It is dominated by $(K,Na)\text{-}HCO_3$ water-types. The groundwater is also of very good quality and is used to supplement the domestic/industrial water needs of the Douala Municipality.

In some areas (e.g. Bonanjo, Bonaberi and Massoumbou) in the Douala basin, groundwater upon aeration forms reddish-brown flocs. These flocs have been identified as iron hydroxide (Mafany, 1999). Iron fixing bacteria, associated with sedimentary environments and decaying organic matter, usually cause iron contamination in groundwater (Effeotor and Odigi, 1983). Dissolved iron

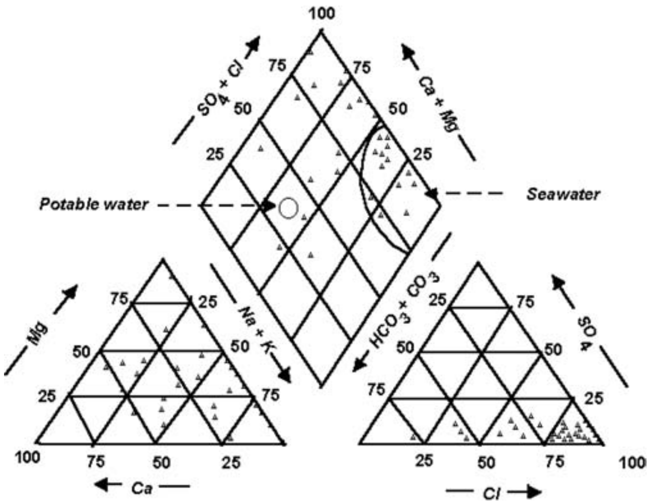

Figure 4.2. Trilinear plot of groundwater from unconfined aquifer in Douala showing seawater contamination. *(Source: Mafany, 1999).*

concentrations range from 1.5 to 9 mg/l. Borehole information reveals the presence of decaying plant matter and pyrite nodules in the sandstone aquifers. Hence it is suggested that the dissolved iron is derived from the bacterially catalysed oxidation of pyrite, as shown in the reaction:

$$2FeS_2 + 2H_2O + 7O_2 \leftrightarrow 2Fe^{2+} + 4SO_4^{2+} + 4H^+$$

This reaction produces large amounts of dissolved iron (Heathcote and Lloyd, 1985) and it is also one of the most prolific acid-producing (H^+) reactions in nature (Appelo and Postma, 1993, Domenico and Schwartz, 1998). In the presence of sulphate reducing bacteria, as is the case in some areas in Douala, the released hydrogen ion combines with sulphate to produce hydrogen sulphide gas H_2S. This explains the repugnant sulphurous smell that is characteristic of groundwater in some parts of the basin. When the iron rich water is pumped to the surface where it is exposed to gas phase oxygen, the dissolved ferrous iron (Fe^{2+}) is oxidised to the ferric state (Fe^{3+}) (Younger, 1995) in the reaction:

$$2Fe^{2+}{}_{(aq)} + 1/2\, O_{2(g)} + 5H_2O_{(aq)} \leftrightarrow 2Fe(OH)_{3(s)} + 4H^+{}_{(aq)}$$

This results in the spontaneous precipitation of the iron oxyhydroxide flocs (Davies, 1994) which are observed in the groundwater.

3.1.2 *Garoua basin*

The Garoua basin is the eastern extension of the Bima sandstone in north-eastern Nigeria. Petrographic studies show that silica in the form of quartz and iron hydroxide are the main cementing material of the sandstone. Groundwater in this basin is basically acidic, with a pH-range of 4–7.1 mg/l. Chloride values may be as low as 5.9 mg/l and as high as 410 mg/l, giving a mean of 137 mg/l (Obiefuna *et al.*, 1999). The sulphate content of the water indicates a mean of 30 mg/l in a bracket of 11–60 mg/l. Groundwater in two villages in the south-western part of the basin is characterised by very high fluoride concentrations, causing mottled teeth in the riparian human population.

3.1.3 *Mamfe basin*

Groundwater quality in the Mamfe basin is greatly influenced by evaporite deposits that constitute part of the sediments of the basin. The model proposed by Eseme *et al.* (2003) entails meteoric water recharge through fractures (Tanyileke, 1994) dissolving the deep-seated evaporite deposits, principally of Na-Cl composition. The water then discharges to the surface as highly mineralised springs dominated by the Na-Cl waters. Concentrations of TDS are slightly higher than seawater and range from 13,000 to 36,000 mg/l (Marechal, 1976).These springs are localised only on the northern and southern limits of the basin (Marechal, 1976), probably along faults bordering the basin. As the rest of the basin hosts water of good quality (dominated by (Ca,Na)-HCO_3) the influence of the springs on groundwater quality in the basin is limited to the margins of the basin.

3.1.4 *Chad basin*

The aquifers surrounding Lake Chad are in hydraulic continuity with the lake water. Within the past 30 years, the surface area of the lake has reduced from 25,000 km^2 to 2500 km^2 as a result of hydro-climatic changes and anthropogenic activity that have dramatically reduced the surface water contribution to the lake. The supply of water into the lake is now mainly being sustained by baseflow from the local groundwater system.

The cation content of the lake waters is typified by Ca, Na, Mg and K with Ca + Mg > Na + K (Roche, 1980). The main anion is HCO_3, with a weak presence of Cl (0.02 meq/l) and SO_4 (0.1 meq/l). Ca-HCO_3 is therefore the principal chemical facies. This trend is reversed in the ambient groundwater, where Na increases while Ca and Mg decrease with depth. This results in an essentially Na-HCO_3 water type. SO_4 concentrations are also observed to increase with depth.

3.2 *Volcanic formations*

The Cameroon Volcanic Line (CVL) is the principal watershed of the country. It is dominated by basaltic rocks with the occasional occurrence of intermediate to acidic rocks such as phonolites, trachytes and rhyolites on the continental part. Also found on the flanks and foothills of the volcanic centres on the CVL are large quantities of loose pyroclastic and tuffaceous ash deposits. These formations form aquifers that store large quantities of water. The numerous springs that emerge from the CVL are typified by Fe-Mg-Ca-HCO_3 or Na-HCO_3 compositions (Aka *et al.*, 2001; Endeley *et al.*, 2001). Marechal (1976) described 97 thermo-mineral springs along the CVL. The HCO_3 anion constitutes more than 90% of the anionic content of 65 of these springs.

The chemical facies can be further sub-divided into the following:

- Na-HCO_3 (4 springs),
- Na > Ca > Mg-HCO_3 (31 springs),
- Na > Mg < Ca-HCO_3 (2 springs),
- Ca > Na > Mg-HCO_3 (5 springs),
- Ca > Mg > Na-HCO_3 (9 springs),
- Mg > Ca > Na-HCO_3 (13 springs),
- Na-(HCO_3 > SO_4) (23 springs),
- Na-(HCO_3 > Cl) (9 springs).

3.3 *Crystalline formations*

The hydrogeology of the crystalline formations is the least studied and least known. Paradoxically, groundwater in crystalline formations is the most spatially abundant in the country and is accordingly the most exploited by the rural communities. Extraction is mostly through hand-dug wells and the principal use is for drinking. In many rural areas, groundwater is the sole source for domestic use. The water quality presented here is based on work carried out in the south-central part of Cameroon by the Ministry of Mines, Water and Energy (MINMEE) in 1990. A significant limitation of this work

Table 4.1. Range of concentrations of Mg and HCO_3 ions in various crystalline aquifers in south-central Cameroon.

Aquifer type	Mg(mg/l)	HCO_3 (mg/l)
Migmatite	3.53–11.35	7.82–19.55
Gneiss	7.83–22.70	11.73–31.25
Schist (mica)	11.35–34.06	11.23–50.82

is that the Na concentrations were not determined. A study is therefore needed to establish the concentrations, at least of all major ions, some trace elements as well as a few isotopes. The discussions here are therefore based on the available data.

Beta (1976) and Djeuda Tchapnga *et al.* (1987) identified two main aquifers: a top, shallow aquifer and a deep aquifer. The thick, weathered lateritic blanket and the highly altered and fractured rocks form the top aquifer. The thickness of this aquifer ranges from 8 to 20 m. The deep aquifer is composed of low permeability fractured rocks. The lithology in both aquifers is either migmatitic, gneissic, quartzitic or schistose. The top aquifer is the most widely exploited.

Groundwater in this crystalline, fractured terrain has low mineralization. The TDS rarely exceeds 300 mg/l. This correlates well with conductivities in the range of 31–345 μS/cm, although exceptionally high values of above 1000 μS/cm have been noted in isolated cases. The water type is principally of the Mg-HCO_3 facies. Mg values range from 3.53 to 34.06 mg/l, while the HCO_3 bracket is between 7.82 and 50.82 mg/l. The Mg and HCO3 concentrations in groundwater generally show a relative increase in concentration from the migmatites through the gneisses to the mica schist, with an overlap among the various groups (Table 4.1).

4 GROUNDWATER VULNERABILITY TO POLLUTION

4.1 *Bacteriological pollution*

Bacteriological pollution of groundwater has an immediate impact on human health. Cases of water-borne diseases such as typhoid, cholera, and ameobic dysentry are recurrent in most of the urban areas in Cameroon. Between 1984 and 1993 for instance, 8000 cases of cholera, 11,500 cases of typhoid fever and 46,400 cases of ameobic dysentry were recorded (Nola, 1996). The most populated towns in the country, Douala and Yaounde, were the most affected. This pollution has been traced to the use of water from shallow, unprotected hand-dug wells. The geology of these two towns is different: Yaounde is located on migmatites (Dumort, 1968; Regnoult, 1986) which are overlain by a beige-coloured weathered blanket, 20 m thick on average. Groundwater flow in Yaounde is controlled by secondary porosity. The faults, fractures and joints that control the permeability in this crystalline aquifer were formed by an extensional tectonic event (Ngako *et al.*, 2003; Mvondo *et al.*, 2003). Areas of extensional tectonics are very good targets for groundwater exploitation because of the high permeability of these structures. This also improves recharge rates significantly. The thick weathered blanket may slow down recharge, but the impedance is insignificant because the weathered material is composed of porous, highly interconnected lateritic duricrust, which facilitates the percolation of water into the subsurface. A contaminant source from the surface will therefore easily reach the groundwater system.

The geohydrological conditions in Douala are different. The municipality is underlain by alluvium. In Douala, primary porosity dominates (Mafany, 1999). The sands and gravels are angular to subangular, indicating short periods of transportation. These sands are generally cross-bedded (Regnoult, 1986) with frequent intercalations of ferallitic indurations and red, white and grey to black clays culminating in a multi-aquifer system. The top unit is an unconfined, shallow sandy aquifer. Most of the hand-dug wells are limited to this surface aquifer, while the boreholes abstract from the deeper confined aquifers. More than 60% of the population relies on these unprotected hand-dug wells, which are highly susceptible to contamination from percolating surface water.

In spite of the different geologies of the main urban centres in Cameroon, microbial pollution is similar in the two cities. Minimum fecal coliform counts of 300 FCU/100 ml have been noted in surface and shallow groundwaters around major hospitals in Yaoundé and Douala (HOSWAP, 2005). Nola (1996) did fecal coliform counts for wells and springs in Yaoundé and obtained a range of 0 to 14,000 FCU/100 ml. Although the geology may play a very important role in the incidence of pollution, the situation in these two settlements is primarily a function of the population, the per capita income of the population and the waste disposal facilities. Highly populated, low income neigbourhoods dispose of their domestic waste at the surface or in unprotected landfills and latrines. Most inhabitants cannot afford the capital to build water cistern toilets or protected wells. These are the areas where pollution is rife. Neighbourhoods inhabited by high income inhabitants have low population densities and well-protected wells and toilets. These areas show the lowest levels of pollution.

The fecal coliform signatures are generally high in May, drop in June and July, start increasing in August and soar in either September, October or November, when the highest counts are observed (Nola, 1996). The high values in May are explained by the first rains flushing the surface system. The run-off water carries concentrated leachate from surface landfills and flooded latrines directly into the groundwater system through the open mouths of unprotected hand-dug wells. As more rain falls, dilution occurs and results in a drop. Percolating rainwater reaches the groundwater system from September, peaks in October and November and starts dropping in December. Accordingly the fecal budget of the groundwater system peaks at this period. Contaminated water is therefore introduced into the groundwater system via two main routes: directly through the apertures of unprotected hand-dug wells and indirectly through the normal recharge routes.

4.2 *Chemical pollution*

The potential of chemical pollution is higher in the northern parts and the south-western coast of Cameroon. This is due to the combination of anthropogenic activities and the geology of these areas. The south-western coast of Cameroon is characterised by intensive agro-industrial activity with extensive use of fertilizers and pesticides. Douala, the main town in the south-western coast of Cameroon, has the highest concentration of industries in the country. Effluent from these industries is discharged untreated onto the surface, where it flows into rivers (Eneke, 2001) or percolates down to the groundwater system (Ketchemen *et al.*, 2001). Pesticides and nitrates represent the main sources of aquifer contamination in agricultural zones (Dupuy *et al.*, 1997a; 1997b), especially if the geology is favourable. The geology of the south-western coast of Cameroon begins with sediments in the east, changes onto an intercalation of sediments and volcanics in the centre, and ends with volcanics in the west. The northern part of the country is typified by Cretaceous sandstones overlain by recent alluvial sediments. These are highly porous media that readily allows the introduction of contentious concentrations of fertilizer and/or pesticide components.

Inorganic fertilizers usually contain potash, nitrogen and phosphorus compounds (UNESCO, 2004). The application of these compounds enriches the soil with potassium, calcium, chloride, nitrate and phosphate. Most potash and nitrogen compounds are highly soluble and could reach the water table if applied in excessive amounts. Hence, when applied in excessive amounts on agricultural land or improperly stored in stockpiles, inorganic fertilizers may lead to unacceptable or even toxic concentrations of chemical constituents in groundwater (Nonner, 2004). Pesticides may not pose a threat in regions where environmental regulations are adequate (Khan, Sial and Mahmood, 1998). In a place like Cameroon, where environmental regulations are inadequate and those that do exist are loosely applied, the use of pesticides is a threat to the quality of groundwater resources in agricultural areas. Circumstantial evidence inferred from the extensive application of fertilizers in the many agro-industrial plantations distributed all over the country, and the indiscriminate disposal of industrial and domestic waste, shows that inorganic products have a high contaminant potential to the groundwater system.

It is suspected that artisan mining of gold in south-eastern Cameroon might be releasing substantial quantities of arsenic in the groundwater of the region. The primary source of arsenic in nature is the oxidation of arsenic sulphides such as arsenopyrite (FeAsS) and pyrite (FeS_2) (Bhattacharya *et al.*, 2004). Arsenic is a relatively mobile element in hydrothermal systems (Zhou, 1987; O'Connor and Gallagher, 1987) like those of east Cameroon where values of up to 2640 ppm have been observed in the hydrothermally altered granitic wall rock (Suh *et al.*, in press). Common artisan mining practices in the area involve the sinking of pits or shafts and the drilling of tunnels in pursuit of the gold-bearing quartz veins and highly fractured hydrothermally altered wall rock. The water table may be as shallow as 1 m along the banks of rivers and as deep as 20 m in higher elevations. The shafts range in depth from 5 to 25 m, commonly intercepting the water table so that access to the gold-bearing lithologies is impeded. To circumvent this, the miners use submersible pumps to get rid of the water. Pumping and dewatering facilitate the release of arsenic into the groundwater (Nonner, 2004) as these processes introduce gas phase oxygen into the groundwater, resulting in the oxidation of arsenic sulphides to release arsenic.

5 CONCLUSION

The most mineralized groundwater is found in the volcanics, while the least mineralized is in the crystalline formations. The sediments are of intermediate composition. A deviation from this general trend are the salt springs of Mamfe, which, although found in sediments, have the highest groundwater mineral content in the country. The good quality of groundwater in the volcanics has resulted in the exploitation of the resource by five of the six water bottling plants in the country.

In Douala, groundwater abstracted from deep confined aquifers is expected to be of good quality. The problems of iron and H_2S may be overcome by aeration. When the water is aerated, the oxygen will cleanse it of iron through oxidation and the iron hydroxide flocs precipitating through this process can easily be filtered. The H_2S exists in solution only at the subsurface. At the surface, where pressures are lower, it is easily exsolved out of solution. Human activity has not yet had any significant impact on the groundwater resources. Given the vulnerability due to the specific geology and the industrial activity in the town of Douala, it is expected to show the highest level of groundwater contamination. This is however not the case, as Mafany (1999) has identified seawater intrusion as the main source of chemical contamination in the shallow unconfined aquifer. Ketchemen *et al.* (2001) and Eneke (2001) underpinned the occurrence of anthropogenic pollution in surface waters, but did not advance any arguments for the incidence of anthropogenic-related chemical contamination in the groundwater system: not even in the phreatic aquifer, into which surface run-off percolates easily. Asaah (2004) has identified negligible groundwater pollution by a study that focused on soil and stream sediment samples in the main industrial centres of Douala. Major ions and trace elements concentrations were very low (TDS 370 mg/l). From a physico-chemical standpoint therefore, groundwater in Douala has been negligibly affected by human activity. It is inferred that if Douala, the most industrialised and highly pollution-susceptible area in Cameroon, shows no significant pollution of groundwater, the rest of the country will have even less pollution.

The situation is however not the same from a bacteriological perspective. Shallow groundwater in densely populated areas of Cameroon, such as Yoaundé and Douala, has fecal coliform signatures higher than 300 CFU/mL. Indiscriminate disposal of domestic, industrial and hospital waste is responsible for this contamination.

For now, the geology is the principal factor that controls the chemical quality of groundwater. Relating the major groundwater chemical facies to their geologic occurrence shows that the volcanics are characterized by (Ca,Na)-HCO_3 water-type, the sediments by (Na,Ca,Mg)-HCO_3 waters and the crystalline by (Mg,K,Na)-HCO_3 water types. The sediments and volcanics sometimes show important pockets of Na-Cl facies. The volcanic occurrence of Na-Cl waters is explained by subsurface igneous activity (Marechal, 1976; Tanyileke *et al.*, 1994; Fantong, 1999; Aka *et al.*, 2003) while the sedimentary signatures are accounted for by the dissolution of evaporites by meteoric waters (Marechal, 1976; Eseme, 2003).

This study has demonstrated that knowledge on the major ion composition of groundwater, the mineralogy of the aquifers and the biological regime of groundwater can provide insight in the quality of groundwater in Cameroon. With the changing hydro-climatic regime and the increasing impact of human activity on the environment, there is a need for a more systematic study on groundwater quality in the country (beginning with the areas under high anthropogenic and natural pressures, such as Douala and the grand north). Such a study, if extended to include trace elements and isotopes, will be indispensable to the sustainable use of the groundwater resources of Cameroon and the Central-African sub-region.

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REFERENCES

Appelo, C.A.J. and Postma, D. 1993. Geochemistry, Groundwater and Pollution. Balkema, Rotterdam.

Aka, F.T., Kusakabe, M., Nagao, K. and Tanyileke, G. 2001. Noble gas isotopic compositions and water/gas chemistry of soda springs from the islands of Bioko, Sao Tome and Annobon, along with Cameroon Volcanic Line, West Africa. *Applied Geochemistry* 16, pp. 323–338.

Asaah, A.V. 2004. Heavy metals concentrations and distribution in the soils of the Bassa Industrial Zone in Douala, Cameroon. M.Sc. Thesis, Ibadan Univ. Nigeria 90p.

Betah, S.S. 1976. Complement sur la monographe nationale de l'eau (aspects eaux souterrraines). Ministère des Mines et Energie, Yaounde.

Bhattacharya, P., Welch, A.H., Ahmed, K.M., Jacks, G. and Naidu, R. 2004. Arsenic in groundwater of sedimentary aquifers. *Applied geochemistry* 19: pp. 163–167.

Davies, J. 1994. The hydrogeochemistry of alluvial aquifers in central Bangladesh. In: *Groundwater Quality* (Edited by Nash, H. and Wells, C.J).

Djeuda Tchapnga, H.D., Tanawa, E. and Ngnikam, E.C. 2001. L'eau au Cameroun. Press Universitaire de Yaounde.

Domenico, P.A. and Schwartz, F.W. 1998. *Physical and Chemical Hydrogeology*. John Wiley and sons, New York: 497–499.

Dupuy, A., Razack, M. and Banton, O. 1997a. Groundwater pollution by nitrates in a heterogenous agricultural watershed I: Evaluation of inputs to the water table (AgriFlux model). *Revue des Sciences de l'eau* 1, pp. 23–24.

Dupuy, A., Razack, M. and Banton, O. 1997b. Groundwater pollution by nitrates in a heterogenous agricultural watershed II: Evolution of concentrations in the aquifer. *Revue des Sciences de l'eau* 2, pp. 185–198.

Effeotor, J.O. and Odigi, M.I. 1983. Water supply problems in eastern Niger Delta. *Nig. J. Min. and Geol.*, 20: 190–193.

Eseme, E., Agyingi, C. and Foba. 2003. Geochemistry of salts springs in Mamfe basin Cameroon. *J. African Earth Sc*.

Endeley, R.E., Ayonghe, S.N. and Tchuenteu, F. 2001. A hydrogeochemical baseline study of water sources around Mount Cameroon and the possible effects on human health. *J. Geosc. Soc. Cameroon*, 1: 46 (abstract).

Eneke, G.T. 2000. Possible effects of industrial waste on surface and groundwater quality in the Bassa industrial zone Doaula. Unpubl. M.Sc. thesis Buea. Univ. Cameroon.

Fantong, W.Y., Ayonghe, S.N., Tchuente, F., Suh, C.E. and Ntasin, E. 2001. Major chemical characteristics of groundwater around the Lobe hot spring on the northwest flank of Mount Cameroon. *J. Geosc. Soc. Cameroon,* 1: 49 (abstract).

Heathcote, J.A. and Lloyd, J.W. 1985. Groundwater chemistry in southeast Suffolk (U.K) and its relation to Quartenary geology. *J. Hydrol*., 75: 143–165.

HOSWAP. 2005. Impact of hospital waste on water resources in Yaoundé, Hospital Waste Project report no 1. Compiled by Mafany, G.T, Fantong, W.T and Nkeng, G.E. 15p.

Khan Sial, J. and Mahmood, S. 1998. Contamination of groundwater from agro-chemicals in irrigated areas of Pakistan. In *Water: a looming crisis? Proc Int Con World Water Resources at the Beginning of 21st Century*. Edited by Zebedi, H. Unesco.

Ketchemen-Tandia, B., Emblanch, C., Ntamack-Nida., Kamta-Fotio, B.M.J.G. and Faye, A. 2001. Première approche de la différenciation physico-chimiques des aquifères du bassin de Douala. J. Geosc. Soc. Cameroon, 1: 59 (abstract).

Mafany, G.T. 1999. Impact of the geology and seawater intrusion on groundwater quality in Douala. M.Sc. Thesis, Buea Univ. Cameroon 100p.

Marechal, A. 1976. Geologie et geochimie des source thermominerale du Cameroun. *Travaux et documents de l'ORSTOM*. N^{o} 56, 176p.

Mvondo, H., den Brok, S.W.J. and Mvondo Ondoa, J. 2003. Evidence for symmetric extension and exhumation of the Yaounde nappe (Pan-African fold belt, Cameroon). *J. Afr Earth Sc.* 36: pp. 215–231.

Ngako, V., Affaton, P., Nnange, J.M. and Njanko, Th. 2003. Pan-African tectonic evolution in central and southern Cameroon: transpression and transtension during sinistral shear movements. *J. Afr Earth Sc*. 36: pp. 207–214.

Nola, M. 1996. Peuplement bacterien des eaux souterraines de Yaounde (Cameroon): bioindicateurs et pathogenes opportunistes. Doctorat de 3e Cycle, Yaounde Univ. Cameroon, 175p.

Nonner, J.C. 2004. Sources of groundwater contamination. In *Groundwater contamination inventory*. A methodological guide Ed. by Zaporozec, A. IHP-VI, series on groundwater 2: 160p.

Nonner, J.C. 2004. Arsenic contamination in groundwater in Bangladesh. In *Groundwater contamination inventory*. A methodological guide Ed. by Zaporozec, A. IHP-VI, series on groundwater 2: 160p.

Obiefuna, G.I., Nur, A., Baba, A.U. and Bassey, N.E. 1999. Geotechnical assessment of selected gully sites, Yola Area, Northeast Nigeria. *J. Envtal Geol*., 7.

O'Connor, P.J. and Gallagher, V. 1987. Gold prospectivity in the Caledonides of southeast Ireland: application of the upper-crustal reservoir model. *Trans. Instn Min Metall. Sec B: Appl. Earth sci.*, 103: pp. 175–187.

Regnoult, J.M. 1986. *Synthèse géologique du Cameroun*. Ministère des Mines et Energie, Yaounde: 17–23.

Roche, M.L. 1980. Tracage naturel salin et isotopique des eaux du systeme hydrologique du lac Tchad. *Travaux et documents de l'ORSTOM*. N° 117, 383p.

Suh, C.E., Lehmann, B. and Mafany, G.T. (In press) Primary gold mineralisation at Dimako-Mboscorro, southeast Cameroon.

Tanyileke, G.Z. 1994. Fluid geochemistry of CO_2-rich lakes and soda springs along the Cameroon Volcanic Line. Ph.D. Dissertation, Okayama Univ, Japan 193p.

Younger, P.L. 1995. Hydrogeochemistry of mine waters flowing from abandoned coal workings in county Durham. *Quart. J. Eng. Geol.*, 28: 101–113.

Zhou, J. 1987. Lithogeochemical exploration for copper and gold in Kilmelford district, Argyllshire, Scotland. *Trans. Instn Min Metall. Sec B: Appl. Earth Sci.*, 96: pp. 187–193.

5

Groundwater contamination in Ghana

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ABSTRACT: Until the last decade, groundwater contamination was not a priority concern for water resource managers in Ghana. The potential for groundwater contamination is now acknowledged widely after recent studies highlighted the hazard.

Groundwater contamination in Ghana is widespread and varied. Naturally occurring contamination is mostly the result of rock-water interactions and the concentrations of chemical constituents are often exceeding drinking water limits. Groundwater contamination by anthropogenic activities is mostly as a result of poorly designed hazardous waste disposal facilities, leakage from underground storage tanks and mine tailings and accidental spills. The application of fertilizers and pesticides in agriculture and their subsequent removal by running water and/or by leaching into the subsurface water resources also contribute to the menace.

The chemicals trapped in the subsurface constitute a major long-term contamination source for a groundwater system, resulting in a threat to the groundwater supply and a direct risk to human health, by volatilization of toxic compounds, for example. Once the groundwater system becomes contaminated, it is almost impossible to clean.

1 INTRODUCTION

In the last decade, there has been an increase in the exploitation of groundwater for the water supply needs of many small communities in Africa. Groundwater is not only feasible, but also the most economic source of potable water for scattered and remote communities. In Ghana, about 68 per cent of the population lives in rural communities. To meet the present and future challenges of population expansion in relation to the observed declining rainfall in most parts of Africa, including Ghana, it has become necessary to assess the groundwater resources of Ghana in order to manage it efficiently and utilize it sustainably.

Until the last decade, groundwater contamination was not a priority concern for water resource managers in Ghana. The potential for groundwater contamination is now acknowledged widely after recent studies highlighted certain issues.

Groundwater has become contaminated mostly as a result of poorly designed hazardous waste disposal facilities, leakage from underground storage tanks and mine tailings and accidental spills. Groundwater pollution can also be caused by the application of fertilizers and pesticides in agriculture. The chemicals trapped in the subsurface constitute a major long-term contamination source for a groundwater system, resulting in a threat to the groundwater supply and a direct risk to human health, by volatilization of toxic compounds, for example. Once the groundwater system becomes contaminated, it is almost impossible to clean.

This paper seeks to provide a general outlook of the situation as it currently pertains to Ghana, reported in a number of publications by individuals within and outside the country, who have conducted various research programmes over the last decade.

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2 STUDY AREA

Ghana is located on the west coast of Africa, with a total land area of 238,000 km^2. It lies between latitude 04°30′N and 11°25′N and longitude 01°E and 03°30′W (Fig. 5.1). The physiographic

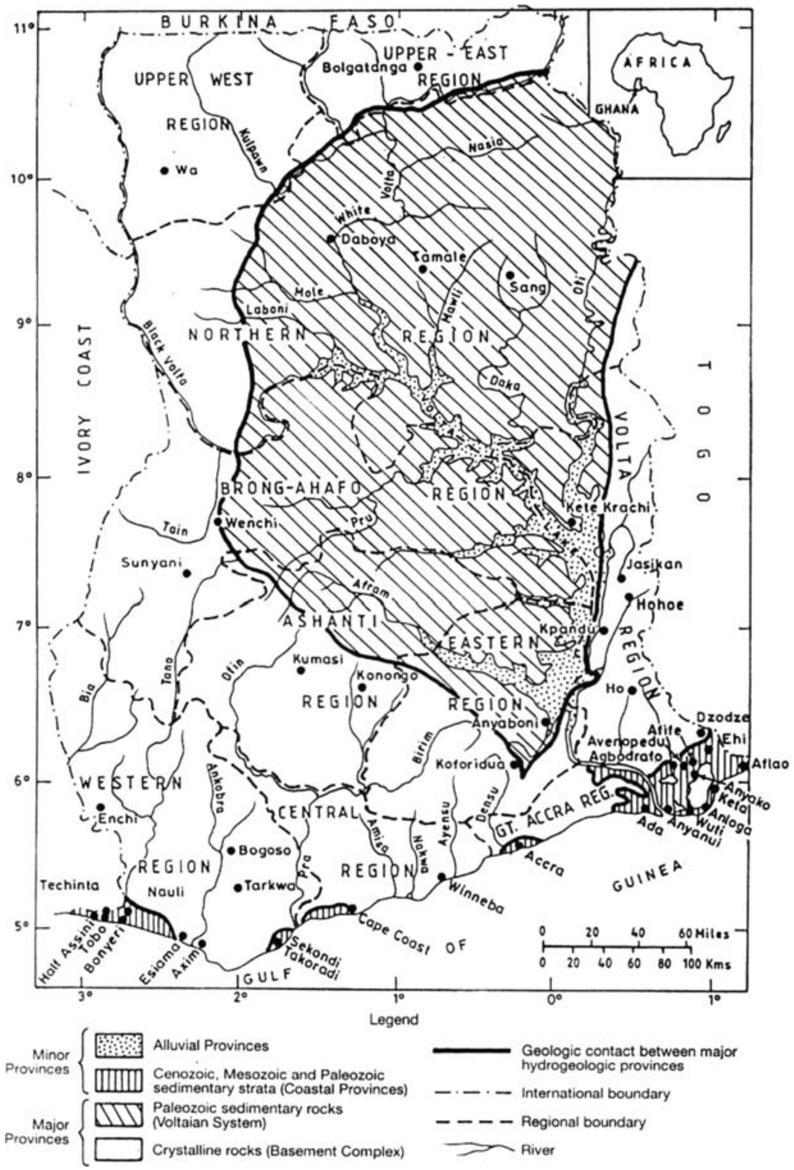

Figure 5.1. Hydrogeological provinces and river systems of Ghana (Geological survey of Ghana, 1969).

features are made up of the coastal plains in the south, a forest dissected plateau in the south-west, savannah plains in the north, the Voltaian basin covering large portions of the north-east to the central parts and the Togo-Buem ranges stretching from the coast to the north-east. The climate is largely influenced by the north-south shifts of the inter-tropical convergence zone of West Africa, with two distinct seasons; the dry (November–March) and wet (April–October) seasons. Mean annual rainfall ranges from 800 mm to 1500 mm, with monthly temperatures varying from 24°C to 28°C. The vegetation types are made up of the equatorial forest zone in the south-west, through a transitional zone, to a moist semi-deciduous tropical forest in the mid-north. In the north the vegetation is largely made up of a dry savannah-type, degrading to a semi-arid transition. In the south-east, the even banding of vegetation is interrupted because of poor rainfall.

3 HYDROGEOLOGY

Groundwater is abstracted for supply through either of the following:

- hand-dug wells, with or without hand pumps,
- boreholes with hand pumps or submersible pumps (depending on the yield) and
- protected springs.

Currently there are over 15,000 boreholes countrywide, and about 80,000 hand-dug wells in use.

Two major hydrogeological provinces have been identified in Ghana: (1) the Basement Complex, composed of Precambrian crystalline igneous and metamorphic rocks, and (2) Paleozoic sedimentary formations (Fig. 5.1).

Minor provinces consist of (1) Cenozoic, Mesozoic, and Paleozoic sedimentary strata along narrow belts on the coast; and (2) Quaternary alluvium along the major stream courses.

The Basement Complex underlies about 54 per cent of the country and is further divided into subprovinces on the basis of geologic and groundwater conditions (Gill, 1969). Generally, these subprovinces include the metamorphosed and folded rocks of the Birimian System, Dahomeyan System, Tarkwaian System, Togo Series, and the Buem Formation. The Basement Complex consists mainly of gneiss, phyllite, schist, migmatite, granite-gneiss, and quartzite. Large masses of granite have intruded the Birimian rocks.The distribution of these units is shown in Figure 5.2.

The Paleozoic sedimentary formations, locally referred to as the Voltaian Formation, underlie about 45 per cent of the country and consist mainly of sandstone, shale, arkose, mudstone, sandy and pebbly beds, and limestone. The Voltaian System is further subdivided, on the basis of lithology and field relationship, into the following sub-provinces (Junner and Hirst, 1946; Soviet Geological Survey Team, 1964–1966): (1) Upper Voltaian (massive sandstone and thin-bedded sandstone); (2) Middle Voltaian (Obosum and Oti Beds); and (3) Lower Voltaian (mainly sandstone).

The remaining 1 per cent of the rock formations is made up of two coastal provinces; namely the Coastal Block-Fault Province and the Coastal-Plain Province, and the Alluvial Province (Fig. 5.1). The Coastal Block-Fault Province consists of a narrow, discontinuous belt of Devonian and Jurassic sedimentary rocks that have been broken into numerous fault blocks and are transected by minor intrusives (Kesse, 1985). Semi-consolidated sediments, ranging from Cretaceous to Holocene in age, underlie the Coastal Plain hydrogeologic Province in south-eastern Ghana and in a relatively small isolated area in the extreme south-western part of the country. The Alluvial hydrogeologic Province includes narrow bands of alluvium of Quaternary age, occurring principally adjacent to the Volta River and its major tributaries, and in the Volta Delta.

In most parts of the Basement Complex, the hydrogeological setting is dominated by unconfined and semi-confined aquifer regimes. It is difficult to obtain large or even adequate groundwater supplies because of the lithology and structure of the crystalline basement rocks. Where sufficient fractures and lineaments exist, aquifers of considerable extent may occur. In areas of extensive weathering, especially with overlying coastal sediments, a considerable amount of groundwater can be utilised for various purposes. In the latter situation, aquifers will generally occur at shallow depths. However, a combination of weathering and fracturing generally characterise the occurrence of aquifers

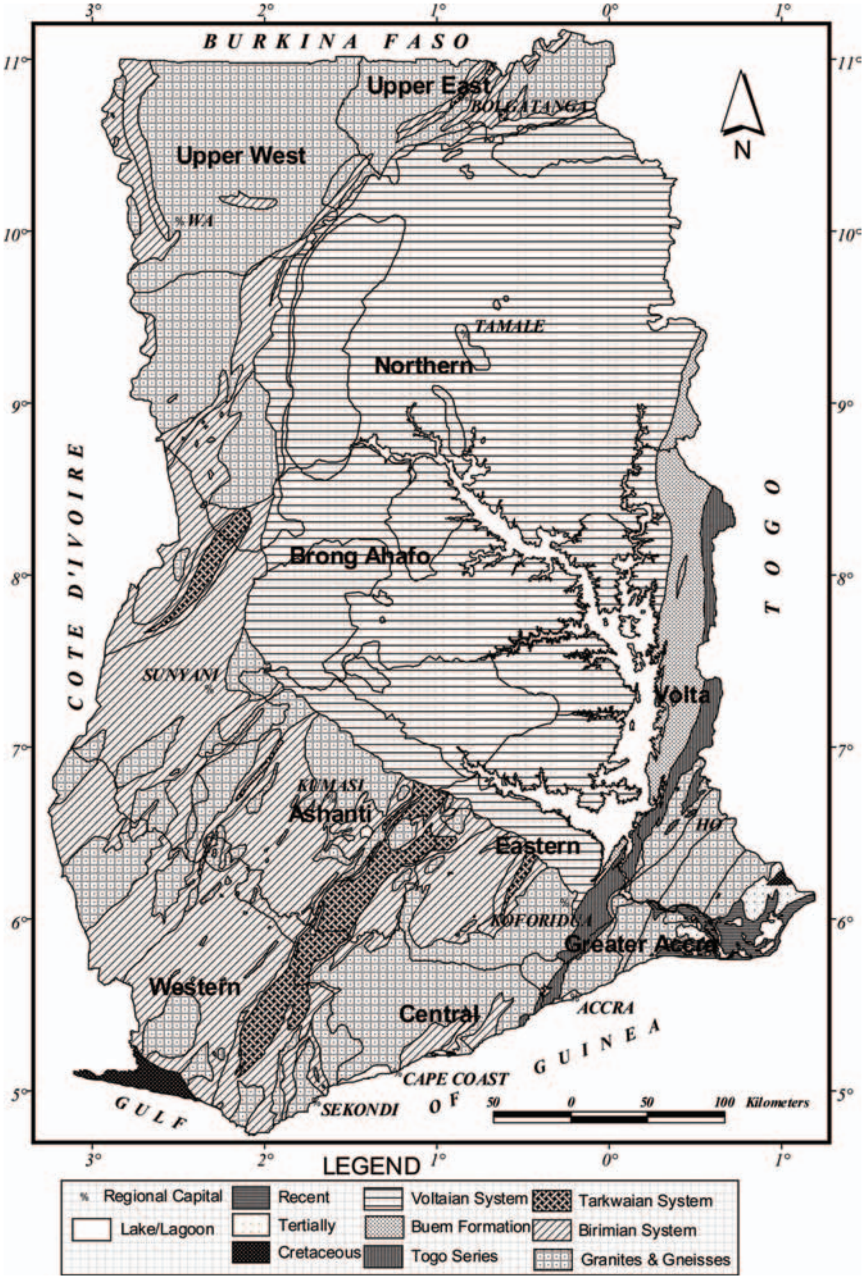

Figure 5.2. Geological provinces of Ghana (Geological survey of Ghana).

in the Basement Complex formations in the country. Most such aquifers are however low-yielding. There are however high-yielding boreholes in areas where drilling sites have been properly selected by means of suitable geophysical investigations.

4 GROUNDWATER QUALITY

In general, the quality of groundwater in Ghana is considered good for multi-purpose use (domestic, agriculture and industry), except in some cases where low pH (3.6–6.0) is found. Groundwater in the forest zones of southern Ghana has high levels of iron. In fact, it is known that about 30 per cent of all boreholes in Ghana have high iron content, making it the most prominent groundwater quality problem in Ghana. High iron concentrations ranging from 1–64 mg/l have been observed in boreholes in all the hydrogeological provinces. Potential sources of iron include corrosion of pump parts by low pH waters as well as natural groundwater-host rock interactions. High levels of manganese and fluoride occur in certain localities as well as high mineralization, with Total Dissolved Solids (TDS) in the range of 1458–2000 mg/l in some coastal aquifers, particularly in the Accra Plains. Fluoride levels range from 1.5–5.0 mg/l and are found in boreholes located in the granitic formations of the upper east, upper west and northern regions.

5 GROUNDWATER CONTAMINATION

Knowledge about the natural hydrogeological and geochemical processes, as well as the associated anthropogenic effects on a groundwater resource is required for a complete scientific understanding of groundwater vulnerability to contamination. The word ‘contaminant’ can include any natural or anthropogenic chemical or physical property of the groundwater resource in question that is not desirable from a health or other perspective, such as interference with water-treatment practices. Potential anthropogenic influences on contaminant sources, fate, and transport must be considered with the natural, inherent interconnection of the geochemical system and the groundwater flow system (Back *et al.*, 1993). A water resource can become more vulnerable to a naturally occurring contaminant if land-use practices affect the groundwater flow system in a way that enhances the solubility or mobility of the contaminant beyond the ambient geochemical conditions.

Pollution sources in Ghana include:

- poorly designed hazardous waste disposal facilities, such as landfills,
- leakages from underground storage tanks,
- mine tailings and
- the consequences from the application of fertilizers and pesticides in agricultural practices.

A literature review of the history of groundwater contamination in Ghana shows that the problems reported in the past include the following:

1. Chemical pollution due to excess iron, manganese, fluoride, nitrate and chloride concentrations
2. Biological pollution from faecal coliforms in hand-dug wells
3. Pollution due to leachate from mining waste disposal
4. Pollution due to leakage from underground petroleum storage tanks.

6 CHEMICAL POLLUTION OF GROUNDWATER SUPPLIES

6.1 *Excessive concentration of iron in groundwater*

An analysis of 750 borehole records from a 3000-borehole drilling programme in central and southern Ghana in 1984 indicated that 68.8 per cent contained iron in excess of the WHO guideline value of 0.3 mg/l (Bosque-Hamilton *et al.*, 2004). The mean reported value was 0.69 mg/l. A maximum value of 50 mg/l total iron was also reported. Amuzu (1974) presenting a countrywide survey of 375 boreholes in rural areas reported 31 per cent of supplies above the WHO guideline value. A value of 26.5 mg/l was reported in Gambaga in the northern region of Ghana. Pelig-Ba (1988) also reported iron concentrations of up to 4.5 mg/l in borehole supplies. The Canadian International Development Agency (CIDA) drilled 2600 boreholes in the upper east region of Ghana and reported that data on 22 representative boreholes, analysed from all geologic formations, showed that 36 per cent of the water supply contained iron well above 0.3 mg/l. The problem of high iron concentrations in groundwater supplies in Ghana is therefore generally widespread, particularly in the forest zones.

6.2 *High manganese concentrations in groundwater*

Records from the 3,000-borehole drilling programme also indicate from the analysis of 849 borehole records that 54.5 per cent contained manganese concentrations in excess of the WHO guideline value of 0.1 mg/l. The values ranged from 0 to 30.0 mg/l. Amuzu (1974), again reports that 17 per cent of 375 boreholes analysed had manganese concentrations above the recommended WHO guideline value.

6.3 *Contamination due to high nitrate levels*

Surveys conducted in the upper east region of Ghana in 1959, 1977 and 1980 indicated a significant increase of nitrate levels in groundwater between 1977 and 1980. Between 1959 and 1977, when there was not much agricultural activity in the area, no appreciable increase in nitrate levels was recorded (Akiti, 1982). Over 2000 boreholes were drilled for domestic water supply towards the end of 1980. At the same time, there was increased agricultural activity, with the result that tons of fertilizers were used yearly to grow corn, millet and rice, as well as an increase in animal husbandry practices. These activities were reported as accounting for the increased nitrate levels between 1977 and 1980. Studies in other parts of the country showed similar trends. Recent studies in the Accra Plains of south-eastern Ghana show that very high concentrations of nitrate are usually localised and these were traced to areas of high agricultural activity, e.g. Sam & Sam dairy farms (Malejor) and Vidas farms (near Oyarifa) where values of up to 83.0 mg/l of nitrate were recorded (Darko *et al.*, 1995).

6.4 *Salinity problems in groundwater supplies*

Along the coastal areas of Ghana, particularly the eastern coast, excessive salinity is quite widespread. In the Accra Plains an analysis of 61 boreholes by Amuzu (1978) indicated that chloride concentrations ranged from 23.5 mg/l to 7000 mg/l. In recent times studies have shown that in the Tema and Ashaiman areas of the Accra Plains, the mean value of chloride concentration in borehole supplies is as high as 1480 mg/l (Darko *et al.*, 1995). Boreholes drilled along the coastline show high levels of salinisation. These are attributed to sea spray and/or saline water intrusion with a $Na^+/(Na^+ + Cl^-)$ equivalent ratio of approximately 0.5 (Banoeng-Yakubo *et al.*, 2003).

6.5 *Fluoride contamination of borehole water supplies*

The Bongo district in the upper east district of Ghana has been associated with a high incidence of dental cavities and mottled teeth. In 1993 a Dental Health Service survey of 1558 students indicated that 62 per cent had fluorosis. This was attributed to the quality of their drinking water supplies (Dwamena-Boateng *et al.*, 1995). A monitoring programme undertaken by the regional Ghana Water and Sewerage Corporation laboratory in Tamale on 93 boreholes and shallow well water supply sources in the region indicated that about 63.5 per cent of the wells had fluoride concentrations in excess of 1.0 mg/l. The problem has been attributed to the geology of the area, i.e. Bongo granite (Smedley *et al.*, 1994), but this is not conclusive. In recent times, there have been reports of excess fluoride concentrations in the groundwater supplies in the northern region of Ghana, the geology of which is predominantly the Voltaian sandstones, mudstones, limestones and shales. This has raised many questions regarding the source of the fluoride in the water supplies.

6.6 *Biological contamination of hand-dug wells from faecal matter*

Coliform bacteria is unwanted in water since their presence indicates faecal contamination and eventually, possible contamination by pathogenic organisms. Pathogenic bacteria are transmitted to water from different waste products, broken drains, refuse dumps, etc. Most cases of groundwater contamination are caused by poor well construction. Increasing concurrently with groundwater resource development is the provision of on-site sanitation facilities. Sanitation provision is of great importance in any community. However, the choice of sanitation technology depends on economic, social and technical issues. Generally, in many rural communities the choice of on-site sanitation happens to be the choice of excreta disposal. In the past, unlined pit latrines have been sited close to wells, resulting in the migration of faecal coliforms into the wells. A case study in the Ashanti region in 2003 provided the following conclusions. On-site sanitation is a major potential source of pollution for groundwater sources. The study showed that pollution levels in groundwater supplies depend on the distance between groundwater supplies and pit latrines (Odai & Dugbantey, 2003). Biological contamination, however, is not widespread in Ghana. The rule of thumb is to site wells at a minimum distance of 50 m from sanitation facilities, cemeteries, refuse dumps, landfills, etc.

6.7 *Contamination of groundwater supplies due to mining activities*

This type of contamination is experienced in most of the mining areas in Ghana. A project funded by the UK-DFID was undertaken between 1992 and 1995 (DFID Project R5552) by a collaboration between BGS and the Water Resources Research Institute, (WRRI), now CSIR, Water Research Institute (WRI), Accra, Ghana. The project involved sampling streams, shallow dug wells and tubewells used for drinking water in a 40×40 km area around Obuasi Town in the Ashanti region (the biggest mine in Ghana owned by Anglogold-Ashanti). Samples of deep groundwater (70–100 m depth) from mine exploration boreholes, as well as mining effluent were collected.

Arsenic concentrations in water from streams, shallow wells and boreholes were found to range between <2 and 175 μg/L. The main sources are mine pollution and natural oxidation of sulphide minerals, predominantly arsenopyrite. Streamwaters have apparently been most affected by mining activities and contain some of the highest arsenic concentrations observed. They are also of poor bacteriological quality. Some of the streams have relatively high As (III) concentrations ($As(III)/As_T > 0.5$), probably as a result of methylation and reduction reactions mediated by bacteria and algae. Concentrations of As in groundwater reach up to 64 μg/L, being highest in deeper (40–70 m depth) borehole supplies. The As is thought to build up as a result of the longer residence times undergone by groundwater and the increasingly reducing conditions in the deeper parts of the aquifer. The proportion of As present as As (III) is also higher in the deeper groundwater. Deep mine exploration boreholes (70–100 m) have relatively low arsenic concentrations (5–17 μg/L), possibly as a result of arsenic adsorption onto precipitating hydrous ferric oxides or to localized low concentrations of arsenic-rich sulphide minerals. The maximum level of As, according to the WHO guideline for drinking water, is 10 μg/L.

There have been similar studies on the contamination of ground and surface water resources in other mining towns. Tarkwa and its environs in the Wassa west district of Ghana was considered an ideal location for another study, since it is a major gold-mining area, where the contamination of water resources due to toxic elements such as arsenic, lead, cadmium, mercury and spillage of cyanide has been reported. The presence of toxic metals in the area were traced to sulphide minerals such as arsenopyrite, galena and pyrite present in the underlying Birimian basement rocks of the area (Sida-SAREC, 2004–2005; Kortatsi, 2000).

6.8 *Contamination from underground storage tanks*

The problem of groundwater contamination from underground storage tanks, mainly from oil and petroleum products, is not widespread in Ghana. They affect mostly shallow wells that are generally hand dug in urban towns, where oil companies are very active. A few urban towns rely on groundwater for their water supplies and a few cases have resulted in gas stations being accused of groundwater contamination. Two recent cases in Accra and Tema have been investigated by the CSIR WRI, during which the source of contamination was traced to leakages from underground storage tanks holding petroleum products. These cases occurred in Adenta, a suburb of Accra where a private hand-dug well supply was reported to be contaminated by leakage from a storage tank. The other case, in Tema, was reported by the Tema Oil Refinery (TOR) after drilling a number of boreholes within their premises for monitoring probable hydrocarbon leakages into the local groundwater regime. In both cases the investigations proved positive and advice was given for mitigating measures to avert any harmful consequences to the groundwater system (CSIR WRI, 2004).

7 CONCLUSION

Groundwater contamination in Ghana is widespread and varied. Naturally occurring contamination is mostly the result of rock-water interactions and the concentrations of chemical constituents are often exceeding drinking water limits.

The other main source for groundwater pollution is caused by anthropogenic activities. The pollution is induced mainly due to agricultural practices and mining activities.

Neither remedial measures nor protective measures for those cases that could be avoided were assessed in this paper. Some of these problems pose a direct risk to human health for example by volatilization of toxic compounds.

Once a groundwater system becomes contaminated, it is almost impossible to clean up. Many contaminants are persistent and remain hazardous even at low concentrations.

REFERENCES

Akiti, T.T. 1982. Nitrate levels in some granitic aquifers from Ghana. Published at the International Symposium on ‘Impact of Agricultural Activities on Groundwater,’ Prague, Czechoslovakia.

Banoeng-Yakubo, B. *et al.*, 2003. Assessment of Pollution Status and Vulnerability of Water Supply Aquifers of Keta, Ghana, UNEP December 2003 Activity Report. Accra, Ghana.

Bosque-Hamilton, E.K., Duah, A.A., Darko, P.K. (Unpublished). Physico-Chemical Quality Problems Associated with Rural Water Supplies in Ghana.

CSIR WRI. 2004. Geophysical Investigations to locate source of hydrocarbon contamination in groundwater in observation well at Tema Oil Refinery (TOR). Technical report on consultancy conducted for the TOR in December, 2004 by CSIR WRI, Accra, Ghana.

Dwamena-Boateng, P., Larmie, S.A. 1995. Excess Fluoride levels in groundwater supplies in the Upper East Region of Ghana. Proceedings of the National Workshop on Water Quality, Sustainable Development and Agenda 21, GNC for IHWRP, Accra, Ghana.

Darko, P.K. *et al.*, 1995. Groundwater Assessment of the Accra Plains, CSIR WRI publication, Accra, Ghana.

Duah, A.A. and Gyau-Boakye, P. 2002. Groundwater Assessment and Exploitation in Ghana. Conference paper at the International Conference on Hydrology and Watershed Management, 18th–20th December, 2002 at Hyderabad, India.

Kortatsi, B.K. 2000. The Impact of mining on groundwater quality in the Tarkwa area, Ghana; Groundwater; past achievements and future challenges. Proceedings of the 30th IAH Annual Congress, Cape Town, South Africa, Nov. 26–Dec. 1, 2000 published in the Netherlands.

Odai, S.N. and Dugbantey, D.D. 2003. Towards Pollution Reduction in Peri-Urban Water Supply: A Case Study of Ashanti Region in Ghana. Diffuse Pollution Conference, Dublin 2003. Poster paper. Dublin, South Africa.

Asklund, R. and Eldvall, B. 2005. Contamination of Water Resources in Tarkwa Mining Area of Ghana: Linking Technical, Social-Economic and Gender Dimensions. Research Project, Dept. of Engineering Geology, Lund University, Lund 2005, Sweden. Published by KFS AB, Lund 2005, Sweden.

Smedley, P.L., Edmunds, W.M., Pelig-Ba, K.P. 1994. Mobility of Trace Elements (As, I, F) in Groundwater from Ghana and their Impact on Health. Western Africa Water & Environment Conference, May 1994, Accra, Ghana.

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Characterization of diffuse pollution of shallow groundwater in the Harare urban area, Zimbabwe

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ABSTRACT: Diffuse pollution of urban groundwater can be a threat to domestic water supplies and a challenge to water and land management. Diffuse pollution of shallow groundwater in the Harare urban area, Zimbabwe, was characterised through an investigation of seven sites within the metropolitan area: two industrial sites, a semi-formal settlement, a sewage works, two landfills and a cemetery. Boreholes were drilled, groundwater sampled and chemical and microbiological analyses carried out. It was determined that industrial sites raise problems of metals and acidity, whilst the other sites studied showed problems with nutrients and coliform bacteria. As Harare expands, it is essential that spatial control of development be governed by geotechnical and environmental considerations, and not by land availability, in order to avoid such problems recurring elsewhere. This requires integrating land-use planning and geotechnical mapping, for better protection of the subsurface and surface environment from diffuse pollution.

1 INTRODUCTION

1.1 *Diffuse pollution of urban groundwater*

The supply of clean water is one the most important of southern Africa's concerns, with demand rising at around three per cent per annum (Laisi and Chenje, 1996). Increasing populations and an improved quality of life (leading to greater personal water use) have reduced the quantity of water available per person. Groundwater could supply at least some of this demand and is widely utilized in some areas, but it is felt that data on groundwater in the region is insufficient. This is certainly true in Zimbabwe, where detailed studies of groundwater have only been undertaken in a few areas on a large scale. The sustainable management of groundwater is dependent upon, amongst other factors, the availability of good data on the quality of the groundwater together with an understanding of the geological and chemical processes that underlie the interactions between humanity, environment and water systems. Diffuse pollution of groundwater is such an interaction and represents a threat to sustainable water supplies as well as a challenge to water and land management. This is particularly true in the urban environment, where many municipal facilities and industries contribute to substantial diffuse pollution of groundwater. At the same time, the demand on urban groundwater as a water supply or strategic resource rises with the increase of populations. In fact, urban groundwater is thought to supply up to half of the world's urban population (Foster, 1999), especially those in peri-urban and semi-formal settlements with their water needs (Butcher, 2003; Dahiya, 2003). This makes the diffuse pollution of shallow urban groundwater a serious problem – and an increasingly serious problem as urbanization increases (Aldrick *et al*., 1999).

Urban sewage systems (disposal works and leaking pipes) tend to release micro-organisms, ammonia, nitrate and phosphate (Fetter, 1994; Mangore and Taigbenu, 2004). Stormwater and surface run-off is often contaminated with metals, nutrients and oils (Magombeyi *et al*., 2005; Mvungi *et al*., 2004) and can often infiltrate to shallow groundwater. Informal settlements are associated with high levels of nitrate, nitrite, phosphate and organic compounds (Gwebu, 2003; Ren *et al*., 2003; Wright, 1999). Landfills release the widest suite of contaminants: sodium, potassium, ammonia, nitrate, nitrite, chloride and heavy metals such as iron, manganese, lead, mercury and chrome (Dutova *et al*., 1999; Zhu *et al*., 1997) and industrial areas release acidity, ammonia, phosphates, nitrates, organic contaminants and heavy metals (King, 1996). Graveyards are associated with bacterial growth, phosphate and nitrate (Dent *et al*., 2004; Trick *et al*., 2002).

1.2 *The Harare urban area*

Harare, Zimbabwe's capital city, has been identified as the study area for this paper. The city has a rapidly rising population: from 665,000 in 1982 to 1.4 million in 2002 (CSO, 2003) to over 1,400,000 in 2005. This population rise has been accompanied by increased industrialisation, land development (now over 570 km^2) and the expansion of municipal facilities. Groundwater has traditionally been used to supply recreational facilities such as parks and sports clubs. It is also utilized for small scale domestic/residential and industrial purposes (Rakodi, 1995). However, private use of groundwater has also increased especially in peri-urban areas. Groundwater is also used as a strategic resource during frequent drought periods.

Furthermore, it is well established that shallow groundwater contamination leads to the contamination of local streams (Winde and van der Walt, 2004). Thus shallow groundwater contamination in the Harare urban area can lead to contamination of the headwaters of the Mukuvisi, Marimba, Gwebi and associated rivers, which drain into the dams providing the water supply of the metropolitan area (Fig. 6.1).

Groundwater in the Harare urban area occurs largely in secondary aquifers, with strong stratigraphic and lithological controls on the occurrence (Baldock *et al*., 1991). The authors report the greatest yields of groundwater in phyllites, the lowest yields in rhyolites and dacites and highly variable yields in the granites in the southern half of the city (Fig. 6.2), which is localized to areas of decomposition or heavy jointing.

Whilst diffuse pollution of the surface water in Harare has been studied, little is known about groundwater pollution in Harare. A transdisciplinary study (the Harare Urban Groundwater Project) involving earth scientists, biologists, engineers, chemists and geographers from the University of Zimbabwe, along with international colleagues was initiated. The aim is to provide information on the quality of shallow groundwater in areas of concern and to provide options for pollution control and the sustainable management of groundwater in the Harare urban area.

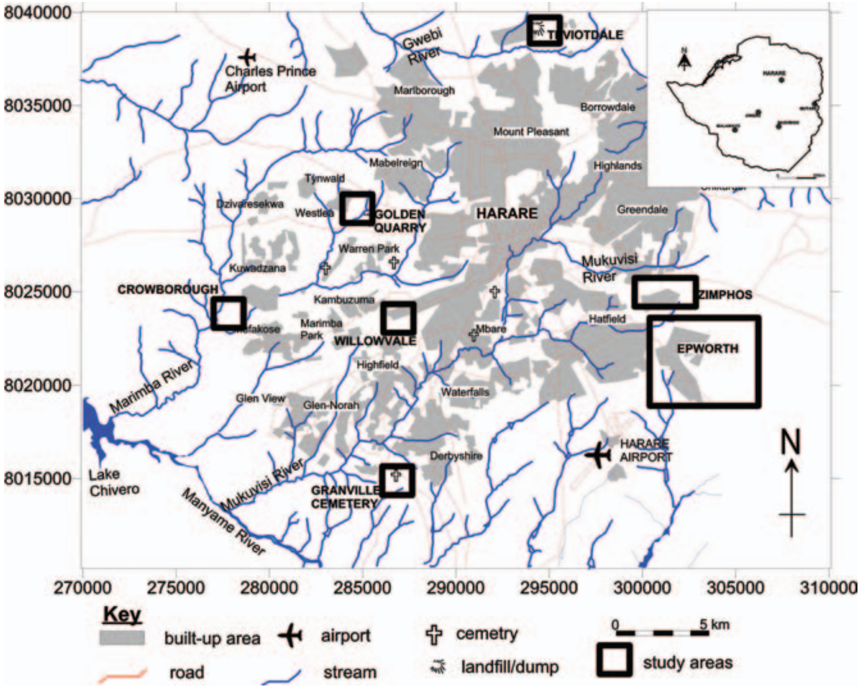

Figure 6.1. The Harare urban area, showing detailed study sites.

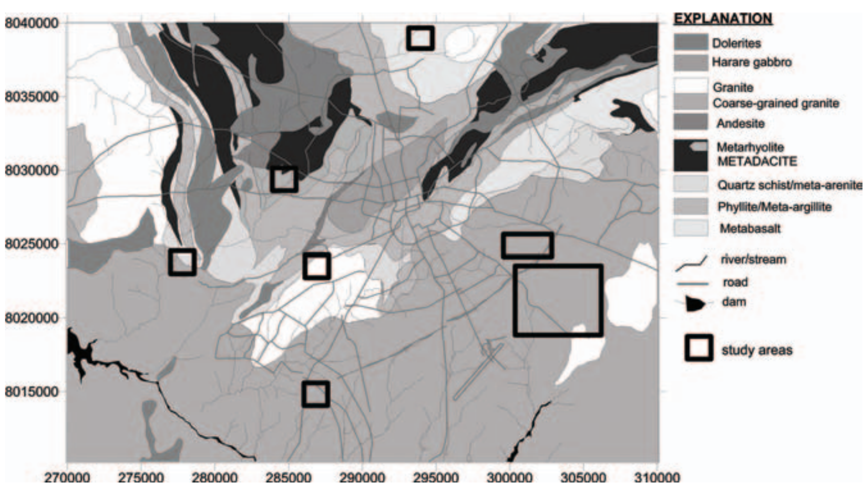

Figure 6.2. Geology of the Harare urban area, adapted from Baldock *et al*. (1991).

2 METHODOLOGY

Potential sources of diffuse pollution were identified on the basis of the literature review discussed above. Since it was not possible to study all diffuse pollution sources in the metropolitan area within the time and budget available, six sites were selected for investigation (see Fig. 6.1). At each site shallow boreholes using a vonder rig were drilled. Borehole depths varied between 5 and 10 m, depending on where the water table was reached. Groundwater sampling campaigns were conducted and chemical and microbiological analyses carried out. Laboratory analyses of water samples were performed at the water quality laboratories of the University of Zimbabwe, within 24 hours of sampling. Analyses were done according to the procedures of Standard Methods (Clesceri *et al*., 1995).

3 RESULTS

3.1 *Zimphos industrial site*

Zimbabwe Phosphate Industries (Zimphos), is part of a heavy industrial site located downstream of the Cleveland Dam on the Mukuvisi River. At Zimphos, water is used in processing sulphide and phosphate ore for the manufacturing of fertilisers and acids. The waste consists of fine material, heavy metals and other harmful chemicals used in the extraction of the desired products (Ravengai *et al*., 2004). The site has eight dumps. One dump consists of mixed tailings and aluminium sulphate; two of the dumps are gypsum dumps, three are tailings dumps and two are aluminium sulphate ponds. The most recent dumps (two tailings dumps, one gypsum dump and two aluminium sulphate ponds) have been lined and are equipped with return water facilities. The older dumps are unlined.

The comparative results of the two seasonal campaigns are shown in Figure 6.3. The levels of all chemical parameters, except the pH, dropped during the rainy season when compared to the dry

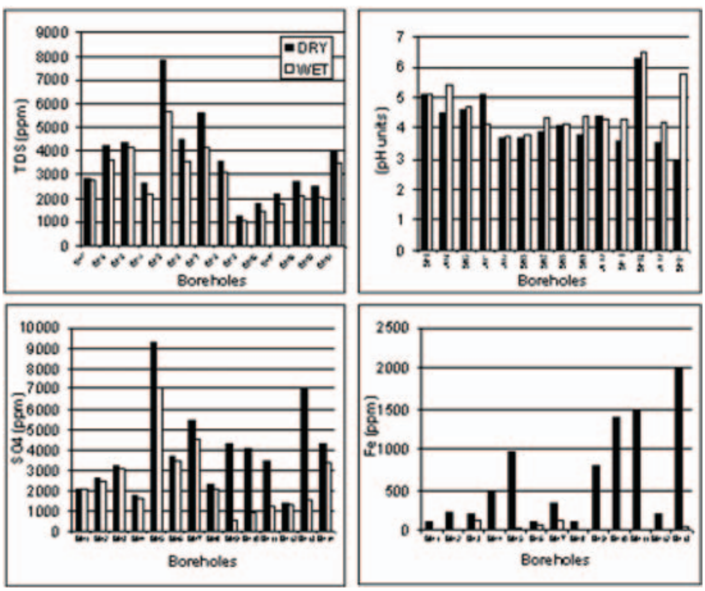

Figure 6.3. Seasonal variations in groundwater chemistry, Zimphos Industrial site, 2003–2004. Boreholes not depicted for the dry season had no water at the time of sampling.

season.The pH is higher in the rainy season, indicating lower acidity levels. This drop can be attributed partly to dilution of the pollutants as a result of recharge to groundwater. Despite the rainy season recharge to groundwater and resultant dilution, all the parameter levels render the water unsafe for use (either for domestic or agricultural purposes) in almost all boreholes throughout the year (DWAF, 1996). Since Zimphos has lined all dumps built during the last fifteen years with impermeable high density geopolymer, it is to be expected that recent dumps (monitored by BH1-2,14) are not contributing pollutants to the groundwater. The pollution is rather likely to be predominantly from the older dumps (monitored by BH3-8,11-13) or the industrial plant itself (monitored by BH9-10). Therefore, the high levels of acidity, metals and anions in groundwater suggest that the historic contamination of the aquifer will remain a problem for some time.

3.2 *Epworth semi-formal settlement*

Rapid urbanisation and the lack of low-cost accommodation in the city of Harare, Zimbabwe, have led many people to settle (formally or otherwise) on previously cultivated land in Epworth, south-east of the city (Zingoni *et al.*, 2005). The Zimbabwe cabinet's 1983 decision to upgrade the settlement rather than demolish it (the latter has been the more usual government response to informal settlements since independence), was followed by a limited upgrading infrastructure programme. Piped water and sanitation were made available to some small parts of the settlement. Half of the population of Epworth makes use of groundwater resources for their domestic water supply – as high as 77 per cent in the most recently-settled area. Well over half of the population (above 75 per cent in all sections of Epworth) makes use of pit latrines (either drop pits or 'Blair' ventilated improved pit toilets), so the health risk is high (Love *et al.*, 2005). The population were estimated at over 110,000 in 2002 (CSO, 2003).

The Epworth settlement thus showed major problems with levels of nitrogen (representing nitrates) and coliform bacteria in groundwater (Table 6.1). This is cause for concern, since the area has a high water table and high population density, leading to an elevated risk of contamination for shallow wells supplying half the population of the settlement with water. Such shallow wells are notoriously vulnerable to coliform pollution (Love *et al.*, 2005; Zingoni *et al.*, 2005). Apparently high levels of cadmium may be due to the WHO guidelines being below the detection limits of the

Table 6.1. Groundwater quality data, Epworth, 2003. Highlighted figures are above guidelines for domestic use (DWAF, 1996; WHO, 2004). *nd* = Not detected.

	Sampling point (Well)								
Parameter	W11	W12	W13	W14	W15	W16	W17	W18	W19
Conductivity μS/cm	208	44	496	290	100	199	66	36	892
pH	6.6	6.7	6.7	7.5	6.8	7.4	7.6	7.6	7.0
Cl mg/L	0.22	0.18	0.35	0.17	0.27	0.09	0.13	0.18	0.23
Total P mg/L	1	*nd*	*nd*	*nd*	1	2	*nd*	1	*nd*
Total N mg/L	20	30	20	30	20	20	20	30	20
FC 10,000 cfu	30	0.1	0.1	10	*nd*	0.2	0.3	*nd*	*nd*
TC 10,000 cfu	80	10	1000	27	10	200	0.1	10	0.1
Ca mg/L	0.371	6.172	0.048	0.145	6.914	7.182	5.856	5.401	0.950
Cd mg/L	0.022	0.094	0.078	0.088	0.060	0.060	0.043	0.038	0.050
Co mg/L	0.624	0.571	0.636	0.745	0.786	0.776	0.782	0.923	0.967
Cu mg/L	0.076	0.076	0.105	0.085	0.094	0.131	0.086	0.107	0.099
Fe mg/L	12.002	11.633	*nd*	10.158	8.527	15.187	9.133	13.899	11.014
K mg/L	6.04	4.08	14.61	5.70	4.12	6.01	3.08	3.64	16.46
Mg mg/L	0.449	0.694	5.216	2.898	1.135	3.100	0.177	0.657	1.435
Na mg/L	24.56	14.70	37.92	26.59	19.28	23.34	16.87	13.28	36.06
Zn mg/L	0.429	0.294	0.495	0.233	0.583	0.554	0.276	0.319	0.665

Table 6.2. Groundwater quality data, Crowborough Sewage Treatment Works, 2003. Highlighted figures are above guidelines for domestic use (DWAF, 1996; WHO, 2004). *nd* = Not detected.

	Sampling point (Borehole)						
Parameter	1	2	3	4	5	Average	6 (Control)
pH	7.5	7.2	6.9	7.0	6.7	7.1	11.6
TDS mg/L	1009	310	861	650	679	886	855
Nitrate as N mg/L	0.33	0.16	0.11	0.27	0.23	0.22	0.25
Total P mg/L	0.14	0.030	0.130	0.140	0.540	0.200	0.040
Ortho P mg/L	0.10	0.010	0.040	0.110	0.640	0.180	0.020
Faecal coliforms cfu/100 mL	295	188	812	960	1170	685	*nd*
Cd mg/L	0.09	0.070	0.080	0.080	0.070	0.080	0.080
Cr mg/L	3.623	5.001	4.051	4.610	3.591	4.171	3.990
Cu mg/L	0.040	0.040	0.090	0.030	0.050	0.050	0.050
Fe mg/L	4.162	4.163	11.097	2.701	3.601	5.143	1.380
Ni mg/L	1.310	1.830	1.910	1.780	1.960	1.760	1.660
Pb mg/L	0.410	0.490	0.700	0.650	1.140	0.680	0.650
Zn mg/L	0.120	0.160	0.180	0.170	0.230	0.170	0.150

instrumentation. High levels of iron are common in groundwater in Zimbabwe, and may have been released from the local rock (Love *et al.*, 2004).

3.3 *Crowborough sewage treatment works*

Sludge and effluent disposal has been taking place for over 30 years on land at Crowborough Sewage Treatment Works (CSTW), which is one of the two major wastewater treatment facilities of the City of Harare. It treats combined municipal and industrial effluents from approximately 400,000 people. Since the 1970s, CSTW has been disposing part of its treated effluent and digested sludge on a specifically designated pasture farm area. The local aquifer is not currently used as a water source, but it recharges the adjacent Marimba River, which flows to the Lake Chivero, one of the major sources of drinking water to the City (Hranova *et al.*, 2005; Moyce *et al.*, 2005).

Groundwater at the sewage works and sludge farm had higher levels of coliforms, nitrate, phosphate, chromium, and iron – although only coliform and chromium levels exceeded guideline values (Table 6.2). The source of chromium and the nutrient parameters are most likely to be the sewage effluent mixture, containing effluent from both domestic and industrial sources. Levels of lead, cadmium and nickel were also high, but not significantly higher than for the acceptable limit; suggesting that their presence may be natural.

3.4 *Golden Quarry Landfill*

Harare has two active landfills: Golden Quarry in the Warren Hills area and Teviotdale in the Pomona area. Solid waste also used to be dumped close to the Mukuvisi River, immediately south of the main railway station (Zaranyika, 1997). The Golden Quarry Landfill site was a gold mine, which was pegged in 1923 and operational from 1924 to 1952. Dumping of municipal waste (domestic and industrial) started at the abandoned mine in 1985, with temporary suspension in 1998, due to continuous outbreaks of fire. Dumping was done to reclaim the land by filling the shafts and pits. There was no lining in place for the landfill. It received 90 per cent of all the waste disposed through landfill operations in Harare and currently waste dumping has been suspended. Gum-tree planting has been undertaken at the edges of the dump (Moyce *et al.*, 2005).

Levels of coliforms, cadmium (Cd), iron (Fe), lead (Pb) and nitrate were well above water quality guidelines throughout Westlea (Table 6.3), hence the water is generally unsafe for domestic use.

Table 6.3. Groundwater quality data, around Golden Quarry Landfill, 2004. Highlighted figures are above guidelines for domestic use (DWAF, 1996; WHO, 2004). *nd* = Not detected.

	Sampling point (Borehole)								
Parameter	BH1	BH2	BH3	BH4	BH5	BH6	BH7	BH8	Control
pH	5.79	6.35	6.82	5.97	6.24	7.03	6.15	6.37	7.01
Conductivity μS/cm	1920	1400	290	350	130	520	100	110	320
Cd mg/L	0.53	0.39	0.32	0.46	0.37	0.22	0.46	0.38	0.01
Fe mg/L	11.05	1.88	3.79	1.48	2.26	2.52	4.72	1.38	1.21
Pb mg/L	0.21	0.14	0.16	0.12	0.19	0.19	0.15	0.19	0.01
Zn mg/L	0.06	0.04	0.05	0.03	0.02	0.04	0.03	0.03	0.01
Nitrate as N mg/L	16.55	10.13	10.89	3.92	1.49	10.2	0.4	4.1	0.2
SO_4 mg/L	47.73	45.3	28.43	26.53	19	21.83	13.63	32.54	31.51
Cl mg/L	7.9	3.5	2.11	2.64	2.35	4.4	3.7	2.6	13.2
CO_3 mg/L	273	129	34.05	32	71	46	31	19	62
Total Coliforms cfu/100 mL	23,000	14,000	*nd*	4000	*nd*	3000	*nd*	*nd*	*nd*
Faecal Coliforms cfu/100 mL	*nd*	6000	*nd*	1000	*nd*	*nd*	*nd*	*nd*	*nd*

Levels of metals decrease westwards, following the groundwater flow direction. Pb concentrations decrease down-gradient, with high concentrations in Boreholes BH5, BH6 and BH8. Levels of Pb, Fe and Cd were particularly problematic, with high concentrations in all boreholes above water quality guidelines, making the groundwater unsuitable for drinking throughout Westlea. They can be related to the disposal of industrial waste at the landfill. Coliform levels also decrease westwards with groundwater flow. Nitrate levels are the highest near the north side of the suburb (probably due to food and other residues in the landfill) and decreased with the groundwater flow direction.

3.5 *Teviotdale Landfill*

The Teviotdale Landfill in northern Harare receives much of the capital city's urban refuse and is located in the headwaters of the Gwayi River, a tributary of the highly polluted upper Manyame River. The landfill has been operational since 1982 and all the domestic waste from Harare as well as some of the industrial waste is disposed of here (Tevera, 1991).

The landfill showed problems with total coliform bacteria and nitrates (Table 6.4). These are likely to be the result of decomposing household refuse. Levels of metals were much lower than recorded at Golden Quarry, probably due to the substantially less disposal of industrial waste at Teviotdale than at Golden Quarry.

3.6 *Granville Cemetery*

Granville Cemetery is located on a gentle slope dipping to the south-west. Due to the gentle slopes, surface drainage is not prominent and where it does occur it forms a vlei. The vlei is slightly moist during the dry season and saturated in the wet season. Granville Cemetery opened in 1995 when the national government allocated the land to the City Council of Harare. The cemetery is divided into two sections; 'A' and 'B' which are based on the type of burial and the value of the grave. Burials at section B average at 10,000 persons per year, buried at between 1.4 m for children and 1.8 m for adults (Moyce *et al.*, 2005).

The results showed the presence of microbiological indicators (total coliform and faecal coliform) in the groundwater (Table 6.5). Faecal coliform levels are supposed to be zero, according to the South African domestic water use standards (DWAF, 1996) and World Health Organisation drinking water guidelines (WHO, 2004). Coliforms were concentrated around the graves, with the highest numbers of total coliforms in boreholes close to more recent graves (GBH7) and faecal coliforms

Table 6.4. Groundwater quality data around Teviotdale Landfill, 2005. Highlighted figures are above guidelines for domestic use (DWAF, 1996; WHO, 2004). *nd* = Not detected.

	Sampling point (Borehole)						
Parameter	BH1	BH2	BH3	BH4	BH5	BH6	Control
Cd mg/L	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*
Ca mg/L	362	226.7	72.9	233.9	9.8	2.69	*nd*
Cr mg/L	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*
Cu mg/L	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*
Fe mg/L	1.39	3.36	21.2	0.85	*nd*	*nd*	17
Mg mg/L	149.9	86.1	536.6	36.3	103.3	77.4	88.9
K mg/L	398	534	410	6.56	nd	0.68	535.5
Na mg/L	1028	1091	1095	1150	1185	1205	*nd*
Zn mg/L	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*
Conductivity μS/cm	3630	9120	4120	1830	3780	460	350
pH	8.06	8.7	7.9	8.5	8.5	9	7.96
Nitrate as N mg/L	50.1	39.8	19.6	10.4	75.7	6.42	2.78
CO_3 mg/L	328	32	26	136	430	76	56
Total Coliforms cfu/100 mL	1100	*nd*	10,000	200	*nd*	*nd*	400
Faecal coliforms cfu/100 mL	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*

Table 6.5. Groundwater quality data around Granville Cemetery, 2003. Highlighted figures are above guidelines for domestic use (DWAF, 1996; WHO, 2004). *nd* = Not detected.

	Sampling point (Borehole)							
Parameter	GBH5	GBH7	GBH1	GBH2	GBH3	GBH8	GBH9	Control
Conductivity μS/cm	49.1	49.3	67.3	114.4	332.0	204.5	56.7	40.0
Temp °C	27.2	27.5	23.9	24.6	23.3	23.5	23.4	22.8
pH	5.4	5.4	5.7	5.5	6.0	6.7	5.7	7.0
SO_4 mg/l	1.7	118.6	38.7	*nd*	*nd*	*nd*	*nd*	*nd*
Cl mg/l	0.09	0.09	0.13	0.09	0.22	0.18	0.22	0.18
CO_3 mg/l	0.004	0.010	*nd*	*nd*	*nd*	*nd*	*nd*	*nd*
PO_4 mg/l	0.800	1.165	0.416	0.602	0.404	0.017	0.080	0.013
Nitrate as N mg/l	0.106	0.093	0.040	0.180	*nd*	0.066	*nd*	*nd*
Faecal coliforms/100 mL	9000	*nd*	*nd*	350,000	*nd*	*nd*	*nd*	*nd*
Total coliforms/100 mL	20,000	7,000,000	*nd*	1,700,000	430,000	320,000	*nd*	*nd*
Ca mg/l	5.097	9.350	4.935	5.196	179.940	0.574	152.240	7.240
Cd mg/l	0.262	0.009	0.029	0.015	0.025	0.483	0.049	0.058
Co mg/l	0.780	0.278	0.245	0.285	0.372	0.356	0.385	0.530
Cu mg/l	0.343	0.057	0.062	0.050	0.058	0.085	0.073	0.108
Fe mg/l	12.080	0.812	13.609	13.749	7.499	5.662	13.550	15.717
K mg/l	4.170	10.167	3.399	9.834	8.102	6.570	1.433	3.123
Mg mg/l	0.572	1.935	0.539	0.689	2.916	2.871	13.190	16.852
Na mg/l	16.484	24.771	21.256	24.287	29.404	18.872	17.662	21.927
Pb mg/l	1.57	1.30	1.40	1.35	1.55	1.61	1.69	1.78
Zn mg/l	0.426	0.215	0.270	0.350	0.343	1.630	0.391	0.728

highest around the older (GHB2) and pauper graves (GBH5). The results from more distant boreholes (GH3 and GBH8) indicate that coliforms have not been carried far in groundwater.

Levels of calcium, iron, zinc and, to some extent, cadmium, are high, but since this is also true of the control, it appears to be a regional problem. Nevertheless, some of these metals could be a product of corpse decomposition. Nitrates were detected at higher levels in samples from boreholes around graves (GHB2, GHB5 and GBH7). Phosphate concentrations were above the control borehole levels and were concentrated in boreholes around the graves, with the highest levels in younger graves (GBH7).

4 DISCUSSION

It is evident that industrial sites raise problems of elevated levels of metals and acidity, whilst the other study sites showed problems with nutrients and coliform bacteria. It is therefore suggested that on-site mitigatory measures and city regulation should focus on ways of decreasing the load to groundwater of these key parameters, perhaps through methods that stabilise the relevant ions in the zone of disposal.

The following site-specific measures could be considered:

1. At Zimphos, the main source of pollution is older, unlined dumps that are still polluting the groundwater. The best way to mitigate this would be to excavate the dumped material and underlying topsoil and redeposit it on new dumps. Another mitigatory measure would be to install an active cover, such as lime, that would allow infiltrating rainwater to react with dump material and stabilise some of the contaminants, such as metals. The Golden Quarry and Teviotdale Landfill would also benefit from a reactive cover.
2. The Epworth semi-formal settlement faces a problem that cannot be properly addressed until networked water and sanitation is installed throughout the area. In the meantime, alternative sources of drinking water are required, perhaps through expanding the system of communal taps.
3. The neutralisation capacity of the soil at Crowborough Sewage Treatment Works has been exhausted, so the only way to prevent further deterioration of the groundwater would be to develop an alternative site for on-land sewage effluent disposal and to carefully monitor the soil and groundwater chemistry at the new site. It is also possible to cut, bale and remove the pasture grass fed by the sewage effluent, and sell it as feed to nearby cattle pens and abattoirs, although this would remove fewer of the effluent contaminants.
4. The pollution load from Granville Cemetery is quite localised and this could be better contained by planting a tree barrier, downgradient of the cemetery, in order to try and reverse the direction of groundwater flow at the edge of the site.

As the Harare metropolitan area expands, it is essential that the spatial control of development be governed by geotechnical and environmental considerations, not land availability, in order to avoid similar problems recurring elsewhere. This requires integrating land-use planning and geotechnical mapping. The necessary data on geology, soil types, hydrology and groundwater levels can be combined in a GIS, land use and groundwater use zones identified and sites for municipal facilities recommended. This can provide for better protection of the subsurface and surface environment from diffuse pollution.

The problems identified at Zimphos and Golden Quarry relate partly to the fact that they are old operations that have been developed before many modern ideas in environmental management were established. Similar problems may exist at other facilities and corporations within the city, established in the first half of the last century. The only way to detect such contamination problems would be on-site investigations of factories in the older industrial areas. This would require cooperation between the Zimbabwe National Water Authority (or the new Environmental Management Agency), the city government and factory operators. This effort should be expedited, perhaps through a budgetary allocation from central government.

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REFERENCES

Aldrick, R.J., Rivett, M.O. & Hepburn, S.L. 1999. Urban groundwater and environmental management: Hull. In: Chilton, W. (ed.) *Groundwater in the Urban Environment: Selected City Profiles*. Rotterdam: Balkema.

Baldock, J.W., Styles, M.T., Kalbskopf, S. & Muchemwa, E. 1991. *The geology of the Harare Greenstone Belt and surrounding granitic terrain*. Zimbabwe Geological Survey Bulletin 94, Harare: Zimbabwe Geological Survey.

Butcher, C. 1993. Urban low-income housing: a case study of the Epworth squatter settlement upgrading programme. In: Zinyama, L.M., Tevera, D.S. & Cumming, S.D. (eds) *Harare: The Growth and Problems of the City*. Harare: University of Zimbabwe Publications.

Clesceri, L.S., Greenberg, A.E. & Eaton, A.D. (eds). 1989. *Standard methods for the Examination of Water and Waste Water.* 20th Edn. Washington D.C.: American Public Health Association.

CSO. 2003. *Demographic and Health Survey*. Harare: Central Statistics Office.

Dahiya, B. 2003. Peri-urban environments and community driven development: Chennai, India. *Cities*, B, 341–352.

Dent, B.B., Forbes, S.L. & Stuart, B.H. 2004. Review of Human decomposition processes in soil. *Environmental Geology*, 45, 576–585.

Dutova, E.M., Nalivaiko, N.G., Kuzevanov, K.I. & Koylova, J.G. 1999. The chemical and microbiological composition of urban groundwater, Tomsk, Russia. In: Chilton, W. (ed.) *Groundwater in the Urban Environment: Selected City Profiles*. Rotterdam: Balkema.

DWAF 1996. *South African Water Quality Guidelines. Volume 8: Field Guide*. Pretoria: Department of Water Affairs and Forestry.

Fetter, C.W. 1994. *Applied Hydrogeology*. Englewood Cliffs: Prentice-Hall.

Foster, S.S.D. 1999. The interdependence of groundwater and urbanisation in rapidly developing countries. *Urban Water*, 3, 185–192.

Gwebu, T.D. 2003. Environmental problems among low income urban residents: an empirical analysis of old Naledi-Gaborone, Botswana. *Habitat International*, 27, 407–427.

Hranova, R., Love, D. & Amos, A. 2005. Irrigation with treated effluent and sludge at Crowborough farm, Harare – impacts on ground water. In: Hranova, R. (ed.) *Diffuse pollution of water resources: principles and case studies from southern Africa*. Leiden: Taylor and Francis.

King, A.S. 1996. Water pollution and management. In: Chenje, M. & Johnson, P. (eds) *Water in Southern Africa Harare*. Harare: Southern African Development Community, International Union for the Conservation of Nature and Southern African Research and Documentation Centre.

Laisi, E. & Chenje, M. 1996. Management of water resources. In: Chenje, M. & Johnson, P. (eds) *Water in Southern Africa Harare*. Harare: Southern African Development Community, International Union for the Conservation of Nature and Southern African Research and Documentation Centre.

Love, D., Hallbauer, D.K., Amos, A. & Hranova, R.K. 2004. Factor analysis as a tool in groundwater quality management: two southern African case studies. *Physics and Chemistry of the Earth*, 29, 1135–1143.

Love, D., Zingoni, E., Gandidzanwa, P., Magadza, C. & Musiwa, K. Impacts on groundwater quality and water supply of the Epworth semi-formal settlement. In: Hranova, R. (ed.) *Diffuse pollution of water resources: principles and case studies from southern Africa*. Leiden: Taylor and Francis.

Magombeyi, M., Love, D., Hranova, R.K. & Hoko, Z. 2005. Stormwater quality in the City of Harare – implications for managing stormwater pollutants in the urban water cycle. In: Mathew, K. & Nhapi, I. (eds)

Water and Wastewater Management for Development Countries. IWA Water and Environmental Management Series. London: IWA Publishing.

Mangore, E. & Taigbenu, A.E. 2004. Land-use impacts on the quality of groundwater in Bulawayo. *Water SA*, 30, 453–464.

Moyce, W., Love, D., Nyama, Z., Musiwa, K., Mangeya, P., Ravengai, S., Wuta, M & Zingoni, E. 2005. Impacts of a solid waste disposal site and a cemetery on groundwater quality in Harare. In: Hranova, R. (ed.) *Diffuse pollution of water resources: principles and case studies from southern Africa*. Leiden: Taylor and Francis.

Mvungi, A., Hranova, R.K. & Love, D. 2003. Impact of home industries on water quality in a tributary of the Marimba River, Harare: implications for urban water management. Physics and Chemistry of the Earth, 28, 1131–1137.

Rakodi, C. 1995. *Harare: Inheriting a Settler-Colonial City: Change or Continuity?* Chichester: Wiley.

Ravengai, S., Love, D. & Owen, R.J.S. 2004. Seasonal variations of groundwater pollution from a heavy industrial site: Zimphos, Zimbabwe. In: *Abstract Volume, 5th WaterNet-WARFSA Symposium*, Windhoek, Namibia, 50–51.

Ren, W., Zhong, Y., Meligrana, J., Anderson, B., Watt, W.E., Chen, J. & Leung, H.-L. 2003. Urbanization, land use, and water quality in Shanghai, 1947–1996. *Environment International*, 29, 649–659.

Tevera, D.S. 1991. Solid Waste Disposal in Harare and its effects on the environment: some preliminary observations. *Zimbabwe Science News*, 25, 9–13.

Trick, J.K., Klinck, B.A., Coombs, P., Chambers, J., Noy, D.J., West, J. & Williams, G.M. 2002. *Groundwater impact of Danescourt Cemetery*. National Groundwater and Contaminated Land Centre Report NC/99/72.

Winde, F. & van der Walt, I.J. 2004. Gold tailings as a source of waterborne uranium contamination of streams – The Koekemoer-spruit Klerksdorp goldfield, South Africa as a case study. Part II of III: Dynamics of groundwater-stream interactions. *Water SA*, 30, 227–232.

Wright, A. 1999. *Groundwater contamination as a result of developing urban settlements*. Water Research Commission Report 514/1/99, Pretoria: Water Research Commission.

WHO 2004. *Guidelines for Drinking Water Quality, Volume 1: Recommendations*. 3rd Edition. Geneva: World Health Organisation.

Zhu, X.-Y., Xu, S.-H., Zhu, J.-J., Zhou, N.-Q. & Wu, C.-Y. 1997. Study on the contamination of fracture-karst water in Boschan District, China. *Ground Water*, 35, 538–545.

Zaranyika, M. 1997. Sources and levels of pollution along Mukuvisi River: a review. In: Moyo (ed.) *Lake Chivero: A Polluted Lake.* Harare: University of Zimbabwe Publications.

Zingoni, E., Love, D., Magadza, C., Moyce, W. & Musiwa, K. Effects of a semi-formal urban settlement on groundwater quality. Epworth (Zimbabwe): Case study and groundwater quality zoning. Physics and Chemistry of the Earth, 30, 680–688.

7

Groundwater contamination from pesticides in Africa – A review

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ABSTRACT: An increasing need for food security together with the exploitation of groundwater resources are happening in Africa. Increased food production inevitably leads to an increase in the application of pesticides. This paper reviews the main sources and processes leading to groundwater contamination from pesticides, the status of groundwater contamination from pesticides in Africa and in particular South Africa. It then also discusses possible mitigation measures for groundwater protection from pesticide contamination. Pesticides in African groundwaters mainly originate from high-income crop plantations (like sugarcane, subtropical fruit orchards, vegetables) and insect disease vectors control in houses. A thorough understanding of the mechanisms of groundwater contamination from pesticides is based on the knowledge of vadose zone and hydrogeological properties, as well as the properties of specific pesticides. Although recent studies indicated that groundwater could be exposed to more risks than surface waters, measurements of pesticides in African surface waters are much more common than in groundwater. A lack of monitoring data as well as education and training in pesticide management is a major concern in Africa. It is not only pesticides currently in use that could have an impact on groundwater but also persistent pesticides that have been banned as well as obsolete stockpiled pesticides. A wide range of mitigation measures can be applied to limit groundwater contamination from pesticides namely improved on-farm management, integrated pest management, biological control, bioremediation and phytoremediation, vegetative buffer strips as well as water purification. The implementation of these procedures depends on ecological, economic and social aspects, as well as possible adverse side-effects in specific countries.

1 INTRODUCTION

An estimated 14 African countries currently suffer from water scarcity and a further 11 will face this prospect in the next 25 years (UNEP, 2002). Despite the complex hydrological terrain of a large part of Africa (weathered basement complex and consolidated sedimentary rocks, particularly in sub-Saharan Africa) the utilisation of groundwater is on the increase, as this is the only feasible option for water supply, especially in rural areas (UNEP, 2005). Groundwater resources are under increasing pressure from new developments together with point- and non-point sources (NPS) of pollution. Some of the major NPS pollutants and nine out of the 12 Persistent Organic Pollutants (POPs), targeted by the Stockholm Convention, are pesticides. The use of pesticides is considered essential in order to overcome pest-related problems and ensure food security in Africa (Gressel *et al*., 2004). However, a survey of research projects done in developing countries indicate that sustainable food production is possible without the widespread use of pesticides (Pretty *et al*., 2003).

Only limited research, throughout the world, has been done on groundwater contamination from pesticides. This is mainly due to the high cost of analysis and the wide range of pesticide products on the market. In developing countries, the main sources of groundwater contamination from pesticides are high-income crop plantations (sugarcane, subtropical fruit orchards, vegetables) and insect disease vectors control in houses (UNEP, 2005). Stocked, outdated pesticides also represent a hazard. The FAO (Food and Agricultural Organization) of the United Nations estimated that more than 50,000 t of obsolete pesticides are stockpiled in Africa, with estimated clean-up costs exceeding $100 million (FAO, 2004). The main reasons for stockpiling were identified as inappropriate procurement, untimely distribution, inadequate storage and stock management, donation in excess of needs, lack of coordination between donor agencies and product bans.

The aim of this paper is (i) to review the sources and processes leading to groundwater contamination from pesticides, (ii) the status of groundwater contamination from pesticides in Africa and particularly in South Africa, and (iii) possible mitigation measures for groundwater protection from pesticide contamination.

2 CATCHMENT PROCESSES LEADING TO GROUNDWATER CONTAMINATION FROM PESTICIDES

A thorough understanding of the main catchment processes implicated in groundwater contamination from pesticides is required in order to assess the extent of the hazard and problem, as well as to design monitoring and mitigation strategies. In order to evaluate potential leaching and groundwater contamination, it is necessary to know the type of chemical, possible pathways and expected concentrations in groundwater. The time and extent of contamination depends on the amount of pesticides applied to the soil surface, frequency, weather conditions, pesticide properties, as well as hydrogeological conditions (UNEP, 2005).

Sigma Beta (2004) identified the following primary processes concerning the fate and behaviour of pesticides in the environment:

- Water and pesticide run-off (stormflow).
- Vertical water and pesticide fluxes, including drainage and leaching.
- Preferential flow (macropore or by-pass flow and funneling).
- Throughflow (subsurface lateral water and pesticide fluxes).
- Plant uptake of pesticides.
- Processes linked to specific pesticide properties. The main properties were identified to be volatilisation, sorption and degradation (decay).

In this study, particular attention is given to processes that are directly linked to groundwater contamination, namely vertical fluxes (drainage and leaching), preferential flow, throughflow and associated pesticide properties.

Together with run-off, leaching is the most direct process involved in the contamination of groundwater resources. Leaching is the amount of pesticide dissolved in the solution, percolating through the bottom of the active root zone. It is therefore directly linked to vertical water and solute fluxes. Vertical pesticide fluxes in the soil profile include movement by convection, mechanical dispersion due to variations in velocity through pores of different size, and diffusion governed by Fick's law, based on differences in concentration gradients over limited distances (Bresler *et al.*, 1982). The main process that causes the downward movement of contaminants in the unsaturated zone is infiltration, which is driven by large suction gradients between the wetting front and the dry media. The downward flushing of solutes generally occurs at the edge of the wetting front. In this way, a center of mass of solutes is generated and transported downwards during infiltration events.

Vertical fluxes of pesticides can cause contamination of groundwater, particularly in areas with negligible slope, high rainfall and soils having high infiltration rates. The depth of the groundwater table is also very relevant, as indicated by Johnson *et al.* (2001). Vertical fluxes depend on rainfall/irrigation conditions (Schierholz *et al.*, 2000), irrigation method (Perillo *et al.*, 1999; Blackwell, 2000; Reichenberger *et al.*, 2002) and tillage (Malone *et al.*, 2003). Generally, the sooner rainfall/irrigation occurs after application of pesticides, the better the chances of groundwater contamination, as travel time through the vadose zone is then reduced.

The rate of pesticide transfer and transformation in sub-soils is often unknown and difficult to establish (Vanclooster *et al*., 2000). In some cases, it is justified to assume that the properties of the sub-soil are similar to the properties of the overlying soil. In other cases, however, different properties between soil and weathered bedrock were measured (Hubbert *et al*., 2001). The vadose zone generally differs from the overlying soil root zone, because of a lack of organic matter, different temperatures, microbial activity, water content, texture and structure. It is, however, clear that this part of the system may represent an important defensive barrier to groundwater contamination. For example, Rae *et al*. (1998) demonstrated that sand in sandy aquifer sediments has a powerful sorption capacity for pesticides.

Spatial variability plays a major role in vertical fluxes. Lennartz (1999) and Netto *et al*. (1999) investigated the variability in the transport characteristics of pesticides, coming to the conclusion that this was affected by spatial variability, in particular through preferential flow. Fetter (1993) classified preferential flow as short-circuiting, fingering and funneling. Short-circuiting occurs due to the movement of infiltrating water along preferential paths (e.g. plant roots, soil cracks etc.). Fingering occurs due to pore-scale variations in permeability, especially at boundaries where finer sediment overlies coarser sediment. Funneling occurs whenever water is funneled on sloping, impermeable layers, and concentrated at the end of these layers where it percolates vertically. Funneling is therefore typical for layered soil or sediment profiles.

Renaud *et al*. (2004) considered five processes to explain different patterns of pesticide leached loads observed in soils and placed preferential flow at the top of the list. Larsson *et al*. (1999) found that leaching to tile drains with shallow groundwater increased by about 80% because of preferential finger flow (from 1.2% to 2.2% of the applied amount) in a sandy soil. The effect of soil properties and preferential flow on pesticide leaching was observed both in breakthrough curve experiments carried out by Kamra *et al*. (2001) and in field experiments (Novak *et al*., 2001). Haria *et al*. (2003) found contamination of shallow groundwater by pesticides due to high rates of both matrix and preferential flow, whilst where the groundwater was deeper, 'intermediate' storage sites remained empty and unsaturated water potential profiles showed that rainfall pulses were attenuated, as these sites absorbed the downward water fluxes. Ray *et al*. (2004) reported that the volume fraction of the macropores affects the amount of pesticides leaving the root zone. They developed a dual-permeability model (S_1D_DUAL) and ran it for a test case to show that water flux in the preferential flow domain was three times more than in the matrix for selected storm events. Bergstrom (1995) indicated that water flow in cracks may move some of the pesticides rapidly through the topsoil to the sub-soil. Once the compound reaches the sub-soil, degradation rates are reduced and the pesticide residues are stored for later leaching.

Zehe and Flühler (2001) found deep penetrating earthworm burrows to be a cause of preferential flow and that pesticides also move attached to mobile soil particles in macropores and soil cracks. They indicated that the main pre-condition for the occurrence of preferential flow events is the presence of sufficient deep penetrating macropores interconnected to the soil surface. In an experiment carried out on large Alfisol monoliths during long-duration simulated rain events, Lægdsmand *et al*. (1999) observed that the first flush of water mobilised loosely bound colloids that had a high organic content relative to the bulk soil. After the initial release, the high ionic strength in the percolating water limited the mobilisation. For prolonged leaching, the diffusion of colloids from the macropore walls appeared to limit the rate of the mobilisation process. During the late leaching phase, the rate of colloid mobilisation was positively correlated with flow velocity.

Christiansen *et al*. (2004) reported that the spatial variations of macropore flow caused by the variation in topography and depth to groundwater table are so large within a catchment that this has to be accounted for by upscaling process descriptions from point scale to catchment scale. In general, studies on preferential flow are very complex to carry out under field conditions, so modelling is used as a tool for predictions and risk assessment (Usher *et al*., 2004).

Contamination of waters can also occur via interflow (throughflow), particularly in layered soils where movement of water and pesticides may occur on sloping impermeable sediment layers (Pan *et al*., 2004). Nimmo *et al*. (2002) showed that low-permeability sediment layers in the unsaturated zone divert some flow horizontally, but do not prevent rapid transport to the aquifer, and that transport rates under these conditions may exceed 14 m/d. Sovik *et al*. (2002) reported that knowledge about sedimentary structures in the unsaturated zone is important for monitoring of contaminant transport and for remediation purposes.

Besides vertical water and solute fluxes, preferential flow and throughflow, other processes may play an important role in determining pesticide leaching. For example, Leib and Jarrett (2003) indicated that plant uptake needs to be quantified not only to improve crop production and the efficacy of pesticides, but also to gain insight into the fate and transport of pesticides. Also, pesticide properties are relevant to pesticide attenuation. In the unsaturated zone, sorption, volatilisation and degradation are the main attenuation factors for soluble chemicals. In the saturated zone, an additional attenuation mechanism is dilution. Volatilisation is the process where solutes move from the medium where they are dissolved (soil, plant and water surfaces) into the atmosphere. Adsorption is the adhesion of pesticide molecules to soil solid particles, which retards pesticide migration. Pesticide sorption occurs mostly on organic matter rather than soil mineral particles. The factors controlling the degradation rate of pesticides are not always well understood (Boesten, 1999). The ideal environmental conditions will induce enough activity for the pesticide to have the desired effect on pests, but enough short persistence to limit leaching and contamination of groundwater. Degradation (microbial decomposition, photodecomposition and hydrolisis) is generally expressed through half-life. Half-life is the amount of time (in days) it takes for one-half of the original amount of the chemical to be deactivated. The half-life of pesticides mainly depends on microbial activity, pH, temperature and moisture.

3 GROUNDWATER POLLUTION FROM PESTICIDES IN AFRICA

Gressel *et al*. (2004) indicated a number of pest-related regional problems with regard to ensuring food safety in Africa:

- parasitic weeds throughout Africa;
- grass weeds of wheat in North Africa;
- stem borers and post-harvest grain weevils in sub-Saharan Africa;
- white fly as the vector of the tomato leaf curl virus in vegetable crops in North Africa; and
- fumonisins and aflatoxins in stored grains.

Wilson and Otsuki (2004) indicated that restrictions on the use of pesticides have major implication for food security and exports of agricultural commodities from developing countries and that a lack of international standards and regulations contributed to high production costs. On the other hand, contamination of food and waters from pesticides may occur with serious consequences to people's health. For example, Adeyeye and Osibanjo (1999) found pesticide residues in raw fruits, vegetables and tubers from markets in Nigeria even though the concentration levels were below the maximum limits established by the FAO. Darkoh (2003) indicated that the use of pesticides also contributes to the degradation of the soil and vegetation, as well as the decline in biodiversity. The loss of biodiversity due to the introduction of intensive agricultural technologies in Ethiopia was discussed by Tsegaye (1997).

Analyses for pesticides in African surface waters are much more common than for groundwater. For example, Bennasser *et al*. (1997) found high organic contamination in the Sebou River basin in Morocco. Kishimba *et al*. (2004) assessed pesticide residues in large areas of Tanzania. They found relatively high concentrations of pesticide residues in waters, sediments and soils, and particularly organochlorines in former storage areas. The Vikuge farm in the coast region, a former storage area, was earmarked for a proposed project on POP remediation of impacted soils. Manirakiza *et al*. (2002) found low concentrations of organochlorines in fish from Lake Tanganyika in Burundi. Similarly, El Nemr and Abd-Allah (2004) detected organochlorines in fish, although the concentrations were relatively low, whilst Khaled *et al*. (2004) detected organichlorines in mussels from the Red Sea. Shereif and Mancy (1995) found that fish grown in treated sewage effluents had lower levels of toxic chemicals than fish harvested from polluted surface waters.

Interactions in pesticide residues between Lake Qarun and the Wadi-El-Rayan wetlands in Egypt were discussed by Mansour and Sidky (2003).

Several studies indicated that groundwater could be exposed to more risks than surface water. For example, El-Kabbany *et al*. (2000) found much higher concentrations of pesticide residues in drainage water than in canal supply water in El-Haram (Egypt). Alemaw *et al*. (2004) carried out a preliminary assessment of groundwater vulnerability to pollution in the Kanye wellfield in Botswana, where agricultural pesticides are some of the potential pollutants. They concluded that 58% of the wellfield area is highly vulnerable to pollution. According to a State of the Environment Report, approximately 200 metric tonnes of obsolete pesticides, including DDT, dieldrin and other chlorinated pesticides, were stocked in different parts of Zambia, with a very high risk of polluting groundwater (Chisupa, 2005).

A lack of monitoring data is one of the major setbacks in pesticide management in Africa. For example, Barakat (2004) indicated that there is lack of data on persistent toxic substances in Egypt. Most of the data available originated from limited studies or hot-spot situations, and the implementation of monitoring programmes was therefore recommended. Mansour (2004) evaluated the use, factors and impacts of pesticides in Egypt, indicating that persistent chemicals like DDT and organochlorines banned 25 years ago, are still detected in waters, fish sediments and foodstuffs.

A lack of education and training in pesticide handling was also mentioned as a major concern. Matthews *et al*. (2003) stressed the need for training and dissemination of information in Africa, based on a survey of pesticide application in Cameroon. The need for farmer education was also stressed in studies involving Zimbabwean cotton and vegetable farmers (Sibanda *et al*., 2000; Maumbe and Swinton, 2003), and Ugandan agroforestry farmers (Nyeko *et al*., 2002). Amr (1999) stressed the importance of monitoring, training and control of pesticides in Egypt. Clarke (2004) indicated that one of the reasons for accidental human poisoning is the mishandling and unsafe storage of pesticides. As a result of these findings, a poison control centre was established in Ghana. Amongst other roles, the poison centre trains agricultural personnel in prevention and first-aid management of pesticide poisoning.

4 GROUNDWATER POLLUTION FROM PESTICIDES IN SOUTH AFRICA

London and Myers (1995) investigated the chemical usage patterns in specific farming sectors in the southern region of South Africa. They identified a widespread and diverse use of agricultural chemicals and reported that different sectors require different pesticide groups. Deciduous farming is the principal consuming agent of insecticides, herbicides, fungicides and growth regulators. Vineyards, wheat and vegetable production involve the use of primarily fungicides, herbicides and nematicides, respectively. Organophosphates are the main insecticides used, particularly azinphos-methyl, chlorpyrifos and dimethoate. Fungicides include phthalic acid derivatives, copper salts, sulphur and in particular dithiocarbamates. Dominant herbicides include dipyridyls (particularly paraquat), chlorphenoxyacetic acids (particularly 2,4D) and triazines. Widely used nematicides are ethylene dibromide and aldicarb.

Sigma Beta (2004) carried out a comprehensive literature review of the most commonly detected and/or studied pesticides in South Africa. It was found that the seven pesticides most frequently reported in South African waters are chlorpyrifos (organophosphate), endosulfan (organochlorine), azinphos-methyl (organophosphate), atrazine (triazine), simazine (triazine), deltamethrin (pyrethroid) and penconazole (azole), based mainly on results of target analyses. In order to get more information on priority pesticides in South Africa, sales figures were used by assuming that the largest portion of pesticides purchased is eventually used and could potentially represent a threat to groundwaters. A tendency of high sale and usage was found in the period from 1994 to 2000 for organophosphates (e.g. azinphos-methyl, chlorpyrifos and prothiofos), organochlorines (e.g. endosulfan), carbamates, hydrocarbons, triazines (e.g. atrazine), amilime/acetamilide and phenolics (Jan Kleynhans, AVCASA, personal communication). Similar trends in pesticide usage can be expected in the future. The groundwater impact based on the toxicity of the particular pesticide species requires evaluation.

As reflected by the sales figures, triazine herbicides are the most widely used class of pesticides. This is probably since the two most common examples, namely atrazine and simazine, are used extensively for maize, orchards and vineyards, which represent a large portion of South Africa's agricultural market. Deltamethrin and penconazole were reported frequently in the literature, yet sales values reflected low usage. This could be due to the fact that some compounds may be dominant in localised areas where studies have been done, whilst usage elsewhere may be minimal. The use of deltamethrin, for example, is concentrated in the Western Cape and Kwazulu-Natal Provinces, and penconazole is concentrated in the Western Cape. Alternatively, factors other than usage, for example pesticide properties, could be the reason why these pesticides have received research attention.

It is to be noted that not only pesticides currently in use could impact the environment, but also pesticides that have been banned, but have a long persistence time (e.g. DDT). A project involving an assessment of 30 years of available data on organochlorine residues in South Africa was carried out by the United Nations Environmental Programme (Bouwman *et al.*, 2003). Records showed that organochlorine pesticides are dominant, with DDT and dieldrin originating from past agriculture and control of Tsetse flies, as well as atrazine residues reflecting current agricultural use. The importance of adequate disposal facilities and effective management of obsolete pesticides in South Africa was widely discussed by Naidoo and Buckley (2003).

Similar to the rest of Africa, most South African studies were done on the contamination of surface waters rather than groundwaters. A general lack of data on groundwater pollution from pesticides is due to (i) surface waters being the main source of water supply in the country, (ii) cost and difficulty to measure organic contaminants, and (iii) private companies often being sensitive to make pollution-related data publicly available.

Solomons *et al*. (2003) and London *et al*. (2000) investigated the contamination of surface and groundwaters from 1997 to 1999 in areas of the Western Cape that are intensively used for agriculture, namely, the Hex River Valley, Grabouw and Piketberg. Endosulfan and chlorpyrifos were the dominant pesticides found in surface and groundwater samples. Out of 382 samples, 30% for chlorpyrifos and 37% for endosulfan were found to be in excess of the European Drinking Water Standard of 0.1 µg L-1. Other pesticides detected include azinphos-methyl, fenarimol, iprodione, deltamethrin, penconazole and prothiofos. Sereda and Meinhardt (2003) investigated water pollution levels in several areas of Kwazulu-Natal and found residues of pyrethroid, organophosphate, organochlorine and carbamate insecticides. Residues were attributed to the high usage of insecticides in the area, for agriculture and the control of malaria (DDT and deltamethrin).

The role of hydrogeological and soil characteristics in pesticide leaching was highlighted by Weaver (1993), who conducted a preliminary survey of pesticide levels in groundwater and soil profiles in two areas of the Western Cape. The Hex River Valley, studied for pesticide levels in groundwater and the Vaalharts Irrigation Scheme, studied for atrazine levels in the soil profile, are both areas of intensive agriculture and are considered to be vulnerable to contamination. The author recommended that pesticides be closely monitored in the future and that further research should be done on a larger scale.

The implementation of a pesticide water monitoring program in South Africa is limited mainly due to a lack of financial and analytical resources (Dalvie *et al.*, 2004). However, Dalvie *et al*. (2004) indicated methods such as solid-phase micro-extraction (SPME) that could provide fast, sensitive and reliable monitoring of pesticides (endosulfan and chlorpyrifos) at relatively low cost in the long term.

5 MITIGATION MEASURES

Groundwater contamination from pesticides can primarily be reduced through appropriate management and application. Comprehensive pesticide programmes at farm level, aimed at both increasing pesticide efficiency and reducing leaching, are already in place in some important South African production systems. Scheduling of pesticide applications based on weather conditions and pest forecasts can optimise the usage and reduce the environmental impact. In addition, appropriate management practices, like irrigation scheduling, choice of the irrigation method (Perillo *et al.*, 1999; Blackwell, 2000; Reichenberger *et al.*, 2002), choice of the pesticide application method and timing to reduce spray drift due to unfavourable climatic conditions, appropriate maintenance of spraying equipment, tillage (Malone *et al.*, 2003) etc. can reduce the hazard of groundwater contamination.

Integrated Pest Management (IPM) programmes are one of the most well-founded prevention measures for groundwater pollution from pesticides. However, this practice requires high-level skills; particularly a knowledge of the biology and ecology of the pest and associated crop that must be protected, as well as monitoring of pest populations in target areas. In addition, this needs to be matched with the need to produce more food and income. Orr and Ritchie (2004) indicated that the implementation of an IPM programme in Malawi could only be successful if increased yields and cash income were achieved. Similar conclusions were drawn by Adda *et al.* (2002), who applied IPM to maize storage systems in Togo. On the other hand, Maumbe and Swinton (2003) indicated that hidden health costs could offset the benefits earned from pesticide applications by Zimbabwean cotton farmers.

Silvie *et al.* (2001) investigated pesticide spraying programs for cotton in West Africa. They concluded that procedures, sampling plans and target insects, considered as key pests, are specific to each country. Improved, country-specific practices and management can yield both ecological and economic benefits. Wiktelius *et al.* (1999) carried out field trials in Algeria, Nigeria, Tanzania, Uganda and Zambia to determine the effects of organochlorine pesticides on non-target organisms. They concluded that African countries need to develop procedures for testing pesticides in order to acquire correct information concerning adverse side-effects.

Showler (2002) suggested that emerging technologies should be applied for locust outbreak control systems in Africa, whilst Muller *et al.* (2002) suggested a participatory approach and biological control of grasshoppers in Benin. Biological control was also suggested by Charleston and Kfir (2000), who investigated Indian mustard as a trap crop for diamondback moths, Olckers and Hulley (1995) for the control of Solanum weeds in South Africa, as well as Maniania *et al.* (2003) for the control of onion thrips in Kenya. Ahmed *et al.* (2001) investigated alternative strategies for pesticide management to control aphids in wheat in Sudan. They concluded that the strategy using imidacloprid was adequate and less hazardous to the environment compared to a standard mixture of lindane and thiram.

In recent years, advances have been made in the fields of phytotransformation, rhizosphere bioremediation, phytostabilisation, phytoextraction and rhizofiltration (Susarla *et al.*, 2002). A number of plants and microorganism species could potentially be used for uptake and *in-situ* degradation of pesticides. In addition, plants stimulate the degradation of organic chemicals in the rhizosphere through the release of root exudates, enzymes, and the build-up of organic carbon in the soil (Erickson *et al.*, 1995). Hydrofracturing techniques are being developed to improve the delivery of nutrients to micro-organisms in low-permeability geologic media, to use microbial produced biopolymers as *in-situ* plugging agents, or to create passive treatment systems (biofilters) (Knapp and Faison, 1997). Bio-bed technologies are also being developed (Vischetti *et al.*, 2004), where pesticides resulting from spillages during filling and rinsing spraying equipment are trapped and contained in pits that are kept moist, slightly acidic and enriched with organic matter.

Contours and buffer strips are common mitigation measures to protect surface waters interacting with groundwater, (Dabrowski *et al.*, 2002). Pesticide transported through interflow can be intercepted by the root zone of vegetative filter strips (Watanabe and Grismer, 2003). Alternatively, high-tech treatment of groundwater is possible to remove pesticides and produce drinking water. For example, Van der Bruggen *et al.* (2001) indicated the treatment of groundwater with nanofiltration as a feasible technology with realistic costs.

6 CONCLUSIONS

Africa is faced with the need to produce enough food to ensure food security but at the same time protect natural resources, particularly groundwater. Pesticides in African groundwaters mainly originate from high-income crop plantations (like staple food crops, sugarcane, subtropical fruit orchards, vineyards, vegetables) and insect disease vectors control in houses. Pest control through the usage of pesticides is an important practice aimed at ensuring affluent livelihoods. This could present a problem in developing countries due to a lack of technologies, regulations, resources and education, as well as weak import control and poor management of pesticides.

Contamination of African surface waters from pesticides is better documented compared to that of groundwaters. This is mainly due to surface waters being the main source of water supply in many countries, the poor accessibility, high cost and difficulty of measuring organic contaminants in groundwaters, as well as the sensitivity and lack of awareness in terms of pollution problems. Besides the lack of monitoring, a lack of education and training in pesticide handling and hazard remains a major issue in Africa.

A number of catchment processes (run-off and stormflow, drainage and leaching, preferential flow, throughflow) were related to NPS pollution from pesticides. The main factors were identified to be the local hydrogeological set-up, as well as specific pesticide properties (volatilisation, sorption and decay). These processes need to be quantified in order to predict their impacts on groundwater and to design related management strategies. Management strategies should be tailored to specific sites depending on ecological, economic and social aspects, as well as possible adverse side-effects.

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REFERENCES

Adda A., Borgemeister C., Biliwa A., Meikle W.G., Markham R.H. and Poehling H.-M. 2002. Integrated pest management in post-harvest maize: a case study from the Republic of Togo (West Africa). *Agriculture, Ecosystems & Environment* 93(1–3) 305–321.

Adeyeye A. and Osibanjo O. 1999. Residues of organochlorine pesticides in fruits, vegetables and tubers from Nigerian markets. *The Science of The Total Environment* 231(2–3) 227–233.

Ahmed N.E., Kanan H.O., Inanaga S, Ma Y.Q. and Sugimoto Y. 2001. Impact of pesticide seed treatments on aphid control and yield of wheat in the Sudan. *Crop Protection* 20(10) 929–934.

Alemaw B.F., Shemang E.M. and Chaoka T.R. 2004. Assessment of groundwater pollution vulnerability and modelling of the Kanye Wellfield in SE Botswana – a GIS approach. *Physics and Chemistry of the Earth, Parts A/B/C* 29(15–18) 1125–1128.

Amr M.M. 1999. Pesticide monitoring and its health problems in Egypt, a Third World country. *Toxicology Letters* 107(1–3) 1–13.

Barakat A.O. 2004. Assessment of persistent toxic substances in the environment of Egypt. *Environment International* 30(3) 309–322.

Bennasser L., Fekhaoui M., Benoit-Guyod J.L. and Merlin G. 1997. Influence of tide on water quality of lower Sebou polluted by Gharb plain wastes (Morocco). *Water Research* 31(4) 859–867.

Bergstrom L. 1995. Leaching of dichlorprop and nitrate in structured soil. *Environmental Pollution* 87(2) 189–195.

Blackwell P.S. 2000. Management of water repellency in Australia, and risks associated with preferential flow, pesticide concentration and leaching. *Journal of Hydrology* 231–232 384–395.

Boesten J.J.T.I. 1999. From laboratory to field: uses and limitations of pesticide behaviour models for the soil/plant system. *Weed Res.* 40 123–138.

Bouwman H., Vosloo R., Maboeta M. and Bester B. 2003. An assessment of 30 years of environmental organochlorine residue data from southern Africa. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21–23 January 2003, University of Cape Town, Cape Town, South Africa.

Bresler E., Mcneal B.L. and Carter D.L. 1982. Saline and sodic soils. Springer-Verlag, New York.

Charleston D.S. and Kfir R. 2000. The possibility of using Indian mustard, *Brassica juncea*, as a trap crop for the diamondback moth, *Plutella xylostella*, in South Africa. *Crop Protection* 19(7) 455–460.

Chisupa K. 2005. Toxic waste dumping in Zambia. No. 6, Sustainable Development Part 2. Voices from Africa, UN-NGLS. http://www.unsystem.org/ngls/documents/publications.en/voices.africa/number6/vfa6.04.htm

Christiansen J.S., Thorsen M., Clausen T., Hansen S. and Refsgaard J.C. 2004. Modelling of macropore flow and transport processes at catchment scale. *Journal of Hydrology* 299(1–2) 136–158.

Clarke E.E.K. 2004. The experience of starting a poison control centre in Africa – the Ghana experience. *Toxicology* 198(1–3) 267–272.

Dabrowski J.M., Peall S.K.C., Van Niekerk A., Reinecke A.J., Day J.A. and Schulz R. 2002. Predicting runoff-induced pesticide input in agricultural sub-catchment surface waters: linking catchment variables and contamination. *Water Research* 36(20) 4975–4984.

Dalvie M.A., Sinanovic E., London L., Cairncross E., Solomon A. and Adam H. 2004. Cost analysis of ELISA, solid-phase extraction, and solid-phase microextraction for the monitoring of pesticides in water. *Environmental Research* Available online 17 November 2004.

Darkoh M.B.K. 2003. Regional perspectives on agriculture and biodiversity in the drylands of Africa. *Journal of Arid Environments* 54(2) 261–279.

El Nemr and Abd-Allah A.M.A. 2004. Organochlorine contamination in some marketable fish in Egypt. *Chemosphere* 54(10) 1401–1406.

El-Kabbany S., Rashed M.M. and Zayed M.A. 2000. Monitoring of the pesticide levels in some water supplies and agricultural land, in El-Haram, Giza (A.R.E.). *Journal of Hazardous Materials* 72(1) 11–21.

Erickson L.E., Davis L.C. and Narayanan M. 1995. Bioenergetics and bioremediation of contaminated soil. *Thermochimica Acta* 250(2) 353–358.

FAO. 2004. http://www.fao.org/newsroom/en/news/2004/50119/index.html

Fetter C.W. 1993. Contaminant hydrogeology. Prentice-Hall Inc., New Jersey.

Gressel J., Hanafi A., Head G., Marasas W., Babatunde Obilana A., Ochanda J., Souissi T. and Tzotzos G. 2004. Major heretofore intractable biotic constraints to African food security that may be amenable to novel biotechnological solutions. *Crop Protection* 23(8) 661–689.

Haria A.H., Hodnett M.G. and Johnson A.C. 2003. Mechanisms of groundwater recharge and pesticide penetration to a chalk aquifer in southern England. *Journal of Hydrology* 275(1–2) 122–137.

Hubbert K.R., Graham R.C. and Anderson M.A. 2001. Soil and weathered bedrock. Components of a Jeffrey pine plantation substrate. *Soil Sci. Soc. Am. J.* 65 1255–1262.

Johnson A.C., Besien T.J., Bhardwaj C.L., Dixon A., Gooddy D.C., Haria A.H. and White C. 2001. Penetration of herbicides to groundwater in an unconfined chalk aquifer following normal soil applications. *Journal of Contaminant Hydrology* 53(1–2) 101–117.

Kamra S.K., Lennartz B., Van Genuchten M.Th. and Widmoser P. 2001. Evaluating non-equilibrium solute transport in small soil columns. *Journal of Contaminant Hydrology* 48(3–4) 189–212.

Khaled A., El Nemr A., Said T.O., El-Sikaily A. and Abd-Allah A.M.A. 2004. Polychlorinated biphenyls and chlorinated pesticides in mussels from the Egyptian Red Sea coast. *Chemosphere* 54(10) 1407–1412.

Kishimba M.A., Henry L., Mwevura H., Mmochi A.J., Mihale M. and Hellar H. 2004. The status of pesticide pollution in Tanzania. *Talanta* 64(1) 48–53.

Knapp R.B. and Faison B.D. 1997. A bioengineering system for in situ bioremediation of contaminated groundwater. *Journal of Industrial Microbiology & Biotechnology* 18(2–3) 189–197.

Laegdsmand M., Villholth K.G., Ullum M. and Jensen K.H. 1999. Processes of colloid mobilization and transport in macroporous soil monoliths. *Geoderma* 93(1–2) 33–59.

Larsson M.H., Jarvis N.J., Torstensson G. and Kasteel R. 1999. Quantifying the impact of preferential flow on solute transport to tile drains in a sandy field soil. *Journal of Hydrology* 215(1–4) 116–134.

Leib B.G. and Jarrett A.R. 2003. Comparing soil pesticide movement for a finite-element model and field measurements under drip chemigation. *Computers and Electronics in Agriculture* 38(1) 55–69.

Lennartz B. 1999. Variation of herbicide transport parameters within a single field and its relation to water flux and soil properties. *Geoderma* 91(3–4) 327–345.

London L. and Myers J. 1995. General patterns of agrichemical usage in the southern region of South Africa. *South African J. Sci.* 91 509–514.

London L., Dalvie M.A., Cairncross E. and Solomons A. 2000. The quality of surface and groundwater in the rural Western Cape with regard to pesticides. Water Research Commission Rep. No. 795/1/00, Pretoria, South Africa.

Malone R.W., Logsdon S., Shipitalo M.J., Weatheington-Rice J., Ahuja L. and Ma L. 2003. Tillage effect on macroporosity and herbicide transport in percolate. *Geoderma* 116(1–2) 191–215.

Maniania N.K., Sithanantham S., Ekesi S., Ampong-Nyarko K., Baumgartner J., Lohr B. and Matoka C.M. 2003. A field trial of the entomogenous fungus *Metarhizium anisopliae* for control of onion thrips, *Thrips tabaci*. *Crop Protection* 22(3) 553–559.

Manirakiza P., Covaci A., Nizigiymana L., Ntakimazi G. and Schepens P. 2002. Persistent chlorinated pesticides and polychlorinated biphenyls in selected fish species from Lake Tanganyika, Burundi, Africa. *Environmental Pollution* 117(3) 447–455.

Mansour S.A. 2004. Pesticide exposure—Egyptian scene. *Toxicology* 198(1–3) 91–115.

Mansour S.A. and Sidky M.M. 2003. Ecotoxicological Studies. 6. The first comparative study between Lake Qarun and Wadi El-Rayan wetland (Egypt), with respect to contamination of their major components. *Food Chemistry* 82(2) 181–189.

Matthews G., Wiles T. and Balegeul P. 2003. A survey of pesticide application in Cameroon. *Crop Protection* 22(5) 707–714.

Maumbe B.M. and Swinton S.M. 2003. Hidden health costs of pesticide use in Zimbabwe's smallholder cotton growers. *Social Science and Medicine* 57(9) 1559–1571.

Muller D., De Groote H., Gbongboui C. and Langewald J. 2002. Participatory development of a biological control strategy of the variegated grasshopper in the humid tropics in West Africa. *Crop Protection* 21(4) 265–275.

Naidoo V. and Buckley C.A. 2003. Survey of pesticide wastes in South Africa and review of treatment options. Water Research Commission Rep. No. 1128/1/03, Pretoria, South Africa.

Netto A.M., Pieritz R.A. and Gaudet J.P. 1999. Field study on the local variability of soil water content and solute concentration. *Journal of Hydrology* 215(1–4) 23–37.

Nimmo J.R., Perkins K.S., Rose P.E., Rousseau J.P., Orr B.R., Twining B.V. and Anderson S.R. 2002. Kilometer-scale rapid transport of naphthalene sulfonate tracer in the unsaturated zone at the Idaho National Engineering and Environmental Laboratory. *Vadose Zone J.* 1 89–101.

Novak S.M., Portal J.M. and Schiavon M. 2001. Effects of soil type upon metolachlor losses in subsurface drainage. *Chemosphere* 42(3) 235–244.

Nyeko P., Edwards-Jones G., Day R.K. and Raussen T. 2002. Farmers' knowledge and perceptions of pests in agroforestry with particular reference to *Alnus* species in Kabale district, Uganda. *Crop Protection* 21(10) 929–941.

Olckers T. and Hulley P.E. 1995. Importance of preintroduction surveys in the biological control of *Solanum* weeds in South Africa. *Agriculture, Ecosystems and Environment* 52(2–3) 179–185.

Orr A. and Ritchie J.M. 2004. Learning from failure: smallholder farming systems and IPM in Malawi. *Agricultural Systems* 79(1) 31–54.

Pan L., Wu Y.S. and Zhang K. 2004. A modeling study of flow diversion and focusing in unsaturated fractured rocks. *Vadose Zone J.* 3 233–246.

Perillo C.A., Gupta S.C., Nater E.A. and Moncrief J.F. 1999. Prevalence and initiation of preferential flow paths in a sandy loam with argillic horizon. *Geoderma* 89(3–4) 307–331.

Pretty J.N., Morison J.I.L. and Hine R.E. 2003. Reducing food poverty by increasing agricultural sustainability in developing countries. *Agriculture, Ecosystems and Environment* 95(1) 217–234.

Rae J.E., Cooper C.S., Parker A. and Peters A. 1998. Pesticide sorption onto aquifer sediments. *Journal of Geochemical Exploration* 64(1–3) 263–276.

Ray C., Vogel T. and Dusek J. 2004. Modeling depth-variant and domain-specific sorption and biodegradation in dual-permeability media. *Journal of Contaminant Hydrology* 70(1–2) 63–87.

Reichenberger S., Amelung W., Laabs V., Pinto A., Totsche K.U. and Zech W. 2002. Pesticide displacement along preferential flow pathways in a Brazilian Oxisol. *Geoderma* 110(1–2) 63–86.

Renaud F.G., Brown C.D., Fryer C.J. and Walker A. 2004. A lysimeter experiment to investigate temporal changes in the availability of pesticide residues for leaching. *Environmental Pollution* 131(1) 81–91.

Schierholz I., Schafer D. and Kolle O. 2000. The Weiherbach data set: An experimental data set for pesticide model testing on the field scale. *Agricultural Water Management* 44(1–3) 43–61.

Sereda B.L. and Meinhardt H.R. 2003. Insecticides in the South African water environment of the KwaZulu-Natal malaria endemic area. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21–23 January 2003, University of Cape Town, Cape Town, South Africa.

Shereif M.M. and Mancy K.H. 1995. Organochlorine pesticides and heavy metals in fish reared in treated sewage effluents and fish grown in farms using polluted surface waters in Egypt. *Water Science and Technology* 32(11) 153–161.

Showler A.T. 2002. A summary of control strategies for the desert locust, *Schistocerca gregaria* (Forskål). *Agriculture, Ecosystems and Environment* 90(1) 97–103.

Sibanda T., Dobson H.M., Cooper J.F., Manyangarirwa W. and Chiimba W. 2000. Pest management challenges for smallholder vegetable farmers in Zimbabwe. *Crop Protection* 19(8–10) 807–815.

SIGMA BETA 2004. Knowledge Review of Modelling Non-point Source Pollution in Agriculture from Field to Catchment Scale. Report to the Water Research Commission, Pretoria, South Africa.
Silvie P., Deguine J.P., Nibouche S., Michel B. and Vaissayre M. 2001. Potential of threshold-based interventions for cotton pest control by small farmers in West Africa. *Crop Protection* 20(4) 297–301.
Solomons A., Dalvie M.A., Cairncross E. and London L. 2003. Contamination of rural surface and groundwater by endosulfan in farming areas of the Western Cape, South Africa. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21–23 January 2003, University of Cape Town, Cape Town, South Africa.
Sovik A.K., Alfnes E., Breedveld G.D., French H.K., Pedersen T.S. and Aagaard P. 2002. Transport and degradation of toluene and *o*-xylene in an unsaturated soil with dipping sedimentary layers. *J. Environ. Qual.* 31 1809–1823.
Susarla S., Medina V.F. and Mccutcheon S.C. 2002. Phytoremediation: An ecological solution to organic chemical contamination. *Ecological Engineering* 18(5) 647–658.
Tsegaye B. 1997. The significance of biodiversity for sustaining agricultural production and role of women in the traditional sector: the Ethiopian experience. *Agriculture, Ecosystems and Environment* 62(2–3) 215–227.
UNEP 2002. Action plan for the implementation of the environment initiative of the new partnership for Africa development. Part 1. United Nations Environment Programme, Working Draft 1, 7 June 2002.
UNEP 2005. Groundwater and its susceptibility to degradation. A global assessment of the problem and options for management. UNEP/DEWA/RS. 03-3. http://www.unep.org/DEWA/water/groundwater/
Usher B., Pretorius J.A., Dennis I., Jovanovic N.Z., Clarke S., Cave L., Titus R. and Xu Y. 2004. Identification and Prioritization of Groundwater Contaminants and Sources in South Africa's Urban Catchments. Water Research Commission Report No. 1326/1/04, Pretoria, South Africa.
Van Der Bruggen B., Everaert K., Wilms D. and Vandecasteele C. 2001. Application of nanofiltration for removal of pesticides, nitrate and hardness from groundwater: rejection properties and economic evaluation. *J. Membr. Sci.* 193(2) 239–248.
Vanclooster M., Boesten J.J.T.I., Trevisan M., Brown C.D., Capri E., Eklo O.M., Gottesburen B., Gouy V. and Van Der Linder A.M.A. 2000. A European test of pesticide-leaching models: methodology and major recommendations. *Agric. Water Manage.* 44 1–19.
Vischetti C., Capri E., Trevisan M., Casucci C. and Perucci P. 2004. Biomassbed: a biological system to reduce pesticide point contamination at farm level. *Chemosphere* 55(6) 823–828.
Watanabe H. and Grismer M.E. 2003. Numerical modeling of diazinon transport through inter-row vegetative filter strips. *Journal of Environmental Management* 69(2) 157–168.
Weaver J.M.C. 1993. A preliminary survey of pesticide levels in groundwater from a selected area of intensive agriculture in the Western Cape. Water Research Commission Rep. No. 268/1/93, Pretoria, South Africa.
Wiktelius S., Chiverton P.A., Meguenni H., Bennaceur M., Ghezal F., Umeh E.D.N., Egwuatu I., Minja E., Makusi R. and Tukahirwa E. 1999. Effects of insecticides on non-target organisms in African agroecosystems: a case for establishing regional testing programmes. *Agriculture, Ecosystems and Environment* 75(1–2) 121–131.
Wilson J.S. and Otsuki T. 2004. To spray or not to spray: pesticides, banana exports, and food safety. *Food Policy* 29(2) 131–146.
Zehe E. and Fluhler H. 2001. Preferential transport of isoproturon at a plot scale and a field scale tile-drained site. *Journal of Hydrology* 247(1–2) 100–115.

8

Degradation of groundwater resources under African cities: Technical and socio-economic challenges

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ABSTRACT: We examine the state of groundwater quality and management in mid-sized African cities, where significant proportions of the population are dependent on groundwater. African urban centres are rapidly expanding and the groundwater underlying these areas needs proper protection. Case study cities from Mozambique, Kenya and Mali are used for the discussion. In these cities, groundwater monitoring programs were established employing portable chemical and microbiological monitoring equipment. Widespread faecal contamination and elevated nutrient levels (with respect to surrounding rural areas) was commonly detected. Possible solutions that are highlighted and evaluated include various technical, socio-economic and water policy strategies as well as the role of the private sector.

Keywords: aquifer management, faecal contamination, urban groundwater, water policy.

1 INTRODUCTION

The transmission of disease through contaminated groundwater resources is well documented (e.g. Macler & Merckle, 2000), with over 100 known viral, bacterial and protozoan pathogens that can contaminate groundwater from faecal and other sources (Kramer *et al.*, 1996). These adverse health issues are especially significant in parts of Africa where large sections of the population depend on untreated groundwater supplies.

Proper aquifer protection is complicated by rapid urban expansion. For the first time in human history, the majority of the world's population will be urban dwellers by mid-2007.

Water supply and management in urban areas is complicated not only by increases in population and density, but also due to the nature of urbanisation itself. The large number of multiple pollution sources (e.g. sewers/latrines, cemeteries, domestic waste, landfills, etc.) found in urban areas complicate urban groundwater protection. These multiple sources are compounded by variable dynamic distributions in hydraulic head due to complex abstraction patterns and often a multitude of abandoned boreholes in urban areas (Cronin *et al.*, 2003). In addition to these technical problems, there are socio-economic issues such as under-investment in the infrastructure needed for water provision (Sinclair, 2000).

This paper documents efforts to monitor and understand the extent of microbiological pollution in urban groundwater systems in selected African cities, with case studies from Mozambique,

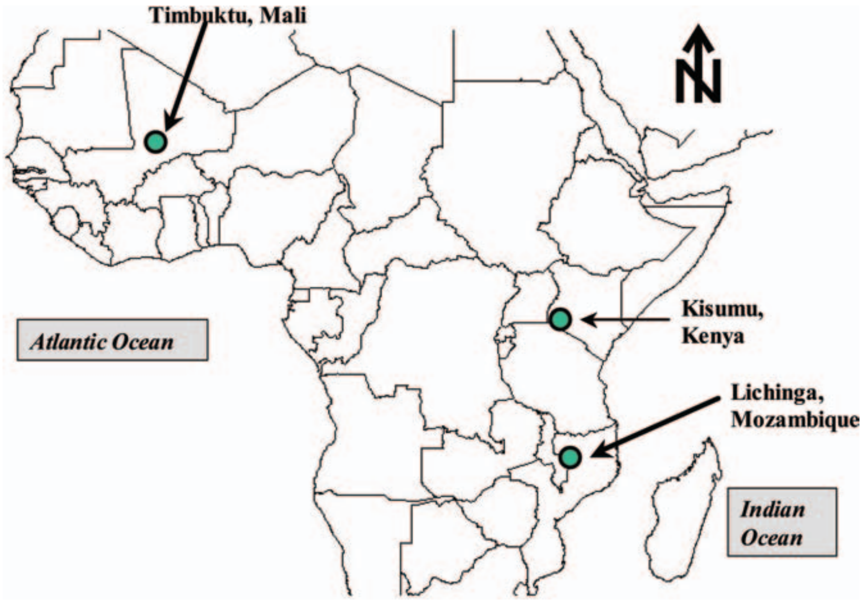


Figure 8.1. Location of case study cities in sub-Saharan Africa.

Kenya and Mali. Selected technical and socio-economic strategies to best protect groundwater resources from increasing urbanisation are also described. Such integrated approaches are vital if the Millennium Development Goals of halving the number of people without access to safe drinking water or basic sanitation by 2015 is to be achieved, not to mention the additional target aiming to achieve a significant improvement in the lives of at least 100 million slum dwellers by 2020.

2 GROUNDWATER QUALITY MONITORING STUDIES

2.1 *Background*

Monitoring programs in the towns of Lichinga (Mozambique), Kisumu (Kenya) and Timbuktu (Mali) have highlighted common problems of groundwater contamination in many parts of urban Africa (Fig. 8.1). Some of these towns' key statistics are compared in Table 8.1. The shallow depth, presence of fractures, and lack of thick soil cover in the study area result in a highly vulnerable aquifer system.

Microbial water quality monitoring was conducted using the portable *Delagua* water testing kit; this is a robust, relatively low-cost method of assessing sewage contamination by analysing the concentration of thermotolerant coliform bacteria (TTC) in water samples. Sanitary risk inspections were also carried out and proved a useful tool (WHO, 2004). These inspections entail the systematic logging of observable faults in the wellhead vicinity that may lead to the degradation of water quality (Lloyd and Bartram, 1991). Each fault is considered one point on the sanitary risk inspection score. Chemical water quality parameters – principally nitrates and chlorides – were analysed using a Palintest Photometer Model 5000 (Palintest UK) to assess the affects of on-site sanitation on groundwater quality.

Table 8.1. Comparison of key parameters for Timbuktu, Mali and Lichinga, Mozambique.

	Timbuktu	Kisumu	Lichinga
Country	Mali	Kenya	Mozambique
Co-ordinates	16° 46′N, 3° 00′W	00° 05′S, 34° 47′E	13° 18′S, 35° 15′E
Town area	880 hectares	40,000 hectares	12,500 hectares
Population	~38,000	~400,000	~100,000
Average Rainfall	230 mm/yr. (June to Sept.). Values are highly variable	1200–1400 mm annual rainfall (650 mm between March and May)	1,120 mm/yr. (mainly between October and April)
Geology	Extensive sandy areas at northern edge of the Sahel and southern edge of the Sahara	Rift volcanics (phonolitic lava) of tertiary age	Crystalline basement complex
Water receptors	Traditional shallow to deep dug wells. Drilled and dug wells to depth increasing from 3 m near the Niger River to over 35 m at 45 km distance, fitted with Mark II hand pumps	Small piped system but mainly traditional shallow hand dug wells, hand pumps and foot pumps	Small piped system but mainly traditional shallow hand-dug wells; improved windlass wells, Afridev hand pumps, rope pumps, local swamps and springs
Sanitation types	Pit latrines (traditional and improved) and seepage pits in urban areas. Latrines are rare in rural areas, except at schools/health centres	Pit latrines; Piped system near town centre that discharges to Lake Victoria	Pit latrines (traditional and improved and some ecological sanitation), some septic tanks
Monitoring Program	September 2002 to May 2003 at 94 wells, biannual sampling	February 2002 to November 2004; 14 wells sampled on 13 occasions	April 2002 to August 2004 at 74 wells quarterly, 159 other samples also taken

Water quality data was evaluated with the results from the sanitary risk inspections to:

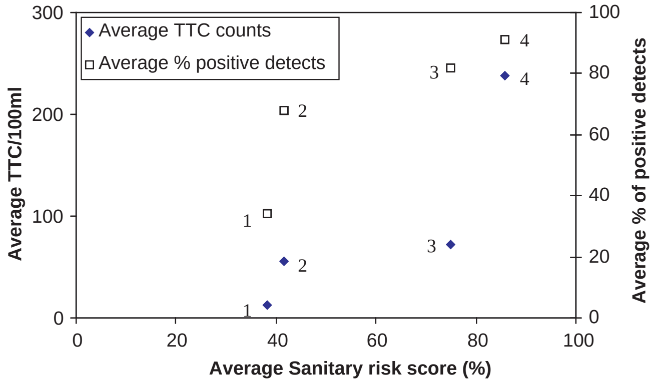
(a) identify possible sources of faecal contamination;
(b) identify potential contaminant pathways to the well;
(c) raise awareness among stakeholders with regard to the impacts of unsanitary conditions or practices on groundwater quality.

2.2 *Results*

Several main conclusions can be drawn from the microbiological data. In Lichinga it is obvious that the level of protection at the wellhead strongly influences the quality of the wellwater (Fig. 8.2). This is important, as it shows that wellhead protection is a vital aspect in protecting groundwater quality. Indeed, in correlations of risk versus TTC carried out in order to prioritise interventions, poor wellhead construction and maintenance were consistently more strongly correlated with higher TTC counts than the proximity of latrines (Cronin *et al*., 2004).

Another important issue is the driving mechanisms for contamination to enter the well. Obviously poor wellhead construction and/or stagnant water on the wellhead can facilitate contamination ingress. Rainfall recharge will therefore also be an important driving factor. In Lichinga, the average TTC of all receptor type analyses rose from 39 TTC/100 ml to 121 TTC/100 ml in the wet season (Cronin *et al*., in review a).

In Kisumu city, wellhead protection and recharge also influence microbial water quality. The magnitude of contamination is also strongly affected by the population density and socio-economic setting. As example the two contrasting suburbs, Migosi and Manyatta, can be compared.



1 = Handpumps (average of 73 samples)
2 = Improved traditional wells with windlass and concrete plinth (average of 128 samples)
3 = Traditional wells (no wellhead protection) (average of 85 samples)
4 = Swamp and unprotected spring water (average of 21 samples)

Figure 8.2. Average sanitary risk scores for the four different receptor types in Lichinga, Mozambique versus the average Thermotolerant coliform colony forming units/100 ml for these receptor types. Also plotted are the average % of positive TTC detects (i.e. counts of >0) for each receptor type, adapted from Cronin *et al.*, in review a).

Migosi	Manyatta	
Middle income	Low-income	
Low to medium density area (~80 people/ha)	High density estate (>200 people/ha)	
97% of houses have flush toilets with water shortage and sewerage problems		
70% of the houses with flush toilets use pit latrines		
350 pit latrines/km^2	957 pit latrines/km^2	(Drangert *et al*., 2002)
4 boreholes and 66 wells/km^2	321 shallow wells/km^2 and 0 boreholes	(Drangert *et al*., 2002)

At present 60% of wells in Migosi and Manyatta remain unprotected. Average water demand for the two estates is ~1600 m^3/d with only 11% supplied by piped water systems (Okotto-Okotto, 1999). No routine water quality monitoring was in place prior to this study. Figure 8.3 demonstrates that much higher counts were found in the poorer and more densely populated area of Manyatta than Migosi, although the counts in both of these areas were exceptionally high (Cronin *et al*., 2004). The comparison between Migosi and Manyatta (Fig. 8.4) shows that the concentration of nutrients in the aquifer can vary over short distances, and that it is influenced by the density of the overlying urban centres.

The chemical quality of urban groundwater is also of concern, especially nitrates derived from on-site sanitation. Sampling of nitrates and chlorides in Lichinga and Timbuktu has clearly shown the effect of the additional nutrient loadings from pit latrines. Nitrate values of at least 3 times more elevated with respect to neighbouring rural areas were found for both cities (Cronin *et al*., in review b). This result provides clear evidence of how urbanisation is affecting groundwater chemical quality. These processes have serious implications for groundwater management in Africa.

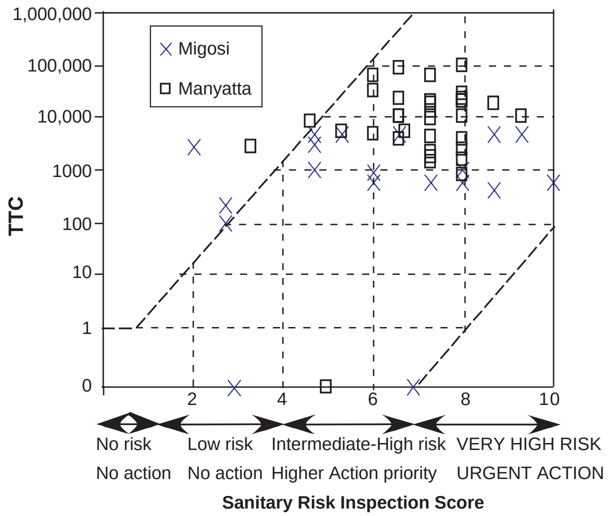


Figure 8.3. TTC (colony forming units/100 ml) vs. sanitary risk score, Kisumu; N = 53. Results plotting outside the diagonal parallel lines require further investigation (from Cronin *et al.*, 2004).

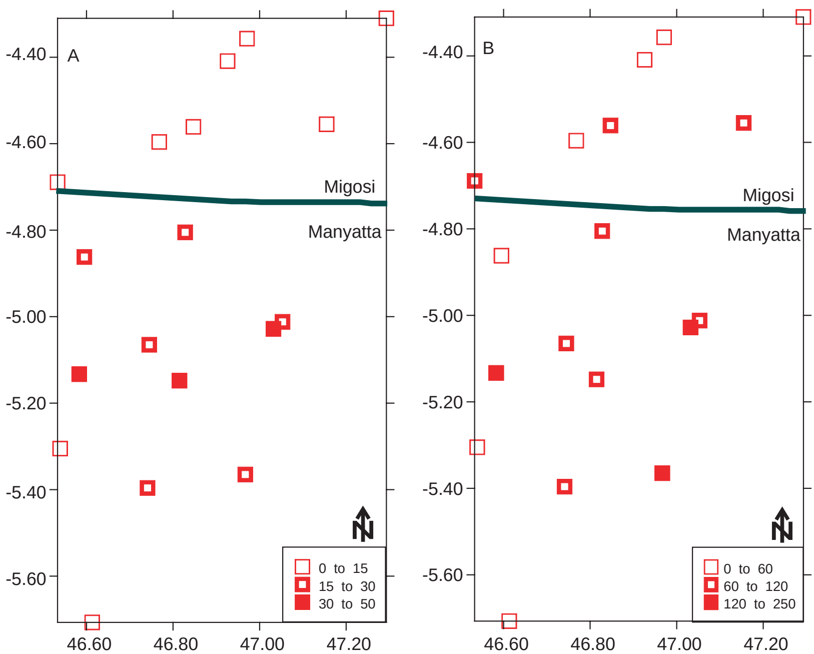


Figure 8.4. Map showing sampling results from the suburbs of Migosi and Manyatta, Kisumu. The thick black line represents the road dividing the two areas. Map co-ordinates are seconds west of 34°W and seconds south of the equator. Map A shows mg/L of nitrate as N and map B shows mg/L of chloride from Nov. 2004.

A typical strategy for reducing excess nitrates in drinking water supplies in developed countries has been to blend high nitrate waters with lower concentration waters from an alternative source. This however is not a viable option in the predominately unpiped water systems of sub-Saharan Africa. Hence, prevention of excessive N loading to the aquifers is the only practical control option.

3 ADDRESSING THE ISSUES: TECHNICAL AND POLICY VIEWPOINTS

Proper water management is central to groundwater protection. In developed countries, water management policies can be divided into three distinct historic time periods. The first stage, *supply management*, involved utilising previously undeveloped sources to satisfy growing demand requirements. As it became evident that such sources were increasingly difficult to find, or transportation costs became expensive, the water sector in developed countries entered into a period of *demand management*, i.e., to influence the demand for water by various measures (Lundqvist and Gleick, 1997). This incorporated such initiatives in developed countries as metered billing, repair of leaking mains, low-water use toilets, the treatment and re-use of water by industries in their processes, etc.

Hence, water demand management has a role to play not in limiting average consumptions levels, but in ensuring that existing water supplies are more effectively used (Mwendera *et al*., 2003), although the supply-side augmentation focus still generally dominates over demand management. Some authors argue that demand management in Africa thus far has been driven more by international environmental and development agencies rather than local interests (Gumbo, 2004; Swatuk & Rahm, 2004). However, there is little doubt that this approach has the potential to avoid high infrastructure costs and improve water and sanitation provision via such initiatives as leakage control, proper communal tap management, appropriate water tariffs, community participation and most importantly, education of all water users as to the value of water (UN-HABITAT, 2003).

With the continuing focus on supply augmentation and demand management, there is an additional third option for meeting increasing water resource needs while simultaneously helping to protect existing water supplies. This is termed *re-use management* and has been successfully used by the industry to treat and re-use its process water. The separation of the different domestic wastewater streams, including drainage waters, allows the decoupling of nutrient content and water quantity, while also facilitating wastewater recycling (Otterpohl *et al*., 2003). Re-use of nutrients in human waste has excellent benefits as crop fertilisers, and it has been estimated that urine-diverting toilets in conjunction with plant fertilisation could reduce N loading to the urban aquifers by over half if properly undertaken (Cronin *et al*., in review b). Proper storage and/or treatment of the faecal wastes are necessary to eliminate pathogenic risks. Further research and, importantly, economical analyses, are vitally needed to test this in different country settings. Small-scale wetlands for the treatment of domestic greywater have been tried in Kisumu city with reasonable success, and this technology could still be refined for use in urban gardening (Emmanuelsson & Linderholm, 2000).

Demand management and re-use management are closely interlinked as separation and recycling of waste waters can reduce the demand for fresh water significantly (Otterpohl *et al*., 2003). Hence, wastewater and drainage waters could be considered an important potential resource in many urban areas, if properly managed. A greater diversity of re-use management systems needs to be actively considered for developing country cities (especially their peri-urban areas), as rapid urbanisation, increasing volumes withdrawn from the underlying urban aquifers, and the high capital and maintenance costs associated with conventional piped systems mean that they are not viable options for the near future (Drangert & Cronin, 2004). Careful and informed choices are vital.

Another important factor when considering future water supply and sanitation options in many African cities is the role of independent water supply and sanitation providers. In much of the developing world, domestic water supply and sanitation is dominated not by officially sanctioned water and sanitation companies, but by independent, frequently small-scale operators who may hold more than 80 per cent of the water supply market (Collignon & Vézina, 2001). They are of particular importance for the urban poor, who frequently lack formal title to their land, and those living in peri-urban areas.

The influence of water vendors can be highlighted with Kisumu city as a case study. Interviews with householders (primarily the female heads of the families) in Migosi and Manyatta during November 2004 led to the following generalisations. The average family size was 5 (2 adults and 3 children), who use ~25 L/person/day. Maximum and minimum per capita usage was 50 and 15 L litres respectively. Water supplied by vendors costs an average of 6 Kenya Shilling (US$ 0.08) for a 20 L jerry can. Hence, the average family could spend up to $0.50 per day on water alone. This is a very substantial proportion of the often less than $1/day income. However, this picture is substantially complicated by the fact that households can use several different water sources. First, in the wet season many households collect rainwater for use. Also, many households have dug their own wells to supply a large amount of their own needs, and then may sell water to neighbours at a reduced rate (often ~10% of vendor prices).

The water vendors themselves may be selling water from similar wells (often with very poor quality, as outlined above) or charging extra for water taken from the piped water system that is perceived to be of better quality. The independent vendors will also charge according to the distance over which they transport water to the household. This can vary from only a few metres to several kilometres. Much of the independent water vendors' profits may go to merchants who lease, at very high rates, the handcarts used to transport the water.

This example from Kisumu underlines the complexity of the urban water sector in a typical African town. It also shows that economic difficulties, together with pressure from multilateral financial institutions, has led to major reassessments of the role that government now plays in the delivery of services such as water and sanitation, once considered to be the realm of government. The official water supply authorities frequently have monopoly rights to water supply services within their area of jurisdiction, while independent operators frequently trade without any official legal sanction. Although the urban poor often pay more for their water than their fellow citizens served by piped systems (e.g. UN-HABITAT, 2003), there is enormous variability in terms of quality of service and prices offered by independent water and sanitation providers (Solo, 1999). Indeed it has been argued that independent operators deliver their services without any subsidies or monopolistic conditions, while official water and sanitation providers tend to focus upon the most profitable urban areas (Collignon & Vézina, 2001; Solo, 1999). The legal and regulatory framework that such suppliers operate within needs further research. Additionally, any policy options considering the re-use of wastewaters or other methods to reduce nutrient loadings to groundwater from pit latrines, must consider the number and diversity of players involved in the urban water sector and the potential limitations of centralised control.

5 CONCLUSIONS

Africa's urban population, many of whom are dependent on groundwater, is growing rapidly. Monitoring of thermotolerant coliforms and faecally-derived nutrients has highlighted that urbanisation is adversely affecting groundwater quality in three typical urban African aquifers (Lichinga, Timbuktu, and Kisumu). The quality of well construction and community education and participation in wellhead maintenance are two important mechanisms to tackle these principal risks. Other important factor is rainfall recharge driving the surface pollution into the aquifer. Average nitrate concentrations were found to be at least three times more elevated in the highly populated urban areas with respect to surrounding rural values, with on-site sanitation a major contributor.

A potential mechanism to tackle this issue is urine diversion with its subsequent use for crop fertilisation. Indeed, this and other innovative systems must be attempted to improve upon the current situation of up to 150 million urban Africans without access to adequate provision of water, and up to 180 million in the case of adequate sanitation provision. Additionally, any policy options considering waste re-use and/or alternative water supply measures must consider the number and diversity of players involved in the urban water sector, including independent vendors, and must actively consult and involve these local people. Such integrated approaches are vital if the Millennium Development Goals dealing with water and sanitation are to be achieved.

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REFERENCES

CDSC 2000. Surveillance of waterborne disease and water quality. January to June 2000. *Communicable Disease Report* CDR Weekly 2000; 319–322.

Collignon, B., Vezina, M. 2001. *Independent water and sanitation providers in African cities*: full report of a ten-country study. Water and Sanitation Program.

Comly, H.H. 1945. Cyanosis in infants caused by nitrates in wellater, *J. Am. Med. Assoc*., Vol. 129, 112.

Cronin, A.A., Hoadley, A.W., Gibson, J., Breslin, Kouonto Komou, F., Haldin, L., Pedley, S. (in review a) Urbanisation effects on groundwater chemical quality: findings from 2 African cities reliant on on-site sanitation and focusing on the nitrate problem, submitted to *Journal of Water and Health.*

Cronin, A.A., Breslin, N., Gibson, J., Pedley, S. (in review b) A methodology to prioritise interventions to reduce waterborne disease in Northern Mozambique, submitted to *Environmental Geochemistry and Health*.

Cronin, A.A., Taylor, R.G., Powell, K.L., Barrett, M.H., Trowsdale, S.A., Lerner, D.N. 2003. Temporal trends in the hydrochemistry and sewage-related microbiology of an urban sandstone aquifer – Nottingham, United Kingdom. *Hydrogeology Journal*, Vol. 11, No. 2, p205–216.

Cronin, A.A., Breslin, N., Taylor, R., Pedley, S. 2004. Assessing the risks to groundwater quality from on-site sanitation and poor sanitary well completion, in '*Ecosan-closing the loop*', Proceedings of the 2nd International conference on Ecological Sanitation, Lubeck, Germany, April 2003, edited by C. Werner *et al.*, ISBN 3-00-012791-7, p431–436.

Drangert, J.O., Okotto-Okotto, J., Okoto, L.G., Auko, O. 2002. Going small when the city grows big: New options for water supply and sanitation in rapidly expanding urban areas, *Water International*, International Water Resources Association, Vol. 27, No. 3, p354–363.

Drangert, J.O., Cronin, A.A. 2004. Use and Abuse of the Urban Groundwater Resource – Some policy implications, *Hydrogeology Journal*, Vol. 12, No. 1, p94–102.

Emmanuelsson, K., Linderholm, C.J., 2000. Small grey water treatment systems in Kenya. Master of Science in Engineering Biology, Linkoping University Sweden.

Ferro, B.P.A., Bouman, D. 1992. The hydrogeological map of Mozambique (scale 1:1,000,000), *Hydrogeologie,* No. 1–2, p17–27.

Gumbo, B. 2004. The status of water demand management in selected cities of southern Africa. *Physics and Chemistry of the Earth* 29, 1225–1231.

Kramer, M.H., Herwaldt, B.L., Craun, G.F., Calderon, R.L., Juranek, D.D. 1996. Waterborne disease: 1993 and 1994. *Journal of the American Water Works Association,* 88 (3): p66–80.

Lloyd, B. & Bartram J. 1991. Surveillance solutions to microbiological problems in water quality control in developing countries. *Wat. Sci. Tech.* 24(2): p61–75.

Lundqvist, J. & Gleick, P. 1997. *Sustaining our waters into the 21st century. Background document prepared for Chapter 4 of the Comprehensive Freshwater Assessment*. Stockholm Environment Institute. Stockholm, Sweden.

Macler, B.A., Merckle, J.C. 2000. Current knowledge on groundwater microbial pathogens and their control. *Hydrogeology Journal*, Vol. 8, p29–40.

Mwendera, E.J., Hazelton, D., Nkhuwa, D. Robinson, P., Tjijenda, K., Chavula, G. 2003. Overcoming constraints to the implementation of water demand management in southern Africa. *Physics and Chemistry of the Earth*, Vol. 28, p761–778.

Nomura, A. 1996. Stomach Cancer, in: Scottenfeld, D., Fraumeni, J.F. (Eds.), *Cancer Epidemiology and Prevention*. Oxford University Press, New York.

Okotto-Okotto, J. 1999. A Preliminary Assessment of Groundwater Recharge and Well Yields in peri-urban Manyatta and Migosi of Kisumu, Kenya. Kisumu Ecological Sanitation and Well Water Project (Kesawwa), Final Report, Part II.

Otterpohl, R., Braun, U., Oldenburg, M. 2003. Innovative technologies for decentralised water, wastewater and biowaste management in urban and peri-urban areas. *Water Science and Technology*, Vol. 24, p23–32.

Sinclair, I.C. 2000. 'Better Laws = Better Environment: The Role of the Environmental Lawer in the Reconstruction and Modernisation of the Water Sector in Private Sector Participation', *Water, Air and Soil Pollution*, Vol. 123, p353–360.

Solo, T.M. 1999. 'Small-Scale Entrepreneurs in the Urban Water and Sanitation Market', *Environment and Urbanization*, Vol. 11, No. 1: p117–131.

SPHERE, 2004. *The Sphere Project, Humanitarian Charter and Minimum Standards in Disaster Response,* Oxfam Publishing ISBN 92-9139-097-6.

Swatuk, L.A. Rahm, D. 2004. Integrating policy, disintegrating practice: water resources management in Botswana. *Physics and Chemistry of the Earth*, 29, 1357–1364.

Thompson *et al.*, 2001. *Drawers of water: 30 years of change in domestic water use and environmental health in East Africa*, IIED, London UK.

UN-HABITAT, 2003. *Water and Sanitation in the World's cities: Local Action for Global Goals*, 304pp, Published by Earthscan London, UK and Stylus Publishing LLC, Vermont USA, ISBN 1-84407-004-2.

VanDerslice, J., Briscoe, J. 1995. Environmental Interventions in Developing countries, *Am. Journal of Epidemiology*, Vol. 141, No. 2, p135–144.

WHO 2004. Guidelines for drinking water quality, Third edition, Volume 1: Recommendations, ISBN 942-154638-7.

Groundwater pollution status and aquifer vulnerability

Editorial comment: Groundwater pollution status and aquifer vulnerability

This section deals with the identification of pollution sources and the vulnerability assessment of the aquifers supplying African cities or urban centers. The bulk of the contributions originate from a UNEP and UNESCO project 'Assessment of Pollution Status and Vulnerability of Water Supply Aquifers of African Cities'. The emphasis in these papers varies and the approach to addressing the problem is unique to each country. The contribution made by these papers lies in the first-hand knowledge dissemination of conditions in many of the continent's urban areas. It is encouraging and notable that these efforts represent the addressing of African issues by the people of Africa. Groundwater vulnerability, and the factors contributing to it, provides a window through which African hydrogeology and the acute problems arising from rapid urban development can be appreciated. The wide diversity of geological conditions from fractured hard rock (e.g. Addis Ababa, Ouagadougou and parts of Bamako and Niamey) karstic marbles and dolomite (Lusaka) and primary aquifers indicate a wide array of conditions; despite this the issues highlighted are relatively consistent. It is sincerely hoped that the results presented here will play a meaningful role in raising awareness at managerial level and trigger the political will for action.

The bulk of the papers adopted or adapted the widely used DRASTIC approach. Despite the inherent limitations in this approach, the results presented here give the reader a sense of the vulnerability in terms of its nature, extent and urgency. In many cases such as in Abidjan, Dakar and Mombassa the classic vulnerability map corresponds well to the observed groundwater pollution. Several papers (e.g. Alemayehu *et al.* and Yameogo *et al.*) put the overall vulnerability in context with additional data and delve into the links between the sources, the hydrogeology and the observed water quality. The limitations of a DRASTIC approach are outlined in Cotonou where seawater intrusion is as great a risk as surface contamination and in the Lusaka case where karstification over much of the area makes the traditional vulnerability indicators redundant.

A few recurring aspects are common throughout the African continent:

- Rapid urban development has resulted in many informal settlements in the urban areas of Africa.
- In the bulk of these areas, groundwater from shallow wells is widely used as the source of water.
- There is a lack of formal domestic waste disposal, sanitation and sewerage/effluent systems and the pit latrines and other sources are often in close proximity to wells and seldom designed to minimize groundwater impacts.
- The result is that nitrate (values in excess of 1000 mg/l recorded) and bacteriological contamination (values in excess of 10^6 total bacterial counts in certain wells and 10^6 faecal coliforms in surface waters around Addis Ababa) are the most pervasive parameters giving rise to poor groundwater quality.
- The health effects of this contamination appear to be significant based on data provided by the case studies from such as Mombassa, Lusaka and Addis Ababa.

The evidence of the groundwater quality deterioration in these communities is overwhelming. With few exceptions, the papers presented suggest that groundwater protection strategies are rare, the communities are not always educated on the groundwater quality issues and significant public health risks to the poorest inhabitants exist. Several papers in this section emphasize the need for continued monitoring.

The pervasiveness of these results, and the correlation to results from rapidly urbanized areas elsewhere in the developing world show that groundwater protection strategies should be a very important consideration across the globe.

Groundwater vulnerability in the urban areas of Africa is a pressing problem that urgently needs to be addressed. Authors such as Orange and Palangié (Mali), Alemayehu *et al.* (Ethiopia), Nkuwa *et al.* (Zambia) Boukari *et al.* (Benin) provide several actions that could be taken to mitigate the problem, in the immediate and longer term. These actions include aspects such as education, mobilizing political will, protection zoning and better understanding of relationships using tools such as isotopes, regular water level and quality monitoring and geophysics.

9

A new cartographic approach to determine the groundwater vulnerability of the Abidjan Aquifer

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ABSTRACT: A new approach relating to the vulnerability mapping of pollution in urban environments has been developed and implemented for the aquifer in Abidjan. The parameters identified in this new approach are both intrinsic and external. The digitising and the scanning of the parametric maps were carried out using MapInfo and ArcView. Each map was transformed into a grid and classified into four classes. Each class is based on an index according to its importance in the process of aquifer contamination. Combining these maps using ArcView enabled us to obtain the vulnerability map of the Abidjan Aquifer to pollution. This map reveals two zones: a zone of 'medium vulnerability' located at the east and the west of Abidjan and a high zone of 'high vulnerability' located at the south and north in the area of recharge of the Abidjan Aquifer.

1 INTRODUCTION

The project entitled 'Assessment of pollution status and vulnerability of water supply aquifers of African cities' relates primarily to the question of aquifer vulnerability and the need for protecting the quality of groundwater resources on the continent. The aquifer systems, and in particular those of the town of Abidjan, are currently being contaminated. The problem is especially alarming for the aquifer systems located near the urban centres.

The inefficiency or absence of purification stations in these centres and zones of industrial activities make it possible to understand the extent of the catastrophe that threatens the groundwater resources. These inefficiencies, generally, cause the contamination of groundwater by transporting the contaminants from its initial location, resulting in contaminated water recharge. The phenomenon becomes uncontrollable to such an extent that it often causes the shut down of the boreholes intended for the domestic water supply. For instance, the boreholes in the south of the Abidjan revealed nitrate concentrations reaching 100 mg/l (in N) (Boreholes IFAN, 120 mg/l; Plateau C4, 110 mg/l). All these boreholes were abandoned. The objective of this study is to produce a vulnerability map of the Abidjan Aquifer for the purpose of protection.

The traditional methods (DRASTIC, etc.) of evaluating the vulnerability of aquifers do not enable us to evaluate the vulnerability of the aquifers in urban environments, as they do not take into account certain factors related to large African cities. These factors include human activities, the density of the population, and the type of sanitation used. They are significant parameters that control the vulnerability of groundwater in Africa. Consequently, we propose a new approach for evaluating the aquifer vulnerability in African urban environments.

1.1 *Study area*

The town of Abidjan is located at the south of the Côte d'Ivoire and lies between the latitudes 5°00′ and 5°30′ NR and longitudes 3°50′ and 4°10′ W (Fig. 9.1). It is the economic and administrative capital for the Côte d'Ivoire, with a surface extent of 57 735 ha. Abidjan is made up of ten communes. The population has been increasing rapidly, from about 2, 9 million in 1998 (INS, 2001) to nearly 3, 7 million in 2003, due to the socio-political crisis that the Côte d'Ivoire had undergone since September 2002.

Geologically, the town of Abidjan belongs to the coastal sedimentary basin of the Cretaceous to Quaternary age. This basin has good groundwater potential. Groundwater resources are composed of three aquifer systems, of varying importance (Quaternary, Continental Terminal and Maastrichtian), and only the aquifer of the Continental Terminal, commonly called 'aquifer of Abidjan', is exploited for the drinking-water supply.

The zone of study belongs to an equatorial, transitional climate zone, with two rainy seasons and two dry seasons. The town of Abidjan has a high annual rainfall (1600 mm/year), which enables good recharge of the Abidjan Aquifer. This recharge could be limited significantly, taking into account the high rate of urbanisation and anarchistic installation of certain activities such as industrial discharge, breakage, plantations, etc. These activities can also be sources of groundwater contamination. It is therefore necessary to develop a methodology enabling us to map the vulnerable zones of the Abidjan Aquifer.

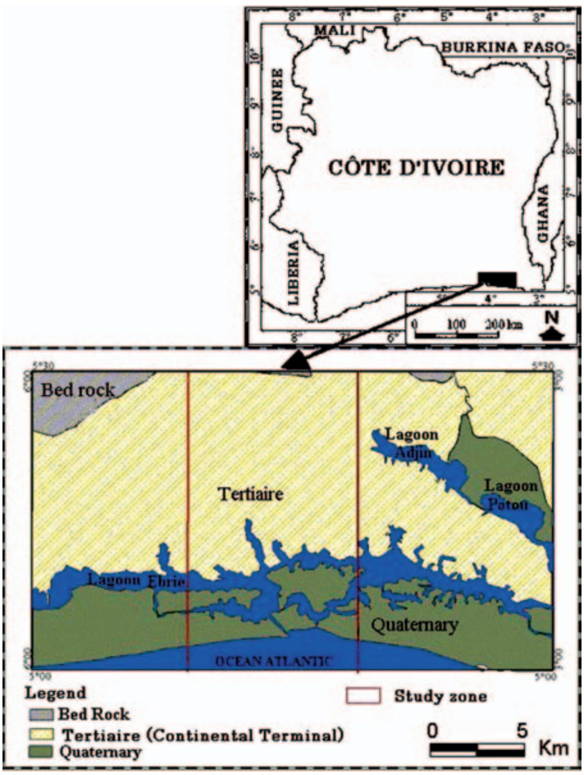

Figure 9.1. Location of study zone.

2 METHODOLOGY

The vulnerability of an aquifer to contamination quantifies the feasibility of its pollution potential. This pollution can be generated by either a point or a diffuse source (Mohamed, 2001). The accuracy of its evaluation depends primarily on the nature, quantity and reliability of the data used.

The term 'assessment of vulnerability' of the aquifers implies two levels of evaluation (Murat *et al.*, 2000):

– the intrinsic vulnerability, which considers the natural physical conditions of the aquifer; the DRASTIC method is typical in this case;
– the specific vulnerability, which utilises not only the natural parameters, but also the properties of the contaminant.

Although several techniques allow the evaluation of the vulnerability of aquifers (DRASTIC, GOD, etc.), the development of another approach to the evaluation of vulnerability within the framework of this study would be appropriate (Kouamé, 2003). This method takes into account the external and intrinsic parameters of the aquifer, inspired by Brian *et al.* (1998).

2.1 *Data and materials*

Our approach included geological and hydrogeologic data from the Abidjan Aquifer. The software used for data processing is MapInfo 5.0 for digitalisation and ArcView 3.2 for the combination and synthesis of maps.

For the development of the vulnerability map of the Abidjan Aquifer, we followed a method allowing us to set up a consequent database for the appropriate requests.

2.2 *Components of vulnerability*

The vulnerability of the aquifer to contamination is regulated by several parameters, such as the natural protection of the aquifer, the type of aquifer, the thickness of the unsaturated zone, importance of the groundwater resource and the intensity of the contribution in polluting products (Guillemin and Roux, 1991). For this study, internal and external parameters are identified and applied to the aquifer of significance. These parameters include:

– Slope.
– Thickness of the unsaturated zone.
– Capacity of infiltration.
– The rate of connection to the drainage collection network.
– Density of population.
– Occupation of the ground.
– Rainfall.

Having only one measuring site for rainfall in the zone of study, located at the international airport of Abidjan (Port-Bouët), we assumed that rainfall is uniform over the whole town of Abidjan. However, this parameter was not used in this study.

2.3 *Preparation of the various layers implied in vulnerability*

Each parameter of vulnerability was transformed into a layer, since the method is based on the principle of the superposition and combination of layers. The layers generated include:

- Cover of the slopes: realised starting from the Digital Terrain Model (DTM) generated under ArcView with the topographic map;
- Cover thickness of the unsaturated zone: realised starting from the combination of the topographic map and the piezometric under ArcView;

Table 9.1. Constitution of the database.

Data	Nature of the data	Data description
Piezometry	Vector	Water level
Topography	Vector	Nature of the curves and value of the level lines
Slope	Raster	Gradient of the slopes
Occupation of the ground	Raster	Nature of the activities
Unsaturated zone	Raster	Thickness of the unsaturated zone
Connection to the sewage network	Raster	Percentage of connection to the network of sewage
Density of population	Raster	Value of the density of population per commune
Capacity of infiltration	Raster	Description of the zones and capacity

- Cover of the capacity of infiltration: realised starting from the geotechnical map defining the capacity of infiltration of the various types of grounds met under MapInfo transferred under ArcView;
- Cover of the rates of connection to the sewage network: generated using the rates of connection to the network by commune under ArcView;
- Cover of the density of population: realised starting from the population by commune and their surface under ArcView;
- Coverage of landuse: generated starting from the sources of pollution identified under ArcView.

All the layers generated under the ArcView environment are related to table's assignees, where all information is contained. They constitute the database installation for the development of the vulnerability map of the Abidjan Aquifer. The database includes the data of Table 9.1.

2.4 *Classification and coding of the layers*

The various parameters were gathered in 4 classes:

- Class 1: very low
- Class 2: low
- Class 3: medium
- Class 4: high

With each class, we have then affected an 'indice', which is a function of the importance of the class in the process of the contamination of groundwater. The purpose of classification and coding is to facilitate the analysis of the requests. Table 9.2 indicates the classification and the coding of the various parameters of vulnerability. The assignment of indices 1, 10 and 100 was adopted to facilitate the distinction after overlaying the layers. Indeed, after the overlaying, the distinction is facilitated in this case, because it is easily seen that for example, the class '110' comes from the overlaying of the classes '10' and '100'. The numbers 1, 10, and 100 are assigned to the low classes and 4, 40, and 400 correspond to the high classes. The operation of assignment of the indices to the classes of the various maps was carried out with the tool 'Reclassify' of the module 'Spatial Analyst' of ArcView.

3 DESIGN OF A CONCEPTUAL MODEL OF DATA (MCD)

To arrive at the development of the vulnerability map, we initially carried out two intermediate maps, as elaborated in Figure 9.2.

Table 9.2. System of classification and coding for the development of the vulnerability map of the Abidjan Aquifer.

Parameters of vulnerability	Components of the various parameters	Vulnerability classification	Coding of classes
Capacity of infiltration	Null	Very low	100
	Medium	Low	200
	Good	Medium	300
	Very good	High	400
	>66 m	Very low	10
Thickness of the unsaturated zone	37–66 m	Low	20
	8–37 m	Medium	30
	<8 m	High	40
Rates of connected to the network of sewage	29–59%	Very low	10
	15–27%	Low	20
	3–14%	Medium	30
	$<2\%$	High	40
	$<3{,}000$ Hbts/km^2	Very low	1
Density of population	3,000–5,000	Low	2
	5,000–10,000	Medium	3
	$>10{,}000$ Hbts/km^2	High	4
	Space frame	Very low	1
	Industrial plantations	Low	2
Land use	Industrial parks	Medium	3
	Scrap yard of Adjamé		
	Storage of the household of Abidjan cemetery	High	4
Slope	$11 < i > 15\%$	Very low	100
	$7 < i < 11\%$	Low	200
	$4 < i < 7\%$	Medium	300
	$i < 4\%$	High	400

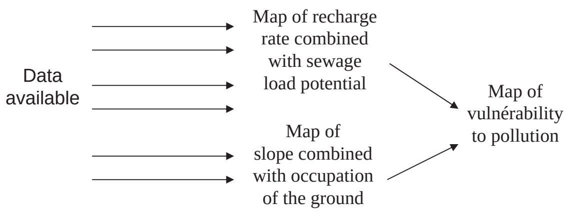

Figure 9.2. Evolution of the various parameters in the vulnerability map.

- Map of recharge rate combined with sewage load potential: This map takes into account parameters such as: capacity of infiltration and the unsaturated zone, and the rate of connection and density of population;
- Map of slope combined with occupation of the ground: This map takes into account the parameters of occupation of the ground (sources of pollution) and the slope.

The crossing of the two maps made it possible to work out the vulnerability map of the Abidjan Aquifer. The algorithm of methodology is summarized in the Conceptual Model of Data (MCD) (Fig. 9.3).

All the crossings of the layers were carried out with a tool called 'Map Calculator' of the module 'Spatial Analyst' of ArcView. The layers were crossed two by two to facilitate the interpretation

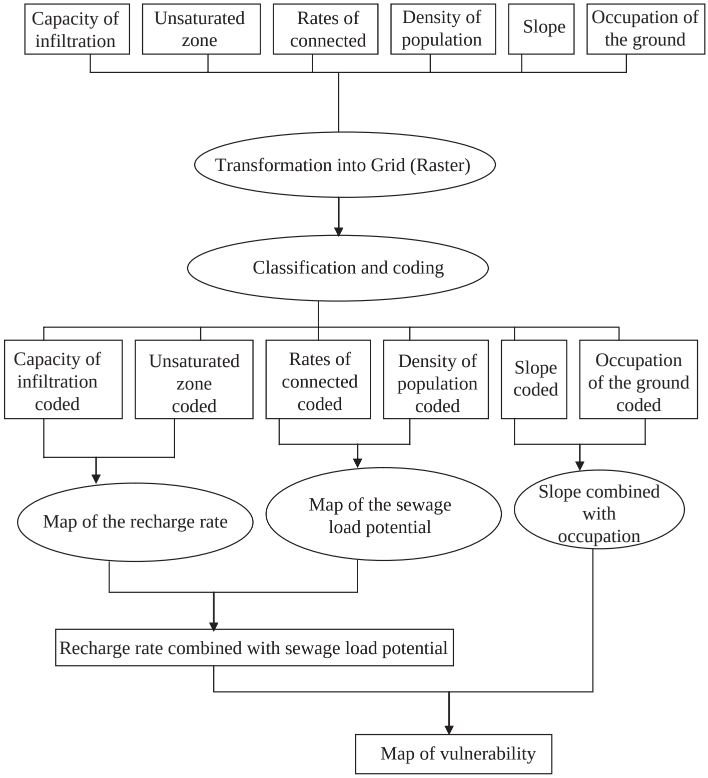

Figure 9.3. Conceptual model of the data for the realisation of Abidjan Aquifer vulnerability map.

of the resulting map. The choice of the two maps to be crossed was carried out by taking into account the concept of aquifer vulnerability.

4 RESULTS AND DISCUSSIONS

4.1 *Recharge rate combined with sewage load potential*

This map highlights the zones of the aquifer, translating the predisposition of the Abidjan Aquifer to contamination.

Two intermediate maps were established to facilitate the development of the sensitivity map to contamination for the Abidjan Aquifer.

The map showing the recharge rate is related to the combination of two parameters (Table 9.3): the thickness of the unsaturated zone and infiltration capacity. This combination is based on the assumption according to which, if for example a zone located on 'good' infiltration capacity with an unsaturated zone 'low' (shallow), then the risk with the contamination is 'high'.

The second map of sewage load potential results from the combination of two other parameters (Table 9.4): the rate of the connection to the network of sewage and population density. This combination is based on the assumption according to which, if for example the common rate of

Table 9.3. Recharge rate.

	Thickness			
Infiltration	10	20	30	40
100	110	120	130	140
200	210	220	230	240
300	310	320	330	340
400	410	420	430	440

Table 9.4. Sewage load potential.

	Density of population			
Rates of connected	1	2	3	4
10	11	12	13	14
20	21	22	23	24
30	31	32	33	34
40	41	42	43	44

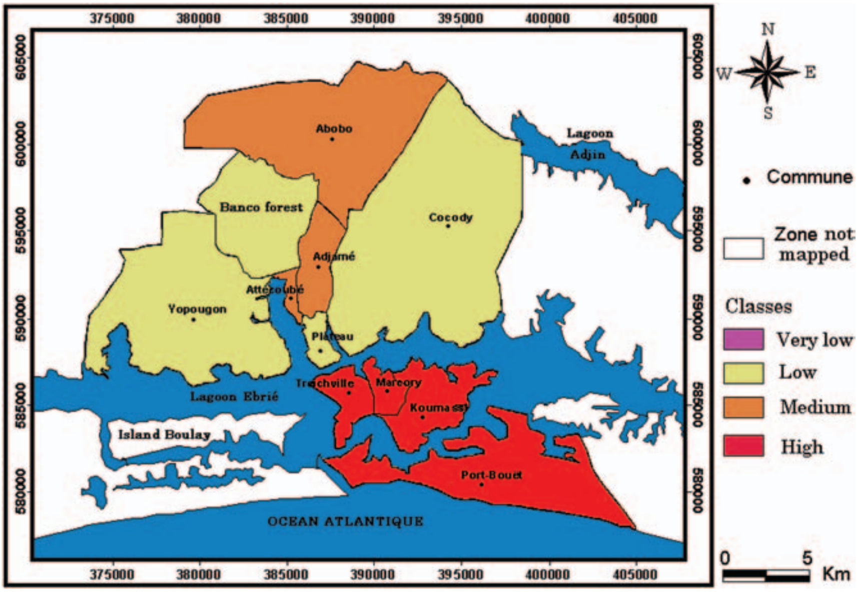

Figure 9.4. Map of the recharge rate combined with the sewage load potential.

connection is 'low' with a population density 'high', then the potential risk of contamination will be 'high'.

After all these maps were overlain, the result in Figure 9.4 was obtained. This map is characterised by the presence of four classes, varying from 'Very low' to 'High'.

Class 1, which characterises a 'Very low' risk of pollution, does not appear on the map. There are thus three great major classes which apply to this area.

Class 2, which characterises a ‘Low’ risk of pollution, is the prevalent class. Indeed, the zones of this class have a low capacity for infiltration and the thickness of the unsaturated zone is also significant (between 8 and 37 m). This class includes/understands the communes of Yopougon, the Plateau, Cocody and the forest of the Banco.

Class 3, which characterises a ‘Medium’ risk of pollution, features the communes of Abobo, Adjamé and Attécoubé. These zones, located in the northern part and centre zone of study, have a limited connection to the network of connection to the network of sewage (14% for Abobo, 23% for Adjamé and 0% for Attécoubé), a high density of population and a small unsaturated zone (between 8 and 37 m).

Class 4 characterises a ‘High’ risk of pollution. This class includes the communes of Treichville, Marcory, Koumassi and Port-Bouët. It is the southern part of the agglomeration of Abidjan that is also of greatest concern. Indeed, the capacity of infiltration in these zones is good and the thickness of the unsaturated zone very small (between 1 and 11 m). So pollution on the surface could reach the aquifer easily and one also notes a system of individual sewage developed with a population density that is high (7906 for Treichville, 9842 for Marcory and 15 918 for Koumassi), except Port-Bouët (2304).

4.2 *Slope combined with occupation of the grounds*

This map, related to the parameters of slope and occupation of the ground, highlight the zones on which activities could have a deteriorating effect on the groundwater quality. The combination of these two parameters delivers the map of the Abidjan Aquifer (Fig. 9.5).

From this, four (4) classes are obtained.

- Class 1, which characterizes a ‘very low’ risk of pollution, relates to the only zone of the Banco Forest. The activities in this forest do not have an impact on the quality of groundwater.
- Class 2 characterises a ‘Low’ risk of pollution. It occupies the totality of the zone of study. This means that urbanisation in itself is not a threat for groundwater, but it must be accompanied by sewerage and sanitation infrastructure.
- Class 3, which characterizes a ‘Medium’ risk of pollution, is represented by spaces corresponding to the industrial plantations located in the Cocody Commune.
- Class 4, which characterises a ‘High’ risk of pollution, represents zones located in the communes of Yopougon, Adjamé, Abobo, Cocody, Koumassi and Port-Bouët.

They are the breakage of Adjamé, the cemeteries of Yopougon, Adjamé, of Abobo, the storage of the Abidjan household with Cocody and the industrial parks of Yopougon, Koumassi and Port-Bouët. These areas constitute threats to groundwater.

4.3 *Vulnerability map for pollution of the Abidjan Aquifer*

The vulnerability map for pollution of the Abidjan Aquifer is obtained with the combination of the two preceding maps (Fig. 9.6). This map made it possible to target the zones at risk. Four classes were determined:

- Class 1 characterizes a ‘Very low’ degree of vulnerability to pollution. This class only appears close to the Banco Forest.
- Class 2 characterizes a ‘Low’ degree of vulnerability to pollution. It contains the zones with a ‘Medium’ sensitivity to pollution, but no activity on the surface involves contamination of subsoil water (this is the case in the Banco Forest). This zone must be protected and managed, especially with regard to future installation.
- Class 3, which characterizes ‘Medium’ vulnerability to pollution, contains the zones whose risk of contamination is high. The zones concerned are the communes of Yopougon, Plateau and Cocody. The zones with ‘Medium’ vulnerability also coincide with the zones of ‘High’ sensitivity. This means that boreholes belonging to these zones must profit from the perimeters of protection. The extension of these communes must also take account the vulnerability of spaces. This vulnerability may be due to the rate of connection, which is average, and to the average population.

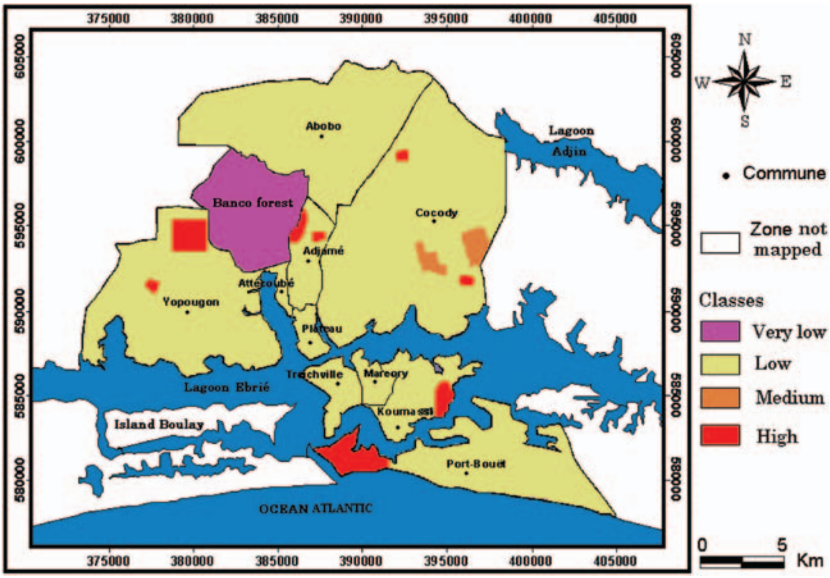

Figure 9.5. Map of the slope combined with landuse.

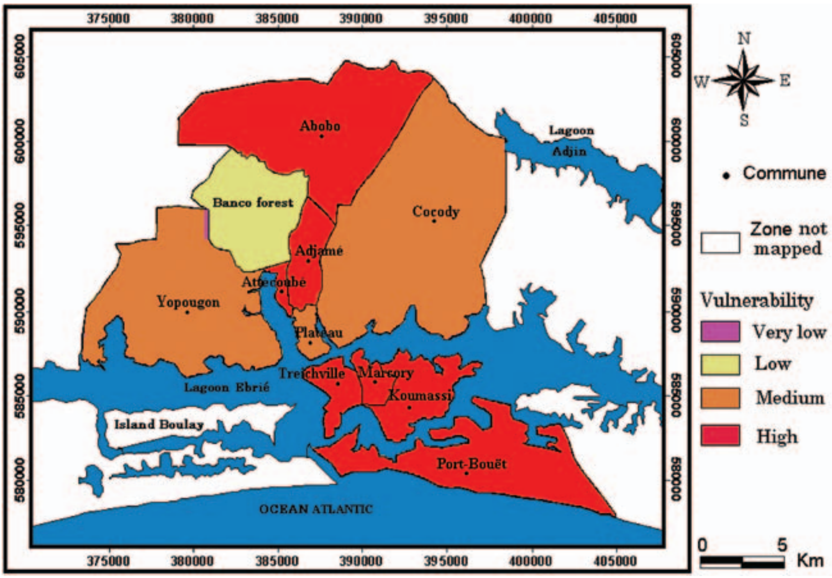

Figure 9.6. Vulnerability map of the Abidjan Aquifer.

– Class 4 characterizes 'High' vulnerability to pollution. In these zones, the risk of groundwater contamination is very high. Zones concerned are Port-Bouët, Koumassi, Treichville, Attécoubé, Adjamé and Abobo. In these communes, the unsaturated zone is small (boreholes are not deep) and the degree of connection to the network of sewage is very weak with a high population. These conditions show that the south of Abidjan is a very vulnerable zone, just as the north (Abobo), which is the recharge zone of the Abidjan Aquifer. The north of the study zone will profit from a very particular follow-up, with suggested best land use.

4.4 *Validation of the vulnerability map*

The new approach to the cartography of the vulnerability zones of the Abidjan Aquifer reveals two (2) main classes:

1. Class with 'Medium' vulnerability, including the communes located in the west (Yopougon), the centre (Plateau) and the east (Cocody). The presence of the Cocody Commune in this class may be due to the weak rate of connection to the network of sewage (46%). Indeed, we estimate that the Cocody Commune has almost all its dwellings connected to the collective network of sewage. So for more reliability on the degree of vulnerability of this commune, the data on the rate of connection must be brought up to date;
2. A second class of 'High' vulnerability, including the communes located in the north (Abobo), the centre (Adjamé, Attécoubé) and the south (Treichville, Marcory, Koumassi and Port-Bouët). In this 'High' class, we distinguish:
   – The highly affected zones (Adjamé and Attécoubé) including those in the centre;
   – the zone not reached yet (Abobo, in the north), but which is vulnerable, making it necessary to protect by adequate policies.

These degrees of contamination can be explained by insufficiency, the failure or absence of a sewerage system in the majority of the popular districts of the communes of Abidjan, where thousands of inhabitants unceasingly inject effluents in septic tanks and French drains dug in the aquifer of Abidjan (Aghui and Biémi, 1984). Chemical analyses on the water of the Abidjan Aquifer were carried out in this study. The analyses focused on the nitrogen rate (NO_3-N) in water of certain boreholes in the aquifer. A map of nitrate contents was produced in 2001 (Fig. 9.7).

On this map, four classes of nitrogen (NO_3-N) are observed. In the south, centre, west and east the nitrogen rate exceeds the international standard of 50 mg/l (European standard). The zones with a rate higher than 50 mg/l NO_3-N are localised near the lagoon. In these zones, the water level is shallow. The high nitrogen concentration could originate form anthropogenic contamination, because of the proximity of water to these zones. This part of the nitrate map corresponds to High vulnerability (Fig. 9.7), with a low thickness of the unsaturated zone (between 1 and 11 m). We also note a progression of this high nitrogen value from the south to the north, east and west, which are respectively High and Medium vulnerability zones.

Studies formerly carried out by Kouadio *et al.* (1998) on the quality control of water in the Abidjan Aquifer during the period from 1993 to 1996 revealed an increasing contamination of nitrogen from the south to the north, i.e. towards peripheral urbanisation. This same observation was made by Yacoub (1999). The aim of these water analyses was to identify the zones presenting risks of contamination for the Abidjan Aquifer. The method used in this study enabled us to map the vulnerable zones of the Abidjan Aquifer and the results conform with those released by the studies on the chemical analysis of water from this aquifer.

5 CONCLUSIONS

The new cartography approach to vulnerability made it possible to obtain reliable results, based on good data. These results reveal that the degree of vulnerability to pollution of the Abidjan Aquifer

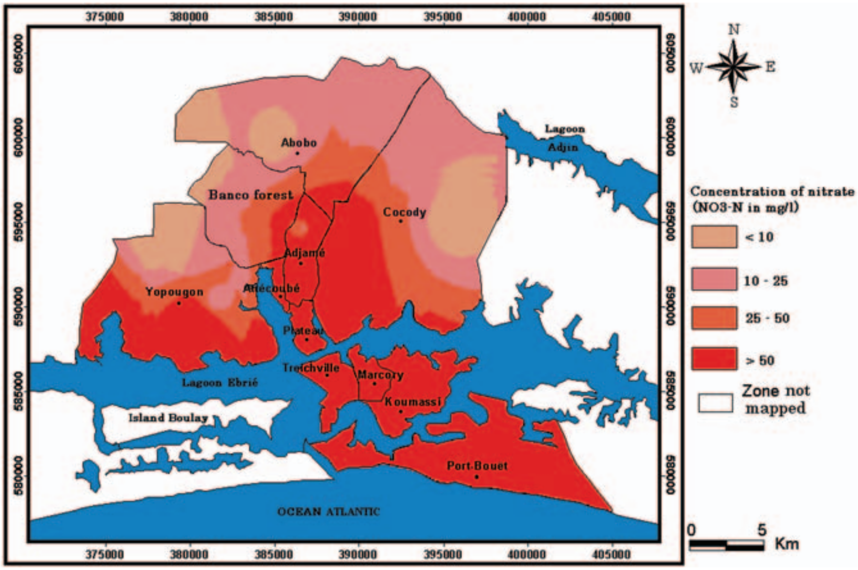

Figure 9.7. Map of nitrates in 2001.

varies from low to very high (the very low one being unimportant on the map). The 'Low' vulnerability is represented by the Banco Forest value.

The Abidjan Aquifer presents two highly vulnerable zones. A zone with 'Medium' vulnerability is located at the east and west side of the zone of study and another with 'High' vulnerability located at the south and north in the part of recharge. This new approach can be used in a comparative cartography study of the vulnerable zones, with the traditional methods such DRASTIC, etc. This new method is a synthesis of the cartography methods of intrinsic and specific vulnerability.

ACKNOWLEDGEMENTS

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REFERENCES

Aghui N. et Biémi J. 1984. Géologie et hydrogéologie des nappes de la région d'Abidjan et risques de contamination. Annales de l'Université Nationale de Côte d'Ivoire, série c, tome 20, pp. 331–347.

Brian M., Litvak G. et Ahmed K. 1998. Urban groundwater protection and management: lessons from 2 developing city case studies in Bangladesh and Kyrghystan. Groundwater protection and management for developing cities, BGS, DFID, 25p.

Guillemin C. et Roux J.C. 1991. Pollution des eaux souterraines en France. Bilan des connaissances, impacts et moyens de prévention. Manuels et méthodes, n°23, éd. par BRGM, 262 p.

INS 2001. Recensement Général de la Population et de l'Habitation (RGPH) 1998. Données socio-démographiques et économiques des localités, résultats définifs par localités, région des lagunes, vol.III, tome 1, 43 p.

Kouadio L., Abdoulaye S., Jourda P., Loba M. et Rambaud A. 1998. Conséquences de la pollution urbaine sur la distribution d'eau d'alimentation publique à Abidjan. Cahiers de l'Association Scientifique Européenne pour l'Eau et la Santé, vol.3, n°1, pp. 61–75.

Kouamé K. J. 2003. Apports d'un Système d'Information Géographique à la réalisation de la carte de vulnérabilité de la nappe du Continental Terminal au niveau de l'agglomération d'Abidjan. Mémoire de DEA, Université de Cocody (Abidjan) 64 p.

Mohamed R. M. 2001. Evaluation et cartographie de la vulnérabilité à la pollution de l'aquifère alluvionnaire de la plaine d'El Madher, Nord-Est algérien, selon la méthode DRASTIC. Sciences et changement planétaires/sécheresse, vol. 12, n°2, pp. 95–101.

Murat V., Paradis D., Savard M. M., Nastev M., Bourque E., Hamel A., Lefebvre R. et Martel R. 2003. Vulnérabilité à la nappe des aquifères fracturés du sud-ouest du Québec: Evaluation par les méthodes DRASTIC et GOD. Ressources naturelles Canada, Commission Géologique, 16 p.

Yacoub I. 1999. Analyse de l'évolution qualitative et quantitative des ressources en eaux souterraines du grand Abidjan. Mémoire de DEA, Université d'Abobo-Adjamé, 58 p.

10

Vulnerability assessment of the Abidjan Quaternary Aquifer using the DRASTIC method

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ABSTRACT: This study presents an assessment of a selected quaternary's groundwater vulnerability to pollution. The area is situated in the District of Abidjan with the target area the Canal de Vridi to Grand-Bassam. The approach used for the DRASTIC method is based on the characterisation of several parameters. This provides an indication of the sensitivity of the groundwater resource to pollutants based on its physical properties. Moreover, the physico-chemical parameters of the groundwater (pH, electric conductivity, temperature, nitrate, nitrite, saltiness) have been evaluated to make a comparison between the results of the DRASTIC method and the current pollution status. The results show that the zones affected by high vulnerability are situated in the townships of Grand-Bassam, Port-Bouët, Vridi, Koumassi, Treichville (the southern and eastern parts) and the villages of Anani, Modeste and Azuretti. Chemical analyses also reveal that these zones generally correspond to the sectors where nitrate and nitrite contents are the most elevated, with concentrations reaching 690 mg/L (NO_2^-) and 3493 mg/L (NO_3^-). This shows that the DRASTIC method provides an accurate indication of the vulnerability of aquifers.

1 INTRODUCTION

Groundwater pollution represents a major environmental problem. The progressive degradation of the quality of this water could cause it to be unfit for domestic use over the medium- to long term.

In the area around Abidjan, the coastal sedimentary basin overlies three aquifers known as the Continental Terminal's Aquifer, the Cretaceous Aquifer and the Quaternary Aquifer. The vulnerability of the latter to pollution was determined by former studies (Aghui and Biémi, 1984; Jourda, 1987; Kouadio, 1997). These studies found that this alluvial aquifer has a very shallow water table, making it particularly susceptible to pollutants from various origins. However, an identification of the high-risk sectors has not been done to date. This aspect is now very important, since groundwater is abstracted for drinking and other purposes. In addition, the studies relating to groundwater sensitivity to pollution in similar settings such as the groundwater of Haouz de Marrakech, in Morocco (Sinan, Maslouhi and Razack, 2003) and the El Madher plain in Algeria (Menani, 2001) have shown that the awareness about principal zones of risk should play a key role in land-use planning and the implementation of protective measures in these areas.

In this area, with its high population growth, increased occupation and industrialisation the medical and social conditions are often very poor. The identification of the principal zones of risk in these aquifers has therefore become of paramount importance.

The purpose of the present study is to evaluate and map the vulnerability of the aquifers in the Canal de Vridi – Grand-Bassam area. From this, for the first time, the zones of vulnerability using

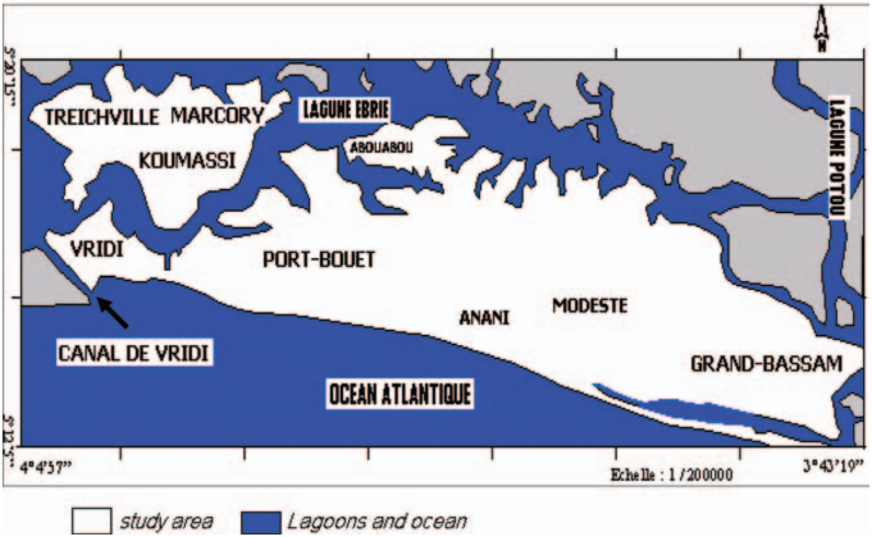

Figure 10.1. Location of the study area.

the Drastic index will be determined. Included is a study of the physico-chemical quality of the groundwater and a map of the vulnerability risks. From this a comparison between the conclusions of the vulnerability assessment method and quality of the groundwater as measured in the field can be obtained.

2 STUDY AREA

The study area lies in the southern part of the lake area around Abidjan, between the latitudes 5°12′5″ N and 5°20′15″ N and longitudes 4°4′57″ W and 3°43′19″ W. It includes the communes of Port-Bouët, Koumassi, Marcory, Grand-Bassam, Treichville and the villages of Abouabou, Anani, Modeste and Azuretti (Fig. 10.1). The population is estimated at 881,000.

The study area is located in the sedimentary basin of the Côte d'Ivoire. It is characterised by a generally flat relief. Various geological formations of Quaternary age are found in the area. These include clayey sands reaching 15 to 30 m thickness, mud and sands of the fluvio-lagunal depressions, and marine sands reaching up to 45 m thickness (Delor *et al.*, 1992). It is a sandy aquifer that receives good recharge. The permeability of the surface deposits of this aquifer system ranges between 10^{-4}ms^{-1} and 10^{-3}ms^{-1} for the clayey sands and 10^{-3}ms^{-1} for marine sands. Groundwater flow in the aquifer is rapid and can be regarded as relatively uniform.

Many domestic wells utilise this resource, due to the shallow piezometric levels to satisfy multiple uses, including drinking water for certain populations of the area.

3 METHODOLOGY

3.1 *Vulnerability assessment*

The vulnerability assessment was done using the DRASTIC method developed in the USA (Aller *et al.*, 1987). It is based on the estimate of seven (7) parameters relating to recharge, soil (slope and nature), and the unsaturated and saturated zones. It concerns the depth to water, the total recharge,

Table 10.1. DRASTIC parameters with their weights.

Parameter	Weight
Depth to water	5
Net recharge	4
Aquifer media	3
Soil media	2
Topography	1
Impact of the unsaturated zone	5
Hydraulic conductivity	3

Table 10.2. Ratings for depth of water and topographic slope.

Depth of water (m)		Topographic slope (%)	
Interval	Rating	Interval	Rating
0–1,5	10	0–2,0	10
1,5–4,5	9	2,0–6,0	9
4,5–9,5	7	6,0–12,0	5
9,5–16	5	12,0–18,0	3
16–24	3	⩾18,0	1
24–32	2		
>32	1		

aquifer media, soil media, topography, the impact of the unsaturated zone (lithology and thickness) and hydraulic conductivity of the aquifer.

Each parameter has a fixed weight that reflects its relative importance to vulnerability (Table 10.1). The most significant parameters (depth of water and impact of the unsaturated zone) have weights of 5 and the least significant (topography) a weight of 1. In addition, a value between 1 and 10 was assigned to each parameter, depending on local conditions. As an example, Table 10.2 represents the system of rating for the 'depth of water' and 'topography'. High values represent higher vulnerability. Next, the local index of vulnerability (Id) is computed through multiplication of the value attributed to each parameter by its relative weight, and adding up all seven products.

$$\mathrm{Id} = (\mathrm{Dc} \times \mathrm{Dp}) + (\mathrm{Rc} \times \mathrm{Rp}) + (\mathrm{Ac} \times \mathrm{Ap}) + (\mathrm{Sc} \times \mathrm{Sp}) + (\mathrm{Tc} \times \mathrm{Tp}) + (\mathrm{Ic} \times \mathrm{Ip}) + (\mathrm{Cc} \times \mathrm{Cp})$$

D, R, A, S, T, I and *C* represent the DRASTIC parameters, while *c* and *p*: represent respectively the rating and the weight attributed to each parameter.

For purposes of interpretation, we subdivided the possible values of the DRASTIC index into four classes of vulnerability, according to the range of indices defined by Lobo, Novo and Oliveira (2004):

– DRASTIC indices above 199 were considered to be of very high vulnerability;
– DRASTIC indices ranging from 160 to 199 were considered to be of high vulnerability;
– DRASTIC indices ranging from 120 to 160 were considered to be of medium vulnerability;
– DRASTIC indices below 120 were considered to be of low vulnerability.

3.2 *Physicochemical quality of Quaternary's groundwater*

The physicochemical characteristics were studied with the aim to compare the results of the DRASTIC method with the contamination observed in the field. The physical parameters taken into account are the temperature, conductivity and pH. These were measured *in situ*.

As for the chemical parameters, the nitrate and the nitrite values where used as pollution indicators. The methods of analysis used included reduction to cadmium (for nitrates) and the diazotisation method (for nitrites).

Water samples were collected from 59 domestic boreholes. The samples were directly collected into small glass bottles and preserved at 4°C for a maximum of 48 hours.

4 RESULTS AND DISCUSSION

4.1 *DRASTIC vulnerability mapping*

The vulnerability maps of the Quaternary's aquifer results from the combination of the seven (7) sets of themes layers (Examples are given in Fig. 10.2).

Only two of the parameter maps are shown here, since the majority correspond closely to the aquifer media zones or a function of these geologic zones. The topography is largely similar and the hydraulic conductivity falls within the same zones.

In addition, it enables the identification of the principal high risk zones, which are related to high indices. In Grand-Bassam, Koumassi and at some places of Port-Bouët and Vridi, the DRASTIC index exceeds the value of 200. In the villages of Modeste, Anani, Azuretti and the communes of Marcory, Treichville, Port-Bouët and Vridi it is in the range 180–199. These values indicate a high vulnerability to pollution in these localities. These zones are underlain by sands and are characterised by high aquifer permeability. In addition, the water level is very shallow. The majority of the sectors of the Marcory Commune and the Abouabou Village have an index ranging between 120 and 160. This confers a medium vulnerability. The areas are located in the clayey sand zones. Where the fluvio-lagunal depressions occur, the values of the index are lower than 120. These represent low vulnerability areas. This is associated with the compact structure and fine granulometry of sandy clays and mud, which results in far lower permeability and a lower capacity for infiltration.

The results of this vulnerability assessment of the groundwater are in conformity with work from former studies. According to Isabel *et al.* (1990), the minimum and the maximum level of the DRASTIC index of 23 and 226 respectively are seldom reached. According to these authors, the calculated indices usually range from 50 to 200, which is in line with the findings of the current study.

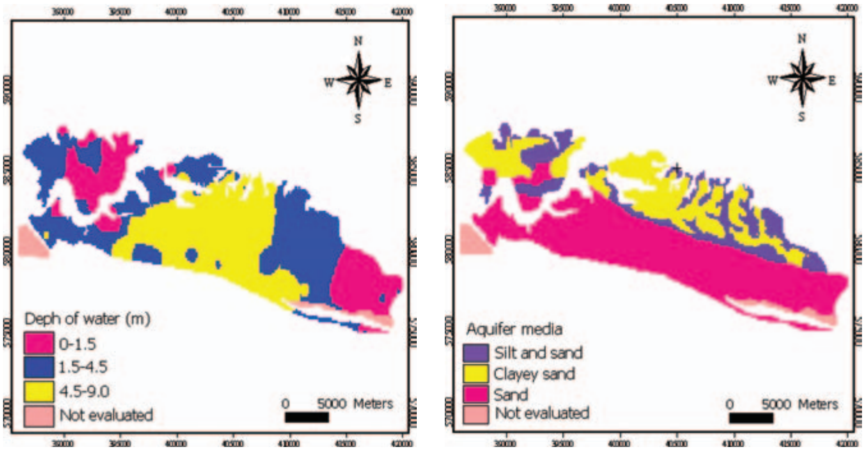

Figure 10.2. Map of the depth to groundwater and aquifer media.

4.2 *Physicochemical characteristics of the groundwater from the Quaternary Aquifer*

The groundwater in the area has an electric conductivity that varies from 44.5 µS/cm to 4360 µS/cm. The samples show that a relatively high average temperature of 29.9°C with pH-values ranging from 5.34 to 7.24 (Table 10.3). The nitrate content of the water samples vary from 3.5 to 690 mg/L (Fig. 10.4). The averages values vary from 207 mg/L in the Treichville/Marcory/Koumassi area, 114 mg/l in Port-Bouët/Vridi/Abouabou/Anani and 36 mg/L in Modeste/Grand-Bassam. Of the 59 boreholes, 76 per cent have nitrate concentrations higher than the WHO maximum permissible limit for drinking water of 50 mg/L nitrate. The highest values recorded are: 353 mg/l in Treichville; 250 and 690 mg/L in Marcory; 260 and 328 mg/L in Koumassi; 220, 250 and 390 mg/L in Port-Bouët and Vridi; 280 mg/L in Grand-Bassam.

The nitrite contents vary between 0.0071 mg/L and 3.493 mg/L. The majority of the boreholes have values in excess of drinking water limits, especially in Treichville, Marcory, Koumassi and Port-Bouët. In Modeste and Anani, the nitrite values are generally lower and within acceptable limits. The nitrite and nitrate distribution therefore follow roughly the same trends.

These high concentrations are according to Biémi (1992) a strong indication that the groundwater in the areas is affected by organic pollution, probably from on-site sanitation.

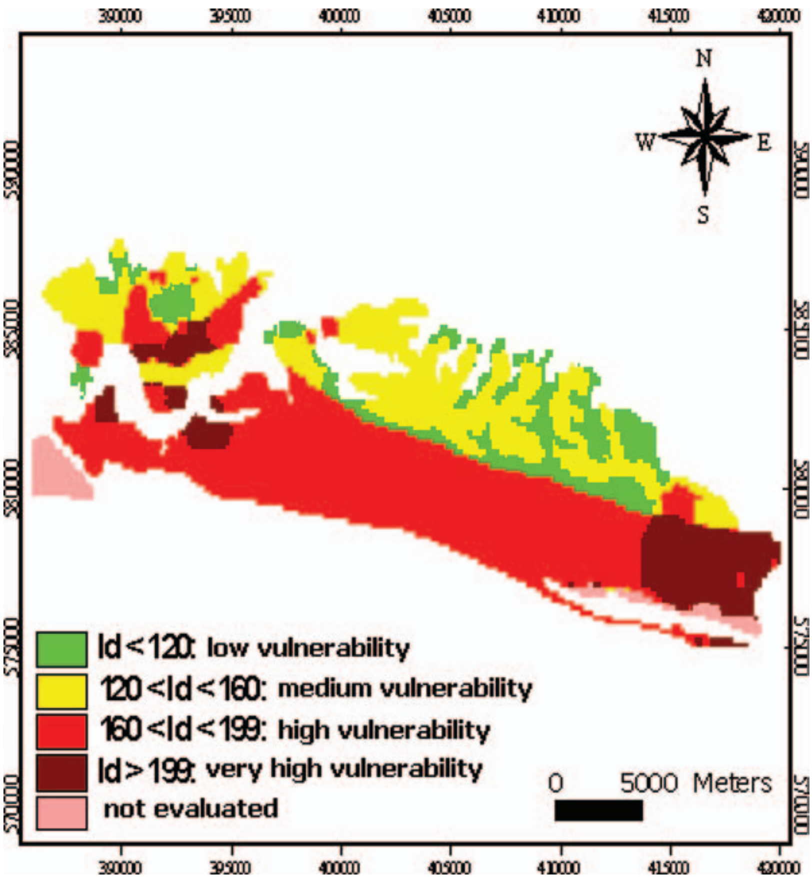

Figure 10.3. Map of DRASTIC groundwater vulnerability.

Table 10.3. Physicochemical parameters of the Quaternary's groundwater.

WELL	X	Y	pH	Temp. (°C)	Cond. (µS/cm)	Sal (‰)	NP (m)	NO_3^- (mg/L)	NO_2^- (mg/L)
B1	389 280.00	585 800.00	5,93	28,4	429	0,2	3,5	92.09	0.107
T1	388 280.00	586 713.00	5,61	28,4	491	0,2	2,9	116.93	0.072
T2	388 333.33	585 280.00	5,4	28,5	876	0,4	3,7	353.13	0.074
T3	387 500.00	586 100.00	6,28	28,3	1001	0,5	2,25	74.75	0.224
T4	388 526.10	584 457.33	6,48	27,9	998	0,5	2,64	176.45	0.535
M 1	389 840.00	587 580.00	6,93	27	1427	0,7	1,4	75	0.214
M 2	389 806.00	586 486.67	6,18	28,9	1150	0,5	3,07	281	0.405
M 3	391 740.00	586 006.67	6,84	28,8	1925	0,9	3,74	250	0.363
M 4	391 766.00	585 040.00	5,66	29,4	589	0,3	1,27	98.7	0.099
M 5	392 380.00	586 006.67	5,95	29	810	0,4	4,34	159.6	0.519
M 6	390 740.00	585 973.33	5,71	28	916	0,4	1,93	196	0.168
M 7	390 806.00	585 426.67	7,24	27,9	1299	0,6	1,4	123.2	0.604
M 8	390 866.67	584 400.00	6,92	28,9	4260	2,1	1,78	690	3.48
K 1	391 714.29	586 500.51	6,12	29,3	1057	0,5	1,5	179	0.271
K 2	392 220.00	587 100.00	7,17	28,7	1491	0,7	1,2	126	0.19
K 3	393 633.33	586 260.00	6,4	28,8	1286	0,6	1,62	164	0.766
K 4	393 566.67	585 326.67	7	28,5	495	0,2	2,04	196	0.084
K 5	394 680.00	584 813.33	6,86	28,4	1266	0,6	1	260	0.304
BO 1	393 620.00	583 906.67	6,55	28,1	1087	0,5	2,27	328	0.422
P1	393 617.69	582 000.00	7,25	29,2	564	0,2	1,13	63	0.16
P2	394 620.00	581 646.67	6,31	27,8	410	0,2	0,8	133	0.581
P3	396 180.00	581 460.00	6,46	29,6	356	0,2	6,54	73	0.497
P4	395 100.00	580 306.60	6,73	29,3	408	0,2	7,22	156	0.124
P5	398 493.33	581 500.00	7,2	28,5	645	0,3	7,3	175	0.113
P6	399 446.00	581 687.00	5,73	29,3	531	0,2	6,9	94	0.044
P7	400 666.70	581 793.33	6	29,6	629	0,3	5,26	151	0.047
P8	396 633.17	579 940.00	6,76	29,8	483	0,2	4,65	187	0.637

P9	398 033.20	579 600.00	6,05	29	672	0,3	4,41	250	0.176
P10	399 520.00	579 046.67	6,86	29,1	475	0,2	6,89	109.1	0.024
P11	401 073.40	579 086.80	5,48	29,8	130,1	0,1	5,74	39.8	0.032
P12	401 300.00	579 753.33	6,08	29,5	222	0,1	4,83	41.5	0.049
P13	402 720.00	578 940.00	6,4	29,5	302	0,1	7,8	42.9	0.049
V1	389 400.67	582 500.00	7,02	29,8	4360	2,1	1,47	101	0.303
V2	389 453.00	581 373.33	6,33	30,6	728	0,3	4,19	106	0.273
V3	391 126.67	581 102.00	5,49	29,5	663	0,3	4,97	220	0.05
V4	388 460.00	581 053.33	6,16	29	1329	0,6	5,51	390	0.429
GB1	414 320.00	577 500.00	6,42	28,4	349	0,2	1,02	56	0.021
GB2	415 288.00	576 693.00	6,48	28,2	131,2	0,1	1,21	4.8	0.026
GB3	415 767.00	578 587.00	6,57	28,2	500	0,2	1,9	23.2	0.104
GB4	417 288.00	577 750.00	6,85	29,2	316	0,1	1,55	24.8	0.075
GB5	418 253.00	577 140.00	6,32	28,3	1113	0,5	2,49	280	0.274
GB6	418 627.00	578 240.00	6,56	29,3	365	0,2	1	22.7	0.115
GB7	419 598.00	577 600.00	6,6	28,3	277	0,1	2,46	11	0.028
GB8	418 385.00	576 095.00	6,63	28,8	387	0,2	2,15	23.1	0.145
GB9	418 529.00	577 750.00	6,72	28,3	142,5	0,1	1,68	11.3	0.03
GB10	417 700.00	576 650.00	6,83	29	256	0,1	2,24	18.6	0.084
GB11	416 223.00	575 209.00	7,02	28,7	299	0,1	2,08	40.1	0.0071
AN1	404 600.00	578 380.00	6,45	30,3	218	0,1	7,34	45.5	0.05
AN2	403 633.00	579 200.00	5,34	30,1	198,9	0,1	5,27	56.6	0.013
MO1	408 346.60	577 566.50	6,22	29,4	159	0,1	6,07	3.5	0.02
MO2	409 500.00	577 493.30	5,69	29,2	123	0	6,38	19.3	0.065
MO3	408 060.00	579 196.90	5,52	29,6	78,6	0	4,95	14.9	0.027
MO4	411 426.80	577 640.00	5,85	29,6	44,5	0	5,7	3.5	0.024
MO5	410 246.00	579 442.30	5,65	29,8	65,2	0	5,33	6	0.06
AZ1	413 280.00	575 646.80	5,85	28,6	420	0,2	2,15	14.5	0.043
AZ2	414 620.00	575 545.00	6,79	29,1	450	0,2	4,62	66.5	0.101
AB1	399 542.00	584 917.00	6,95	30	53,2	0	1,35	28.4	0.341
AB2	399 174.00	584 900.00	6,8	28,7	170,3	0,1	0,92	7.4	0.071
AB3	400 179.00	584 500	6,58	29	98,5	0	7,27	35.1	0.048

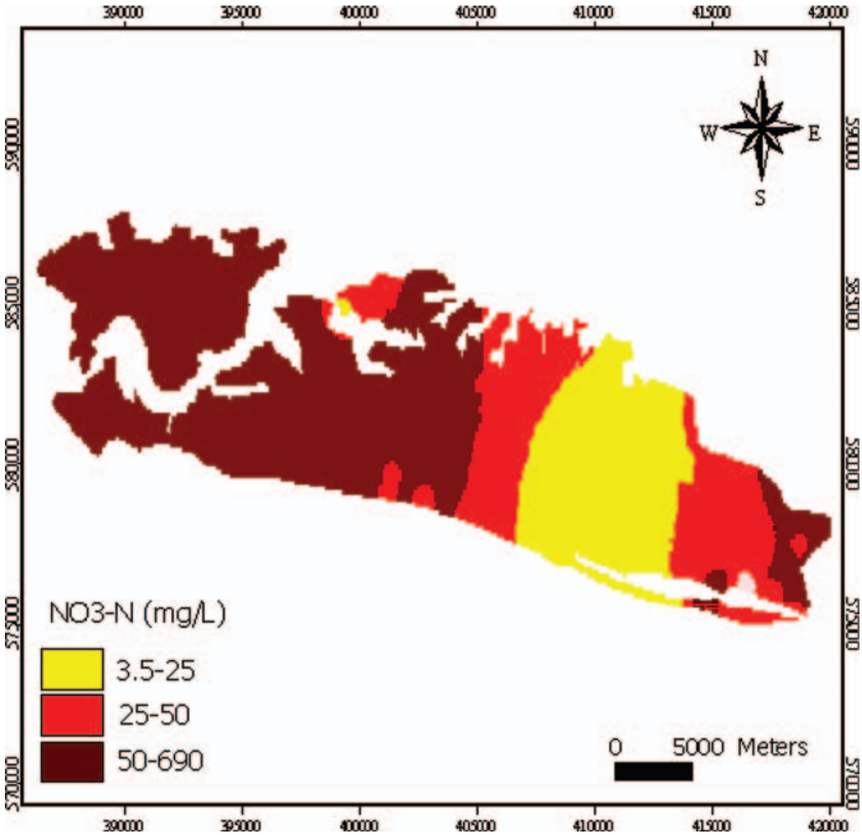

Figure 10.4. Distribution of the nitrate contents of the Quaternary's groundwater.

The apparent sources of this pollution are associated with the manner in which on-site sanitiation is handled, the design of the pit latrines, and the distance separating the boreholes from the latrines. Indeed, all the boreholes in Treichville (B1, T1, T2, T3 and T4) and some in Marcory (M1, M2, M4 and M8), Koumassi (BO1) and Port-Bouët (P1et P6) are located in close proximity to latrines. The water from these boreholes has nitrate contents generally higher than the allowed standard of 50 mg/L. This influence of the latrines on the nitrate contents of the boreholes is confirmed by the results of work by Tandia *et al.*, (1997). These authors showed that there is a significant variation of the load and concentration of nitrate between the closest boreholes and those furthest away from the latrines, which have far lower concentrations.

The second factor at the origin of nitrogenised pollution in the boreholes could relate to the poor siting or location of these boreholes. Indeed, within the immediate surroundings of several of boreholes in the study area, household refuse and other waste was found. Wastewater and grey water is also disposed of directly on the surface or into holes in the ground. These practices are a possible source of several of the nitrogen species in the soils that can lead to groundwater degradation. According to Tandia *et al.* (1997), the nitrate contents of the soil are related *inter alia* to the decomposition of the nitrogen-containing organic waste disposed of on the soils, which leads to the formation of ammonium in the ground, which oxidises to form nitrate.

The groundwater from these boreholes is used by the local population for drinking. The consumption of this water is a serious threat to populations, as the nitrates and the nitrites are at the origin of serious illnesses such as cyanosites cyanosis and especially methyglobinemia.

4.3 *Comparison of vulnerability and pollution maps*

The vulnerability map highlights the sensitive zones of the aquifer to pollution while the distribution maps of the nitrate and nitrite make it possible to visualise the areas of the aquifer affected by pollution. The evaluation of vulnerability revealed that the sand zones are most likely to be polluted. This high vulnerability relates to the entire southern part of the study area, including Vridi to Grand-Bassam, and the communes of Koumassi and Treichville. In these localities, except the villages of Anani and Modeste, the chemical results of analysis show that the groundwater is unsuitable for human consumption. Of importance for the validity of the DRASTIC approach in this area is the fact that the contaminated zones correspond to those where the indices of vulnerability are highest ($160 < Id < 200$). However, the relation between the vulnerable and polluted zones is not completely aligned. Anani and Modeste are areas where the groundwater is not affected by nitrogen-containing contaminants, although these have a high vulnerability. A low population density and consequently little anthropogenic activity in these areas provides the probable explanation for these observations.

5 CONCLUSION

The application of the DRASTIC method to the area of Canal of Vridi – Grand-Bassam allowed the generation of a vulnerability map of the Quaternary Aquifers in this area. The correlation between the zones of high vulnerability with those where cases of nitrogenised pollution is observed, confirms the results of the method. The communes of Grand-Bassam, Port-Bouët, Vridi, Koumassi, Treichville and the Azuretti Village, where high indices have been obtained, also show excessive nitrate and nitrite contents, exceeding the allowed standards for the water intended for human consumption.

The interest of this map thus goes well beyond the visualisation of the high-risk zones of vulnerability. It will provide decision makers with an adequate tool in the interest of the population. Nevertheless, this water is unfortunately used for drinking, despite the risks involved. Vulnerability mapping therefore provides a good indication of where these risks could occur, but does not provide any means with dealing with such situations once they have arisen.

REFERENCES

Aller, L., Bennet, T., Lehr, J.H., Petter, R.J. & Hachett, G. 1987. DRASTIC: a standardized system for evaluating groundwater pollution potential using hydrogeologic settings. U. S. Environmental Protection Agency, 455 p.

Aghui, N. & Biemi, J. 1984. Bassin sédimentaire de Côte d'Ivoire: Géologie et hydrogéologie des nappes de la région d'Abidjan et risque de contamination. Série c (Sciences) tome XX 27 p.

Biemi, J. 1992. Contribution à l'étude géologique, hydrogéologique et par télédétection des bassins versants subsahéliens du socle précambrien d'Afrique de l'ouest: hydrostructurale, hydrodynamique, hydrochimie et isotopie des aquifères discontinus de sillons et aires granitiques de la Haute Marahoué (Côte d'Ivoire). Th. ès-sciences naturelles. Univ. nationale Côte d'Ivoire 424 p, 49 tabl.

Delor, C., Diaby, I., Tastet, J.P., Yao, B., Simeon, Y., Vidal, M. & Dommanget, A. 1992. Notice explicative de la carte Géologique de la Côte d'Ivoire à 1/200000, Feuille Abidjan, Mémoire de la Direction de la Géologie de Côte d'Ivoire, no˚ 3, Abidjan, Côte d'Ivoire, 26 p.

Chalbaoui, M. 2000. Vulnérabilité des nappes superficielles et subaffleurantes du Sud-Ouest tunisien. SECHERESSE, vol 11, no˚ 2, 95–101.

Isabel, D., Gelinas, P. & Aubre, F. 1990. Cartographie de la vulnérabilité des eaux souterraines au Québec. Rev . Sci. Tech Eau, n˚ 23, 255–258.

Jourda, J.R.P. 1987. Contribution à l'étude géologique et hydrogéologique de la région du Grand-Abidjan (Côte d'Ivoire). Th. Doct. 3ieme cycle, 319 p.
Kouadio, B.H. 1997. Quelques aspects de la schématisation hydrogéologique: cas de la nappe d'Abidjan. Mémoire de DEA. Univ.d'Abidjan, 68 p, 30 tabl. 19 fig.
Lobo, P., Novo, E. & Oliveira, M. 2003. Groundwater vulnerability mapping: application to the Minho watershed in northern Portugal's celtic Region; 7 p
Menani, M. 2001. Evaluation et cartographie de la vulnérabilité à la pollution de l'aquifère alluvionnaire de la plaine d'El Madher, Nord-Est algérien, selon la méthode Drastic. Sécheresse, vol 11, no° 2, 95–101.
Sinan, M., Maslouhi, R. & Razack, M. 2003. Utilisation des SIG pour la charactérisation de la vulnérabilité et de la sensibilité à la pollution des nappes d'eau souterraine. Application à la nappe du Haouz de Marrakech, Maroc, 15 p.

11

Groundwater pollution from urban development in Cotonou City, Benin

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ABSTRACT: In the framework of the UNEP/UNESCO project entitled 'Pollution assessment and aquifer vulnerability of big African urban cities', a physico-chemical analysis and cartography of shallow aquifer system vulnerability in the western region of Cotonou, the main metropolis of Benin, was started in 2001. The goal was to prevent/minimize the quality and quantity deterioration of water resources. The results of borehole sample analyses show that the groundwater in the study region is of a sodiumchloride nature and that the average chemical composition is generally still within the internationally recommended levels of potability. However, the vulnerability map developed according to the DRASTIC method for most shallow aquifers indicates that the aquifer system is moderately vulnerable to vulnerable. This result implies that, despite the fact that the current water quality is good, this situation can deteriorate if no measures are taken to protect the aquifer. Finally, the relationship of the elements within the study area in terms of land use, to meet a constant increasing drinking water demand, the drawdown of water levels and the degradation in quality of the shallow groundwater is illustrated in the hydrogeological cross-sections and discussed.

1 INTRODUCTION

The project entitled 'Assessment of pollution status and vulnerability of water supply aquifers of African cities' relates primarily to the question of the vulnerability of aquifers and the need for protecting the quality of groundwater resources on the continent. In the case of Benin, the capital Cotonou City was chosen as a target for the study since its drinking water supply relies exclusively on localised water resources.

The aquifer system lies in the continental plain of the great Allada plateau formations and has been exploited since 1956. There has been a constant increase in groundwater usage, particularly in the last 15 years. This has resulted in seawater intrusion in three areas to the north-east of the aquifer.

Included in the study area are regions of human settlement with the highest current population growth rate in the country (INSAE, 1994; MECCAG and PDPE, 2000, 2002).

This settlement was initially uncontrolled and is now at a stage where urbanisation has reached problematic proportions. If nothing is done to properly classify and protect the aquifer system from both a quantity and quality perspective, this important shallow water resource in southern Benin is under threat of over-exploitation.

With this in mind, the main objectives of this study are:

1. The classification of chemical and bacteriological constituents of shallow water resources.
2. The development of an aquifer vulnerability map for the shallow aquifer.

3. The development of synthesising diagrams of the evolution of the shallow water resource. The results should contribute to the definition of a management strategy for this resource to safeguard its integrity as well as the environment.

1.1 *General context of the study*

The identified zone of the current study covers the intense pumping area and its immediate environs, as shown in Figure 11.1. It spreads to the west of the Godomey Bank (west) of Lake Nokoue and is about 10 km long (N-S direction) and 5 km wide (E-W direction). The metropolitan centre of Cotonou

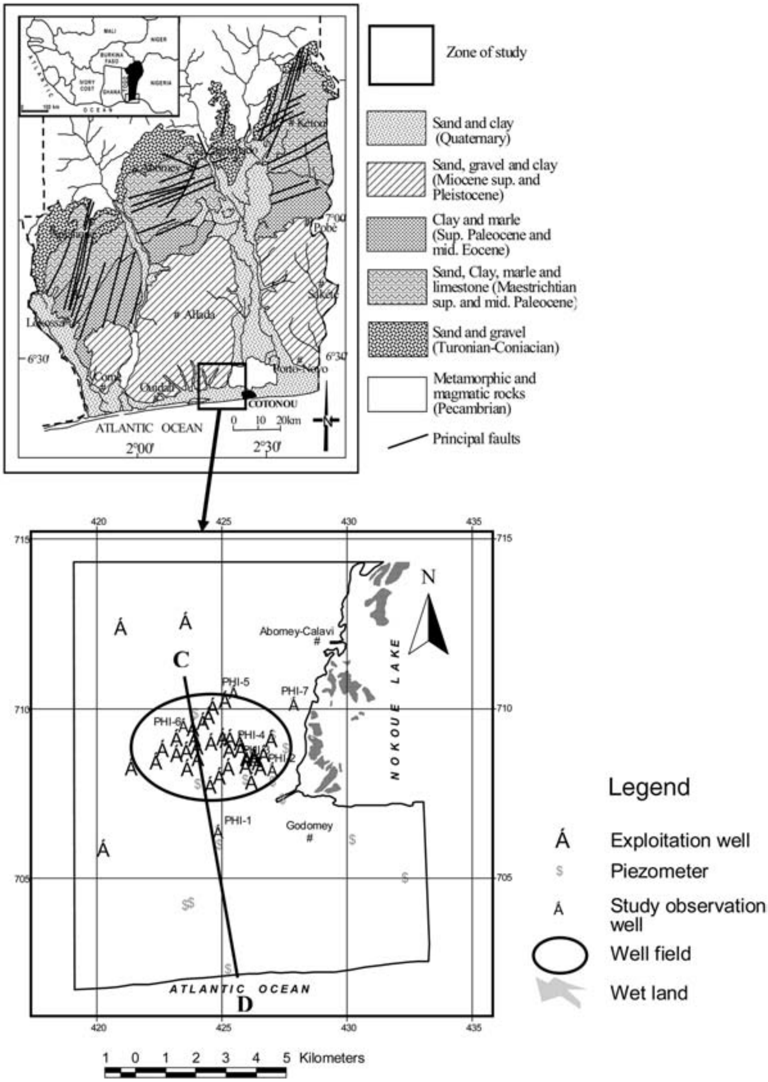

Figure 11.1. Location of study zone.

itself extends on a coastal plain in between the Atlantic Ocean to the south and Lake Nokoue to the north. This plain includes a very vulnerable shallow aquifer (Sagbo, 2000) but which is generally not exploited for the domestic use of the city. As a result, this study has focused on the developing peri-urban area, the Godomey Plateau (or Calavi) and the neighbouring zones situated approximately 10 km to the west of Cotonou. The aquifer system in these areas is pumped extensively and supplies the whole Cotonou agglomeration (approximately 1 million people) with drinking water (RGPH, 2002).

The Godomey site has a sub-equatorial climate with uneven rainfall distribution in space and time (Le Barbe *et al.*, 1993). The average rainfall of the sub-region increases from west to east. Fairly well-marked seasons of two rainy and two dry seasons are identified and distributed as follows:

1. A long dry season from mid-November to the end of March.
2. A long rainy season from the beginning of April up to mid-July.
3. A short dry seasons from mid-July to mid-September, and then rain to mid-November.

The temperature varies between wet monsoon rainfall periods accompanied by south-west winds and a dry and hot wind from the north (the Harmattan winds).

The geological structure of the Godomey site and surrounding area has been the subject of numerous studies (Slanky, 1962; Instituto Recherce Breda, 1987; Oyede, 1991). From a hydrogeological perspective, investigations were carried out for the purpose of an urban hydrocensus (SGI, 1981; IGIP-GKW-GRAS, 1983, 1989) and village prospecting (Geohydraulic, 1985; Nissaku, 1994), within an academic framework, knowledge inventory or assessment of available resources (Palles, 1988; Turkpak International-SCET- Tunisia, 1997; Boukari, 1998, 2002; Gnaha *et al.*, 2001).

Figure 11.2 presents a cross-section of the site's hydrogeological structure. At the regional level it is possible to distinguish several aquifer layers adding up to a thickness of 200 m. This was

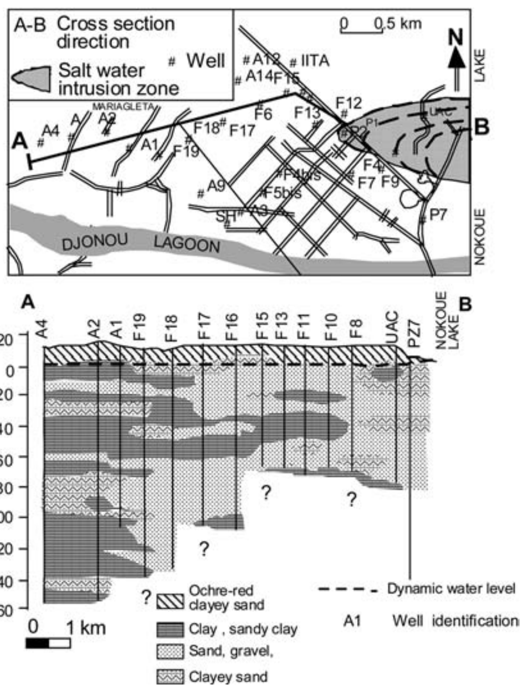

Figure 11.2. (a) Lithological cross-section of the Godomey wells field (b) Following the direction AB (Adapted from Boukari *et al.*, 1995).

investigated and a shallow aquifer identified, followed by three deeper layers which are separated by discontinuous clay layers.

All these horizons are hydraulically connected. In other words; the pollution of one of the aquifers (in this case the shallow aquifer) will most probably influence the others. The main characteristics of these aquifers are given in the sections that follow.

1.2 *Shallow aquifer*

The reservoirs of the shallow aquifers are constituted of:

- A thin sand layer (of 2–5 m thickness) with a coarse grain size which is situated at the base of Terre de Barre.
- Yellow sands, then grey and brown sands with thickness varying from 10 to 20 m to the bottom of the coastal plane. This layer is fairly homogeneous in structure and is unconfined. The shallow aquifer lies on a clayey layer that separates it from the lower aquifer system. This layer continues and is reduced by half in the south of the plain. It discontinues northwards to the plateau level. In the coastal plain, the aquifer is less deep and at times may be artesian on the plateaus.

From the piezometric plan, the following can be inferred (SCGI, 1981; SEREAU/BURGEAP, 1987; Maliki, 1993; Boukari, 1998):

– The drawdown amplitudes of the shallow aquifer varies from a few centimetres to 3 metres, depending on the years of abstraction, climatic conditions and geomorphologic positions.
– The main influx directions enable the distinction of piezometric domes corresponding to recharge zones (shallow and marshy zones) on the plateau to the west of the main wellfield. This situation confirms the existence of a hydraulic interconnection between this aquifer and the lower aquifer system which is pumped extensively.

1.3 *Lower aquifer system*

This system consists of three distinct zones with widely differing aquifer parameters. These aquifer horizons are made up of sand and gravel layers with varying thickness. The aquifers are separated by clay layers that are silty to sandy, leading to hydraulic interactions between the zones.

The available data (SGI, 1981; Pallas, 1988; SOGREAH/SCET TUNISIA, 1997; Boukari, 1998) indicates that the general water-flow direction is towards the SSE. However, towards the south drainage or anthropogenic influences alter this general direction.

Less water quality information is available for the shallow aquifer than for the deeper one. Only the water supply boreholes that are the subject of periodic control by the SONEB organisation have information regarding this, and most of the wells have targeted the deeper aquifer. The available data are obtained from water supply projects for villages, from occasional local- or regional-scale studies (Maliki, 1993) or research covering the entire plateau (Boukari, 1998).

2 METHODOLOGY

To reach the objectives of the current study, physico-chemical and fortnight piezometric measurements were taken from October 2001. Measurements were taken from the waters of seven wells (individual wells of great diameter, Fig. 11.1) with four on the plateau (PHI 3, PHI 4, PHI 5 and PHI 6), two on the oriental edge near the Lake Nokoue bank (PHI 2 and PHI 7) and one on the coastal plain (PHI 1). These sites were chosen due to the absence of supply wells in the surroundings. The samples were collected after purging which allows for the flushing of stagnant water before sampling. In low-yielding boreholes this has the disadvantage of creating a cone of depression. The parameters measured *in situ* include conductivity, pH and water temperature. One hundred and seven *in-situ* samples were sent for laboratory analysis. This included the following chemical elements: Ca, Mg, K, Na, CO_3, HCO_3, Cl, SO_4 and NO_3. Piper diagrams and other graphs were constructed for comparative analysis of data in space and time.

To better understand the evolution of the different parameters, temporal graphs of the fortnightly average rainfall were superimposed on the other graphs. These rainfall figures were from the IITA (International Institute of Agriculture) station, located immediately north-west of the Godomey abstraction field.

Simultaneously, the necessary complementary data for the generation of the vulnerability map were collected from literature and processed following the DRASTIC method (Aller L *et al.*, 1987). This method was preferred to several other available methods such as GOD (analytical, parametric or cartographic methods, Murat *et al.*, 2000) due to the limited extent of the study area (about 100 km^2), the relatively limited number of parameters and the heterogeneity of the area. DRASTIC provides a means to integrate the available data.

The DRASTIC method was applied in hydrogeological units of an overall area of 400,000 m^2 (40 ha). It is based on four fundamental hypotheses:

– The source of potential contamination is found at the soil surface.
– The contaminants are transported from the soil surface to the aquifer by infiltration.
– The contaminant has the same mobility as the shallow water.
– The contaminant type does not influence the determination of vulnerability.

In the DRASTIC method, the pollutant behavior from the soil surface up to the saturated zone is the same as the rainfall water. This method therefore gives a general indication of the distribution of shallow-water vulnerability in a given region, which can also include other parameters and specific information at a particular site (Menani, 2001).

The calculated indicator gives an evaluation of the aquifer contamination risk, which increases with a higher value. The minimum DRASTIC indicator is 23, while the maximum is 226. These theoretical extremes are very rare and the calculated indicators are generally in the range of 50 to 200 (Menani, 2001). The scale of vulnerability proposed by McCormick (1986) in Aubre *et al.* (1990) is the following:

1. DRASTIC indicator $<$ 100, low vulnerability.
2. DRASTIC indicator is between 100 and 150, moderate vulnerability.
3. DRASTIC indicator $>$ 150, elevated vulnerability.

Eight classes of indicator are generally differentiated: <79, 80–99, 100–119, 120–139, 140–159, 160–179, 180–199 and <200. The vulnerability map can also be obtained by the combination of seven spatial distribution maps of the individual parameters, assumed from their respective coasts.

3 RESULTS

3.1 *Physico-chemical parameters of waters*

Table 11.1 presents by site the average and extreme fortnight values of the different parameters measured regularly for the entire period of study (October 2001–November 2003).

Table 11.1. Extreme and average weekly values of water temperature, conductivity and level in the monitoring wells (October 2001–November 2003).

	Temperature (106)*			pH (106)*			Conductivity (106)*			Water level (107)*		
Borehole	Max	Ave	Min	Max	Ave	Min	Max	Ave	Min	Max	Ave	Min
PHI 1	33.9	30.3	28.2	11.46	8.44	6.04	1880	571.59	355	6.87	5.44	4.6
PHI 2	33	29.8	27.4	6.59	5.42	5.07	503	312.84	110.7	7.51	5.82	4.48
PHI 3	31.8	29.7	27.8	5.52	4.92	4.49	226	85.38	66.4	15.93	14.66	13.97
PHI 4	31.7	29.7	28.1	5.69	4.80	4.27	183	74.54	54.7	16.54	14.93	14.59
PHI 5	32.3	29.9	27.8	5.87	4.75	4.18	217	80.83	59.4	17.08	15.81	14.97
PHI 6 (39)*	31	30.2	29.2	4.74	4.44	4.23	199	191	176.6	17.62	16.75	17.27
PHI 7 (38)*	31	30.5	30	5.75	5.5	5.39	441	367.38	330	4.66	4.20	4.32

(38)* = number of measurements.

For static water levels, two zones have been identified; one of them characterised by shallow water levels between 4 and 8 meters, and the other by relatively deep levels, between 14 and 18 metres. The first zone corresponds to that of the coastal plain and the border of the plateau (lower ground) and the second to the plateau.

Temperatures are elevated at the coastal plain and bordering the plateau (27 to 34°C), whereas the temperatures in the relatively deep aquifer in the plateau range from 27 to 32°C. Since this parameter is extremely sensitive to atmospheric conditions, only the range in values can be considered significant.

The plateau zone waters are often acidic and are therefore aggressive with average pH-values between 4.18 and 5.87. On the contrary, on the coastal plain site PHI 1 well has a basic pH averaging at 8.44. These differences can be explained by the difference in lithology. The plateau, being of continental origin, is abiotic in nature, while the coastal plain is biotic in origin.

Overall, the groundwater levels of the coastal plain are slightly higher than those of the plateau. Furthermore, they are more mineralised, but less aggressive. This relatively strong mineralisation could be explained by the proximity to saline coastal waters or lakeside waters contaminated from anthropogenic activities. Other possible sources should not be excluded.

The PHI 2 boreholes located at the edge (lower plateau terrace) are characterised by a chemical composition that is intermediate between the plateau wells and those of the coastal plain.

3.2 *Chemical and bacteriological characteristics of water from the study area*

The average values of the six water samples from the first five boreholes selected for the study are presented in Table 11.2. An analysis of this table confirms the overall sense that the plateau waters are less mineralised than those of the coastal plain. The lower terrace zone between the plateau and the lake demonstrates intermediate contents between those of the plateau waters and the plain.

Table 11.2. Average values of physio-chemical and chemical parameters of groundwater (samples of 2002 to 2003).

			EC	Ca^{2+}	Mg^{2+}	NH_4^+	PO_4^{3+}	CO_3^{2-}	HCO_3^-	Cl^-	NO_3^-	NO_2^-	SO_4^{2-}	F^-	I^-
Points	N*	pH	µS/cm	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
PHI 1	6	7.828	502.3	33.67	6.638	0.047	0.27	17	64.22	43.47	75.65	0.219	0.784	0.137	0.117
PHI 2	6	5.154	271.6	12.52	2.075	0.247	0.282	0	10.78	49.31	31.85	0.038	0.96	0.026	0.122
PHI 3	6	4.666	72.26	1.466	0.647	0.121	0.485	0	5.186	16.33	10.44	0.035	0.314	0.037	0.081
PHI 4	6	4.594	65.18	1.922	0.39	0.043	0.209	0	5.226	19.07	5.844	0.023	0.632	0.037	0.097
PHI 5	6	4.528	82.88	1.154	0	0.061	0.408	0	5.154	20.41	4.796	0.015	0.674	0.04	0.115

Table 11.3. Results of physico-chemical and chemical analyses of the water samples for April 2004.

	PHI 1	PHI 2	PHI 3	PHI 4	PHI 5	PHI 6	PHI 7
pH	6.95	5.21	4.25	4.38	66.00	139.00	318.00
EC	432.00	140.00	92.00	57.00	4.37	4.35	5.25
Na^+	75	20	15	0.8	8.00	25.00	50.00
K^+	02	0.20	0.10	0.10	1.50	0.30	0.70
Ca^{2+}	54.32	6.43	0.71	0.70	0.71	0.71	8.58
Mg^{2+}	3.92	1.31	0.87	0.87	0.87	0.87	4.79
NH_4^+	0.17	0.05	0.08	0.0	0.0	0.58	0.04
Cl^-	63.90	24.85	28.40	21.30	24.85	51.12	92.30
SO_4^{2-}	5.85	6.59	9.20	2.04	4.18	1.72	4.70
CO_3^{2-}	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HCO_3^-	85.40	14.64	1.22	1.83	1.83	1.22	12.20
NO_3^-	96.60	24.00	5.36	2.23	0.48	1.04	35.00
NO_2^-	0.03	0.07	0.01	0.01	0.01	0.04	0.12

A more detailed examination of these results and those of April 2004 is presented in Table 11.3. The following facts are emphasised:

- in all the samples, Na^+ is the dominant cation.
- the cations indicating organic pollution (NH^{4+}, PO_3^{4+}) are present in the same order (0.043 to 0.48 mg/l) as chloride.

The carbonate-bicarbonate system is a major consideration in the anions, followed by Nitrates, with chloride dominating most water. There is only a single sample containing carbonate as a dominant ion, but this is not surprising when the elevated pH levels (between 7 and 11.5) are taken into account. This sample also displays nitrate and nitrite levels higher than the WHO recommended levels of 50 mg/l.

On the plateau the latter elements are present in the water, but still at very low levels. The importance of the role of the thick unsaturated zone explains this result. The mentioned elements are in fact indicators of organic pollution of especially anthropogenic origin. The iodide, fluoride and sulphate concentrations are significant, but not alarming.

Most of the analysed groundwater have a high bacterial count (total bacterial count, total coliforms, faecal coliforms, faecal streptococcus, and clostridium perfringens). All these parameters occur at levels above the physical count capacity of the laboratory, and no sample proved to be fit for drinking. This is not surprising when one considers that the abstraction boreholes are largely open, and therefore vulnerable to organic contamination.

The examination of these parameters' fluctuation over time (with the exception of bacteriological parameters, which are always in excess), sheds more light on the relationships that characterise their evolution.

3.3 *Seasonal and inter-annual fluctuations of frequently studied parameters*

From a quantitative point of view, it generally seems in Figure 11.3 that static water levels fluctuate between 3 m and 7.5 m in the coastal plain and the edge of the plateau (PHI 1, PHI 2 and PHI 7), and between 12 m and a little more than 18 m on the plateau. The water level fluctuations in each of the studied sites do not exceed 1 or 2 m, and we note a general trend in the rise of water levels since the end of the low water mark in 2001. The excessive rainfall of the last season (2004) caused exceptional rises with water levels reaching only 4 m on the plateau and 3 m elsewhere. The

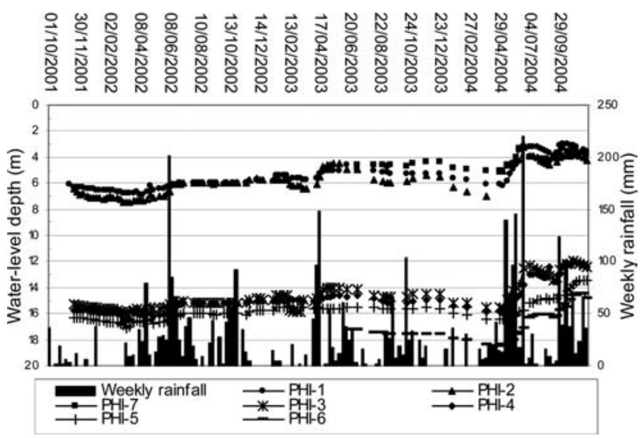

Figure 11.3. Temporal evolution of static water levels in wells.

three-year study period (2002–2004) therefore corresponds to a positive recharge period especially since this trend was maintained in spite of a substantial increase in pumping.

From the groundwater quality assessment qualitative plan, as indicated in Figure 11.4, pH-values were maintained around 5 for most of the samples. The only exception is the boreholes of the coastal plain, where the pH shows large fluctuations but still remain basic. It is however noted that there is an actual upward trend of pH in the PHI 7 wells, which is probably of anthropogenic origin (rehabilitation and construction). Finally, there is a general and substantial decrease of pH at all of the sites at the beginning of long rainy seasons which consequently increase the supply of CO_2 (atmospheric CO_2 and CO_2 linked to biological activity in the soils) through the first infiltration water.

Conductivity values, as presented in Figure 11.5, in most of the plateau wells (PHI-3, PHI-4, PH-5 and especially PHI-6) are around 100 μS/cm. The wells from other parts of the study area

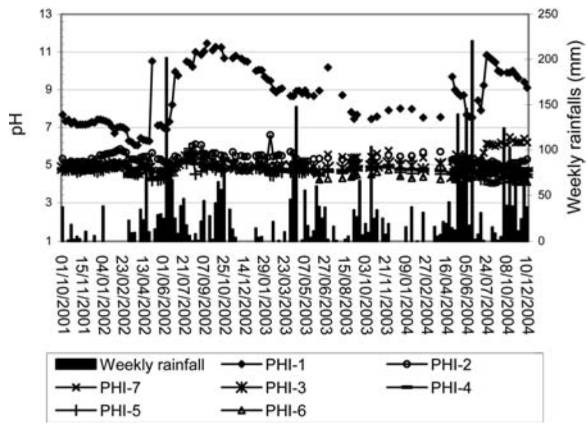

Figure 11.4. Temporal evolution of the water pH of the monitoring wells of the Godomey sector.

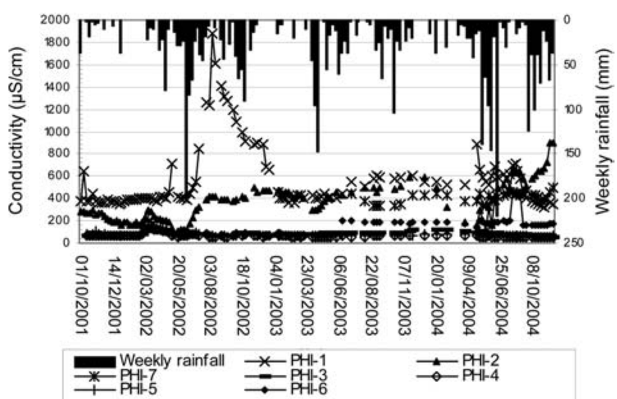

Figure 11.5. Temporal evolution of water conductivity of the sampled boreholes.

have conductivity values around 400 μS/cm with some results around 800 μS/cm. The elevated values registered in PHI 1 in 2002 (between 800 and 900 μS/cm) seem to be of anthropogenic origin (constructed as rehabilitation site).

3.4 *Vulnerability and shallow aquifer pollution*

The DRASTIC indicators determined are between 87 and 191. The vulnerability map developed from the results is shown in Figure 11.6. Only six of the eight classical indicators previously defined were used, with the two extremes (^{3}79 and ^{3}200) not represented. But all three the vulnerability scales of McCormack (1986) are presented: (1) one of elevated vulnerability (DRASTIC indicator $>$150), which exclusively characterises the coastal plain zone; (2) another of moderate vulnerability (DRASTIC indicator registered on 100 and 150), which especially characterises the south and the centre, and also a small northern portion of the plateau zone; (3) one indicator of low vulnerability (DRASTIC indicator $<$100), which characterises the northern portion of the plateau and the field sector aquifer from Godomey.

From the map it is evident that vulnerability generally reduce from the south to the north (the coastal zone towards the plateau). This is not surprising when the thickness and nature of the unsaturated zone are taken into account. These two factors have the strongest influence on the DRASTIC vulnerability calculation and on most of the valuations of the complex analytical methods. In fact, as the thickness of this vadose zone increases from south to north, it becomes finer. This alters the grading from coarse, average or fine sands in the coastal plain to silt and clayey and silt sands, i.e. at the Terre de Barre, essentially on the plateau. It must be noted that the Godomey intensive aquifer field is situated in an area of low aquifer vulnerability. The residual drawdown of the natural water level due to almost permanent intensive pumping in the area explains the transition from moderate vulnerability to low vulnerability relative to areas with natural water level.

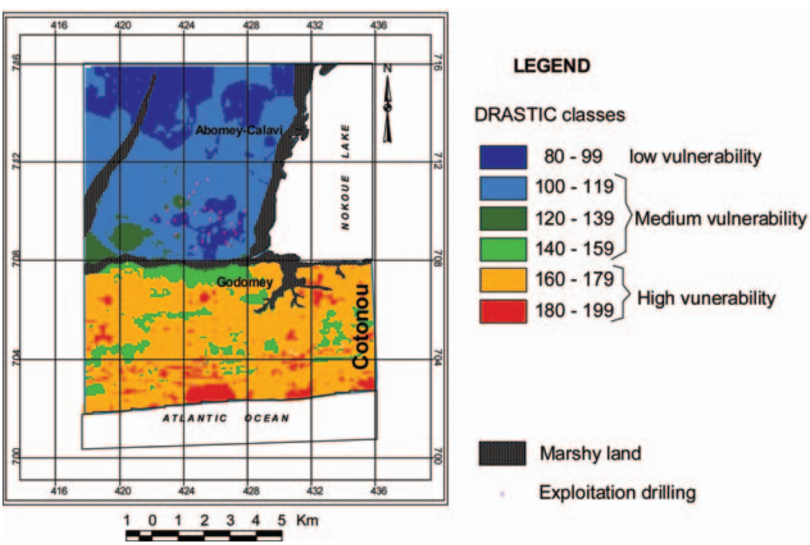

Figure 11.6. Vulnerability map of the Godomey sector and its surrounding area.

3.5 *Aquifer pollution and validation of the establishment map*

All the sampled boreholes on the plateau display contaminant levels (Cl^-, NO_3^-, NH^{4+} and PO_3^{4+}) generally below those recommended by the World Health Organisation (WHO). Only the wells located on the coastal plain (PHI 1) show some distinctly higher values than this recommendation, especially the nitrates. Boreholes located in this plain is known to have high nitrate values, especially those located in the areas with high population densities such as the centre of Cotonou (Boukari, 1988). It is for this reason that it was not considered useful to select study sites here. The PHI 1 wells were analysed and the lowest concentration is located at the plateau edge (PHI 2), in which the nitrate contents sometimes reach the required limits. For the three years of the study, no remarkable upward trends were noted for the five elements listed above, with the exception of the PHI 1 boreholes where nitrate and chloride seem to increase. For the other wells, no constant upward or downward trends were observed. In total, these results correspond well with the information derived from the vulnerability map and confirm the findings.

These boreholes are however characterised by high bacteriological counts (Table 11.3). On-site santitation practices generally seem to be the main cause of this pollution type.

4 DISCUSSION: RESOURCE STATUS AND PERSPECTIVES

As can be seen from the results above, the water quality of the shallow aquifer on the plateau in terms of the inorganic constituents is still within the recommended WHO limits. This is also true for the indicators of anthropogenic contamination elements, especially nitrates and nitrites. However, on the coastal plain the nitrate and nitrite concentrations already exceed the accepted levels. The established vulnerability map shows that even for the water from the plateau sector, which is still of good quality, the shallow aquifer waters' deterioration risk is not isolated. The vulnerability degree is more moderate than low. This situation requires regular study of the pollution element indicators, notably in the aquifer perimeter and its environs.

As for bacterial pollution, if it is evident in all the analysed boreholes this could be linked to the ode and remains limited. The equipped and exploited boreholes following the correct regulations are of no concern.

The chemical pollution from the surface is, as we observe, much lower at the aquifer field sector, even though it is some cause for concern. On the contrary, the most imminent resource threat proves to be saline intrusion. From this perspective, the synthesis diagrams of Figures 11.8 and 11.9 summarise the spatial and temporal evolution of shallow-water resources at the Godomey site, since the opening of the aquifer field. From the start of site exploitation in 1956 to 1980 at least, no saline contamination has been observed in the aquifer perimeter (SGI, 1980). The first indications of salt-water intrusion occurred at the beginning of 1990s, as noted by the National University of Benin, and confirmed by SONEB. From the current database, this contamination seems to originate from Lake Nokoue, as presented in the cross-sections of Figures 11.7 and 11.8. From the southern side of the study site, i.e. the oceanic side, the freshwater-saltwater interface seems to be largely located at the lower aquifer system. As yet, it is only the shallow aquifer that experiences the advanced phenomenon of saline groundwater, which may or may not be linked to pumping in the Godomey Field (Fig. 11.9). All these hypotheses must still be confirmed, especially through modelling (flow and mass transport or variable density flow) and hydrochemical study through isotopic approaches.

While awaiting the results of such studies, it is important to pursue a regular quantitative and qualitative study within the framework of the current study of pollution indicators; notably nitrates, nitrites, ammonium and chlorides. This integration has been done for nitrates since the beginning of this year. It is yet to be expanded to other elements. This is important, since it informs public sectors (regulators and decision makers) of risks related to a lack of protection for the existing aquifer and the consequences of a continued increasings pump volume. Resource protection measures and optimisation of exploitation through the development of a new aquifer field are currently of

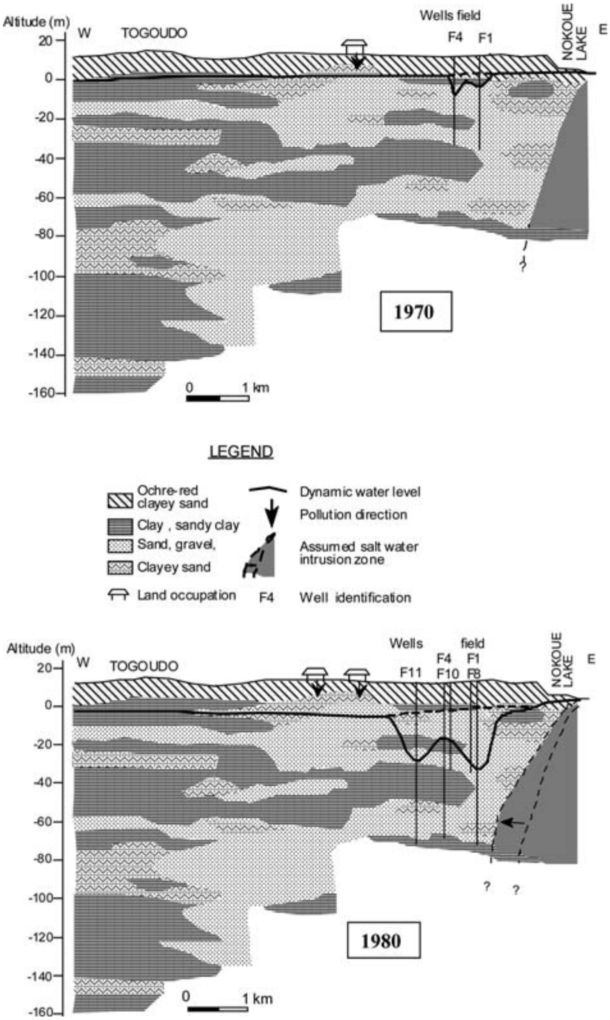

Figure 11.7. Spatial and temporal evolution of the aquifer piezometric surface and schematic representation of saline water intrusion and land occupation evolution on the Godomey site: 1970 and 1980 situations (Boukari *et al*., 1995 reviewed and completed).

great importance. This will have to be done if we do not want to compromise the integrity and durability of the current water supply to the city.

A modelling system (with a contribution of isotopic hydrology for the correct definition of limited conditions) would especially facilitate the judicious location of the new fields to be opened and to define aquifer zones of exploitation for the different durations (1 year, 5 years and 20 years, for instance) in view of protection zones or delimitation around which information and education campaigns for the local population could be undertaken.

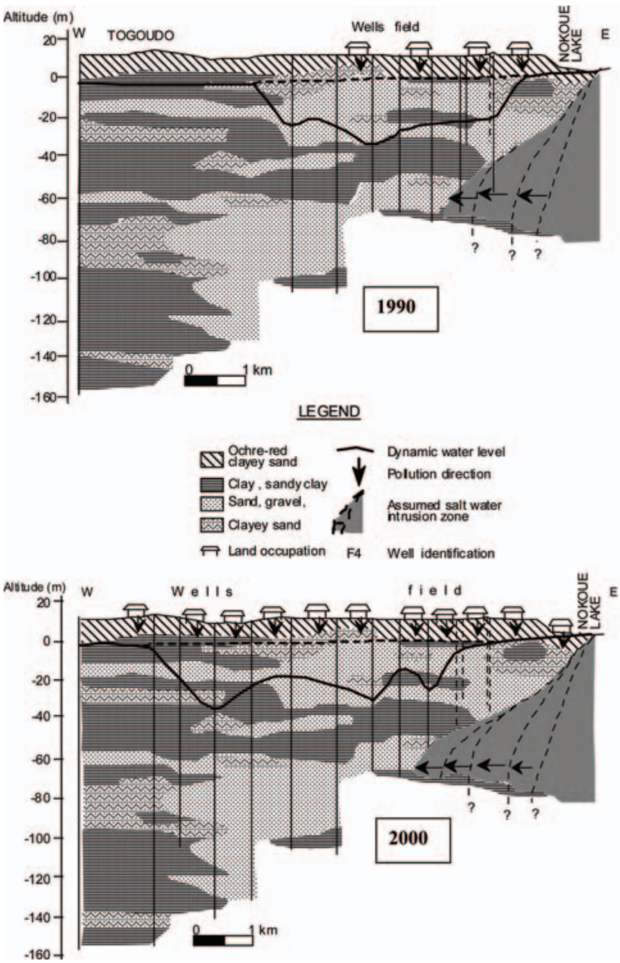

Figure 11.8. Spatial and temporal evolution of the aquifer piezometrique surface and schematic representation of saline water intrusion and land occupation evolution at the Godomey site: 1990 and 2000 situations (Boukari *et al.*, 1995; reviewed and completed).

5 CONCLUSION

This study enabled the qualitative and to a certain extent quantitative observation of the intensively studied aquifer system to the west of Cotonou City and the development of a vulnerability map of the shallow aquifer in the zone.

From the results it can be seen that the aquifers supplying the Cotonou agglomeration are prone to contamination due to its hydraulic interconnection with its immediate environment. Currently the water quality is still acceptable for the physico-chemical and bacteriological constituents. This area has also experienced a high rate of human settlement and therefore exposed even further to negative anthropogenic influences.

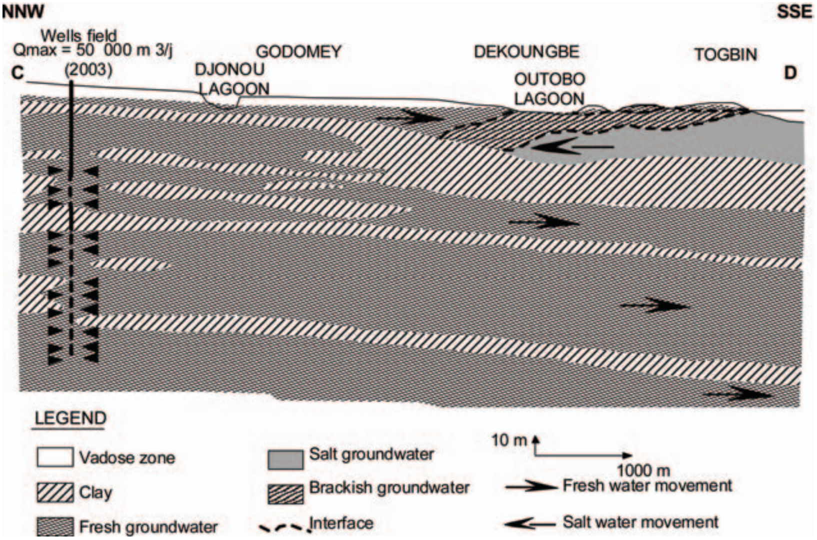

Figure 11.9. Actual situation of salt-water intrusion into the aquifer of the Godomey sector from the ocean following the north-south direction (modified from Boukari *et al*., 1995).

The DRASTIC aquifer indicators obtained are from average to very high. Based on increased settlement and high vulnerability rating, it is clear that pollution is imminent, even if it has not yet been realised. In this context, it is necessary to take short-term measures to enable the protection of these aquifers in order to control the progressive degradation of water quality.

As the Godomey Aquifer is situated in a large hydrogeological basin shared by Ghana, Togo and Nigeria, it is necessary to make a concerted effort for the medium and long term to initiate trans-boundary efforts to protect the groundwater quality in the region.

ACKNOWLEDGEMENTS

We would like to thank the UNEP/DEWA and the UNESCO/IHP for their financial support for making this study possible. Our thanks go to structures such as SODECI (Distributive Society of Water of Côte d'Ivoire), the Directorate of Human Hydraulic (DHH) and the Directorate of the Water Resources (DRE) for their collaboration.

REFERENCES

Aller L., Bennett T., Lehr J.H. *et al*., 1987. DRASTIC : a standarized system for evaluating ground water pollution potential using hydrogeologic settings. National Water Well Association. Rapport EPA-600/2-87-035; 622 p.

Aubre F., Isabel D. et Gélinas P.J., 1990. Revue de littérature sur les methods d'évaluation des risques de contamination des eaux souterraines. Rapport GGL-90-21. Département de Géologie. Univ. Laval. Québec. Canada.

Boukari M., Alidou S., Oyédé L.M., Gaye C.B., et Maliki R., 1995. Identification des aquifères de la zone littorale du Bénin (Afrique de l'Ouest): hydrodynamique, hydrochimie et problèmes d'alimentation en eau de la ville de Cotonou. African Geoscience Review, Paris France. Vol. 2, 1, 139–157.

Boukari M., Gaye C.B., Faye A., et Faye S., 1996. The impact of urban development on coastal aquifers near Cotonou, Bénin. Journal of African Earth Sciences special issue, Vol. 22, issue 4.

Boukari M., 1998. Fonctionnement du système aquifère exploité pour l'approvisionnement en eau de la ville de Cotonou sur le littoral béninois. Impact du développement urbain sur la qualité des ressources. Thèse de Doctorat d'Etat ès Sciences, Univ. C.A. Diop, Dakar Sénégal.
Boukari M., 2002. Réactualisation des connaissances hydrogéologiques relatives au bassin sédimentaire côtier du Bénin. Rap. DH. Appui à la Gestion des Ressources en Eau (AGRE). Cotonou Bénin.
Géohydraulique., 1985. Note explicative de la carte hydrogéologique à 1/200 000 du bassin sédimentaire côtier du Bénin. MET. DH. FED 23 p + 1 carte hors texte, Cotonou, Bénin.
Gnaha E.F.C. & Adjadji Ch. A., 2001. Evolution quantitative et qualitative des ressources en eaux souterraines captées dans le périmètre de pompage intensif de Godomey: impact sur l'approvisionnement en eau potable de l'agglomération de Cotonou. Mém. de Maîtrise de Geol. Appl. Univ. Abomey-Calavi, Bénin.
IGIP-GKW-GRAS, 1983. Plan Directeur Alimentation en eau potable ville de Cotonou. Les ressources en eau. Ministère de l'Industrie des Mines et de l'Energie -SBEE. Vol. 2, Cotonou Bénin.
IGIP-GKW-GRAS, 1989. Plans Directeurs et Etudes d'Ingénierie pour l'alimentation en eau potable et l'évacuation des eaux pluviales, des eaux usées et déchets solides. Ville de Cotonou. 90 p. SBEE. Cotonou. Bénin.
INSAE, 1994. Deuxième recensement général de la population et de l'habitation. février 1992. Projections démographiques 1992–2027. Cotnou, Bénin.
Intituto Recherche Breda ou IRB, 1987. Etude de cartographie géologique et prospection minière de reconnaissance au Sud du 9ème parallèle. Rap. FED-OBEMINES. 80 p. + Annexes.
Le Barbe L., Ale G., Millet B., Borel Y., Gualde R., 1993. Les ressources en eau superficielles de la République du Bénin. Orstom éditions, Institut Français de Recherche Scientifique pour le Développement en Coopération, Paris, Direction de l'Hydraulique du Bénin, Cotonou.
Maliki R., 1993. Etude hydrogéologique du littoral béninois dans la région de Cotonou (A.O). Thèse de Doctorat du 3ème cycle. Version provisoire. UCAD; Dakar, Sénégal. 162 p. + Annexes.
McCormack R., 1986. Devis-type d'une étude de vulnérabilité des eaux souterraines à lapollution utilisant la méthode DRASTIC. Ministère de l'Environnement du Québec; Direction des Eaux Souterraines et de la Consommation. 5p.
MECCAG- PDPE, 2000. La population au Bénin: évolution et impact sur le développement. Policy Project/USAID. Ministère d'Etat Chargé de la Coordination de l'Action Gouvernementale, du Plan, du Développement et de la Promotion de l'Emploi. Cotonou, Bénin.
MECCAG- PDPE, 2002. Etudes nationales de perspectives à long terme NLTPS-Bénin 2025.
Menani M.N., 2001. Evaluation et cartographie de la vulnérabilité à la pollution de l'aquifère alluvionnaire de la plaine d'El Madher, Nord-Est algérien, selon la méthode DRASTIC. Sécheresse n° 2, vol. 12: 95–101.
Murat V., Martel R., Michaud Y., Fagnan N., Baudoin F., et Therrien R., 2000. Cartographie hydrogéologique régionale du piémont laurentien dans la MRC de Portneuf: comparaison des méthodes d'évaluation de la vulnérabilité intrinsèque CGQ/UQ. Québec. Canada.
Nissaku Co. LTD, 1994. Projet pour l'exploitation des eaux souterraines du Bénin, phase III. Rapport final. 204 p. + photos + 3 vol. annexes. Cotonou, Bénin.
Oyédé L.M., 1991. Dynamique sédimentaire actuelle et messages enregistrés dans les séquences quaternaires et néogènes du domaine margino-littoral du Bénin (Afrique del'Ouest). Thèse de Doctorat UnivBourgogne et Univ Nat du Bénin 302p.
Pallas P., 1988. Contribution à l'étude des ressources en eau souterraines du Bassin côtier du Bénin. Confrontation ressources-besoins. 29 p. + annexes A, B, C, et D. Cotonou, Bénin.
RGPH, 2002. Synthèse des résultats. Direction des Etudes démographiques, INSAE, octobre 2003, cotonou. Bénin.
Sagbo O.M., 2000. Pollution des nappes alimentant la ville de Cotonou en eau de consommation: vulnérabilité et urgence des mesures de protection des aquifères.
SEREAU/BURGEAP, 1987. Expérimentation sur l'assainissement à Cotonou: potentialités d'infiltration dans le sol. Rapport de synthèse. 16 p. + annexes. Cotonou. Bénin.
Slansky M., 1962. Contribution à l'étude géologique du Bassin sédimentaire côtier du Dahomey et du Togo. Thèse. Univ. Nancy, série 59, n° 165 et 1962, Mém. BRGM n° 11, 170 p.
Société Générale pour l'Industrie ou (S.G.I.), 1981. Etude de la nappe aquifère de Godomey. Banque Africaine de Développement -SBEE. Cotonou, Bénin.
SOGREAH/SCET Tunisie, 1997. Rapport sur 'Etude de la stratégie nationale de gestion des ressources en eau du Bénin: assistance à la définition de la stratégie nationale de gestion des ressources en eau du Bénin". 7 vol. DH. Cotonou, Bénin.
Turkpak International-SCET-Tunisie, 1991. Inventaire des ressources en eaux souterraines au Bénin. Rapport final. V.1 284 p. + annexes, Cotonou, Bénin.

12

Assessment of water pollution and risks to surface and groundwater resources in Bamako, Mali

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ABSTRACT: Due to the lack of research on wastewater quality in Mali, a specific monitoring campaign focusing on both chemical pollution (trace elements, heavy metals, nitrates, ammonium) and biological pollution in the cities of Bamako and Koulikoro and the Bani agricultural region was undertaken. Monitoring the water quality of 18 vulnerable areas in the Bamako-Koulikoro metropolitan area shows that all of them exceeded the WHO standard for wastewater effluent without surface water treatment on at least three parameters. Parameters requiring priority actions are BOD5, COD, SS and heavy metals. It is further concluded that the run-off is not a significant source of contamination by heavy metals. Rainfall events cause significant volumetric variation, which places the treatment works under stress, particularly in the correction of BOD5 and COD (the main parameters of biological purification). Consequently, an essential intervention would be to separate wastewater sewer networks from the run-off collection system. Finally, the priority is to regulate industrial effluents by imposing treatment and water quality controls on the industries, such as the power plant of Daar-Salam. As for the rest of the Niger River valley, no major risk of water pollution yet exists, thanks to the low rates of population growth and urbanisation. As shown by the water quality monitoring specific anthropogenic pollution problems are regularly observed. Thus water pollution is actually a problem related more to education than it is a technical problem of protecting water resources. However, the very fast population growth, associated with increases in waste production and the development of the Bamako-Koulikoro urban area, will severely worsen the situation. In the short term, if no measures are adopted to treat the effluents, all the water resources could be contaminated. In the near future, industrial effluent must be efficiently controlled and the population educated on the problems and hazards of water pollution. Finally, instead of exploiting the deep aquifer, it seems more profitable to urgently improve the existing infrastructures already in place in terms of water resources, and to develop wastewater treatment facilities. This would ensure that the current resource can be used in the longer term.

1 INTRODUCTION

1.1 *Bamako and the Sudanese African context in Mali*

Most of the literature concerning water problems in Bamako, and generally speaking in developing countries, focuses only on the drinking water supply, very rarely on sanitation concerns and almost never on the treatment of pollution affecting water resources (Tecsult, 1994; Sow, 1997). Furthermore, the lack of research on wastewater quality in an African context, an essential requirement for the design of wastewater purification processes, was underlined by the BCEOM in 1984. Research in this field is still lacking for the city of Bamako. Indeed, bibliographic studies show that only very little measurements are effectively taken. Publications on water quality problems in Bamako (from engineering offices, administrations or scientists) often re-use the same data, which renders it outdated and unreliable, since it became less and less accurately through a number of re-writing exercises.

In this context, the EQUANIS research program of the IRD office in Bamako undertook systematic weekly monitoring of the chemical quality of surface waters on the entire Niger River Valley area and its effluents from 1992 to 1998. This was combined with specific monitoring campaigns for chemical pollution (trace elements, heavy metals, nitrates, ammonium) (Picouet, 1999; Palangié, 1998; Paget, 1999; Derolez, 2001) and biological pollution for the cities of Bamako and Koulikoro (Palangié, 1998) and the Bani agricultural axis (Dessouassi, 1997; Bonnefoy, 1998).

Large-scale population growth took place in Bamako from the early 1960s (Fig. 12.1).

Several socio-political and material factors are responsible for this situation:

– The decision to turn Bamako into a major administrative centre (1908: capital of the French colony; 1960: capital of the new independent state);
– the breaking of the Mali Federation in 1959, which caused the return of the Malian population settled in Senegal;
– the freedom of movement between the countryside and the capital, thanks to the political regime change in 1968;
– the rural depopulation following the pauperisation of the agricultural population;
– the concentration of activities and most of the administrative and education institutions in Bamako.

As a consequence, Bamako had the fastest demographic growth in Mali. The highest growth rate observed was 22.7% from 1958 to 1960 (years of massive repatriation of foreign Malian populations). This rate abruptly fell to 3.9% between 1960 and 1966 (years marked by political policies of regional development). From 1974 to 1976, under the military regime, the growth rose again to reach 7.3%. In 1987, during the last census, the population in Bamako reached 678 000. This growth has been estimated to continue at a sustained rate of 4.2% until 1993.

2 THE WATER QUALITY OF THE NIGER RIVER

2.1 *Uncontaminated surface waters in other areas*

Several studies have been undertaken on the water quality of the Niger River (Iwaco, 1996; EQUANIS, 1997; Gihrex, 2000; Ghenis, 2001) yet it remains, like most of the major African rivers, slightly affected by chemical contamination (Picouet *et al*., 2002). In fact, water in the main bed of the Niger River is not contaminated by any metals, nitrates, nitrites, ammonium, phosphates, biological compounds or agents. This result is unsurprising, considering the lower level of industrialisation in the Niger Basin and the minor use of fertilisers and pesticides in agriculture. However, weekly

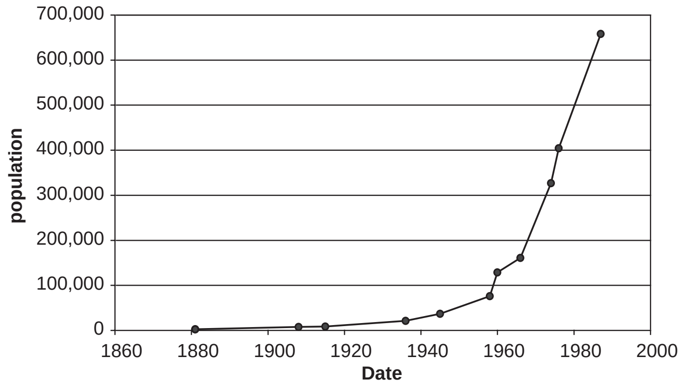

Figure 12.1. Evolution of the population in Bamako between 1881 and 1987 (Source: TECSULT – Plan directeur d'assainissement de Bamako (1993–2003), Volume 1: Rapport Principal, p. 35).

monitoring of wells, boreholes and run-off waters in the CMDT cotton wool production zone in south Mali showed major contamination by specific pesticides, namely DDT, which is linked to problems of storage or product misuse (Bonnefoy, 1998).

Moreover, although the physico-chemical water quality remains generally good in the entire Niger Basin (except in specific places near industrial discharge points on the Bamako-Koulikoro axis, Palangié, 1998), bacteriological pollution in populated areas is common. Thus, the high concentrations of faecal coliforms and streptococci alone make these surface water resources unsuitable for drinking according to WHO standards. This water is, however, still commonly used for drinking, which causes major sanitary and health risks.

2.2 *The Bamako-Koulikoro axis*

Monitoring the water quality of 18 vulnerable points in the Bamako-Koulikoro urban area (Palangié, 1998) shows that they all exceeded the WHO standards for wastewater effluent without surface water treatment on at least three parameters (Table 12.1). Parameters requiring priority action are BOD5, COD, SS and heavy metals. However, the concentrations of these compounds remain equal to or below the average values of wastewater in industrial countries, except for the effluent water from the heavy fuel power plant of Daar-Salam. These latter wastewaters are by far the most heavily contaminated and should be pre-treated in the industrial plant itself in order to prevent the pH and associated heavy metal problems before any further treatment downstream is considered.

BOD5, COD and SS are almost always much higher in rainy weather, due to run-off waters carrying many organic materials. Heavy metal concentrations are higher in dry weather because effluents are not diluted by run-off waters. Thus, one can deduce that the run-off is not a significant source of contamination of heavy metals. The rainfall events cause significant volumetric variation that places the treatment works under stress, particularly in the correction of BOD5 and COD (the main parameters of biological purification). Consequently, an essential intervention would be to separate wastewater sewer networks from the run-off collection system. This is particularly important, especially in the Sudanian climate context, where great contrasts between dry and wet seasons occur (Palangié, 1998).

Table 12.1. Water quality from five selected effluent sites between Bamako and Koulikoro, and comparison with standards for effluent disposal without treatment in surface waters (from Palangié, 1998).

Parameter	Value required by standards[1]	Site 3		Site 4		Site 7		Site 8		Site 10	
		rain	dry	rain	dry	rain	dry	rain	dry	rain	dry
BOD_5 (mg/l)	30	500	60	40	80	255	140	20	100	30	40
COD (mg/l)	125	500	196	219	133	255	162	315	181	96	101
Nitrates (mg/l)	30	11	3.5	12.3	10.5	4.8	6.6	9.68	11	15.8	0
pH	5.5–8.5	7.25	7.2	7.17	8.9	6.96	7.7	10.4	10.3	7.32	7.4
SS (mg/l)	35	284	41.3	318	20	452	131	253	113	323	60
Cu (mg/l)	0.5	<2.2	1.37	<2.2	0.73	<2.2	0.45	<2.2	0.9	<2.2	0.6
Fe + Al (mg/l)	5	1.29	3.82	1.32	2.49	1.7	8.21	0.79	1.25	0.95	5
Mn (mg/l)	1	<0.12	0.11	<0.12	0	<0.12	0.44	<0.12	0	<0.12	0

Site 3: main sewer in Bamako at its entrance to the Niger River;
Site 4: west main sewer in Bamako at its entrance to the Niger River;
Site 7: main sewer of the industrial area of Bamako at its entrance to the Niger River;
Site 8: main sewer of textile and pharmaceutical factories at its entrance to the Niger River;
Site 10: main sewer of the market gardening area.

[1] Transcription in French legislation of European directives concerning wastewater effluent without treatment in surface waters.

In the average urban wastewaters of industrial countries it is estimated that about 66% of BOD_5 consists of particles (Dégremont, 1978). Similarly, the measurements in this study show that an important part of pollution loads is linked to particles larger than 0.7 μm (i.e. SS). Thus, a simple decantation of wastewater in Bamako alone should eliminate a significant part of the contamination. However, sludges resulting from this decantation could contain hazardous compounds. Consequently, they should be stored and used carefully, mainly as fertilisers. Moreover, the ability of SS to decant is directly linked to its granulometry: the finer the particles, the slower their sedimentation. The process is sometimes too slow for processing within reasonable timeframes.

In rainy weather, the contamination loads tend to rise or remain stable while they travel through gutters and main sewers before being disposed into the Niger River. However, in dry weather the evolution of contamination loads is much more uncertain, due to gutters and sewers being cluttered by sediments and plants restraining water flow and significantly reducing the proportion of pollutants arriving into the river. In fact, the sewers open to air and light work as a kind of biofilter and eliminate easily biodegradable substances (BOD5, nitrates and phosphates). As a consequence, cluttered sewers do not only cause drawbacks: they paradoxically also contribute to reducing the pollution disposed of in the Niger River.

All monitored sites show signs of biological contamination. Great quantities of faecal coliforms, streptococci and other bacteria were detected regardless of the weather being dry or rainy. However, the influence of rain and run-off is also significant for biological pollution, since tests indicate much higher micro-organism concentrations in waters sampled during rainy weather. This fact highlights the insufficient protection of sanitary installations, pitlatrines and sewers and the frequent pollution of soils by excrements. Furthermore, domestic wastewaters are less diluted in dry weather, and the detergents they contain have a more powerful inhibiting action on micro-organism growth than during rainy weather.

Even in the middle of the Niger River at the pumping station of the EDM, which produces the drinking water of Bamako, biological contamination exceeds the EEC standards for drinking water. This situation is made worse by the fact that many people continue to use water directly from the river for drinking water. Thus it can be concluded that, if the physico-chemical pollution is specific, the biological contamination of surface waters in Bamako is ubiquitous. The high concentrations of iron in all sampled waters are noteworthy. These concentrations do not change upstream and downstream of Bamako, leading to the conclusion that that the origin is probably geological and not a result of human activities.

2.3 *Implications*

Previous studies and this study's measurements underline the present unstable and fragile balance for the conservation of the quality of surface waters, and even those of the Niger River in spite of its high flow rate. Several major industrialisation projects, agricultural development in the Niger Valley, systematic and untreated disposal of effluents from nearby cities into the river pose the risk of irreversible pollution for the system. According to the Bamako purification plan (Plan directeur d'assainissement de Bamako) (1993–2003), the Niger River flow rate is enough to ensure its auto-purification. It is generally considered that wastewaters can be rejected without treatment in a waterstream if the following conditions are in place:

– no installation (a pumping station for example) that could be affected by contamination should be in close proximity downstream from the effluent point
– the wastewater flow rate cannot exceed 0.2% of the minimum flowrate of the water stream.

When the effluent flow rate is between 0.2% and 1% of the minimum, wastewater has to be pre-treated (rough filtration, sand removal, fast decantation for four hours…). When the effluent flow rate is above 1%, a secondary treatment is required (biological purification…). In the case of Bamako, the minimum Niger River flow rate is $62\,m^3/s$ for a recurrence period of 10 years. Thus, the auto-purification potential permits the disposal of effluent of $0.124\,m^3/s$ wastewater without treatment, which corresponds to an equivalent population of 143,000 inhabitants (based on a water consumption of 75 litres per person per day, which is a high hypothesis of forecasts for 2005). The population connected to the sewer network was estimated at 44,000 in 1998 (4% of the population of Bamako). It is necessary to add to this amount the equivalent population corresponding to effluents generated by the administrative sector (estimated at 53,350 persons) and the equivalent population corresponding to industrial effluents (about 30,000 persons). With a total equivalent population of 127,350 persons, Bamako theoretically remains under the auto-purification potential of the Niger River, which makes the installation of a sewage treatment plant unnecessary. However, the equivalent population should reach this limit in 2006, with an estimated population of 139,250 persons.

This calculation is however incomplete because it only takes into account the extreme minority of the population connected to the sewage network and not, as shown by the study of the rainwater network, that gutters and channels flowing to the Niger River are also used as open-air sewers by a very large part of the population. Therefore, the proposed equivalent population is greatly underestimated. Furthermore, the equivalent load for industries only corresponds to inert or organic pollution (BOD, SS) and completely ignores the impact of toxic or non-biodegradable compounds. The direct use of the Niger River to do laundry, bathe, or do the washing is also not taken into account and as a consequence the population is particularly exposed to water pollution.

3 AQUIFER VULNERABILITY AND POLLUTION

3.1 *Pollution of the shallow aquifer system*

The shallow aquifer in the Bamako area is highly contaminated. This is a result of the following factors:

- numerous specific contamination sources in the Bamako district,
- the lack of attenuation measures,
- the fact that the water level is very shallow,
- the high permeability of the soils in certain places (namely on the right side of the river, with the ground on the left side seeming mostly impermeable) and
- the numerous non-protected wells and improperly designed individual sanitation installations (Zallé and Maiga, 2002) opening direct connections between the surface and groundwater.

Several studies have described this contamination (Alpha *et al.*, 1991; Iwaco, 1996). High concentrations of nitrates, nitrites, phosphates and chloride from a variety of sources (detergents, fertilisers, decaying organic compounds), the presence of pesticides, high levels of bacteriological contamination (faecal coliforms and streptococci and total bacteria) have been reported for this aquifer. The bacterial contamination comes from pit latrines, soakaways and septic tanks that are poorly designed or installed too deep, leaking septic tanks and excreta carried by run-off waters that flow into wells without sanitary seals or other protection mechanisms.

Thus the water of the superficial aquifer is already inappropriate for human consumption according to the WHO standards. A study by IWACO in 1996 showed that the concentrations of nitrates and nitrites rose dramatically in certain wells sampled in 1991.The results were tempered by the fact that sampling during 1991 was done in old neighbourhoods with a high population density, where the pollution is expected to be the most severe. In the market gardening areas, the water quality of the wells is much better, with concentrations of nitrates, nitrites and phosphates usually within permissible drinking water standards. Wells can also often be preferred pathways for pollutants to migrate to the aquifer where incomplete protection measures are in place. This suggests that the water in the wells may not necessarily be representative of the quality of the entire aquifer. From this point of view, dedicated monitoring boreholes should be drilled to ascertain the true quality. Wells in the industrial area have also shown evidence of nitrate and nitrite levels similar to the worst cases seen in the other areas, and in addition they are largely contaminated by heavy metals and toxic compounds (arsenic, chromium).

If no protective measures are implemented, the rampant population growth and the urban expansion of Bamako will aggravate the situation and deprive a majority of the population of its main water resource (66% of global domestic supply in 1998).

3.2 *Threats to the quality of water in the deep aquifer*

The fractured sandstone aquifer is hydraulically connected to the shallow aquifer through fissures and major cracks that are open to a depth greater than 100 m. The deeper aquifer has already been considered as a possible drinking water resource and prospecting boreholes have been drilled (Giorgi, 1995). The quality is excellent, except for some nitrate enrichment. It is not known if these elevated values result from infiltration from the shallow aquifer or are a true reflection of the deeper aquifer's quality. Nonetheless, the generally good quality shows that transfer between the two aquifers is slow enough to prevent widespread degradation under current conditions. This is explained in part by the higher hydraulic pressure in the semi-confined fractured sandstone aquifer, which limits downward migration.

However, the equilibrium is at risk of being disturbed by drilling the deep aquifer. The creation of such pathways provides direct access for contamination if protective measures around these wells are not put in place. If the groundwater quality of the shallow aquifer continues to deteriorate, it is possible that the fractured sandstone aquifer may become badly polluted.

The potential migration of contaminants to the deep aquifer is a risk Bamako cannot afford to take. The risk is even more real, since it has been speculated that the deeper aquifer may be used to supply drinking water to the city through deep boreholes. This risk is an additional and important reason for implementing measures to prevent contamination of the shallow aquifer and treating effluents in surface waters in the Bamako-Koulikoro urban area. The fault to the east of the Bankoni cliff directly connects the deep aquifer waters with the run-off waters. Currently this area is an informal dumping ground for car shells, old batteries, and other waste material that present a high risk to the future groundwater quality of the deep aquifer.

3.3 *Summary of the situation in Bamako*

The situation in the Bamako district can be summarised as an environmental dead end. The growth of the population, urban expansion and waste generation may lead to a breakdown. This situation is characterised by a degradation of the environment with major consequences for the health of the population and the destruction of the entire water resource, all of which are interconnected (except for the reservoir located in the upper plateau, which can unfortunately barely be drilled due to a deep lateritic cover).

Secondary effects could also be observed in terms of the reduction of fishing resources due to the pollution of the Niger River and the reduction of agricultural resources due to the pollution of soils and irrigation waters. Some of these impacts are already becoming evident, and the only question that remains is how much longer it will take for this catastrophic forecast to become a reality.

3.4 *Suggested actions to improve the situation*

The hydrogeologic situation in Bamako shows that the low permeability of the soils on the east side of the river only allows the pollution to reach the shallow aquifer through boreholes and individual wastewater treatment facilities. An initial short-term measure of protection could be to build ledges or sanitary seals in order to prevent run-off waters from flowing directly into the boreholes and to adapt the majority of incorrectly designed effluent facilities, namely by:

– increasing the capacity of the overused septic tanks,
– waterproofing every subsurface facility that collects domestic grey and black wastewaters,
– adapting the depth and the position of pits and soakaways in a way that ensures a protection zone between these facilities and pumping wells, and a barrier to the water table,
– making the population adhere to city regulations.

Other measures to be taken in the interim must contribute to restraining access to polluted waters. It is recommended to:

- cover the stormwater run-off channels with removable slabs to keep people away from polluted water, to prevent plants from growing in the channels, and to prevent the population from using them as garbage dumps,
- put wire netting a short distance upstream and downstream from sewer effluent sites in the Niger River. Currently these places are popular spots on the river bank because they allow easy access and are used for crossing the river. These measures could seem extreme, but they could be temporary until effluents into the river are treated, and until an effective awareness campaign takes place to educate the population.

In the long term, improved wastewater treatment will become essential due to population growth. The following measures are very general guidelines for the management of watewaters:

- install sewage treatment plants near the main effluent points in the Niger River (sewers and rainwater evacuation channels). Even, if it is not ideally suited for this, the stormwater system should be used for the transport of effluent, since it already exists;
- associated with this in the medium to longer term, it would be necessary to separate the effluent network and stormwater network to ensure the stability of the quality and quantity of wastewater to be treated;
- in the medium and longer term, implement small and simple biological treatment plants to treat the wastewaters from a few blocks of houses, in order to decrease the strain on the drainage network, which has not been designed to accommodate the rapid urbanisation of the city. The macrophyte ponds (Kane and Morel, 1998), successfully tested in Dakar and Niamey, or the constructed wetlands, seem to be well adapted to the climatic conditions and allow sufficient purification with a reasonable surface per inhabitant. The benefits of such decentralised facilities would be significant and include the possible *in-situ* re-use of part of treated waters and disposal of much cleaner water into the environment.

Finally, the priority is to regulate the industrial effluents by imposing treatment and water quality controls on industries such as the Daar-Salam power plant.

4 CONCLUSION

The lack of capacity, the inappropriate design and the lack of maintenance of wastewater purification systems in Bamako cause serious physico-chemical, bacteriological pollution of the shallow aquifer and surface waters. The widespread use of these water resources by the local population creates numerous public health concerns. However, the water quality of the Niger River remains chemically acceptable due to its high flow rate, in spite of the disposal without any treatment of all urban wastewater. The water quality of the deep aquifer also seems to be preserved. However, the very fast population and waste production growth, as well as the development of the Bamako-Koulikoro metropolitan area, will severely worsen the situation. In the short term, if no measures are adopted to treat the wastewaters directly disposed in the environment, all of the water resources could be contaminated. Some technical solutions adapted to the Malian context exist, namely macrophyte ponds and wetlands. The most important limitation in this situation remains the absence of a political will to protect the water resources and develop the infrastructures necessary for proper long-term management.

In the near future, industrial effluents must be efficiently controlled and the population educated on the problems and hazards of water pollution. Finally, instead of exploiting the deep aquifer, it seems more profitable to urgently improve the existing infrastructures already influencing the surface water quality, and to develop wastewater treatment facilities. Thus, it would be possible to go on using the potentially sustainable resource provided by the Niger River.

As for the rest of the Niger River valley, no major risk of water pollution yet exists, thanks to the low rates of population growth and urbanisation. However, the conclusions for the Bamako-Koulikoro axis can be applied to the cities of Mopti, Ségou, Diré, and Tombouctou. As shown by the water quality monitoring, specific pollution problems, linked to anthropogenic activities, are regularly observed. Thus, outside big cities, the water pollution problem is to a greater extent related to education than the technical issue of protecting water resources.

REFERENCES

Alpha A., Ait-ssi L., Banton O., Mariko A., Traoré A.Z., Villeneuve J.-P., 1991. Hydrogéologie et contamination de la nappe phréatique alimentant la ville de Bamako (Mali). Rapport ENI, INRS-Eau, Bamako.

Dégremont, 1978. Mémento technique de l'eau. Technique & Documentation, Paris, 734 p. Transcription in french legislation of European directives concerning wastewater effluent without treatment in surface waters.

EQUANIS, 1997. Environnement et qualité des apports du Niger au Sahel. Programme sur l'Environnement de la Géosphère Intertropicale (PEGI), INSU-CNRS-ORSTOM, Opération « Grands Bassins Fluviaux », 1992–1997, IRD, Paris.

Giorgi L., 1995. Etude préliminaire pour l'alimentation en eau de l'agglomération de Bamako à partir de la nappe des grès fracturés. ENI et Mission Française de Coopération et d'Action Culturelle, Bamako.

GIHREX, 2000. Gestion intégrée, Hydrologie, Ressources et Systèmes d'Exploitation. Grand Programme 21, Département Eaux Continentales, 1997–2000, ORSTOM-IRD, Paris.

GHENIS, 2001. Gestion Hydrologique et Environnementale du Niger Supérieur. Direction Nationale de l'Hydraulique et de l'Eau (DNHE), projet Delft Hydraulics, Bamako, 2000–2002.

IWACO, 1996. Projet pilote Guinée-Mali de création d'un système intégré de suivi hydro-écologique du bassin du Niger supérieur. Rapport final, Delft Hydraulics, juillet 1996.

IWACO, 1996. Etude préliminaire de la qualité des eaux souterraines phréatiques des zones maraîchères de Bamako. Rapport final d'évaluation, Bamako.

Kane M., Morel M.A., 1998. Le lagunage à macrophytes, une technique permettant l'épuration des eaux usées pour son recyclage et multiples valorisations de la biomasse. Sud Sciences et technologies, 001, IER, Ouagadougou: 5–16.

Picouet C., Dupré B., Orange D., Valladon M., 2002. Major trace element geochemistry in the upper Niger river (Mali): physical and chemical weathering rates and CO2 consumption. Chemical Geology, 185: 93–124.

Zallé D., Maiga O., 2002. Zones inondables en milieu urbain: conflits fonciers et concurrences pour l'espace maraîcher à Bamako (Mali). In Orange *et al*. (ed.): Gestion intégrée des ressources naturelles en zones inondables tropicales, Colloques et Séminaires, IRD, Paris: 49–60.

13

Statistical assessment of groundwater quality in Bamako City, Mali

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ABSTRACT: The groundwater in Bamako City showed 3 hydrogeochemical facies: HCO_3^-/Ca^{2+}/Mg^{2+}; Cl^-/Ca^{2+}/Mg^{2+} and HCO_3/Na^+/K^+. Statistical methods, including a factor analysis, were applied to determine factors influencing the groundwater quality in Bamako City. Three factors, termed 'Natural', 'Pollution' and 'Biological Contamination' relative to the geology and the anthropogenic processes, were identified and mapped using GIS features.

1 INTRODUCTION

This paper results from the Mali component of the UNEP & UNESCO project which focused on urban pollution of superficial and groundwater aquifers in Africa. The aim of the project is to assess the impact of urban pollution on groundwater, evaluate the evolution of some pollutants, publish a bulletin on water quality and provide decision makers with validated information regarding groundwater quality management.

Bamako, the capital of Mali in West Africa, is located in the basin of the Niger River. The city covers about 240 km^2 (Fig. 13.1). It is located between 12° 29′ to 12° 43′N and 7°55′ and 8°05′ W. Mali is divided in 6 districts. The Niger River crosses the City and divides it into north and south banks. The sandstones hills of the Mont Mandings, with an elevation varying from 30–100 m, surrounds the city. This limits the city's extension from the west. Urbanisation is therefore progressing in the south bank of the Niger River. Some non-perennial streams (Oyanko, Banconi, Korofina, Sogoninko, etc.) flow into the Niger River.

The population of Bamako is 1,016,000 (1998), and the birth rate is around 6.5%, while the availability of drinking water is decreasing from year to year.

Water supply in the city is assured by the following means:

- The National Company of Energy (Energy du Mali EDM) which distributes water from the Niger River after relevant bacteriological and chemical treatment.
- Water directly from the river and stream, without any chemical and bacteriologic treatment.
- Water from shallow boreholes with depths varying from 2 to 10 m. These shallow aquifers are contaminated in historically occupied and populated areas.
- Water from deep boreholes in deep fissured or fractured aquifers.

The climate in Bamako City is sudanian-type, characterised by a rainy season from June to October and a dry season from November to May. Bamako City receives 70 to 80 days of rain per year, with the highest rainfall occurring in August. The average annual precipitation is around 1000 mm, while the yearly temperature average reaches 27 degrees.

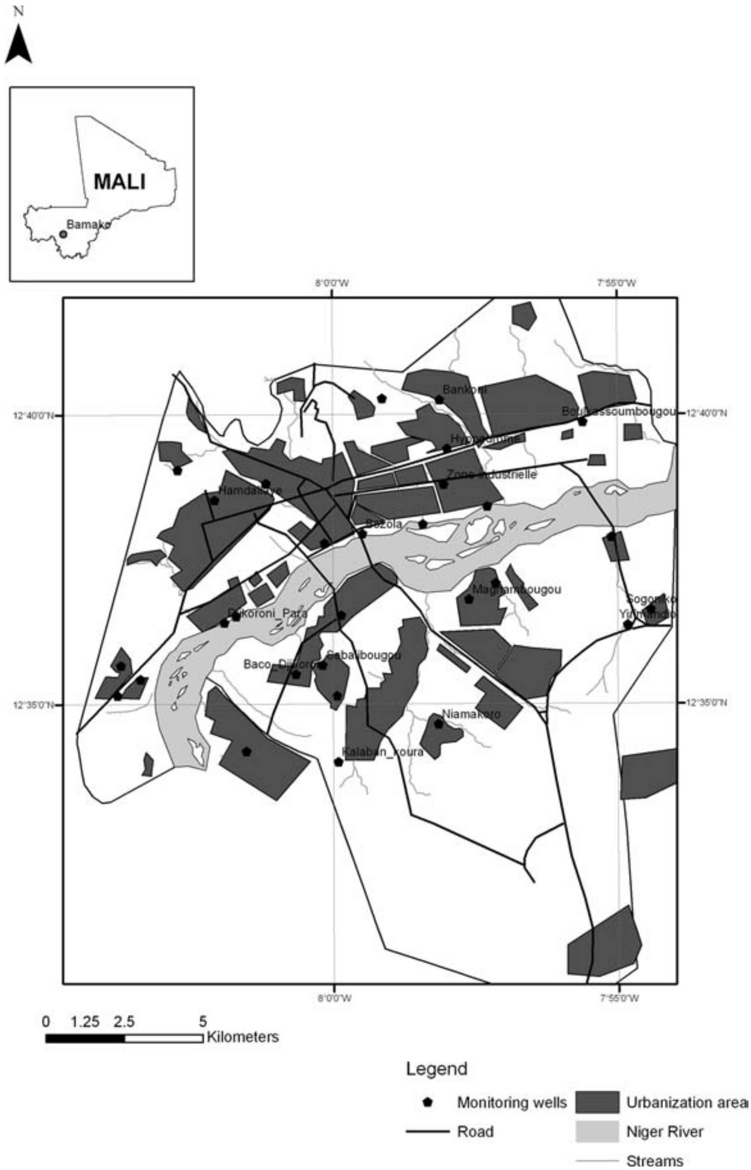

Figure 13.1. Location of study area.

2 GEOLOGICAL SETTING

The geology of Bamako is described by many geologists, including Dars (1960), Carrère (1972), Travi (1980), Giorgi *et al.* (1995) and the Geology Department of the University of Bamako. The area is situated at the southern edge of the Taoudenit Basin, which covers the largest part of

Mali. The area is near the boundary between the sedimentary deposits in the north and crystalline rocks of the south. The crystalline rocks are observed under the sediments less than 50 km south of Bamako.

2.1.1 *Crystalline rocks*

These generally consist of early Precambrian granites and are widely overlain by lateritic deposits.

2.1.2 *Sedimentary formations*

Sedimentary formations are mainly represented by sandstone from the bedrock of quaternary lateritic and alluvium deposits.

The sandstone constitutes two facies groups:

The Sotuba Group, dated from late Precambrian. This group forms the bed of the Niger River and is sometimes directly embedded on the crystalline basement.

The Koulouba Group, of Palaeozoic age, is located in the north bank of the Niger River and constitutes the sandstone hills surrounding the city.

2.1.3 *Dolerites*

Dolerites are intruded by a wide range of faults in the groups of Koulouba and Sotuba.

2.1.4 *Tectonics*

Fractures and faults control the structure of the study area. Their presence vary, therefore there are zones which are highly affected by fractures whereas some areas are not affected at all/very little.

The principal directions of fractures are: N110°–N140°; N0°–N20°, N80°–N100°, N30°–N70° and N150°–N170°. The following directions apply to the main faults:

- The Sikoroni Fault (E–W)
- TheYirimadjo Fault (NW–SE with injection of dolerite)
- The Lassa–Koulou Fault (ESE–WNW)

Based on a cross-section of the site's hydrogeological structure, it is possible to distinguish several aquifer layers at the regional level. The upperlayer of these is made up of two hundred metres thick sands and unconsolidated material, verified through drilling programmes and boreholes. The three deeper layers separated by discontinuous clay zones.

There are hydraulic interconnections between all these horizons. In other words, the pollution of one of the aquifers (particularly the surficial aquifer) will most probably influence the others. The main characteristics of these different aquifers are given in the sections that follow.

2.2 *Hydrogeology*

The hydrogeological conditions are directly related to the geological and tectonic structures of the area. The hydrogeology of Bamako City is composed of two aquifers system, namely a shallow aquifers system in superficial deposits and a deep fractured aquifer system in fractured or fissured sandstone. The two systems are hydraulically linked through leakage from the upper to the lower one.

2.2.1 *Shallow aquifes systems*

The shallow aquifer systems consist of two different topographical areas; the lateritic hills of Koulouba and the basin of the Niger River in Bamako.

2.2.1.1 Aquifer in lateritic hills

The aquifer varies from 5 to 20 m in depth and is highly fissured, resulting in a high permeability. It is one of the most productive aquifers in the area and shows good recharge. Several important springs including Sikoroni, Lido and Ngomi originate from this aquifer.

2.2.2 *Aquifer of the Niger River Basin in Bamako*

The extension is around the banks of the Niger River. The population exploits it through shallow wells and it is influenced by water-level fluctuations.

Through pumping tests in some shallow wells, transmissitivity values were recorded (Aboubacrine *et al*., 1991), varying between 4×10^{-6} and 2.5×10^{-5} m/s in the north bank of the Niger River and 3×10^{-5} to 2.5×10^{-4} m/s in the south bank.

Groundwater level contour maps (Travi, 1981; Aboubacrine *et al*., 1991) have shown that the isohyets are parallel with the Niger River and the groundwater flow converges towards it.

2.2.3 *Fissured and fractured aquifer*

Although the bedrock is consolidated sandstone of the Sotuba or Koulouba group, which is almost impermeable, it becomes a good aquifer due to its structural tectonic alteration, characterised by fractures of varying orientations. This aquifer is highly utilised for drinking water, especially in the suburban area where it is exploited through deep boreholes. The yields of these boreholes can be up to 100 m^3/h. The transmissivity varies from 4×10^{-5} to 3.5×10^{-4} m/s.

3 APPROACH

In order to determine the evolution of groundwater quality, measurements of the electrical conductivity, pH, temperature, and water levels in 8 monitoring boreholes were taken monthly during the dry season and weekly during the rainy season.

From 2002 to 2004 measurements were also recorded in 30 sampling boreholes in both the shallow and deep aquifer to understand the behaviour of water-level fluctuations and the evolution of groundwater quality. The ions analysed in this project include: Ca^{2+}, Mg^{2+}, Na^+, K^+, HCO_3^-, CO_3^{2-}, Cl^-, SO_4^{2-}, total Fe, Zn^{2+}, Cu^{2+} as well as total and faecal coliforms.

The results of chemical analyses are presented in the Piper Diagram (Fig. 13.4) for the major elements. To determine the factors controlling groundwater quality a factor analysis was done and the contribution of each factor (factor scores) determined at every site. This step is very important for the mapping of the geographical distribution of the interpreted factors.

4 RESULTS AND DISCUSSION

4.1 *Groundwater quality*

Figure 13.2 shows the variation of the electrical conductivity (EC) with precipitation. The figure indicates that the EC is higher during the rainy season, which is unexpected. Conductivity is generally lower with higher precipitation values due to dilution. In the case of Bamako City, the unsaturated zone is rich in chemical and organic constituents, which can be easily dissolved in the high recharge period, i.e. during the months of July, August and September. This may explain the positive relationship between higher precipitation values and increased EC levels.

Figure 13.3 indicates the boxplot of the evolution of EC and nitrate from 1990 to 2002. In both cases there are upward trends, indicating an increase of the values over time which has led to the deterioration of the groundwater quality. The data distribution for nitrate particularly was somewhat skewed due to the high values at some sites. The EC results showed a normal distribution. The pH-values vary from 3.5 to 8, with median values between 5 to 6.5, which corresponds to slightly acidic waters.

4.2 *Groundwater classification*

The 30 water samples monitored in 2002 were analysed using the Piper Diagram (Piper, 1940) in Figure 13.4. The ions classified by the diagram are Ca^{2+}, Mg^{2+}, Na^+, K^+, HCO_3^-, CO_3^{2-}, Cl^- and SO^{2-}_4. The concentrations of Na^+ are generally grouped with K^+ and those of HCO_3^- with CO_3^2.

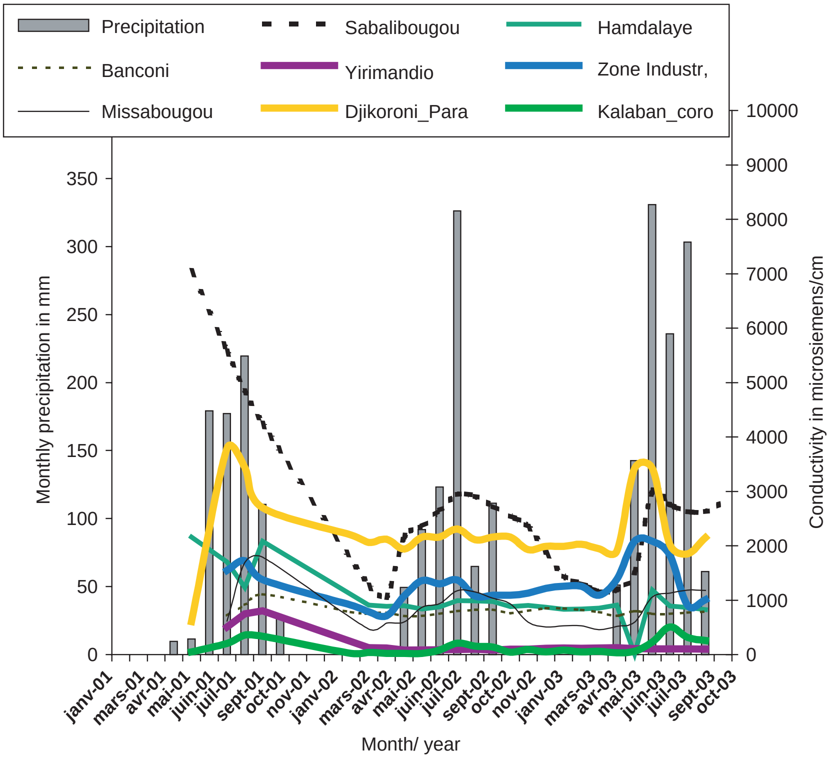

Figure 13.2. Graphs showing the evolution of the EC in eight monitoring wells and rainfall from 2001 to 2003.

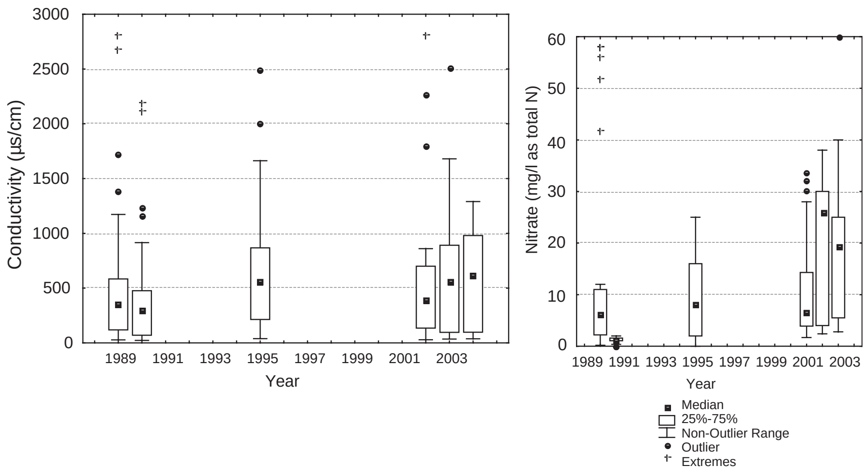

Figure 13.3. Boxplots showing the evolution in values of EC and nitrate (as total N) value from 1990 to 2002.

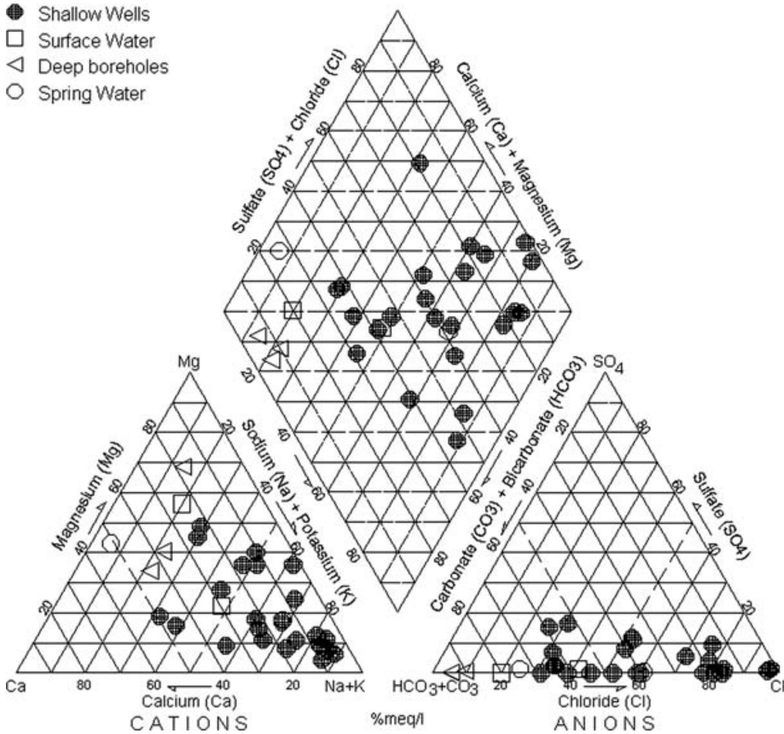

Figure 13.4. Piper diagram showing the hydrochemical facies in groundwater and surface water.

Three groups of groundwater facies were observed:

1. HCO_3^-—Ca^{2+}/Mg^{2+} facies type characterised the three samples for deep boreholes, one sample of the surface and one of the spring waters.
2. Cl^-—Ca^{2+}/Mg^{2+} facies type characterised 50% of sampled boreholes.
3. HCO_3^-—Na^+/K^+ characterise two samples.

5 FACTOR ANALYSIS

A factor analysis is often applied to hydrochemical groundwater data (Usonoff and Gusman, Gusman, 1989; Rao *et al.*, 2001). This is done as follows: the correlation matrix, i.e. the array of correlation coefficients for all possible pairs of variables is calculated. The matrix is then diagonalised and its principal components (eigenvectors) obtained. Factor I will be related to the largest eigen value and can explain the greatest amount of variables in the data set. The second factor (orthogonal and uncorrelated with the first one) explains the greatest of the remaining variables, and so on (Johnson and Whichern, 1992).

The factor analysis was performed using the Statistical package, while factor scores were mapped using ArcGis 8.3.

The Principal Component Analysis was adopted as an extraction method of factors (Harman, 1976). The variables for the factor analysis were Ca^{2+}, Mg^{2+}, Na^+, K^+, HCO_3^-, CO_3^{2-}, Cl^-, SO_4^{2-}, total Fe and total and faecal coliforms. The correlation matrix is shown in Table 13.1.

Table 13.1. Correlation matrix of shallows and deep well data of Bamako City in 2002.

	pH	EC	TH	Alk	Ca	Mg	Na	K	HCO_3	SO_4	Cl^-	NO_3	Total Coli	Fecal Coli
pH	1													
EC	.364	1												
TH	.671	.813	1											
Alk	.664	.646	.923	1										
Ca	.390	.697	.602	.305	1									
Mg	.641	.704	.950	.975	.326	1								
Na	.244	.884	.715	.625	.444	.693	1							
K	.328	.962	.797	.650	.610	.721	.930	1						
HCO_3	.664	.646	.923	1.000	.306	.975	.625	.651	1					
SO_4	.546	.676	.789	.821	.284	.847	.783	.727	.821	1				
Cl	.249	.969	.722	.579	.590	.644	.933	.945	.578	.661	1			
NO_3	−.187	.604	.240	−.057	.672	.031	.592	.606	−.056	.100	.640	1		
Total Coli	−.031	.228	.061	.097	−.12	.133	.512	.312	.096	.346	.329	.273	1	
Faecal Coli	.060	.190	.121	.241	−.27	.261	.307	.232	.240	.236	.268	−.02	.570	1

The factor analysis showed three factors explaining about 85% of total variance. The variance explanations of the factors were 39.15 % in Factor I, 31.46% in Factor II and 13.77% in Factor III. The communalities of all parameters were very high, explaining that the three factor models adequately explain the variance of almost all the parameters.

The HCO_3^-, alkalinity, Mg^{2+}, TH and pH loads were higher in Factor I. The EC, Cl^-, and NO_3^- loads were higher in Factor II and total and faecal coliform loads were higher in Factor III.

The maps of the distributions of factors cores showed that Factor I is relative to the natural factor. It reflects the geology of the area. The positive factor scores show the Koulouba group, and the negatives ones the Sotuba group.

Factor II, which shows higher loads in the values of the EC, chloride and the nitrate, can be expressed as the Pollution Factor. It coincides with the most polluted areas.

Factor III can be expressed as the Biological Contamination Factor, which indicates that the most populated and contaminated areas are characterised by the higher loads of total and faecal coliforms.

The distribution of the factor scores is illustrated in Figure 13.5.

6 CONCLUSION

The values of electrical conductivity of the 8 monitoring boreholes Bamako City increase in the rainy season. All he values, including those of nitrate, showed an upward trend over time, indicating a deterioration of groundwater quality over the time.

The hydrochemical facies showed three types:

- HCO_3^-—Ca^{2+}/ Mg^{2+} facies type
- Cl^-—Ca^{2+}/ Mg^{2+} facies type characterised 50% of sampled wells
- HCO_3^-—Na^+/ K^+

The factor analysis, based on the monitoring of 2002 showed three factors of importance, corresponding to the natural, pollution and biological contamination zones of the area under investigation. The geographical distribution of factor scores showed that the first factor is related to the geology, the second to pollution and the last to biological contamination.

It is hoped that further studies will include more monitoring samples and would extend to the deep aquifer, which remains the basic resource to consider for long-term water supply to Bamako City.

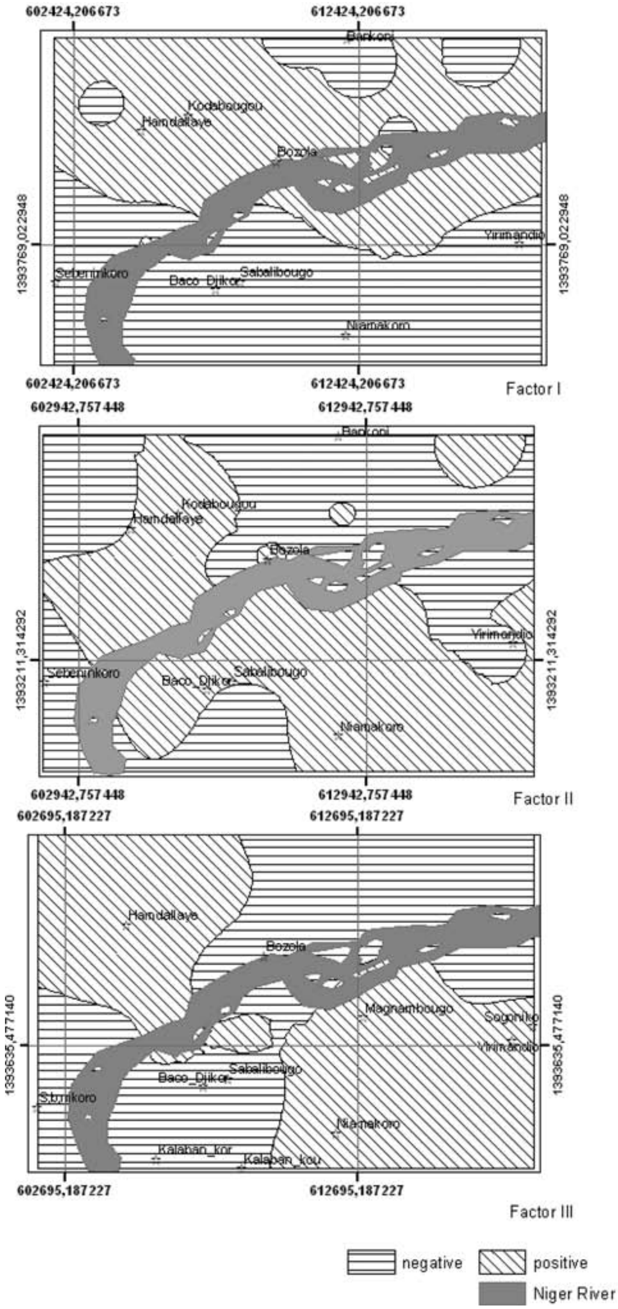

Figure 13.5. Distribution of factor scores data of hydrochemical data of Bamako City in 2002.

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REFERENCES

Aboubacrine Alpha, Amadou Z. Traoré, Adama Mariko, Olivier Banton, J. P. Villeneuve Lahcen Ait-Ssi (1991): Hydrogeologie et contamination de la nappe phréatique alimentant la ville de Bamako (Mali) INRS eau Ste Foy, Quebec Canada

Carrère J.M. (1972) Stratigraphie des formations gréseuses attribuées au protérozoïque supérieur de la région de Bamako, Mali B.S.G.F. (7), XIV

Dars R. (1960) Les formations sédimentaires et les dolérites du Soudan Occidental (Afrique de l'ouest) Thèse de Doctorat és science, Paris

Johnson & Whichern (1992) Applied multivariate statistical analysis 3rd Edition Prentice Hall, Englewood Cliffs, NJ

Loic G. Traoré A.Z. 1995. Etude préliminaire pour l'alimentation en eau de l'agglomération de Bamako à partir de la nappe de grés fracturé (technical Reports)

Piper A.M. 1944. A graphic procedure in the geochemical interpretation of water analysis Trans.Am.geophys.Union Volume 25, 914–923

Rao N. Subba, Rao, J. Praskasa, Devadas D. John, Rao K. Srinivaso Krishna C. 2001. Multivariate analysis for identifying the governing factors of groundwater quality Journal of Environmental Hydrology Volume 9 (16)

Travi Y., Oller G. 1980. Contribution à l'étude hydrogéologique des environs de Bamako Bulletin de l'I.F.A.N T.12, sér. A n° 1, 1980

Usunoff, E.J. & A. Guzman-Guzman 1989. Multivariate analysis in Hydrochemistry and example of the use factor and correspondence analyses. Groundwaters 27 (1):10 27–34

14

Spatial and temporal variations of groundwater pollution in Ougadougou City, Burkina Faso

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ABSTRACT: Ouagadougou City is built on crystalline rocks dating back to the Proterozoïc era. These rocks are covered by weathered material. Demographic explosion, rapid extension of the town, and particularly its undeveloped peripheral slums, lack of appropriate sanitation networks are exposing aquifers to the risk of contamination. Aquifers contribute about 60% of the population's water supply and especially to the poor. With this in mind, the present study evaluated the situation in order to develop an early warning system. The discontinuous nature of the area led to a comprehensive evaluation of 1800 wells and 510 boreholes, which would be used to obtain water for analysis (nitrate content). This would enable the assessment of the range of spatial pollution and its comparison with the theoretical vulnerability map. The surficial aquifer, tapped by wells, is currently the most contaminated. The main factor influencing vulnerability here is the thickness of the unsaturated zone. Regular monitoring indicates that bacteriological contamination is mainly derived from faecal or household origin, and that the contaminating products are periodically washed into the aquifers during months with the high rainfall, i.e. July, August and September. Nitrate concentrations increase before they are reduced by dilution or diffusion during this period. From this, suggestions can be made to amend the monitoring schedule.

1 INTRODUCTION

Ouagadougou City, like many other capital cities of the West African subregion, has seen a rapid demographic growth and spatial expansion, especially around the town where the development of vast peripheral zones has been observed.

These zones are sometimes characterised by very high population densities and a lack of sanitation infrastructure (waste dumps, latrines and septic tanks). The lack of appropriate drainage systems for rain waters and liquid waste pose a great contamination risk for the aquifers providing 60% of the drinking water supply to the city.

Bearing in mind this worrying situation, UNESCO in partnership with UNEP, UNCH and ECA has designated the hydrogeological laboratory of Ouagadougou University within the project entitled 'Urban Pollution Of Surficial And Groundwater Aquifers In Africa', to conduct a study on the quality of Ouagadougou groundwaters in order to identify not only the vulnerable areas, but also the physico-chemical factors that are the source of water quality degradation. The final objective of this study is the establishment of an early warning system for groundwater pollution.

2 NATURAL AND ENVIRONMENTAL CONTEXT OF OUAGADOUGOU

2.1 *Background of study area*

Located in the centre of the country, Ouagadougou City is situated between 1°28′ and 1°36′ west longitude and 12°20′ and 12°26′ north latitude. It currently extends to a 15 km radius and is made

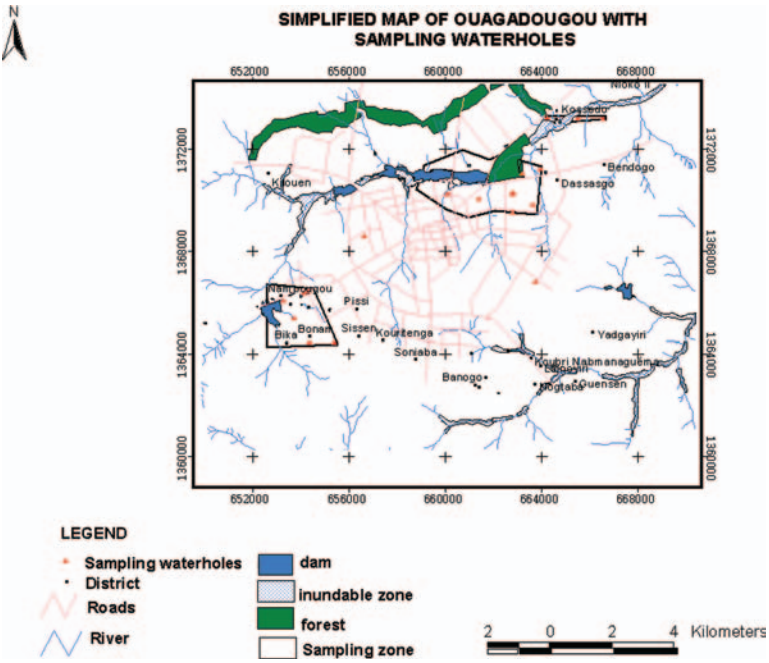

Figure 14.1. Representation of study zones with pollution monitoring sites.

up of six districts and thirty (30) sectors, the populations and areas being unevenly distributed (Fig. 14.1).

Estimated to have 441,514 inhabitants in 1985, the population has increased to 1,200,000 inhabitants by the year 2000 with a growth rate of 8% per year. The most densely populated are the peripheral areas, where the people are engaged in socio-economic activities that are less remunerating and where the provision of clean water networks is limited or non-existent. Populations are thus obliged to obtain water from wells or borehole equipped manual pumps.

The contamination status study and follow-up have been done in three areas, consisting of 27 sites of measurement and sampling for analyses. The sites include an upstream area, a central and a downstream release where a great number of the city factories are found.

2.2 *Topography and climate*

Ouagadougou has flat, slightly undulating topography with an average altitude of 300 m. From the climatic point of view, the seasonal cycle is controlled by the alternating Sahara winds and oceanic monsoons resulting in a climatic regime of the Soudano-Sahelian type characterised by two very contrasting periods:

- Rainy season from June to September;
- Dry season (8 months) from October to May.

In the last few decades, rainfall has been on a downward trend. The inter-annual amplitude variations have however eased with the high recorded rainfall between 600 and 850 mm. It is especially during the very wet months of the year (July–August and September) that aquifer recharge is possible.

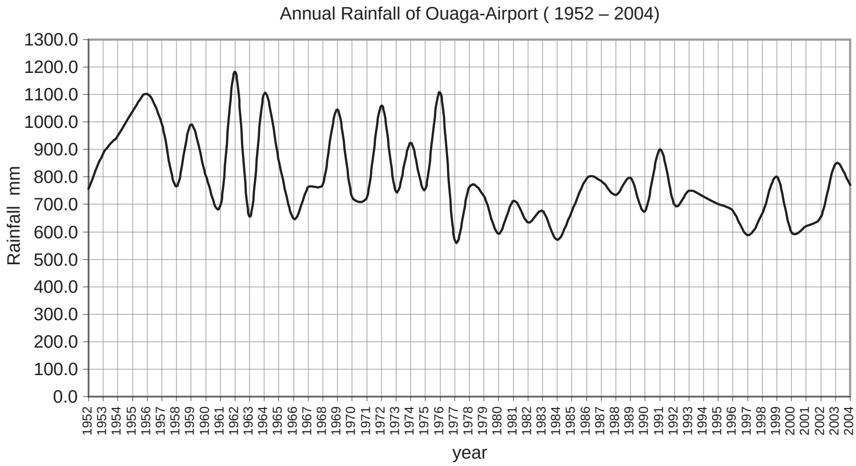

Figure 14.2. Representation of the downward trend of rainfall in Ouagadougou.

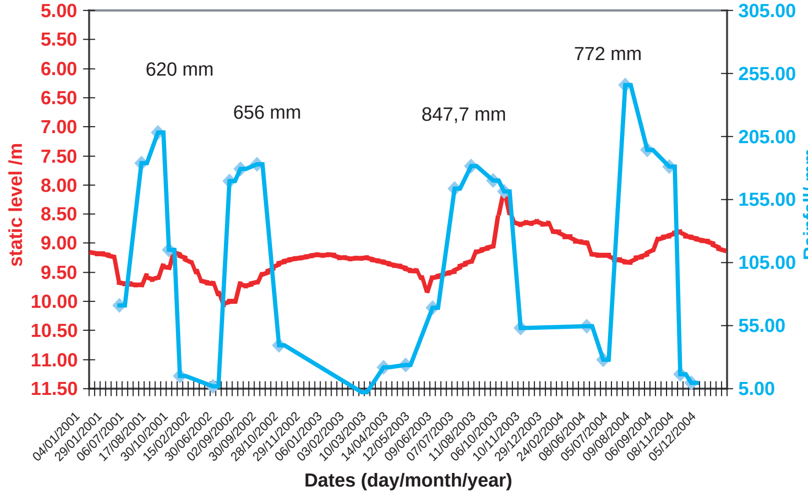

Figure 14.3. Rainfall and piezometric level variations, 2001–2004 (CIEH Piezometer).

The combined effects of temperature and relative humidity lead to high direct evaporation and evapotranspiration, thus reducing rainfall effectively infiltrating into the groundwater. The monthly evapotranspiration was about 165.44 mm and the trough evaporation 249.9 mm per month during the observation period of 30 years (1961 to 1990).

These losses have a negative effect on surface waters (lakes and dams) and the groundwater reserves (drying up of wells).

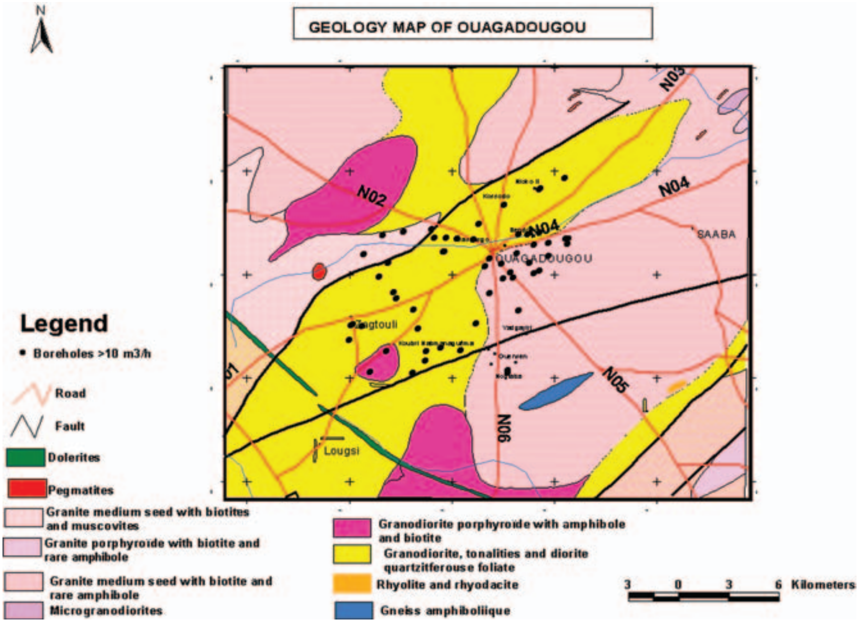

Figure 14.4. Geology Map of Ouagadougou.

2.3 *Geology and hydrogeology*

2.3.1 *Geology*

Ouagadougou lies on crystalline rocks mainly made up of granitoids:

– some granodiorites, tonalities and foliate quartziferous diorites are located in the extremity of the south-west part of the town;
– some granodiorites with amphibole and biotite, combined with a range of gneissic rocks with amphibole and biotite, located in the north-west of Ouagadougou;
– some granites of medium grain biotitic and amphibole and porphyroïde granite with biotite and rare amphibole. These two facies constitute a big entity dominating the west side of the town.

The topography is cut by some intrusions of dolerites, pegmatites, alpites, rhyolites and rhyodacites. This set of intrusions in the Baoulé-mossi domain of 2215 and 2060 Ma in age, have an associated zone of general metamorphism and a syn granitization. The tectonics have resulted in an intense fracturation following two predominant directions: NE-SO and NO-SE (Fig. 14.4).

The number of boreholes with high yields (higher than 10 m^3/h) from the Office National de l'Eau et de l'Assainissement (ONEA) coincides with the major fractures or their intersections as identified by geophysics. This suggests that fractures hidden by the alterites have been greatly underestimated as groundwater targets by surface geological mapping.

2.3.2 *Hydrogeology*

In the areas of the crystalline platform, the only groundwater reserves are linked to alterites and the fractures that affect the bedrock. The level of the alteration profile horizon allows a distinction between the two aquifers. The first is a shallow aquifer overlain with lateritic cuirass or alluvium, while the second is deeper and corresponds to the fragmented band of the substratum (Fig. 14.5).

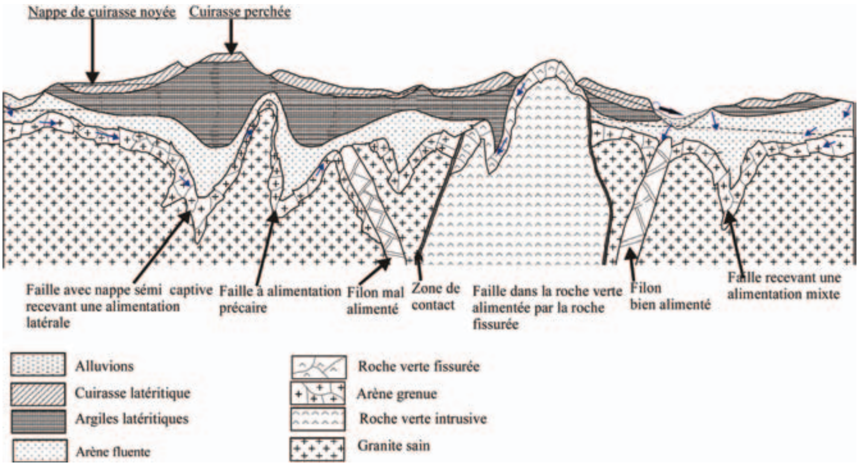

Figure 14.5. The aquifer stratigraphy around Ougadougou (Savadougo, 1984).

At the Ouagadougou area level, the alteration profile is well developed. The lowest levels of lateritic cuirass from the Quaternary age are cut by water courses flowing through the city. These surface waters are the major recharge drivers for the shallow aquifers. The shallow aquifers are widely tapped by many traditional wells and boreholes. Fewer boreholes, most of which are located through geophysical prospecting, exploit the arena aquifers and fractured rock. Where major fractures are encountered, these borehole are very high-yielding and act as a back-up water supply for Ouagadougou's drinking water.

3 METHODOLOGY

Ouagadougou City is divided in two by the Kadiogo River, on which four dams are built to supply drinking water to the city.

The selection of sampling sites to characterize and monitor the quality of groundwater was based on the aquifer stratigraphy, the sewerage and sanitation network and the socio-economic activities in three target areas (Fig. 14.1):

– an entrance area corresponding to the south-west part of the town, the upstream of Boulmiougou Dam, which is the point where waters start flowing towards Dams 1, 2 and 3 contributing to the water supply of the town. It is a under-developed area and there are no industrial activities The principal consideration is that the local inhabitants of the area practice market gardening using fertilisers and pesticides;
– Area 2, located downtown near dam No 3 and the central channel draining rainwaters, the release of the central market and its surroundings;
– Area 3, the discharge system is for almost all of the town's waters in addition to the release of the Kossodo industrial flow. This area is located downstream of the industrial area.

Samples and *in-situ* measurements have been made consecutively during four complete hydrological cycles. Field measurements were taken for pH, temperature, conductivity and water levels. Chemical analyses were done for major elements and nitrates were identified for contamination monitoring. Given the discontinuous nature of the aquifer system, a general study of pollution through the whole hydraulic system (1800 wells and 510 boreholes) of the town has been made, using the concentrations of nitrates as a contamination indicator.

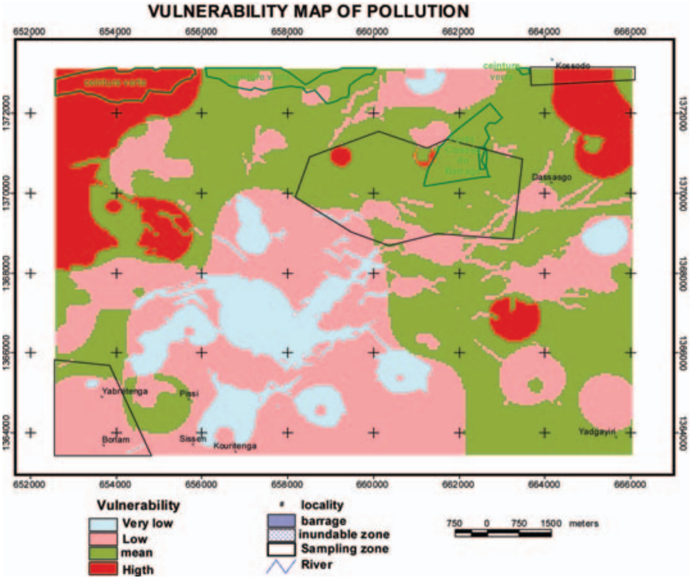

Figure 14.6. Vulnerability map of Ouagadougou aquifers.

4 ANALYSES AND DISCUSSION OF RESULTS

4.1 *Mapping of Ouagadougou groundwater vulnerability*

The vulnerability of the aquifers has been determined through a vulnerability map using the simplified DRASTIC method and taking into account the following parameters:

- the topography
- fracture density
- the thickness of the vadose zone (in this geological context, the thickness of the vadose zone corresponds to the depth of piezometric level).

The vulnerability map obtained (Fig. 14.6) shows that the most vulnerable areas of the town are located in the west, north-west and industrial area of Kossodo, as well as the south-east part of the city. The industrial area is located in a vulnerable area.

4.2 *Distribution maps of nitrate concentrations in the wells and boreholes of Ouagadougou*

Water points have been identified in combining systematic measures of pH, conductivity, the concentration in nitrates and the distance between the wells and the latrines, the distance separating sinks and the increasing volumes of existing household refuse.

It was observed that, except the south center of the town, well waters contain high concentrations of nitrates in areas already identified as vulnerable (Fig. 14.7). The vulnerability map can be considered a relevant tool for contamination prevention. We have also notice that boreholes (Fig. 14.8) are clearly less contaminated than well waters, of which more than 50% are higher than 50 mg/l nitrate. We can deduct that, among the factors influencing the aquifer's vulnerability to pollution, the thickness of the non-waterlogged area constitutes the dominating factor.

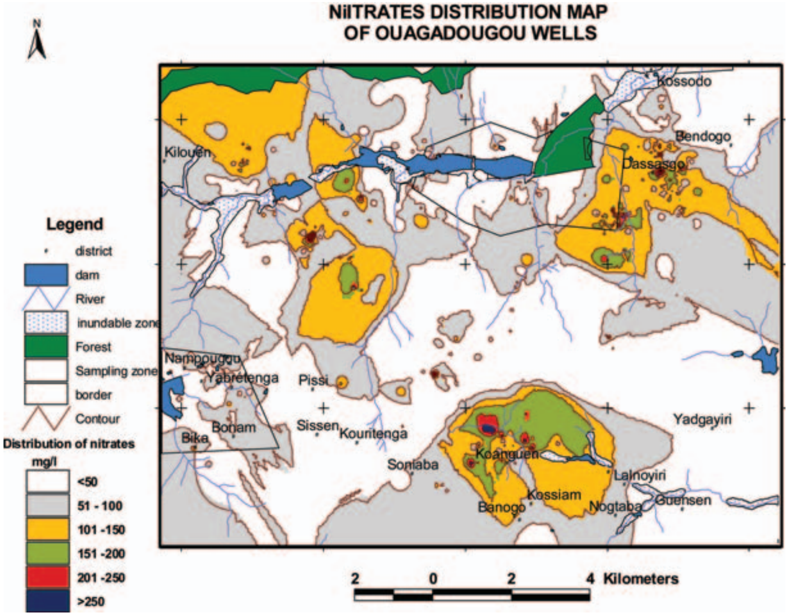

Figure 14.7. Map of spatial distribution of nitrates in 1800 wells in Ouagadougou.

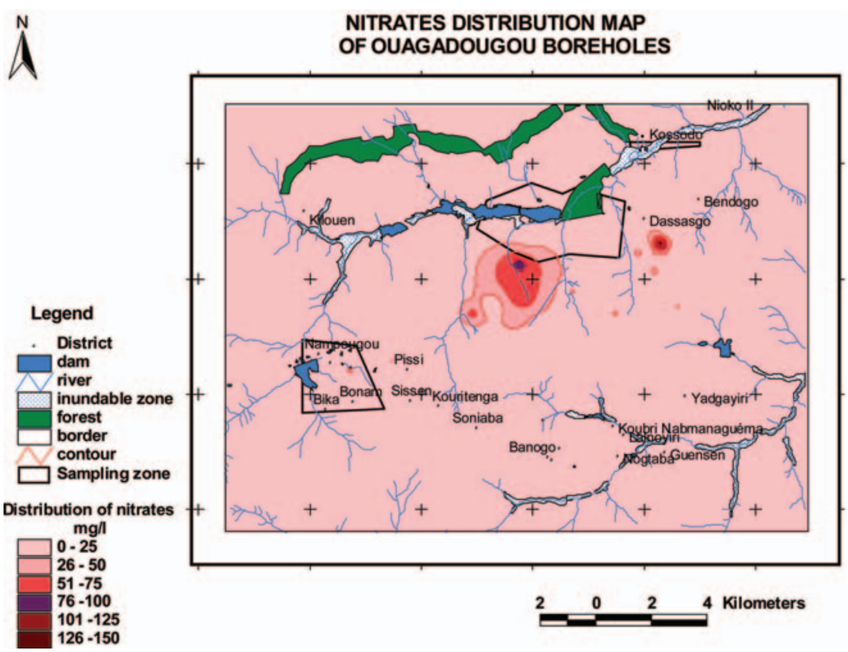

Figure 14.8. Map of spatial distribution of nitrates in 510 boreholes in Ouagadougou.

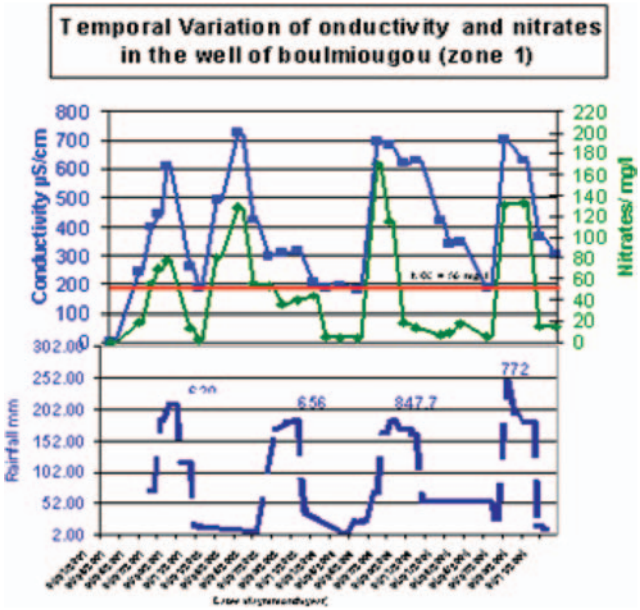

Figure 14.9. Curve of temporal variation of conductivity/nitrate and rainfall (2001–2004) in the wells of Boulmiougou/Zone 1.

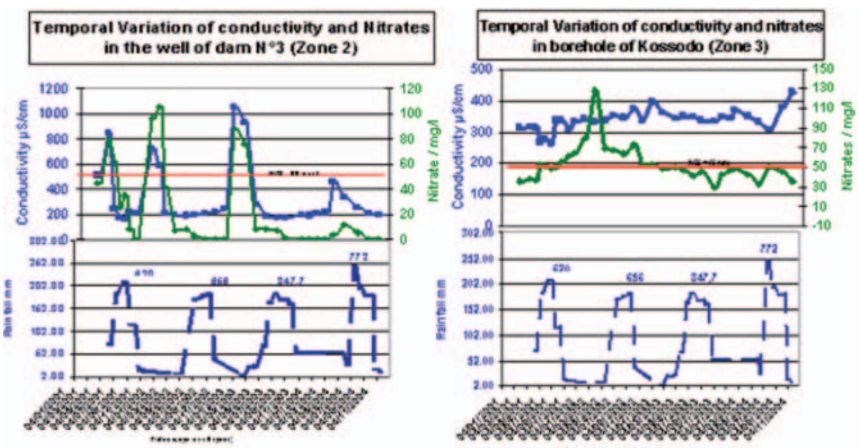

Figure 14.10. Temporal variation of conductivity/nitrate and average rainfall 2001–2004 (In the wells of Dam No. 3 Zone 2) and in the borehole of Kossodo (Zone 3).

4.3 *Migration of pollution and temporal variation*

The permanent follow-up of nitrate concentrations, conductivity and precipitation have allowed the setting up of Figure Graphs 9, 10, and 11. These graphs, together with the one of Figure 14.3, show that there is a good correlation between the concentrations in nitrates, conductivity, variations

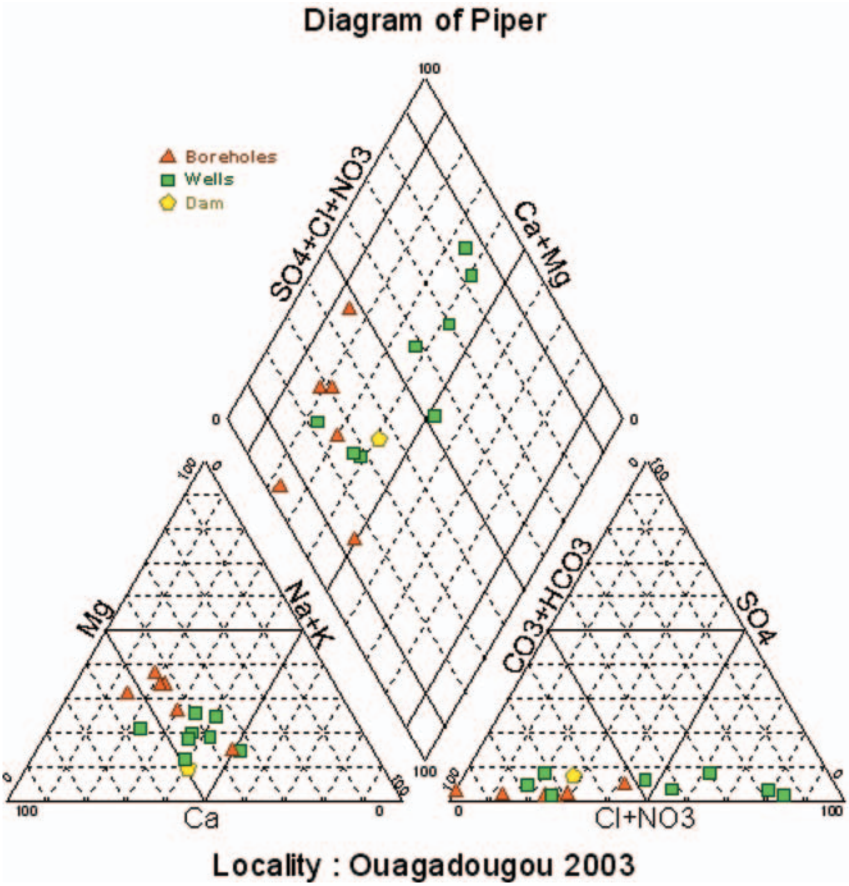

Figure 14.11. Chemical characteristics of Ouagadougou waters.

of piezometric and precipitations. The graphs clearly show that contaminating agents flow to the aquifer during the recharge period some weeks after the rains of July and August.

The nitrate concentrations then decline simultaneously with the piezometric levels, which reduce under the effect of evapotranspiration. Evapotranspiration does not induce a surconcentration of the contaminating agents that seem to dissolve and spread into great reserves of waters compared to the season contributions of the recharge period. Generally speaking, the basic concentrations of nitrates are stable and do not change from one year to the next on the level of wells that tap the superficial aquifer, as well as the boreholes that tap the deep aquifer. The highest peak of contamination everywhere can be observed in high water. The scope of significant contamination does not seem to be associated with annual precipitation, particularly in Areas 2 and 3, where the last good pluviometry has flown towards the aquifers with fewer nitrates. Observations show that these polluting agents mostly come from latrines and household refuse. As Travi (1997) indicated, a significant concentration of nitrates is obviously the result of faeces and household refuse in relation with the type of housing. In Area 3, great concentrations of nitrate originate from the industrial release of tanning and brewing industries.

During the study period, sites were monitored on a monthly basis. From the graphs it is suggested that we can reduce the monitoring frequency to three measure a year: one in May before the peak of the curve, the second on the top of the curve in August and the last one in November, on its lower section.

4.4 *Physico-chemical characteristics of Ouagadougou groundwater*

All water samples of the 27 monitoring sites have been analysed. These waters show a conductivity varying between 100 and 600 S/cm and a pH between 6 and 7.5. Throughout the year, the groundwater temperatures vary between 27 and 32°C. Most of the borehole waters are calcic bicarbonate and lightly sodic. Well waters are also calcic bicarnonate and sodic with more high contents of chloride and mostly nitrates (Fig. 14.11). It can be observed that everywhere the deep borehole waters are clearly less contaminated than exposed well waters.

5 CONCLUSION

This study allows us to show the efficiency of vulnerability maps in the management of groundwater and aquifer pollution risks even in crystalline rocks.

The discontinuous nature of aquifers leads to contaminated areas that are in close relation with their immediate environment. Generally, we notice that well waters that tap superficial aquifers have the greatest risk of contamination. The contaminants reach the aquifer after the high rainfalls of July and August, leading to a highest point of cyclical pollution during the first few months and then decreases whilst the piezometric level drops due to evapotranspiration. The evapotranspiration in the monitored areas does not induce a high concentration of contaminants that are dissolved or spread into large volumes of water compared to the rain waters brought by infiltration. It is the successive rainfalls that increase polluting agents and flows to weak mineralised and calcic bicarbonate or sodic waters. The high nitrate concentrations no doubt originates from faeces and household refuse in relation with housing (Travi *et al*., 1997).

In a crystalline platform area like Ouagadougou, the relationship of different aquifers allows migration of pulses of contamination. The status of each area will depend on its immediate environment and its vulnerability will be linked to the thickness of the vadose zone which protects it. In setting up a monitoring system for early warning it is therefore important to take into account each study area and its related environment. The monitoring of the contamination in sahelian climate reduces to three periods each year: before, during and after the rainfall season.

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REFERENCES

BGS, CNA, SAPAL, & UACH., 1996. Effects of wastewater on urban groundwater resources of Leon, Mexico. Final report february 1996. British Geological Survey. Technical report WD/95/64.

Casting C. Chevremont P. *et al.* 2003 Notice explicative de la carte Géologique à 1/200 000., Feuille ND-30-V de Ouagadougou. 1ère édition.

Forslund J. 1986. Groundwater quality today and tomorrow World. Health. Statist quart., 39p.

Klimas Algirdas A., 1995. Impact of urbanisation and Protection of water resources in the Vilnius District Lithuania : Hydrogéology Journal vol 3, n° 1, pp. 24–35.

Moukolo N., Gaye., C.B. 2001. Problèmes des nappes phréatiques par les rejets domestiques dans les métropoles d'Afrique Noire. Cas de la nappe de Brazzaville au Congo. Science et changements planétaires. Secheresse Vol 12, n° 3, pp. 175–182.

Nkotagu H., 1996. Origin of high nitrate in ground-water in Tanzania. Journal of African Earth Sciences, Vol 21, n° 4, pp 471–478.

OMS., 1986. Directives de qualité pour l'eau de boisson Vol 3, 122 p.

ONUGBA Abraham., Bernard., BLAVOUX et Yves TRAVI., 1990. Pollution par les nitrates des aquifères en Haute Bénoué, Nigéria. Géologie Africaine. Actes du 3è Colloque.

Rogoto Hobah., 1993. Complément d'inventaire des forages et éléments de synthèse hydrogéologique de l'agglomération De Ouagadougou. Mémoire de fin d'étude Sciences et techniques de l'eau. Université de Ouagadougou, 47 pages+ annexes.

Savadogo N.A., 1984. Géologie et Hydrogéologie du socle cristallin de Haute Volta. Etude régionale du bassin versant de la Sissili. Thèse d'Etat, Université Scientifique et médicale de Grenoble, 340 p.

Savadogo N.A., Yameogo S., 2002. The Urban pollution of surficial and ground-water aquifers vulnerability in Africa: Pollution par les nitrates des nappes aquifères de l'agglomération de Ouagadougou. Rapport d'étude 28 p + annexes.

Tandia A., Diop E. S., Gaye C. Pollution par les nitrates des nappes phréatiques sous environnement semi-urbain non Assaini: exemple de la nappe de Yeumbeul, Sénégal. Note technique, Journal of African Earth Sciences, Vol 29, no 4, pp 809–822.

Travi Yves., Jacques Murdy., 1997. Méthode pour l'évaluation et la gestion du risque nitrate dans les aquifères de socle de la zone sahélienne de l'Afrique de l'Ouest. Hydrogéologie, no 1, pp 13–21.

15

Groundwater contamination in the Niamey urban area, Niger

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ABSTRACT: Many residents from peripheral areas and poor city dwellers are usually reliant on untreated drinking water for their daily consumption and household needs. Such is the case of Niamey aquifer with its three main aquifers (discontinuous basic aquifer, alluvial aquifer and continental terminal aquifer) relatively tapped for water supply. Research has confirmed that the groundwater of the Niamey urban area is heavily polluted chemically and bacteriologically. High nitrate, chloride and sulphate concentrations as well as high bacterial counts of different genera (total count, faecal coliforms, feacal streptococci) indicated that this type of pollution was of faecal origin. This does not exclude the possible contribution of other pollution sources. The main pollution sources identified were the numerous pit latrines and septic tanks located in close proximity to domestic, industrial effluents, and numerous illegal domestic disposal sites in the urban areas, run-off waters in the rainy season, and finally a substandard sanitation network. This poses a serious problem, due to the volume of groundwater consumed and its negative effects on human health. Research to better characterise the pollution sources and transport mechanisms will facilitate more effective protection of the water resource, and the urban environment as a whole.

1 INTRODUCTION

Niamey City, the capital of the Republic of Niger, is located in the south-west of the country, along the banks of the Niger River between 13°30″ north latitude and 2°30″ east longitude. It extends to both of the river banks in a low-lying area characterised by plateaus and alluvial plains, with an altitude oscillating between 185 m and 205 m (Fig. 15.1).

The water supply in Niamey City (population 674,950) is estimated to be 70% effective. In the peri-urban areas in suburbs not connected to the reticulation system and in the over-populated, poverty-stricken informal areas, drinking water and domestic supply is derived from the Niger River or from the 121 boreholes drilled in an emergency programme from 1984 to1986.

Unfortunately in Niamey, as well as in many sub-Saharan African cities and other regions around the world, field studies conducted on the groundwater bodies (Joseph and Ousmane, 1988; Joseph and Girard, 1990; Gross, 1999; Schossiers, 1999; Ousmane *et al.*, 2001–2004) have indicated a high level of physico-chemical and bacteriological degradation.

1.1 *National context of the study site*

1.1.1 *Locality*

Niamey City, the capital of the republic of Niger, is located in the South-West of the country, along the banks of the Niger River between 13°30″ North Latitude and 2°30″ east longitude (Fig. 15.1).

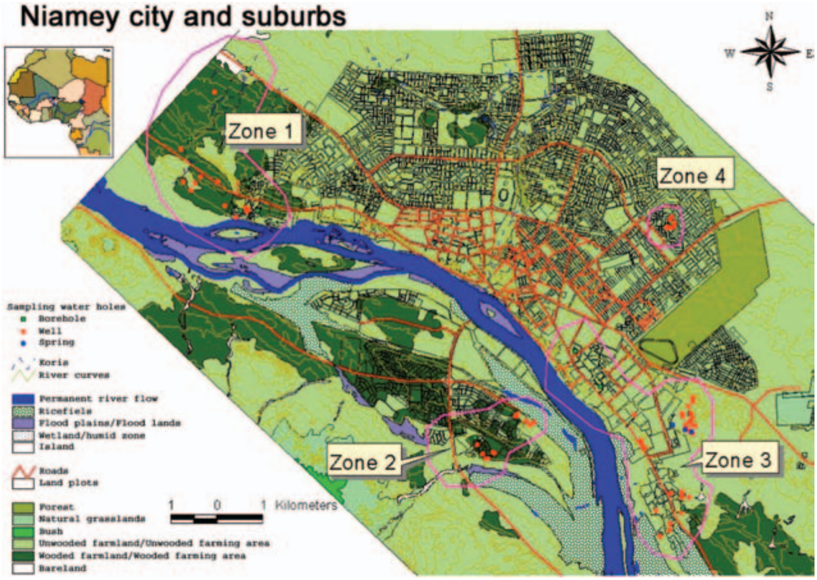

Figure 15.1. Situation of the study site in Niamey.

It extends on both the riverbanks in a low-lying area characterized by plateaus and alluvial plains with an altitude ranging between 185 m and 205 m above sea level.

1.1.2 *Climate*

The Niamey region belongs to the Sahelo-Soudan climate characterised by the alteration of two very distinct seasons: a rainy season of 3 to 4 months (May–June up to September) and a dry season of 8 to 9 months (October to April–May).

The average annual rainfall varies between 400 and 700 mm. This is characterised by strong spatio-temporal variations. Two stations separated by a distance of 5 to 10 km can register differences of more than 50 mm of rainfall during a season. Furthermore, rainfall can vary from 30 to 50% of the annual average rainfall (506 mm for the period 1962–2000), about once every three years.

The average monthly temperature in Niamey varies from 29°C in January to 41°C in April –May.

The potential evapotranspiration (ETP) exceeds 2,000 mm/year. As a result the aridity indicator (rainfall/ETP) varies from 0.2 to 0.5 in different years, confirming that the Niamey region is located in a semi-arid zone of Niger. However, in spite of the high potential evapotranspiration rate, which is 4 to 7 times higher than the annual rainfall, the water levels in Niamey showed that the effective recharge varies from 41 to 66 mm/year (Ousmane, 1988).

1.1.3 *Geology*

The Niamey region is situated on the south-western border of the Iullemmeden Basin and to the extreme north-east of the Man dorsal, Liptako. The geological formations of these regions are represented by rocks of birimian age (2300 to 2000 million years), representing the platform and the cover rocks. Among the latter, we distinguish the formations of neo-proterozoic age, previous to the African orogenesis deposited around 1000 million years ago and those of Mio-phocene age (23 to 2 million years) or continental terminal (CT3) (Greigert, 1966).

Above these sedimentary rocks, locally ferruginous cuirass sands and alluvium are encountered. These surficial formations correspond to recent sediments. The main structural elements found in the Niamey formations are the schistic metamorphism, foliation and fractures.

1.1.4 *Hydrogeology*

The main aquifer systems of the Niamey region include:

- The discontinuous aquifer of granitoids and some meta-basites (meta-gabbros and meta-basalt formations). These are found across Niamey and are tapped by boreholes and large diameter wells. The production rates are low and vary from 0.4 m^3/hr to 23 m^3/hr, with an average value lower than 3 m^3/hr for the 121 boreholes studied in 1985/1986. The transmissivity values are between 0.07×10^{-3} and $2.4 \times 10^{-3}\,m^2/s$.
- The alluvium aquifers are localised to the southern part of Goudel and exploited for small-scale irrigation and domestic purposes in Zone 1.
- The aquifer of the continental terminal (CT3) is located at the west bank of the river, in sandy horizons. The piezometric level of the aquifer rises above the level of the clay layer of the plateau, giving rise to several springs.
- The water quality is fairly consistent within each aquifer, but there are distinct differences between the aquifers.
- Water temperatures vary between 26 and 30°C, the pH-values vary from 4.0 to 8.0 and the electric conductivities are registered between 80 and 3500 μS/cm.

The hydrochemical composition varies greatly. The main hydrochemical facies vary from calcium bicarbonate, magnesium bicarbonate or mixed, with a selection of boreholes and wells characterised by a sulphate and chloride dominance. Furthermore, several boreholes have elevated bacteriological contents (total bacteria, coliforms). A large proportion of the wells have a chemical and/or bacteriological quality that could render these waters unfit for human consumption.

2 METHODOLOGY

The study was carried out in four stages. The first consisted of the selecting zones (Fig. 15.1) and analysing available geological and hydrogeological data in these zones.

2.1 *Zones*

Zone 1

Zone 1 is situated in the peri-urban zones in an upstream of the city centre on the left bank of the sub-zones, which are: the Goudel street, ancient village communalized since long-time, and very densely populated, the second sub-zone which corresponds to a vast empty space occupied by millet cultivation farms, gardens and some dispersed habitats. The morphological units of this zone include a plateau, an alluvium plain whose waters are drained by a Kori oriented towards N–S and a well-developed sand dune between Goudel and Losso Goungou and which is perpendicularly oriented to river Niger.

This zone is equally characterized by a quarry of construction materials (Banco, gravel, sand), of which some are like the one located in the Southern part of Goudel which gave rise to a semi-permanent pond, and finally the River Niger and its different islets.

The borehole's lithological cross-section and the well excavations study demonstrated that the main geological formations are: platform rocks constituted of matabasites (metabsalts, schistose) and the granitoids, the sediments of the CT and the surficial deposits represented by the alluvium and eolian sands.

The aquifer horizons are located in the platform and in the alluvium. Among an inventory of seventeen (17) study sites, four (4) boreholes were numbered F101 to F104, which are functional out of the ten boreholes dug in 1984, and they harness all the discontinuous aquifers of the platform. The remaining thirteen wells were noted P105 to P113, which exploit the platform aquifers.

Zone 2

Zone 2 is situated at the right Bank of the river towards downstream of zone 1, in an ancient peninsula of river Niger. This zone corresponds to Gnalga and Kirkissoye areas, which are very populated. It is an alluvial plain that has a slightly higher elevation than the lake.

The observation of lithological cross-section of the boreholes carried out in 1984/1986 and the well excavation shows that granitorial rocks represent the platform. The CT3 formations are not found in the study zone, and the surficial deposits are made up of alluvium with low thickness.

The nine (9) boreholes are currently non-functional and the fourteen (14) wells numbered P210 to P214 harness all the aquifers of the granite platform.

Zone 3

Zone 3 is situated downstream of the city on the left bank of Niger River. This zone includes Gamkalle, Saga and low lands, and an empty space occupied by orchards, hydro-agricultural establishments and some isolated habitats. Gamkalle and Saga are the former streets while the Pays-Bas streets are relatively recent dating back to less than six years, but are in full expansion.

The zone includes plateaus occupied by the Gamkalle' and Pays-Bas streets whose basins are very abrupt following the excavation of the quarries at banco especially at Pays-Bas, a vast alluvial plain located between Gamkalle and Pays- Bas. The runoffs waters caused by the plateaus and alluvial plains give rise to a small kori, which is thrown into an endoric basin at the center of the alluvial plain. The valley of River Niger is bordered by cornice rocks in the Gamkalle zones.

The lithological cross-sections of the 1984/1986, well excavations and the outcrops of the Gamkalle cornice and in the quarries at banco in Pays-Bas, demonstrate that the geological formations of the zone is constituted by: granite fractures and platform alterites, sedimentary series (sand stone conglomerates sands, gravelly, clayey, and ferruginous cuirass sandstone) of the continental terminal (CT3) and by surficial deposits which are the alluvial and eolian sands.

The lithological sections of the two (2) boreholes of Gamkalle; the four (4), wells of saga and that of Pays-Bas, the excavations of different wells show that the platform granites and sandy clayey levels of the continental terminal contain the two aquifer systems which are exploited in this zone.

The inventoried studies include five (5) boreholes numbered form F301 to F 305, thirty three (33) numbered from P301 to P333 and two (2) sources noted S301 and S302.

Zone 4

Zone 4 is situated at the right Bank of the river in the urban centre at the Route Filingue (Dan Gao Sector). This zone is densely populated, but also possesses a vast fundamental reserve forming a basic point as regards to the habitats. It is an area where uncontrolled runoff, effluent disposal and dumping of domestic waste occurs.

The lithological section of the boreholes and the wells excavations show that mylonitised granites and the sandy-clayey levels, and sandstone of CT3 make up the discontinuous aquifer platform and that of CT3 which is super-imposed but separated by more than 36 meters of alterites.

Among the several studies harnessing the aquifers, the borehole numbered F401 was retained and broken down into six (6) wells numbered P 401 to P 406. The second phase of the study was on the following:

2.2 *Digitisation*

The second phase of the study included the following:

– The digitisation of the topographic maps (to 1:200, 000 scale), the Niamey geological map and of cadastral survey of Niamey, using the ARCVIEW 3.2 software.
– The determination of different groundwater abstraction points using GPS and plotting these features on the digitised maps as an additional layer.

2.3 *Monitoring*

The third stage, involving the selection of monitoring sites from the zones (1, 2, 3, 4) above, included:

- The study of piezometric levels of wells with the help of an electronic dipmeter and the coincidental *in situ* measurement of macro water-quality parameters (pH, temperature and electrical conductivities) with portable meters.
- The sampling of each site for the analysis of chemical parameters in the laboratory. The chemical parameters (HCO_3^-, SO_4^{--}, Cl^-, Na^+, K^+, Mg^{++}, Ca^{++}, NO_3^-, and NO_2^-) were determined in the chemical laboratory of the Geology Department through diverse analytical techniques (alkalimeter, complexometer, and photometer). The samples were collected after purging through pumping to ensure that representative groundwater samples were obtained. The samples were generally clear and consequently not filtered.
- Taking water samples for bacteriological analyses.

2.3.1 *Sampling*

The sample was taken at the beginning, during and/or at the end of the pumping using two (2) washed, dried and sterilised 250 ml bottles at a temperature of 150°C for 45 minutes. The conservation and the transfer of samples to the CERMES laboratory were carried out in conformity with the bacteriological requirements (between 2 and 4°C) in a coolbox containing ice blocks. Once in the laboratory, the samples were kept in a refrigerator at 4°C.

2.3.2 *Analysis methodology*

Using standard analytical protocols, the samples were filtered using four membranes to allow for 100 ml/filter. The membranes are cultured from environmental cultures (Sertorius) according to the bacteria to be determined:

- 1 standard surrounding (total bacteria).
- 1 Azide surrounding (feacal streptococcus).
- 2 Tergitol TTC surrounding (total coliforms) and search for thermotolerant coliforms.

The ambient samples (Standard, Azide and 1. Tergitol TTC) are incubated at 37°C for 24 hrs. The Tergitol TTC surrounding is incubated at 44°C for 24 hrs.

2.4 *Determination of pollution sources*

The fourth stage focused on the location of pollution sites, and their spatial relationship with each zone/aquifer using ARCVIEW.

3 RESULTS AND INTERPRETATION

3.1 *Upstream zone (Zone 1)*

This zone includes the peri-urban zones upstream of the city center. The area is very densely populated in the Goudel area and also includes an area with small cultivation farms, gardens and low population density. A quarry, mined for construction materials (Banco, gravel, sand), gives rise to a semi-permanent pond and the River Niger and small tributaries are found in this zone.

Borehole logs show that the main geological formations are: basement rocks constituted of matabasites (metabsalts, schistose) and the granitoids, the sediments of the CT and the surficial deposits represented by the alluvium and aeolian sands.

The aquifer horizons are located in the platform and in the alluvium. An inventory of seventeen (17) boreholes showed that four (4) are functional. Thirteen boreholes exploit the basement aquifers.

In this area, rainfall seems to play the dominant role in aquifer recharge, while the river and liquid waste (irrigation and household effluent) are not significant contributors to groundwater input.

The electrical conductivity values of the surficial waters in this zone are between 400 and 1350 µS/cm. This variability in conductivity is due to aquifer differences and is also evident in large differences found in seemingly similar aquifer systems. Generally, however, the groundwater of the alluvial aquifer shows a more elevated mineralisation than that of the basement. In the alluvial aquifer this is probably due to anthropological factors, as illustrated by P101 to P104 in the highly populated Goudel area, which has very high EC values. These increases often coincide with the rise in water levels, which is explained by either rapid recharge and leaching of surface and soil salts or direct recharge due to surface run-off infiltrating directly due to inadequate surface controls at the boreholes and wells. The large differences in EC between the groundwater and Niger River support the notion that there is little surface water influence in this zone. The pH-values are fairly consistent in this area, with an average value of 7.5. Water temperature is between 28°C and 30°C.

The monitoring data show a large variability in the ionic content of waters from different study sites. This indicates areas of elevated CL^-, SO_4^{--} and NO_3^- contained in the waters of P101, P104 and P106. These coincide with elevated EC values, confirming the relations between the high street population and surficial water degradation at the harnessed alluvial aquifer. The chemical water quality in the basement aquifer is distinct and generally of better quality, suggesting hydraulic discontinuity between the aquifers.

3.1.1 *Downstream zone I (Zone 2)*

The zone is situated on the south bank of the river downstream of Zone 1, in an alluvial plain of the Niger. This zone includes two densely populated areas; Gnalga and Kirkissoye.

Granitic rocks represent the base of the shallow alluvial aquifer. Nine boreholes are currently not functional and fourteen (14) boreholes abstract water from the aquifers in this area. The fluctuations of water levels in boreholes next to the flood plain indicate surface water contributions to the recharge in these sectors.

The electrical conductivity of shallow groundwater varies between 550 µs/cm and 2000 µS/cm. This large variability is as evident in each individual area as across the zone as a whole. These results suggest that there is limited hydraulic continuity between the aquifers, and that the consequences of localised pollution due to activities in each area are the most important influences on the observed quality. In the period from February to June, the drop in water levels correspond to increases in the salinity of the groundwater. In July water levels rise rapidly again, with a decrease in electrical conductivity. This is due to rapid recharge from rainfall and potential surface water influx into the alluvial aquifer. The latter is not thought to be the dominant mechanism, since the groundwater qualities are significantly more saline than the river.

The pH-values of the groundwater are neutral to slightly alkaline, varying between 7 and 8. Water temperatures are high and correspond to seasonal variations. The groundwater is dominated by nitrate, chloride and sulphate in this area. Of importance is the increase in relative nitrate content during the pumping period. These results suggest the existence of more polluted water deeper in the aquifer. They further confirm that groundwater in the zone is widely contaminated, with nitrate as a key parameter of concern (Joseph and Ousmane, 1989).

3.1.2 *Downstream zone II (Zone 3)*

This zone is situated downstream of the city on the northern bank of the Niger. It includes Gamkalle, Saga and a region with orchards, irrigation plots and low population density. The geological formations in the area are characterised by fractured granite and basement alterites, sedimentary series (sandstone, conglomerates, sands, gravelly, clayey, and ferruginous cuirass sandstone) of the continental terminal (CT3) and by surficial deposits consisting of alluvial and aeolian sands. The water levels in the study sites show a continuous decline between January and June of each year. This drop in water level is quite abrupt between February and March, which could be due to intense water drawing for irrigation purposes during these months. A rise in the aquifer levels is observed between in the irrigation areas April and May. This is due to the major reduction of abstraction and recovery is strongly dependent on sufficient rainfall. The behavior of piezometric levels shows that the rise of the aquifers is strongly dependent upon the rainfall from July onwards. In the Saga area, the proximity of the river could account for the apparent fluctuations though indirect recharge and/or piezometric pressure changes as the river level rises and falls.

The spatial distribution of conductivity values of waters in this zone suggests the presence of two hydrogeological zones:

– Gamkalle-Saga sector, where 69% of the study sites have water conductivities between 500 and 1000 μs/cm, 25% have mineralisations varying between 1000 and 3000 μS/cm and a single water point has a conductivity of less than 500 μS/cm.
– The Pays-Bas sector, where electric conductivity values are between 80 and 150 μS/cm for 75% of the 20 sampled water points, and those remaining below 250 μS/cm.
– There is a good correlation between the aquifer in which the well or borehole is found and the water quality. Furthermore, local influences play an important role in the observed salinity. In the Pays-Bas sector, the water quality is relatively homogenous and not as contaminated as the rest of the area.

The values of water conductivity in the Gamkalle-Saga sector are relatively stable, showing high salinities and strong seasonal variations. The water quality deteriorates at the start of the rainy season, suggesting leaching from the soils and surface activities and direct recharge by contaminated run-off. Most of this contamination appears to be by nitrate and to a lesser degree sulphate. The pH-values of the waters of the continental terminal are acidic, contrary to the basement aquifers and alluvium, where the pH is neutral to alkaline. The groundwater temperatures in Gamkalle-Saga sector are generally lower than those of CT of Pays-Bas.

3.1.3 *Down stream zone III (Zone 4)*

Zone 4 is on the northern bank of the river in the urban centre at the Route Filingue (Dan Gao Sector). This zone is densely populated and plays an important role in groundwater quality due to high artificial recharge from urban run-off and on-site sanitation.

The lithology shows that mylonitised granites, sandy-clayey levels and sandstone of CT3 make up the discontinuous lower aquifer. The fluctuations of water levels in the different study sites show a water-level drop between January and June but rise again between May and June. The water-level decreases reach about 2 m in all the wells. The seasonal recovery is manifested very slowly and furthermore has only been observed in three of the six monitored wells. From this it can be deduced that the construction of paved areas, buildings and roads has an unfavourable impact on aquifer recharge.

The EC values vary between 400 and 800 μS/cm in a radius of less than 150 m in this area. These strong spatial variations are due to localised pollution in this sector (proximity of septic tanks and latrines). The results show that aquifers of the CT of this zone have some hydraulic continuity and the salinity variability of waters is essentially the result of pollution sources close to each groundwater abstraction point. The groundwater quality in the basement aquifer is largely consistent, as demonstrated by a five-day test conducted in the areas, during which the EC varied between 437 and 473 μS/cm despite 36 m of drawdown (Margueron, 2000).

The pH of this zone is relatively low, with values of about 5, possibly due to contamination from latrines and the low buffering capacity of the granites. The temperature of borehole water is close to the average annual temperature of atmospheric air. An important fact confirming the non-hydraulic continuity between the aquifer of the CT and that of the basement aquifer is the clear difference of chemical composition of their waters. The basement waters have very low chloride and nitrate, whereas in the CT groundwater, these values are high.

4 BACTERIOLOGY

4.1 *Results from the groundwater sampling*

Several samples were taken for bacteriological analysis, as shown in Table 15.1.

The following aspects are noteworthy:

- The total bacterial count (GT) provides a first-order indication of groundwater contamination, with high counts as a preliminary indication of water pollution.
- The coliforms are the entero-bacterium. Faecal coliforms (or thermo-tolerant) (FC) form part of this and are indicative of pollution from faecal matter. Escheria coli is a type of a thermo-tolerant coliform.
- The faecal streptococcus (enterococcus) (FS), are also the entero-bacteria from the intestine. The coliforms and the streptococcus are the bacteria whose presence in water constitutes an indication of contact between faecal matter and water.

4.1.1 *Guidelines for interpretation*

The groundwater quality is very good to good when (1) the total count is lower than 1,000 units of the colonies from for each 100 ml (UFC/100 ml), (2) the total coliforms are less than 10 UFC/100 ml, and (3) there are no other bacterial colonies. The waters are of average to poor quality when the total count and total coliform exceed these values and if there is faecal streptococcus or traces of Escheria coli.

4.2 *Interpretation of bacteriological assessment*

The data show that a large majority of the groundwater samples have elevated contents for total bacterial count, faecal coliforms and faecal streptococcus.

These results therefore confirm the existence of significant bacteriological pollution of faecal origin. The samples taken at the end of pumping show that these values are not only present in well water, but are also pervasive in the deeper aquifers. This groundwater therefore poses a health risk to the people that consume it.

5 DISCUSSION

5.1 *Piezometry*

The piezometric measurements carried out on the wells of the four zones show:

A decline in water levels (0.2 to 2 m) in the study sites during the dry seasons as a result of abstraction and outflow to springs.

A rapid rise in water levels as soon as the first rains fall. This rise continues beyond the end of the main rainy season, before decline in the water levels begin.

These reactions of the different aquifers suggest that that they are susceptible to pollution from surface activities.

5.2 *Hydrochemistry*

- The water temperature is close to that of the yearly atmospheric average at all sites. This is probably indicative of shallow groundwater.
- The pH-values vary from 5 to 8 units; therefore aquifer waters of the CT (Zone 3 and 4) are acidic, while the basic waters of the basement and alluvial aquifers are neutral to slightly acidic.
- The water conductivity values show a large variation within the zones, mostly as a function of the presence of localised pollution sources. The groundwater from the basement and the alluvial aquifers shows the most severe contamination.

Table 15.1. Bacteriological content of the groundwater in Niamey during September 2001.

Water points	GTI (UFC/100 ml)	FSI (UFC/100 ml)	TCI (UFC/100 ml)	FCI (UFC/100 ml)	GTII (UFC/100 ml)	FSII (UFC/100 ml)	CII (UFC/100 ml)	FCII (UFC/100 ml)
F101(zone1)	6000	25,000	30,000	0	0	0	0	0
F102(zone1)	2000	100	500	0	0	0	0	0
P104(zone1)	90,000	60,000	5000	100	0	0	0	0
P105(zone1)	30,000	100	0	0	30,000	100	0	0
P106(zone1)	70,000	13,000	50,000	400	60,000	100	40,000	200
P109(zone1)	80,000	40,000	40,000	0	80,000	2000	40,000	0
P204(zone2)	20,000	300	17,000	20	0	0	0	0
P206(zone2)	500,000	10,000	120,000	0	1400	200	200	0
F301(zone3)	6000	2000	200	0	0	0	0	0
F305(zone3)	80,000	23,000	50,000	10	0	0	0	0
F306(zone3)	150,000	3000	16,000	500	150,000	7000	70,000	100
F307(zone3)	15,000	0	10,000	0	0	0	0	0
F308(zone3)	80,000	1400	30,000	1500	0	0	0	0
P314(zone3)	100,000	60,000	15,000	60	30,000	3000	20,000	300
P315(zone3)	30,000	10,000	10,000	60	15,000	1000	7000	200
P331(zone3)	110,000	500	30,000	0	25,000	10,000	10,000	10
P401(zone4)	130,000	1400	30,000	10	60,000	10,000	40,000	60
P404(zone4)	19,000	200	5200	200	5500	400	3500	10

GT – Total bacterial count, FS – Faecal streptocoques, TC – Total Coliformes and FC – Faecal Coliformes. I indicates the analysis of samples taken at beginning of pumping. II indicates the analysis of samples at end of pumping.

- These results further demonstrate that measuring water conductivity is a useful tool in hydrogeological investigation. In this study, the conductivity values enabled the classification of different aquifers, as well as their vulnerability to pollution.
- The high nitrate, sulphate and chloride content found in the groundwater and parameter increases show that the contamination exceeds the attenuation capacity and dilution from recharge in the aquifers (Ousmane, 1988). This is a testimony to the influence of anthropogenic pollution (household wastes, latrines, human or animal wastes etc.) on the aquifers of the urban community in Niamey.

The nitrate values clearly demonstrate that the zones of high settlement density have the highest nitrate concentrations. Although some of the samples show elevated chloride and/or sulphate, nitrate is the parameter of greatest concern and influence in Niamey.

A lack of significant correlation between nitrate contents and bacterial colony concentrations, as witnessed by Pays-Bas waters, suggests that there may be two main pollution mechanisms:

- Direct or indirect infiltration of surface waters and effluents, which results in chemical pollution (especially nitrate) and bacteriological contamination.
- Bacteriological pollution due to pitlatrines and cesspools, informal ablution areas and the lack of sanitation around the wells.

In other words the general hygiene for the urban community still remains a determinant factor to be taken into consideration for fighting against chemical and/or bacteriological pollution of surficial waters.

6 CONCLUSIONS AND RECOMMENDATIONS

This study has demonstrated that groundwater in the study area originates from three types of aquifers, which are actively exploited by the urban community of Niamey for domestic and human consumption purposes. Furthermore, these are the only sources of available water supplies in areas like Pays-Bas. Unfortunately, the aquifers whose waters have already greatly degraded due to chemical and bacteriological contamination also become increasingly vulnerable due to the increase of pollution sources.

Consequently, in the framework of actions and activities aimed at preserving and protecting the environment for sustainable development in developing countries like Niger, specific actions should be taken to protect urban groundwater resources. This is particularly important, since the provision of services, including drinking water, is problematic due to the uncontrolled expansion of towns in developing countries. The main pollution sources have been identified in this study and it is important that the pollution transfer mechanisms be better understood.

Thus, in spite of the interesting results obtained in this study, research must continue in order to better understand the processes of pollution.

A larger scope of investigation should also be undertaken to include:

- Increased study sites beyond the immediate peri-urban zone.
- Further chemical and bacteriological analyses on the more than 50 water points for at least two hydrological years.
- The inclusion of isotopic analyses (nitrogen 15, oxygen 18, deuterium and carbon 14) to determine the origin of nitrogenous nitrates and to better appreciate their recharge and transport mechanisms.
- A further understanding of the hydrogeological regime in the Niamey region.
- The analysis and processing of all the results and the construction of a conceptual model to facilitate better management and protection of the aquifers in the Niamey region.

REFERENCES

BCR: Bureau Central du Recensement 2003. Recensement général de la population et de l'habitat (RGP/H – 2001) Résultats provisoire. Ministère des Finances et de l'Economie, 25p.

Bernert, G.; Dehays, H.; Garin, H. et Zunino C. 1985. Programme d'urgence pour le renforcement de l'alimentation en eau potable des quartiers périphériques de Niamey (Niger). Exécution de 50 forages productifs. Rapport BRGM/MHE, 52 pages et annexes.

Ellis, B. 1999. Impacts of urban growth on surface and groundwater. IAHS. Publication n° 259, 438 p.

GROSS, P. 1999. qualité de l'eau souterraine de la communauté urbaine de Niamey, Compte-rendu des travaux réalisés à Niamey d'Août 98 à février 99. Document CERMES Niamey, 63 p.

Houssier, S. 1999. Rapport de fin de Mission: Etude de la pollution de la nappe de Niamey. Rapport SCAC Ambassade de France à Niamey.

Joseph, A., et Girard, P. 1990. Etude de la pollution en nitrates des aquifères de socle : Exemple, la nappe de Niamey. Rapport, Université de Niamey, 32 p.

Joseph, A., et Ousmane, B. 1988. Pollution nitratée de la nappe alluviale du fleuve Niger à Niamey. A M Université. Niamey, T. III, pp.129–154.

Karbo, A. 1992. Etat des connaissances sur les systèmes aquifères du Niger. Rapport, MHE/OSS, 8 p.

Ousmane, B. 1988. Etude de géochimique et isotopique des aquifères du socle de la bande sahélienne du Niger (Liptako, Sud-Maradi, Zinder-Est). Thèse de Doctorat d'Etat ès Sciences Naturelles, Université de Niamey, 175 p. et annexes.

Ousmane, B.; Soumaila, A.; Margueron, T.; Daddy Gaoh, A. et Boubacar, A. 2001. Aquifères superficiels et pollution urbaine en Afrique: cas de Niamey. Rapport UNESCO/PHI/CEA, 20 p.

Ousmane, B.; Soumaila, A.; Margueron, T.; Daddy Gaoh, A. et Boubacar, A. 2002. Rapport final UNESCO/UNEP-46 p. et annexes.

Ousmane, B.; Soumaila, A.; Margueron, T.; Daddy Gaoh, A. et Boubacar, A. 2004. Rapport final de la 2ème phase UNESCO/UNEP, 35 p. et annexes.

16

Management of nitrate pollution of groundwater in African cities: The case of Dakar, Senegal

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ABSTRACT: The aquifer of the Cap-Vert Peninsula (region of Dakar) is confined in the west (infrabasaltic aquifer) and unconfined towards the east. In this region groundwater supplies the majority of the urban population. Anthropogenic activities result in high nitrate contents widely exceeding the heath standards. The existing nitrate pollution, originating from sanitation, has steadily increased over the years. This is related to the inadequacy of the sanitary installations and garbage collection infrastructure in the densely populated suburban area (east of Dakar). In this zone, the high vulnerability of the shallow, unconfined aquifer under sandy cover facilitates the infiltration of nitrates from household latrines and leachate from of waste disposal. Chemical contamination is accompanied by localised bacteriological contamination, exposing populations directly using wells to risks of sickness. Pollution tends to become widespread towards the infrabasaltic aquifer, since boreholes in to the suburban area are highly contaminated.

Keywords: Dakar, aquifer, vulnerability, chemical pollution, suburban.

1 INTRODUCTION

The region of Dakar (capital of Senegal) is located in the western extremity of Senegal and Africa (Fig. 16.1). It is surrounded by the sea on three of its sides and in a peninsula of about 200 km^2 with a population estimated at 2.5 million inhabitants (Martin, 1970).

As with other African cities, the region of Dakar, collectively called the Peninsula of Cap-Vert, has seen a rapid population growth infused a movement of refugees from nearby countries in conflict. This fast population growth has resulted in an increase in water requirements and is putting even more pressure on an already insufficient sanitation infrastructure. Populations in this area use pit latrines and dump waste on or in the ground, exerting enormous pressure on the quality of the groundwater resources already threatened by seawater intrusion.

This pressure on the aquifer from nitrogenous organic contamination in the area has been addressed by several authors (Collin and Sale, 1989; Tandia *et al*., 1997).

The increasingly evident nitrogenous pollution in the suburban area, coupled with a lack of improved sanitation, has become alarming. The problem results in both socio-economic and public health problems. Indeed, the pollution affects the distribution of water and is the cause of several water-borne diseases for inhabitants directly consuming water from wells.

To address this problem, it is crucial to define adequate plans for the control of pollution and better management of the groundwater resources of the Peninsula. Optimal management can only be achieved if based on a detailed study of the system and the understanding of mechanisms of

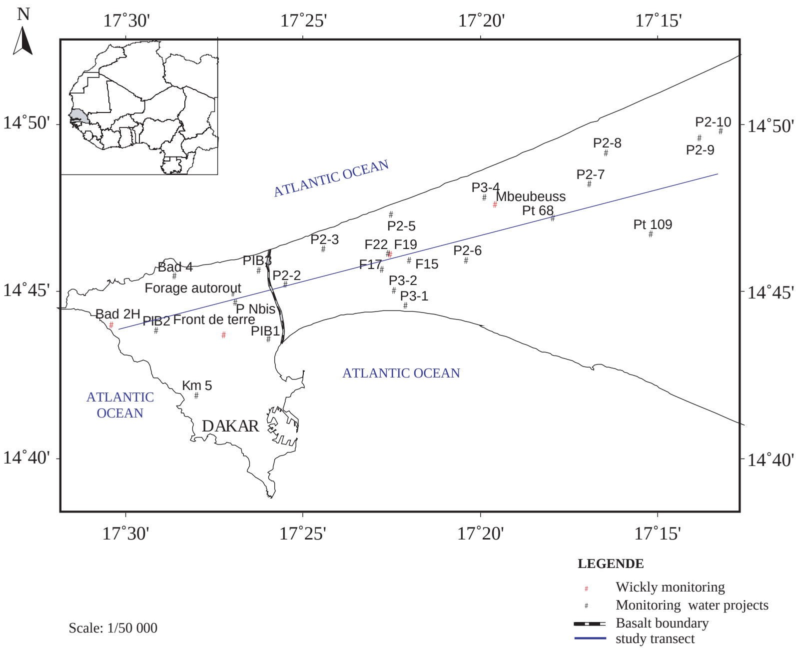

Figure 16.1. Localization map showing the limit of basalts and the sampled projects.

contamination. The objective of this study is to follow the development of the contamination and to propose a vulnerability map to facilitate decision making.

The Cap-Vert Peninsula is characterised by a microclimate of a coastal type. It is strongly influenced by maritime trade winds and the monsoon established respectively from November to June and July to October in the N-NW and S-SE directions (Olivry, 1989). The rainy season lasts four months (from July to October), with an annual rainfall of 500 mm. The annual average temperature is 24°C with relative humidity around 70%. Evaporation and annual evapotranspiration are 1500 mm and 1830 mm respectively.

2 HYDROGEOLOGY

The geology of the Cap-Vert Peninsula is the same as the Sénégalo-Mauritanian Basin, which is the widest of the coastal basins of Africa, with a surface of 500,000 km^2.

The local stratigraphy includes outcrops and oil prospecting and groundwater boreholes is constituted by the formations of the Secondary, Tertiary and Quaternary era.

The stratigraphy shows that the Quaternary sands are the most susceptible areas of the aquifer. The Tertiary consists of marly limestones and very low permeability clays, whereas the Maastrichtien section is essentially clayey in the east.

The formations of the Quaternary age extend over the whole North Atlantic border and particularly over the Dakar Peninsula. An ancient sandy Quaternary section, marked by volcanic 'Mamelles' is distinguished, and a recent Quaternary dominated by sandy deposits and elevated beaches is encountered in Dakar and its environs.

However, the dual maritime and continental origin of Quaternary sands, added to the effects of the wind erosion, make permeability and the yield of the aquifer variable, while its nature greatly depends on basaltic coverage (Gaye, 1980).

In the area of Cap-Vert the aquifer system is divided according to its confined nature into two aquifers (Fig. 16.1):

1. a confined aquifer of infrabasaltic sands at the west and
2. an unconfined aquifer of Quaternary sands in Thiaroye's Basin.

2.1.1 *Confined infrabasaltic sand aquifer*

The aquifer extends along the western boundary of the Peninsula for about 50 km^2. The aquifer is comprised of maritime sands of Quaternary age covered by a basaltic volcanic flow. Several flows are locally interstratified in the sands. Gravelly alluviums and continental sands have settled on the basalts, resulting in sufficient storage in the aquifer. The infrabasaltic and suprabasaltic sands occur under or above basalts.

The infrabasaltic aquifer is based on the marly-argillaceous Tertiary substratum which forms an aquiclude. It is limited in the south by the clays of Eocene (which outcrops here), in the north and west by the sea. The thickness varies according to the morphology of the marly substratum and the basaltic roof. Generally it increases from the south-east to the north varying between 30 and 80 m with an average of 50 m.

The infrabasaltic aquifer is fed by rain water directly infiltrating to the east at the limit of basaltic overburden or indirectly from the sandy supra-basaltic aquifer through permeable to semi-permeable basalts.

The transmissivity in the aquifer varies between $1.05 \times 10^{-2} m^2 s^{-1}$ and $9.4 \times 10^{-2} m^2 s^{-1}$ and the storage coefficient between 1.25% and 1.5%. The permeability of the aquifer is between $1.2 \times 10^{-4} ms^{-1}$ and $2.5 \times 10^{-4} ms^{-1}$ and this follows the variations of the nature of the sands, which are fine (0.175 mm), medium (0.300 mm) or coarse (0.485 mm).

2.1.2 *The unconfined aquifer of Thiaroye's Quaternary sands*

Thiaroye's Aquifer constitutes the continuance of the infrabasaltic aquifer, which extends laterally to east of the Dakar Peninsula.

The aquifer consists of diverse sands: coarse sands with a diameter of 2 to 10 mm are inserted between clayey sands, overlying the marly substratum and aeolian sands with a diameter of 0.2 to 0.5 mm in the upper portions of the aquifer. The aquifer is limited to the south-west by a piezometric high, in the north-west by the Atlantic Ocean and in the south-east by the outcropping marly substratum. The surface of the substratum is marked by depressions filled with sandy deposits. This morphology results in a variable thickness from 30 to 80 m. The aquifer is high yielding in the zones filled with sands but yielding less in the south-east.

The unconfined aquifer of Quaternary sands is fed by the direct recharge. Independent of the origin or genesis, the aquifer formations of the peninsula (infrabasaltic and Thiaroye unconfined aquifer) behave, from geologic point of view, as a single reservoir overlying the marly substratum.

3 METHODOLOGY

A study of the water quality has been done with seasonal sampling (before and after the rainy season) on around thirty water engineering projects of the Dakar groundwater monitoring network (Fig. 16.1) during the years 2001, 2003 and 2004. The physico-chemical parameters (electrical conductivity, pH, temperature) and depth are measured in the field, while the major elements are determined through the ion chromatography (Na^+, Ca^{2+}, Mg^{2+}, K^+, Cl^-, SO_4^{2-}, NO_3^-) and volumetric dosage (HCO_3^-; CO_3^{2-}) in the Laboratory of Hydrochemistry of the University of Dakar.

To better understand the spatial and temporal evolution of nitrate, a weekly follow-up was done on four chosen sites, following an east-west transect (Fig. 16.1). These included a well in the east, close to the waste site (Mbeubeus), two boreholes; one located in the densely populated suburban area (F19) and the other in the intermediate zone (infrabasaltic aquifer), and the piezometer situated to the west of the Peninsula head (Bad 2 H). The follow-up was performed from July 2003 to March 2004. Measured parameters included the physico-chemical parameters, nitrates (NO_3^-), ammonium (NH_4^+) and the faecal coliform counts.

In 2002, additional samples were taken for the bacteriology and parasitology of 21 wells located in the contaminated suburban area. The nitrate contents and bacteriological counts were also measured for soil profiles, to understand the mechanisms of contamination in the unconfined aquifer. Three profiles were carried out: one in the Borehole F19 perimeter of protection (named RSF), one situated near a contaminated well (RSP) and a third placed in a domestic waste discharge area (PY).

The vulnerability map of the area was also refined to provide decision makers with tools allowing them to manage the pollution of water resources.

The proposed model is based on the cartographic method, which is simple and very accessible. The construction of the vulnerability map, utilizing a Geographic Information System (with ARC VIEW software) consists of four stages (Fig. 16.2):

– Identification of the factors of vulnerability which, for this study, include topography, landuse, sanitation network, thickness of the vadose zone, recharge (related to permeability and typology of soils), and nature of the vadose zone.
– Preparation of the various layers involved in the GIS where data were digitised and maps stored in shape files.
– Classification of various factors that influence the definition of thematic maps. These were divided into four classes (very weak, weak, average and strong), every class being affected by a

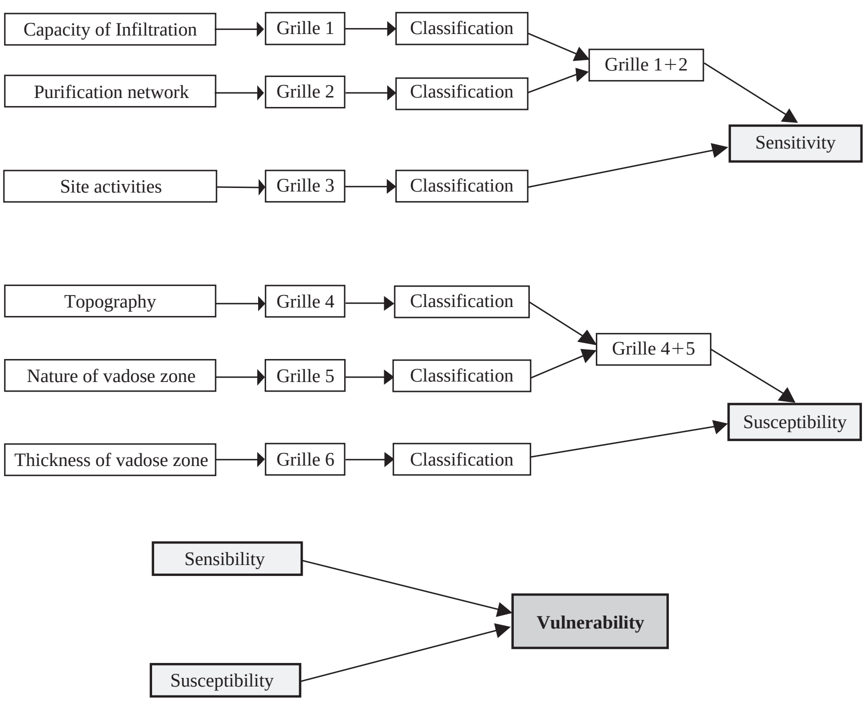

Figure 16.2. Overview of the process of constructing the vulnerability map.

weight; this level character of the classes allows the interpretation of the map obtained after overlaying the different factors.

– Overlaying the thematic maps converted in Raster size (image) results in a composite map outlining the contamination vulnerability map of the aquifer.

4 RESULTS AND DISCUSSIONS

4.1 *Hydrochemical characterisation of the aquifer*

In the aquifer, the physico-chemical parameters are pH, temperature and electrical conductivity. The pH of the aquifer varies between 4.9 and 8.2, temperature between 27.3 and 33.3°C. Electrical conductivity measures ranged between 300 and 4500 $\mu S \cdot cm^{-1}$. The Piper diagram of major ions during July 2004 (Fig. 16.3) distinguishes three types of chemical facies:

– Sodium chloride facies,
– Calcium chloride facies,
– Calcium bicarbonate facies.

The occurence of the sodium chloride facies shows the influence of the sea on the mineralization of the aquifer system. This facies is moderately to highly mineralized with high TDS content observed in the areas adjacent to the ocean. The mineralization of the aquifer reflects the geographic position of the area (coastal environment) and the nature of the aquifer. Mineralisation, apart from seawater influence, is controlled by the phenomena of dissolution, dilution, concentration and cation exchange. Rains originating from this environment are high in marine-based parameters, resulting in the observed sodium chloride nature. The aquifer recharged by rainfall initially reflects this facies. This indirect influence of the sea on mineralisation is more marked in the unconfined aquifer, where depths are shallow.

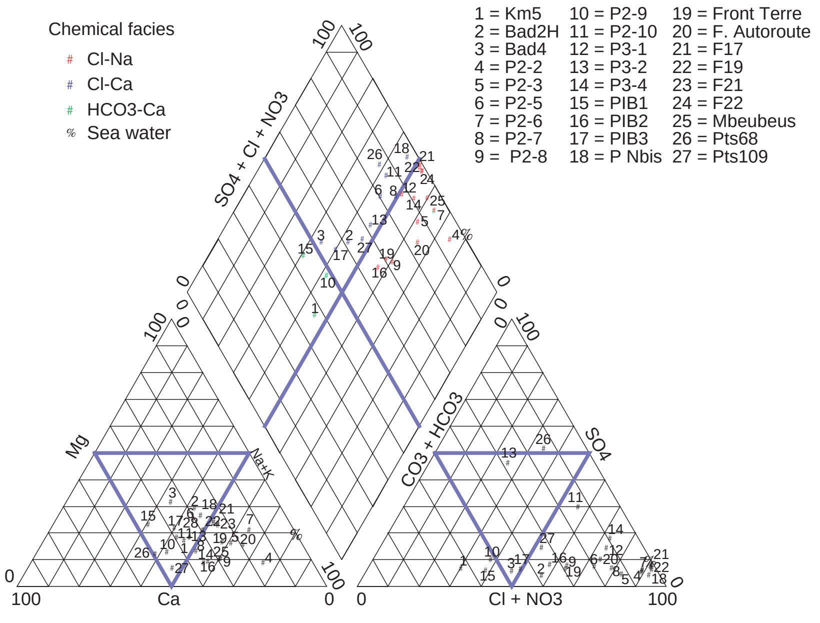

Figure 16.3. Determination of chemical facies of the aquifer (on Piper's diagram).

The calcium chloride facies observed in water engineering projects located in the ocean border is bound to a process of inverse cation exchange, as is evident from the saline intrusion front. When sea-water influenced pollution of the groundwater is limited (less than 15%), there is an enrichment in Na^+, and if the percentage increases, freshwater is increasingly enriched by Ca^{2+} and depleted in Na^+ (Magaritz and Luzier, 1985). The initial sodium chloride facies becomes calcium chloride in nature. The observation of the changes of facies enables an understanding of the development of freshwater contamination by sea water in the coastal aquifers (Banton, 1989).

Because of the influence by sea water, the use of chlorides as indicator of nitrogenous pollution is not suitable.

4.2 *Study of nitrogenous pollution*

4.2.1 *Nitrates in the saturated zone*

The study of contamination was carried out in terms of the spatio-temporal variation of nitrate content since it is a good indicator of contamination from organic origin. This ion, generally at low concentrations in uncontaminated groundwater, results from the mineralisation of organic nitrogen (by oxidation) by intermediate stages (ammonium, nitrites) (Canter, 1997).

4.2.2 *Spatial evolution*

The nitrate ion is the most oxidized from of inorganic nitrogen and values ranged between 5.28 and 790.50 $mg \cdot l^{-1}$ for July 2004 (Fig. 16.4).

More than 30% of sampled water engineering projects have nitrate contents that exceed the standard of 50 $mg \cdot l^{-1}$ and 30% have concentrations exceeding 100 $mg \cdot l^{-1}$. This shows the advanced degree of the contamination in the aquifer. The spatio-temporal graph displays three very different zones:

– In the west at the head of the peninsula, nitrate contents are average, increasing in boreholes situated east of the infrabasaltic aquifer (PN bis and Front de Terre).
– In the densely populated suburban area (Pikine – Thiaroye) nitrate concentrations widely exceed the standard of 50 $mg \cdot l^{-1}$ defined by the WHO (2000). Maximal concentrations (between

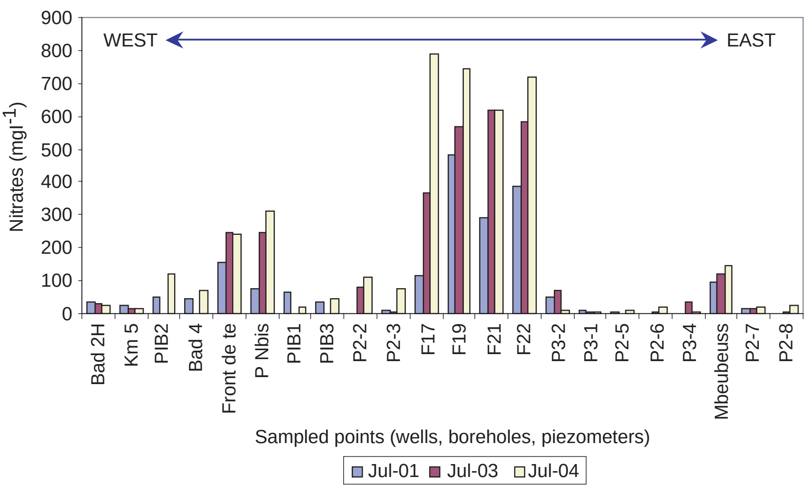

Figure 16.4. Spatio-temporal evolution of nitrate concentrations in the aquifer (2001 – 2003 – 2004).

616.75 and 790.50 mgl^{-1}) are encountered in Thiaroye's impounding area in Boreholes F17, F19, F21.

– In the uninhabited area east of Thiaroye, nitrate contents are lower than 50 $mg \cdot l^{-1}$. This zone corresponds to areas of vegetable farming (called 'Niayes').

The spatial distribution of nitrates shows that the main source of contamination of the aquifer is focused on the densely populated suburban area due to poor sanitation. This is evidence of the domestic origin of nitrogenous contamination. Previous studies (Tandia *et al.*, 1997) showed that these negative effects are closely related to the inadequate systems to contain, purify or dispose. Indeed, the load resulting from traditional latrines (and leaking sewerage and sanitation units) and disposal of domestic waste are at the origin of the contamination in the unconfined and shallow aquifer.

At the head of the peninsula, basaltic formations provide a degree of protection to the aquifer, depending on their thickness and the degree of fracturation. Average nitrate concentrations in this area range around 30 $mg \cdot l^{-1}$. However, elevated nitrate contents in boreholes situated in the interior of the Peninsula (intermediate zone) suggests possible pollution of the infrabasaltic confined aquifer from the unconfined aquifer, since they are hydraulically related and the general drainage is from east to west. These boreholes are indeed the first affected by lateral pollution from the unconfined aquifer of the suburban area.

Besides this lateral contamination, some nitrates can result from vertical infiltration through areas with thinner basaltic coverage.

It was also noted that the group of boreholes studied is strongly affected by nitrogenous pollution, contrary to piezometric gradients, suggesting a continuous influx of nitrates.

4.2.3 *Temporal variation*

In the very polluted areas, nitrate contents increased sharply during the last two years (Fig. 16.4). This is the case for all the sampled boreholes, as well in Thiaroye's impounding area (F17 , F19, F21, F22) and also in infrabasaltic aquifer (PN Bis, Front de Terre). Nitrate contents for example rose from 117.4 $mg \cdot l^{-1}$ (July 2001) to 790.5 $mg \cdot l^{-1}$ (July 2004) in three years, and from 333.7 $mg \cdot l^{-1}$ (in October, 2001) to 810.4 $mg \cdot l^{-1}$ (October 2004) in Borehole of F19. In this same borehole, the nitrate contents measured in 1967 averaged around 26.7 $mg \cdot l^{-1}$. Pollution in the suburban area is related to the rapid and informal settlement in these areas. Due to poverty, these population settled in an area without sanitary installations or formal water supply. The shallow aquifer is therefore under continuous anthropogenic pressure, resulting in high nitrate concentrations of organic origin. This situation is all the more alarming, as contamination is maintained and could worsen if adequate measures are not taken. If the current situation persists, the groundwater-dependent population of the peninsula will be severely affected.

On the other hand, in zones with little contamination, nitrate contents vary only slightly from one year to the next. This included wells and piezometers where there is adequatic basaltic cover or situated in areas with low population numbers. Nitrates in these locations even showed some decline over the past years.

The spatio-temporal evolution of nitrates can be determined from weekly observations of four wells following a transect. Observations were made from October 2003 to March 2004. Although the monitoring was not continuous, it provided a view of the nitrate evolution over time.

There is a sudden increase of nitrates in the F19 borehole, which is revealed by the inter-annual evolution. Indeed, a peak of nitrates is observed just after the end of the rainy season, marking the arrival of a first wave of nitrate-rich waters. The other peak begins in December and persists until March. These concentrations are much higher than those observed in October. In the area of Borehole F19, the unconfined and the shallow aquifer (about 6 m) reacts fairly rapidly to surface pollution. Nearly flooded latrines result in nitrate-rich groundwater, which is intercepted by the cone of depression from pumping.

In the 'Front de Terre' borehole, values stabilise between 116 and 175 $mg \cdot l^{-1}$ before showing a slight increase. They stabilise again from January until March. The external influence is less

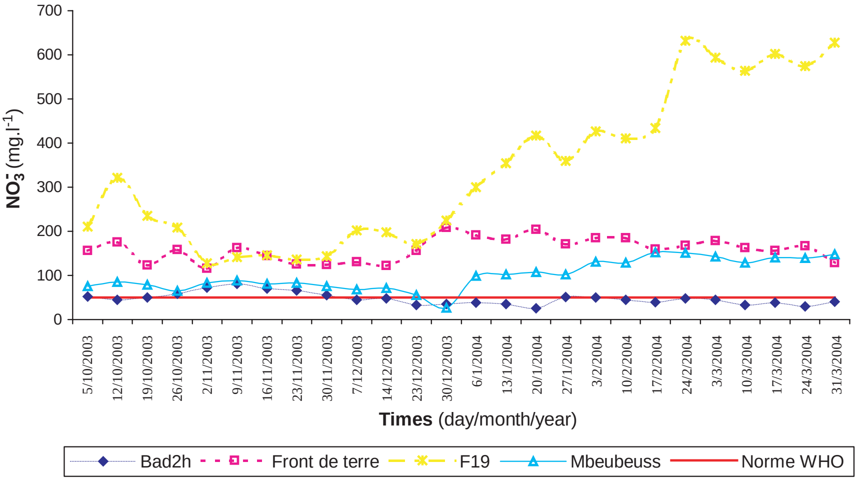

Figure 16.5. Evolution of nitrates in projects of weekly follow-up.

marked, but in the F19 borehole, pumping provokes a convergent stream of water transporting NO_3^- ions.

The Mbeubeus well, in spite of the proximity of the rubbish dump, is less polluted than Boreholes F19 and Front de Terre. The evolution of nitrate contents during the observation period is identical to that of other boreholes.

In the piezometer Bad 2H, nitrate concentrations are just below $50\,\mathrm{mg{\cdot}l^{-1}}$, except in November 2003. Generally speaking, the nitrate contents do not change much during the year.

The shallow, unconfined aquifer has a sandy cover and is far more sensitive to surface pollution. Infiltration waters wash the nitrates from the ground surface and transport them to the water table by piston flow. The infrabasaltic aquifer remains confined by basalts and is less affected by this outside influence.

4.2.4 *Nitrates in the unsaturated zone*

The behaviour of nitrogen in the ground is affected by numerous physical, chemical and biologic processes that can interact with variable intensities, according to conditions of the environment.

In the suburban area where the aquifer is unconfined, the evolution of nitrates is variable in the unsaturated zone (Fig. 16.6), very high in the subsurface and concentrations decrease considerably with depth, down to the level of the capillary fringe measured in the groundwater. This decrease with depth shows that the sources of pollution are localised on the surface. The transport of the nitrate occurs by several processes.

When the water table is shallow, the evaporation favours the accumulation of nitrates on the surface while the capillary rise favours the decrease of concentration. This type of evolution is characteristic of dry and semi-dry areas (Edmunds, 1983; Aranyossy *et al.*, 1991).

On the other hand, when the water table is deeper the evolution of nitrates in the unsaturated zone becomes more complex. If the source is localised in the soil, it moves by convection towards the water table due to the percolation of infiltrating waters. This explains the peaks of nitrates observed on the RSF profile. The processes is assisted by the coarse grained nature of the aquifer. Dilution by capillary rise is less important here than when the water table is shallow. Denitrification produces the inverse effect in the presence of reducing bacteria of organic carbon substrata and leads to the decrease of the oxygen (Reddy and Patrick, 1981).

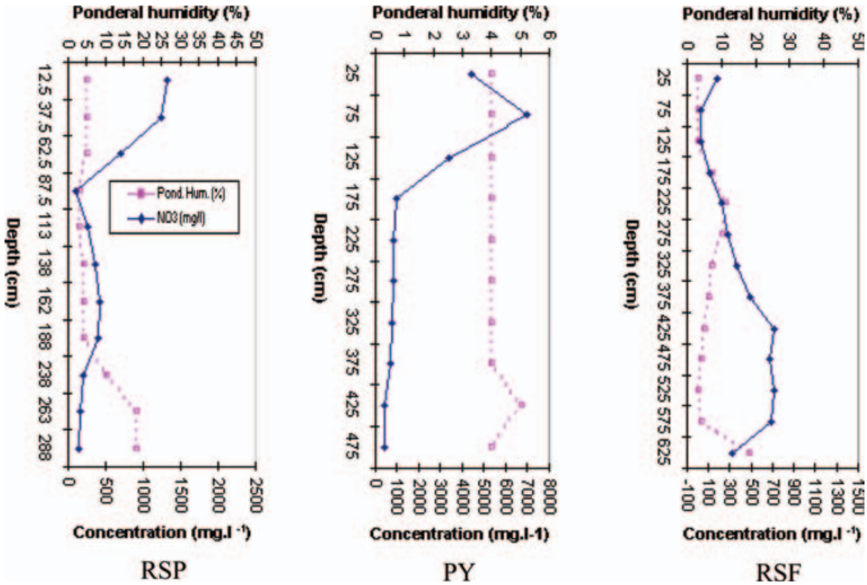

Figure 16.6. Evolution of nitrate concentrations (mg·l^{-1}) and the ponderal humidity/soil moisture content (%) in the unsaturated zone (shots profiles: RSP; PY and long profile: RSF).

Denitrification, inhibited by the frequency and loading, cannot prevent the increase of nitrate in groundwater.

The presence and evolution of the high concentrations of nitrates in the unsaturated zone of the aquifer outside the areas of latrines indicates a certain link between their presence and the disposal of domestic waste and excreta on or in the ground. These waste materials contain large quantities of organic matter that can be leached by infiltrating water. The groundwater contamination by nitrates is the result of leaking domestic latrines that provide vertical streams to evacuate the waste on the surface.

The sediments of the unsaturated zone, with an average water content of 3%, act as a nitrate reservoir that retains the pollution for a long periods, even when organic waste dumping stops (Diop and Tandia, 1997).

5 BACTERIOLOGY

Bacteriological measurements involved coliforms and faecal streptococci, which are non-pathogenic bacteria living in abundance in human faeces. They are reliable indicators of likely faecal pollution and the presence of pathogenic bacteria in water (Rodier, 1996).

Analytical results show the presence of faecal coliforms in 52% of the sampled wells. The contaminated wells are generally shallow. The presence of coliforms confirms the faecal origin of the nitrogenous pollution associated with bacteriological pollution, which is responsible for grievous hydrous diseases. The absence of bacteria in other wells and boreholes (F19) does not mean that they are definitively free of bacteriological contamination. Indeed, the presence of these contaminants on the surface in the F19 borehole perimeter of protection, is evident of a real risk of aquifer contamination, especially in the wet season when water levels are very shallow.

A decrease in coliforms and faecal streptococci with depth is noted where vertical infiltration from onsite sanitation is suspected. Bacterial concentrations are very high at the surface and disappear below two meters from the ground surface. This decrease of bacterial concentrations with depth is due to the soil acting as a biologic filter (Castany, 1982).

Parasites were found in three wells, of depths between 0.4 m and 3.3 m. The parasites are cysts of *Entamoeba histolitica*, *Entamoeba coli* and *Rabditis*. Previous studies (Dieng *et al*., 1997) in the suburban area showed that, of the 705 persons subjected to stool examinations, 298 were the victims of one or several parasites, in other words with a prevalence of 42.26%. The most frequently encountered parasite was *Entamoeba coli*, with a higher prevalence among the consumers of water from wells than those from fountains.

6 AQUIFER VULNERABILITY BASED ON NITROGENOUS CONTAMINATION

The combination of various factors results in the aquifer vulnerability map.

The vulnerability map (Fig. 16.7) shows:

– A zone from very low to low vulnerability to the west of the region at the head of the peninsula, and in the east.

At the head of the peninsula the infrabasaltic confined aquifer seems well protected against pollution from the surface.

In the east the strong slopes favor water run-off from the surface resulting in infiltration towards the saturated zone.

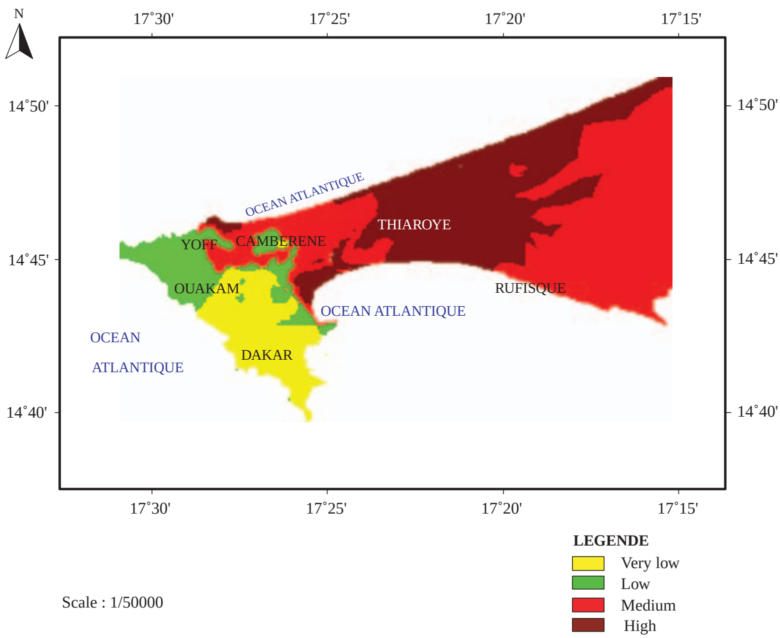

Figure 16.7. Vulnerability map of Dakar aquifer.

– A zone of medium vulnerability in most of the suburban area characterized by a lack of formal or adequate sanitation.
– A zone of high vulnerability localised in Thiaroye-Pikine's basin, characterized by a lack of adequate sanitation and the shallowness of the unconfined aquifer.

The extension of nitrogenous pollution confirms the vulnerability map. It was thus observed that the highly vulnerable zones correspond to areas where nitrogenous pollution is intense.

7 CONCLUSION

The urban aquifer of Dakar contains nitrate that exceeds the standard of $50\,\mathrm{mg \cdot l^{-1}}$. Nitrates found in the unconfined aquifer to the east reaches $300\,\mathrm{mg \cdot l^{-1}}$, as seen in monitored boreholes.

The evolutionary study of nitrates revealed that the densely populated suburban area and inadequate sanitation constitute the source of the nitrogenous compounds. Up to date localised pollution has become widespread towards the infrabasaltic aquifer, where boreholes close to the suburban area are significantly contaminated with nitrates. High NO_3^- contents are related to the rapid population growth, which exerts continuous pressure on groundwater resources. Nitrates increase steadily over time. In Thiaroye's area they increased from $26.7\,\mathrm{mg \cdot l^{-1}}$ in 1967 to $804.10\,\mathrm{mg \cdot l^{-1}}$ in 2004.

The vulnerability study showed that the unconfined aquifer is very vulnerable in surface pollution. The shallow water table and sandy cover, and the inadequate sanitation and domestic waste facilities, which are the causes of nitrogenous pollution, lead to high levels of contamination. The possibility of widespread bacteriological pollution, which has up to now been localised in shallow wells, should be emphasised and appropriate actions taken to prevent the spread of such pollution.

The water supply of the urban population is affected by the gradual degradation of the aquifer. All the studied boreholes are polluted by nitrates, which obliges water provision officials to proceed to dilute waters to lessen nitrates before distribution. This method however is not adequate to manage the severity of upstream pollution. Cleaning up is expensive, with the simplest available method currently being to eradicate the causes of contamination by the improvement of purification and garbage collection systems. It is also necessary to facilitate the connection of the water supply network for drinking water to the suburban areas. These populations, are exposed to risks of several water-borne diseases as they directly use the contaminated groundwater from shallow wells.

The degradation of the groundwater quality in the region has become an environmental, socio-economic and health problem that all role-players (decision makers and academics) have to try to resolve.

REFERENCES

Aranyossy JF., Filly A., Tandia AA., Lourat D., Ousmane B., Joseph A., Fontes JC., 1991. Estimation des flux d'évaporation diffuse sous couvert sableux en climat hyper aride (erg de Bilma, Niger). Isotope Techniques In Water Ressources Development. Processing of a symposium, Vienne, IAEA-SM-319/39, pp. 309–324.

Banton O., Bangoy LM., 1997. Hydrogéologie. Multiscience environnementale des eaux souterraines. Presse de l'univ. du Quebec, AUPELF. ff. Quebec, Canada. 460 p.

Canter LW., 1997. Nitrates in groundwater. Lewis publishers, New York. 263 p.

Castany G., 1982. Principes et méthodes d'hydrogéologie. Ed. Dunod, Paris. 236 p.

Collin JJ., Sale G., 1989. Pollution des eaux souterraines par les nitrates dans les banlieues non assainies des pays en développement : cas de Pikine (Sénégal). Note Technique BRGM, SGN/3E, 89/27, 11 p.

Edmunds WM., 1983. Use of geochimical methods to determine current recharge and recharge history in semi-arid zones. Proc. Ieme symposium arabe sur les ressources en eau. Rabat, pp. 282–301.

Gaye CB., 1980. Etude hydrogéologique, hydrochimique et isotopique de la nappe aquifère infra basaltique de la Presqu'île du Cap-Vert (Sénégal). Thèse 3e cycle, univ. Dakar.

Magaritz M., Luzier JE., 1985. Water-rock interaction and sea water-freshwater mixing effects in the coastal dunes aquifer Coos Bay, Oregon. Geochim. Cosmochim. Acta. 49

Martin A., 1970. Les nappes de la Presqu'île du Cap-Vert, leur utilisation pour l'alimentation en eau de Dakar. Pub. BRGM, Dakar, Senegal, 50 p.
WHO., 2000. Directives de qualité pour l'eau de boisson. Recommendations WHO, 2eme ed. vol. 2. Geneva OMS, 2
Olivry JC., 1989. Le point en 1982 sur l'évolution de la sécheresse en Sénégambie et aux îles du Cap-Vert. Examen de quelques séries de longue durée (débits et précipitations). Cahiers ORSTOM, série, études et thèses, 372 p.
Reddy KR., Patrick WH., 1981. Nitrogen transformations and loss in flooded soils and sediments. in CRC Critical Reviews in Environmental Control, vol. 13, No. 4, pp. 273–303.
Rodier J., 1996. L'analyse de l'eau: eaux naturelles, eaux résiduaires, eaux de mer. 8eme ed. Dunod, Paris.1383 p.
Tandia AA., Gaye CB., Faye A., 1997. Origine des teneurs élevées en nitrates dans la nappe phréatique des sables quaternaires de la région de Dakar, Sénégal. Rev. Sécheresse, No.8, vol.4.
Diop ES., Tandia AA., 1997. Qualité de l'eau de la nappe phréatique à Yeumbeul Sénégal. Bull. CSI info No 3, UNESCO, 27 p.
Dieng Y., Tandia AA., Diop ES., 1997. Les parasitoses intestinales chez les habitants d'une zone périurbaine à nappe phréatique polluée par les nitrates d'origine fécale, (Yeumbeul, Sénégal). cahier santé, vol. 9, pp. 351–356.

17

The application of electrical resistivity techniques in the delineation of saltwater-freshwater in the Keta Basin, Ghana

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ABSTRACT: The Keta Basin forms one of the five coastal sedimentary basins in the southern part of the country. The top sand-plain is an extensive shallow aquifer used for both the domestic and agricultural water supply. Very recently, a number of shallow boreholes were drilled into the shallow freshwater lenses and fitted with submersible pumps. Studies conducted on the water quality suggest an increase in the chloride content of the aquifers and the possibility of salt-water intrusion into this aquifer. The paper deals with the application of electrical resistivity techniques to delineate formations bearing fresh and saline water, distinguish between sandy and clay layers and establish the depth to the freshwater-saline water interface. Results of the VES conducted for test wells AN1, AN2 and AN3 indicate that the northern sector is basically underlain by a four-layer structure. The top layer has an average apparent resistivity of 27.1 Ohm-m and an average thickness of 6 m. The infinitely thick bottom layer has an apparent resistivity of 0.2 Ohm-m, which was confirmed to be due to an extremely thick layer of clay, probably saline water-saturated. An analysis of the geophysical data also indicates that the freshwater-saline water interface in the Anloga area lies at an approximate depth of 40 m, close to the shoreline to a maximum depth of about 100 m at the mid-portion between the lagoon and the sea. Test drilling close to the lagoon has also confirmed that the interface lies beyond 14 m in the vicinity of the lagoon.

1 INTRODUCTION

The Keta Basin is the most extensive sand-plain shallow aquifer in Ghana. Almost all communities in the area have been relying on the shallow sandy aquifers for both their domestic and agricultural water supplies. These aquifers are tapped by hand-dug wells. Extensive irrigation of vegetables for both local consumption and export is transported from boreholes drilled into this aquifer.

In recent years, the agricultural practices along the entire Ada-Anloga-Keta strip had changed from traditional small-scale farming systems to more advanced and modern practices. Presently, a number of wells are drilled into the shallow freshwater lenses and more land is put under cultivation. Consequently groundwater abstraction has significantly increased.

The potential impacts of inadequately controlled groundwater withdrawal on the environment, especially in the coastal zone, include: (1) seawater intrusion, (2) degradation of groundwater quality of both shallow and deep aquifers caused by migration of poor quality groundwater into good quality aquifers, (3) up-coning of the saline water layer at the saline-water-freshwater interface, (4) significant decrease in surface flow, which can unfavorably influence the fauna and flora.

The quality of groundwater used for the irrigation of agricultural lands could affect plant growth and yield. Some serious implications of increasing salinity and high sodium hazards have been reported in certain parts of the world (Ayer, 1975; Hem, 1985).

Several studies have been conducted on water quality (Banoeng-Yakubo *et al.*, 2003, 2004; Hotor, 2004), which suggest an increase in the chloride content of the groundwater in the basin. Jorgensen

Figure 17.1. Location map of the study area.

and Banoeng-Yakubo (2001), using environmental isotopes, showed that a significant excess of chloride in the shallow groundwater is the result of a combination of evaporation and marine sources.

Various hydrochemical characterisations of the groundwater in the area have been done (Anku, 2003; Kortatsi *et al.*, 1999). The main objectives of the present study are to apply geophysical techniques to delineate formations bearing fresh and saline water, distinguish between sandy and clay layers, and establish the depth to the freshwater-saline water interface. These objectives were achieved through the collection and interpretation of primary and secondary geophysical, lithological and hydrochemical data.

1.1 *Location and physical setting*

The Keta Basin forms one of the five coastal sedimentary basins in the southern part of the country and lies at the extreme south-east corner of Ghana, forming a continuous aquifer system into the Republics of Togo, Benin and Nigeria. The study area in Ghana lies within the latitudes 5°.45′N–6°.20′N and longitudes 0°.45′N–1°.15′E (Fig. 17.1).

The basin covers an area of about 3577 km^2, of which 2201.50 km^2 are on-shore and the remainder extends below the Gulf of Guinea. A large portion of this basin lies in a region dominated by the Volta River Delta Complex (Kesse, 1985).

The topography of the Keta Basin is described as generally flat. However, there are areas of undulations with an average maximum height of 7 m above sea level located in the central areas, and less than 1 m towards the coast, particularly to the south-east near Keta.

Climatically, the area lies within the Dry Equatorial Climatic Region, dominated by the tropical maritime air masses (monsoon) from the South Atlantic Ocean. Due to its southward location, the region comes under the influence of the Inter-Tropical Convergence Zone (ITCZ) for most parts of the year. This climate is characterised by two rainfall maxima, mainly between March–July and September–November. The dry season is more marked and the mean annual rainfall is 750 mm. Relative humidity is between 60 and 75 per cent, with a mean annual temperature of 27.5°C (Benneh and Dickson, 1995).

There are a few streams and rivers draining a small portion of the area. Most of these streams are concentrated at the northern fringes of the study area, towards the northern parts of the Keta

Lagoon. Prominent among the rivers are the Tordzei, Balikpa, Uuli and Agali. The rivers and streams generally flow southward at short distances within the area before discharging most of their waters into the numerous lagoons and marshy grounds in the study area and/or eventually die out into the coastal sand and gravel aquifers. With the exception of River Volta, none of the streams ever discharge directly into the Atlantic Ocean (Gulf of Guinea).

The River Volta lies to the extreme south-west of the study area and flows predominantly over the Quaternary-Tertiary sediments. It discharges large volumes of fresh water into the sea and constitutes a major source of fresh surface water supply in the region.

Salt ponds and marshy grounds are very common, some of which are under exploitation for salt mining. Most of the lagoon waters are salty throughout the year and the only fresh water lagoon is the Avu Lagoon.

1.2 *Geology and hydrogeology*

Geologically, the Keta Basin is a continuous basin that spans from the eastern coastal areas of Ghana through Togo to the Republic of Benin. It is underlain by recent unconsolidated beach sands and lagoon clays, which deepen westwards towards the Volta River estuary and at depth by clay, gravels, and interlayers of varying thicknesses of limestone, which form the main deep freshwater resource of the whole basin.

The recent deposits rest on a series of continental beds of Middle Tertiary age. The rocks are unconsolidated limonitic argillaceous sands and gritty sands, with persistent gravelly beds at their base. The gravelly beds are persistent from the Ghana-Togo border.

The Middle Tertiary rocks have very permeable soils and subsoils resulting in low run-off. Large parts of the rainfall infiltrate into them and percolate into the groundwater system. These Tertiary sands rest on Cretaceous and Eocene age marine shale, glauconitic sandstone and limestone. The limestone varies with depth, but can be about 300 m deep.

2 FIELD METHODOLOGY

Geophysical methods are used extensively for groundwater exploration, contamination studies and for aquifer delineation. The thickness of unconsolidated materials, depth to water table, depth to bedrock, the location and extent of gravelly beds and clay layers, as well as the location and extent of fault and shear zones can be evaluated using geophysics (Carruthers and Smith, 1992). The geophysical techniques normally employed in hydrogeological studies depend on the existence of differences in physical properties between the subsurface formations studied (Whiteley, 1985; Parker, 1981). The electrical resistivity method was applied during the study, because it is capable of detecting the slightest changes in pore water conductivity. This characteristic makes it highly responsive to the freshwater-saltwater interface in coastal regions.

In the field, forty (40) stations were selected and sounded accordingly. Two sets of VES were carried out at the same stations for two seasons; first during the dry season (March–April) and the other during the rainy season (June–July). The inter-seasonal measurements were carried out in order to monitor the relative movement of the saltwater-freshwater interface with the seasons.

The VES data was collected with a Signal Averaging System instrument (Abem SAS 4000 Terrameter). Averaging, as used above, is a system specially designed, whereby consecutive readings are taken automatically and the results averaged continuously. SAS results are more reliable than those obtained using single-shot systems (ABEM, 1998). It was carried out using the Schlumberger configuration described by Parasnis (1979) and Zohdy *et al.* (1974). The minimum and maximum half current electrode separations used are AB/2 = 1 metre and AB/2 = 251 metres respectively, purposely designed to track the subsurface resistivity from the surface down to a theoretical depth of about 330 m. The potential electrode separations (MN) are MN/2 = 0.5, 5.0 and 25 m. The depth of investigation increases generally as a function of the distance AB (i.e. C_1C_2) and is given by the relation 0.125 L, where L is the separation between C_1C_2 (Roy and Apparao, 1971). However,

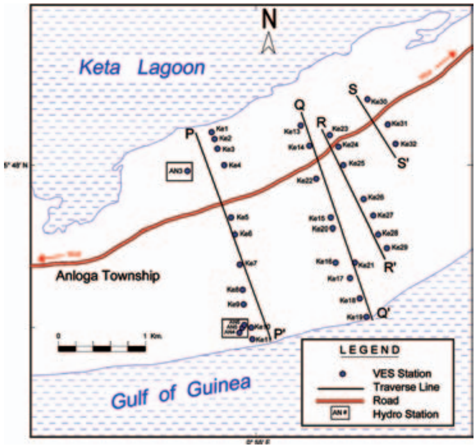

Figure 17.2. Diagram showing spatial distribution of the VES stations in the study area.

based on practical experience, Armah (1995) also stated that the theoretical depth of investigation in electrical sounding is related to C_1C_2 as 0.667 L, and this latter relationship has been used for the interpretation.

The sounding points were chosen along four traverses, with the aim of ensuring a fair distribution of sounding points in the study area. The traverses trend in an approximately north-south direction. Each traverse stretches from the Gulf of Guinea coast in the south to the Keta Lagoon (Fig. 17.2). These traverses are designated PP′, QQ′, RR′ and SS′ and are of varying lengths. The average distance between individual VES points along each traverse is 150 m. The average inter-traverse separation is 450 m. The Schlumberger method was used for the soundings.

In the absence of previous geophysical data, eight Vertical Electrical Soundings (VES) were conducted at the sites of the test boreholes. These measurements preceded the general soundings carried out in the area. The objective was to obtain background hydro-geophysical information that would be used as a basis for the interpretation of subsequent sounding results**.** The maximum half-current electrode separation (AB/2) used is 100 m (Carruthers and Smith, 1992).

The VES results were modelled using the software *RESIST*, which is an iterative inversion-modelling programme. Analysis of the resulting apparent resistivity versus half-current electrode separations yielded layered earth models composed of individual layers of specified thickness and apparent resistivity. The results were then imported into *IPI2win*, a programme for 1D automatic and manual interpretation of VES curves, where curves could be received with different arrays: Schlumberger, Wenner, Dipole-Dipole and Pole-Pole. This programme is advantageous in that besides the manual interpretation, it is capable of automatically generating geo-electric and pseudo-sections. Also, the parameters of models could be changed in different ways.

The positions of the boreholes, the VES stations and the hydro-stations are indicated in Figure 17.2. The VES station located at the center of the three boreholes AN-01, AN-02 and AN-03, is designated as AN1–3 and those at boreholes AN-04, AN-05 and AN-06 are referred to as AN4, AN5 and AN6 respectively.

2.1 *Presentation and discussion of results*

This section discusses the results of the borehole-controlled VES, where the layer resistivities were superimposed on the borehole lithological logs. As a result, the project area was divided into

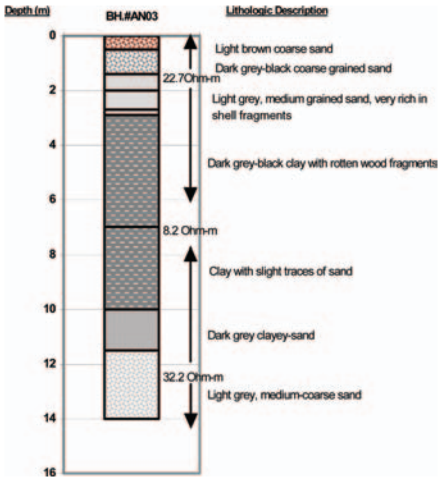

Figure 17.3. A representative composite lithological log with subsurface superimposed layer resistivities (northern sector).

Table 17.1. Summary of resistivity model results for the sounding at borehole sites (calibration).

VES stations	No. of layers	Resistivity of layers (Ohm-m)						Thickness of layers (m)				
		ρ_1	ρ_2	ρ_3	ρ_4	ρ_5	ρ_6	T_1	T_2	T_3	T_4	T_5
AN1–3	4	22.7	8.2	32.2	0.2	–	–	6.2	4.7	9.6	–	–
AN4	5	2,258.5	2,715.3	54.2	5	57.9	–	1.9	0.1	12.6	12	–
AN5	4	3,188.1	91.7	17.8	428.6	–	–	1.2	7.9	28.1	–	–
AN6NS	5	365.8	1,685.3	38.2	4.5	254	–	0.4	0.9	12.3	18	–

two main sections, based on the geology. The area that stretches from the Anloga-Keta road towards the Keta Lagoon is referred to as the northern sector and is basically underlain by the lagoon clays where the sand layer is less than 3 m thick. The area that stretches from the Keta-Anloga road to the Gulf of Guinea coast is referred to as the southern sector and is primarily underlain by the marine sands.

The resistivity of the materials in the zone where the conductive layers terminate ranges from 30 to 3000 Ohm-m, and represents a thick accumulation of medium-coarse-gravelly sand aquifers with very little or no traces of clay intercalations extending beyond a depth of 50 m depth. Two inter-seasonal geo-electric cross-sections constructed during this investigation confirmed the presence of a relatively deeper root of freshwater aquifer in the area that serves as a hydraulic barrier separating the marine from the lagoon clays. The sounding results were calibrated using test boreholes drilled in the vicinity of the sounding points. The results were used to adopt certain ranges of apparent resistivity values to indicate the zones of saline water aquifers, thick clay layers, freshwater aquifers and desiccated layers.

The layer resistivities and thicknesses for the soundings carried out on each borehole were superimposed on the lithological logs of the test boreholes (Fig. 17.3). Due to the fact that the same materials and rock types were encountered, especially at shallow depths at each drilling site, only the deepest test boreholes at each drilling site were used for discussion. The layer resistivities and thicknesses for the borehole calibration curves are summarised in Table 17.1.

The results of the VES conducted for test wells AN1, AN2 and AN3 indicate that the northern sector is basically underlain by a four-layer structure. The top layer has an average apparent resistivity of 27.1 Ohm-m and an average thickness of 6 m. The second and third layers have resistivities and thicknesses of 8.2 Ohm-m and 4 m and 32 Ohm-m and 8.2 m respectively, while the infinitely thick bottom layer has an apparent resistivity of 0.2 Ohm-m.

Correlating this with the drilling results, the top layer is mainly medium to coarse sand. On the whole, the top layer resistivity is comparatively low in this northern sector and could be due to partial saturation of the sand caused by continuous irrigation in the area. The high organic and inorganic fertiliser concentrations could also account for such low resistivities. The second zone corresponds to the relatively thick sandy-clay. The third layer is a medium to coarse freshwater sand aquifer. Drilling was not extended beyond 14 m at this sector. However, based on the analyses of lithological logs of other water supply boreholes in the area, and also geological, hydrogeological and field observations, it was confirmed that the extremely low resistive bottom layer is a thick layer of clay, probably saline water-saturated.

Similarly, Figure 17.4 represents the composite VES/lithological log for the test wells AN4, AN5, and AN6 located near the Gulf of Guinea coast.

From the modelled VES curve, it was observed that the southern sector appears to be underlain by a five-layer resistivity structure with the first and second layers having apparent resistivities of 365.8 and 1685.3 Ohm-m and thicknesses of 0.4 and 0.5 m respectively. Ideally, both layers constitute the unsaturated zone, which from the drilling records, is about 1.5 m thick. For practical purposes, it becomes necessary to combine the layers as one and assign one apparent resisitivity value to it. It is important to note that the choice of the second layer value of 1685.3 Ohm-m for the combined layers is based on in-depth knowledge of the hydrogeophysical characteristics of the rocks in the area and also on the values obtained for similar layers at VES stations close to the borehole. Thus the top layer has an average apparent resistivity of 1685.3 Ohm-m and a thickness of 1.3 m. The second layer has resistivity of 38.2 Ohm-m and a thickness of 12.3 m. The third layer has a resistivity of 4.5 Ohm-m and is 17.8 m thick, and the infinitely thick bottom layer has an apparent resistivity of 253.9 Ohm-m.

The highly resistive top layer represents desiccated medium-coarse sand. The second resistivity layer represents the freshwater shallow aquifers consisting of medium-coarse and gravely sand.

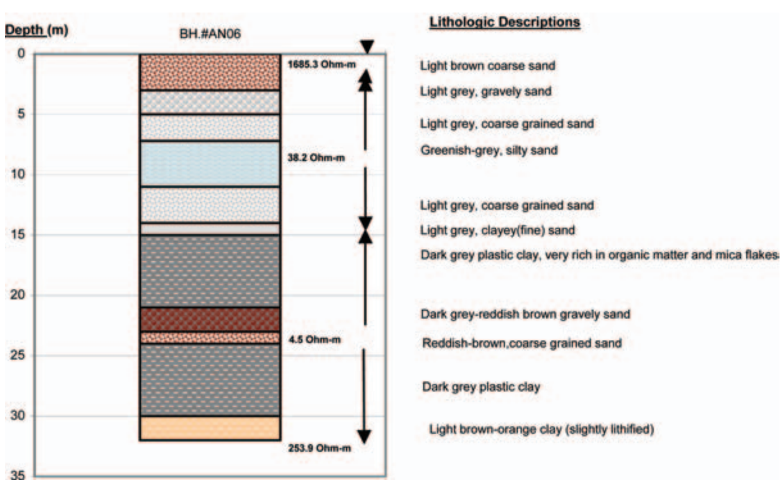

Figure 17.4. A representative composite graph of lithological logs with layer resistivities superimposed (southern sector).

The depth to the top of this second freshwater layer was mapped from the first water strike in the test wells AN4 and AN6 at a depth of 2.3 m. The third layer basically consists of very fine sand and silts that grade sharply into thick layers of clay with intervening sandy, gravelly and calcareous bands (Fig. 17.4). Current borehole information on the area does not exist beyond 34 m. However, the lithological log of the deep exploration well in the area reveals the presence of alternating layers of sand and gravel with occasional silt, clay and calcareous matter intercalations at depths beyond 34 m.

The extent of saline water contamination and lithologic variations with depth was also investigated using VES data. The apparent resistivity values for six different Schlumberger half-current electrode spreads (AB/2) = 1.58, 3.98, 10, 39.8, 63.1 and 100 m were selected. These VES investigations were to find out the extent of saline water intrusion as water pumping is intensified, and how far the fresh water-saline water interface is flushed during the wet season. Figures 17.5a to 17.5d show selected iso-resistivity contour maps for the soundings in the area at various depths. Each map

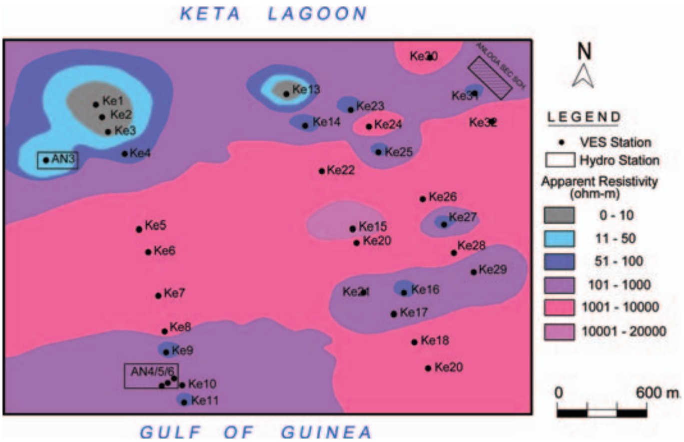

Figure 17.5a. Iso-resistivity map for AB/2 = 3.98 m (at a depth of 5.3 m).

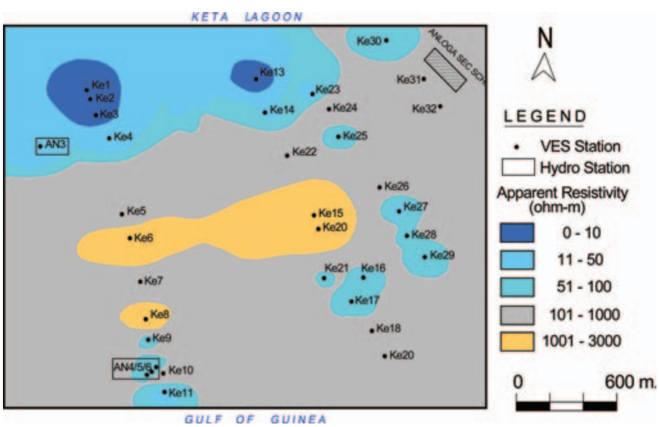

Figure 17.5b. Iso-resistivity map for AB/2 = 10 m (at a depth of 13 m).

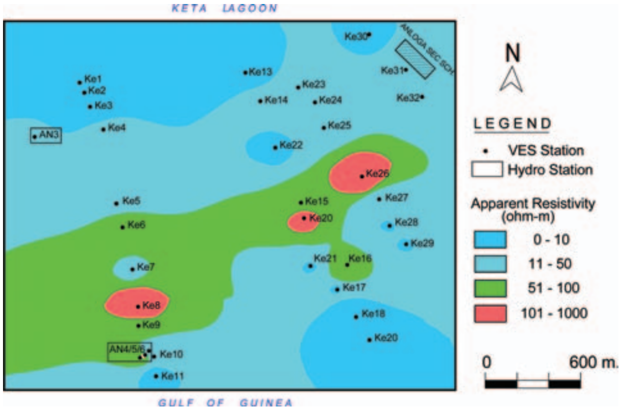

Figure 17.5c Iso-resistivity map for AB/2 = 39.8 m (at a depth of 53 m).

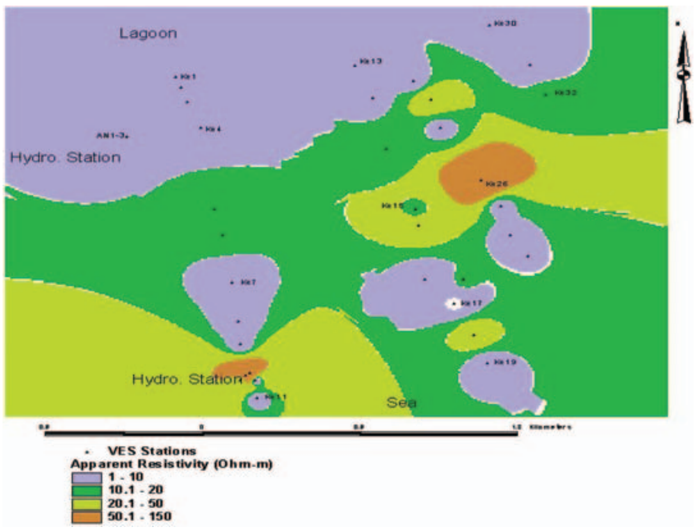

Figure 17.5d. Iso-resistivity map for AB/2 = 100 m (at a depth of 130 m).

represents a slice of resistivity variation at the depth of investigation. Table 17.2 presents the statistical summary of the resistivity results for the individual half-current electrode spreads and their corresponding approximate depths.

From top to the bottom, the layers are arranged in an increasing order of half-current electrode spread (AB/2), indicating mapping of the subsurface with depth. The results show that generally the layer resistivities decrease with increasing depth of sounding, indicating a general increase in the conductive properties of the material with depth. However, the mean resistivities measured at greater depths (53, 84 and 130 m) showed that as the sounding depth increased, the resistivities also increased, except for pockets of low resistivity zones that indicate possible saline zones or clay horizons. Drilling results also indicate that very low resistivity areas are underlain by thick plastic clays, which in part serve as the confining layers to the sand and gravel aquifers. The presence of the clay layers may

Table 17.2. Statistical summary of the resistivity distributions for the AB/2 spreads.

Current electrode spread (AB/2) m	Resistivity range (Ohm-m)	Mean resistivity (Ohm-m)	Standard dev. (Ohm-m)	Average depth (m)
1.38	0.69–18,432	2527.00	4002.99	2.1
3.98	1.38–15,488	1536.46	2870.00	5.3
10	2.12–2957	341.51	613.48	13.0
39.8	2.75–70.13	16.72	13.12	53.0
63.1	1.69–81.28	12.61	15.73	84.0
100	0.62–145.66	17.49	31.15	130.0

also impede the downward movement of the contaminant front. This may have accounted for the presence of fresh water in the sand and gravel layers located beneath thick clay layers in such a saline environment. A typical example of this was observed in the 14 m deep test well (AN3) located close to the flood plains of the lagoon.

Land-use patterns, to some extent, reflect the resistivity measurements and distribution in the area. Results indicate that, where extensive agriculture (crop cultivation) is undertaken, the freshwater sand and gravel aquifers have generally lower resistivities than in the uncultivated areas. This could be due to an increase in conductance of the groundwater due to an increase in the concentrations of chemical fertilisers on farms.

It is clear from the geophysical investigations that, despite the increased water used for crop irrigation in the Anloga area, there is no intrusion of salt water into the fresh water of the confined sandy aquifers. The interface is located at a depth below 50 m. The southward pressure of groundwater recharged from the upper parts of the shallow aquifers has controlled the possibility of saline water inflow into the shallow unconfined freshwater aquifer.

3 CONCLUSIONS

In general, the electrical layering interpreted from VES corresponds satisfactorily with the lithological logs from the test wells, except for small discrepancies in some resistivity values. The slight disagreements between resistivity and lithological results were found to be due to a general problem associated with the electrical resisitivity method when carried out on such a terrain; the problem of lithological suppression.

Based on the analyses of electrical resistivity data, lithological logs and hydrochemical data from the study area, the following conclusions are drawn:

- Resistivity less than 1′Ohm-m characterizes saline contaminated zones. The suspected saline water-freshwater interface is demarcated by resistivities $1 < P_a < 4'$Ohm-m. The clays are characterised by apparent resistivities of $4 < P_a < 10'$Ohm-m, whereas values ranging from 10 to 700′Ohm-m are used to define the freshwater aquifers in the area. Apparent resistivities of $1000 < P_a > 2000'$Ohm-m are assigned to the partially saturated freshwater sands and gravel layers, and values greater than 2000′Ohm-m represent the dry, clay-free top sands.
- The freshwater-saline water interface is not uniquely defined using VES data alone. Test drillings near the flood plains of the Keta Lagoon and close to the Gulf of Guinea coast failed to intercept the saline water aquifers. The freshwater-saline water interface could be located at depths beyond 20 m, near the lagoon and 40 m close to the coast. The interface is, however, inferred to occur at about 70 m at the midpoint between the sea and the lagoon, but could be deeper in areas underlain by VES Stations Ke15, Ke16 and Ke20.
- The northern sector of the project area is characterized by resistivities of less than 10 Ohm-m, interrupted by thin peaks of relatively high resistivities that indicate sandy layers with fresh water. Hence, the low resistivities must be attributed to clayey, poorly permeable layers, which at some locations contain saline water.

- There are few instances where VES curves reflect high resistivities at greater depths by steeply rising apparent resistivities at larger electrode spacing. In the general case, this reflects bedrock of high resistivity at depth. However, the depth to the bedrock in the study area is far beyond the depth investigated during the study. The most probable explanation is the presence of another sand or gravel aquifer containing fresh water.
- A multi-layer aquifer system confined by thin bands of clay and sandy clay layers separating the sand and gravel aquifers exists in the area. The sequence is simple at shallow depths, consisting only of sand and gravel, but generally complex at greater depths where sandstone and limestone formations intercalate with sand and gravel layers. The clays forming the aquitards were not resolved by the resistivity method in some instances, especially where they occur as thin bands. This is probably due to the problem of lithological suppression. Similarly, certain prolific aquifers consisting of sand and gravel beds, but occurring within thick clay layers, could not be distinctly resolved.

REFERENCES

Armah, T.E.K. 1995. Integrated geophysical and remote sensing technique to groundwater exploration in parts of the Voltaian Basin (West Gonja Distrct) – Northern Region, Ghana. M.Phil. thesis, University of Ghana, Legon. 139 pp.

Anku, Y.S. 2003. Groundwater Vunerability Assessment in the Keta Basin, Ghana. Unpublished thesis, University of Ghana, Legon.

Ayers, R.S. 1975. Quality of water for irrigation. Proc. Irrigation, Drain Div; Specialty Cont; Am. Soc. Civ. Eng. August 23–25, Logan, Utah, pp 24–56.

Banoeng-Yakubo, B.K., Danso, S.K. & Tumbulto, W.J. 2003. Assessment of Pollution Status and Vulnerability of Water Supply Aquifers of the Keta, Basin, Ghana. UNEP/UNESCO/HABITAT project.

Banoeng-Yakubo, B.K., Danso, S.K. & Tumbulto, W.J. 2004. Assessment of Pollution Status and Vulnerability of Water Supply Aquifers of the Keta, Basin, Ghana. UNEP/UNESCO/HABITAT project.

Carruthers, R.M. & Smith, I.F. 1992. The use of Ground Electrical Survey methods for Sitting Water-Supply Boreholes in Shallow Crystalline Basement Terrains – British Geological Survey, Keyworth, Nottingham NG12 5GG U.K.

Jorgensen, N.O. & Banoeng-Yakubo 2001. Environmental Isotopes (O18, H2, and 87Sr/86Sr) as a tool in groundwater investigations in the Keta Basin, Ghana, Hydrogeology Journal, 9: 190–201.

Kesse, G.O. 1985. The Mineral and Rock Resources of Ghana, A.A. Balkema/Rotterdam/Boston.

Kortatsi, B.K. & Agyekum, W.A. 1999. Final report on environmental impact assessment of large-scale groundwater abstraction in the Keta strip of the Volta Region of Ghana – MOFA/ASIP.

Helgensen, J.O., Leonard, R.B. & Wolf, R.J. 1993. Hydrology of the Great Plains Aquifer Systems in Nebraska, Colorado, Kansas and adjacent areas: Regional Aquifer Systems Analysis – Central Midwest. United States Geological Survey Professional Paper 1,414-L. U.S. Govt. Printing Office, Washington; 1993.

Hem, J.D. 1985. Study and interpretation of the chemical characteristics of natural waters. 3rd edition, United States Geological Survey Water Supply Paper 2254.

Hotor, V. 2004. Hydrochemical and hydrogeological Investigations in the Keta Basin, Ghana. Unpublished M.Phil thesis. University of Ghana, Legon.

Maathuis, H., Wak, W. & Adi, S. 2000. Hydrogeology of the Coastal basins of Jakarta, Indonesia. In: Sililo et al. (eds), Groundwater: Past Achievements and Future Challenges. A.A. Balkema/Rotterdam/Bookfield/ 2002, pp 209–213.

Parasnis, D.S. 1979. Principles of Applied Geophysics. 3rd edition, Chapman and Hall. 275 pp.

Roy, A. and Apparao, A. 1971. Depth of investigation in direct resistivity processing, Geophysics, 36 (5): 943–959.

Tibbals, C.H. 1990. Hydrogeology of the Floridan Aquifer System in East-Central Florida Regional Aquifer System Analysis – Floridan Aquifer Systems. United States Geological Survey Professional Paper 1403-E.

Whiteley, R.J. 1985. Modern Applied Geophysics for Groundwater Exploration and Development, Conference on geology and mineral Resources Development of the Northeast, Thailand, Khon Kean University, Khon Kean, pp 26–29.

Zohdy, A.A.R. 1974. Application of Surface Geophysics to Groundwater Investigation, US Geological Survey Techniques Of Water Resources Investigations Book Chapter D1.

18

Degree of groundwater vulnerability to pollution in Addis Ababa, Ethiopia

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ABSTRACT: To obtain relevant information on the quality of water and the aquifer response to contamination, pertinent and wide-range hydrological, geological, hydrogeological, chemical and bacteriological data have been collected and analysed in the Addis Ababa area. The main pollution indicators such as pH, nitrate, ammonia, electrical conductivity, bacterial population, etc. have been constantly monitored for two years at selected water points. Due to random disposal of solid and liquid wastes, the streams and rivers that cross the city are extremely polluted and consequently springs and shallow groundwater systems are affected. The results indicate that the rapid development of urbanization, along with industrial expansion, has deteriorated the natural environmental quality of water bodies in the area in particular and the city of Addis Ababa in general. Groundwater vulnerability mapping using the DRASTIC model indicates that a large proportion of the city lies on a high-risk zone and a limited sector of the city falls in the medium vulnerability category. However, in all cases, the quality of groundwater has deteriorated due to infiltration of pollutants from different human activities.

1 INTRODUCTION

Addis Ababa was established as the capital city of Ethiopia in 1886 and has grown to become the largest urban and commercial center of Ethiopia. It is currently host to over 4 million residents. The project area encompasses the city of Addis Ababa and its environs that are specifically located at 9°2′N, 38°42′E. The city lies on a steep slope declining towards the south. The northern elevation is 2,800 meters above sea level (m a.s.l.) and the southern end is 2100 m a.s.l.

For the first 58 years (1886–1944), the water supply for Addis Ababa was derived from groundwater in the form of springs located at the foot of the Entoto Ridge in the northern part of the city, and dug wells located in the central and southern parts of the city. Additional demand necessitated the treatment of surface water derived from three surface reservoirs (Gefersa, Legedadi and Dire) situated in the north-eastern and north-western parts of the city (Fig. 18.1). However, the dramatic population growth in the city necessitated the abstraction of groundwater in different localities and most prominently from the Akaki wellfield located in the southern part of the city.

It has been estimated that per capita water consumption will approximately double over the next 10 years as a result of several factors, including improved living conditions with a greater proportion of backyard plantation and sanitary facilities; good personal hygiene; increase in commercial and industrial demand, etc. The present demand creates a current supply shortfall of 25 per cent.

The impact of the human population on surface and groundwater is increasing with the development of industry and population size in the city. The state of groundwater contamination in the

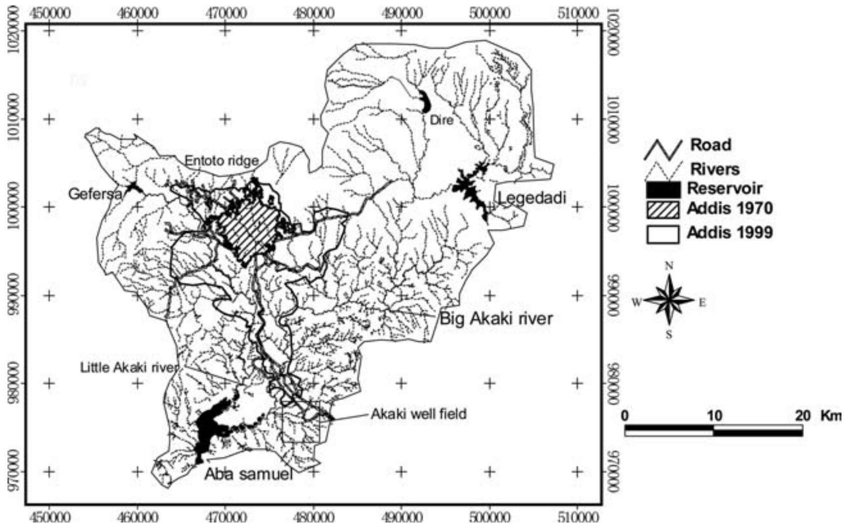

Figure 18.1. Urban expansion from 1970 (shaded) to 1999 within the Akaki River Basin.

city is similar to the reality in most developing countries. The level of water contamination tends to rise with the increasing human population and the low level of economic development. Consequently, the pollution of surface and groundwater is one of the most serious problems affecting the health of city dwellers. In addition, uncontrolled urbanisation and industrialisation, poor sanitation, uncontrolled waste disposal, etc. cause serious quality degradation of surface water and groundwater. One of the measurements of environmental sanitation is the safe and efficient disposal of domestic waste. According to the Central Statistics Authority, 74.1 per cent of the housing units in the city have toilet facilities (CSA, 1994). However, less than three percent of these discharge their wastes into proper sewerage networks, while the vast majority releases into streams and groundwater.

In the early 1970s, the size of the city was about 38 km^2 and by then it had a population of about 800,000, while in the year 1999, the size was 231 km^2 with a population of 2,300,000. Thus, within 30 years the city has expanded more than six-fold (about 193 km^2), with an average increase rate of about 6.7 km^2/yr. This has two major implications in terms of water supply for the city, (1) a sharp increase in the volume of water supply for urban use, and (2) the need for proper waste management to minimise water pollution.

The main objective of this work is to assess the current water quality status and to produce an aquifer vulnerability map for the water supply aquifers of Addis Ababa. From this study it is expected that decision makers or end-users such as industries will then be able to prioritise their actions and propose feasible water management plans. An integrated approach has been employed using mainly conventional methods. These include geological, hydrological and hydrogeological mapping and an exhaustive survey and analysis of hydrochemical, biological and isotope data. Of 328 boreholes used for piezometric mapping, 14 wells and five river points were selected for hydrochemical monitoring on a monthly basis. Major ion and bacteriolog ical analyses were carried out to investigate the anomalies. By converging evidence from the different methods, an aquifer vulnerability map has been developed using a valid point count system model (DRASTIC), created to assess aquifer vulnerability by USEPA (Aller *et al.*, 1987).

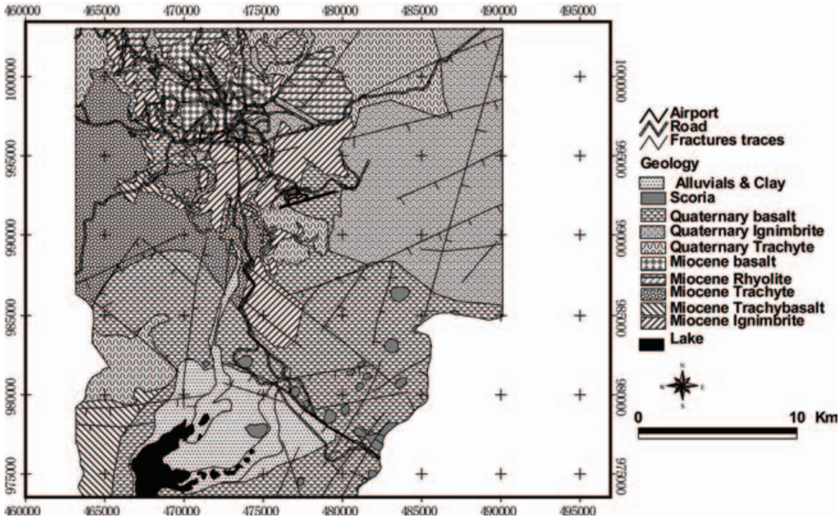

Figure 18.2. General geology of Addis Ababa.

2 GENERAL HYDROLOGY AND HYDROGEOLOGY

2.1 *Geology*

The city of Addis Ababa is located at the western margin of the Main Ethiopian Rift. Geologically, it is dominated by volcanic materials of different ages and compositions (Fig. 18.2). The Miocene-Pleistocene volcanic succession in the Addis Ababa area from bottom to top are: Alaji basalts and rhyolites, Entoto silicics, Addis Ababa basalts, Nazareth group, and Bofa basalts.

The Alaji group volcanic rocks (rhyolites and basalts) show a variation in texture from highly porphyritic to aphyric basalts, and there is an intercalation of gray and glassy welded tuff. The outcrop of Alaji basalt extends from the crest of Entoto (ridge bordering the northern parts of Addis Ababa) towards the north. The age of the rock is 22.8 Ma (Morton *et al.*, 1979).

The Entoto silicics is composed of rhyolite and trachyte, with minor amounts of welded tuff and obsidian (Zanettin *et al.*, 1978; Haileselassie and Getaneh, 1989). The rhyolitic lava flows outcrop on the top and the foothills of the Entoto Ridge. The rhyolites are overlain by feldspar porphyritic trachyte and underlain by a sequence of tuffs and ignimbrites. Tuffs and ignimbrites are welded and characterised by columnar jointing. The Addis Ababa basalt, mainly present in the central part of the city, is underlain by Entoto silicics and overlain by welded tuff of the Nazareth group. The Nazareth units are represented by welded tuff and aphanitic basalt. The Bofa basalts, of the age of 2.8 Ma, outcrop southward from the Akaki River with a thickness of around 50 m. They are restricted and dominant in the south-eastern part of the city, especially in the Akaki wellfield area. These rocks represent the major groundwater supply for large parts of Addis Ababa.

Elongated fault lines, running east-west, cut across the western rift escarpment and uplift its northern block about 8 Ma ago (Zanettin *et al.*, 1978). The fault marks the western margin of the Main Ethiopia Rift north of Addis Ababa. This fault has a down throw to the south towards the city. Another prominent normal fault in the city is the Filowha Fault. This fault has a trend of NE-SW located in the city center. The fault has a down throw to the south. Another fault from the railway station crosses bole road and formed nice graben marked with Filwoha fault. The graben is dominantly filled with transported soil (1–8 m thick). The faults act as a major conduit for deep heated water to the surface and horizontal diversion.

2.2 *Hydrogeology*

The variation in the seasonal distribution of rainfall in Ethiopia can be attributed to the position of the Inter-Tropical Convergence Zone (ITCZ), the relationship between upper and lower air circulation, the effects of topography, and the role of local convection currents. According to Daniel (1977), Addis Ababa is located in the region where the rainy months are contiguously distributed. In this region there are seven rainy months from March to September. The small rains occur in March and April while the big rains are from June to September. The total long-term average annual rainfall of Addis Ababa is 1150 mm. The city of Addis Ababa enjoys temperate and warm climatic conditions. The highest mean monthly maximum temperature occurs in March (24.6°C) and the lowest is in August (20°C). While the mean monthly minimum temperature ranges from 7.5°C in December to 11.6°C in March. Thus, the average temperature for Addis Ababa is 16°C.

Most of the streams emanate from the steeply rugged ridges of Entoto and flow across the city towards the relatively flat areas of the southern sector (Fig. 18.1). There are two major rivers, i.e. the Big Akaki and Little Akaki, which converge at the southern end of the study area. The amount of annual run-off depth for the Big Akaki River sub-basin is 142.4 mm, while the mean annual discharge for the Little Akaki River is 4.3 m^3/s.

The base flow component of the Big Akaki River from the data measured at Akaki Bridge was estimated as 1,020 l/s during the dry season (Tahal, 1992). However, we estimated this value to be 8500 l/s, while the estimated wastewater discharge into the river is about 2500 l/s.

The main aquifers in the project area are:

1. Shallow aquifer: composed of slightly weathered volcanic rocks and alluvial sediments. Depth to groundwater in this aquifer reaches up to 50 m.
2. Deep aquifers: composed of fractured volcanic rocks that contain relatively fresh groundwater. These aquifers are mainly located in the southern part of the project area of the city. The depth in places reaches as high as 180 m.
3. Thermal aquifer: that is situated at depths greater than 300 m and located in the center of the city. The existence of these aquifers is manifested by deep circulating thermal water with a few hot wells drilled along the major Entoto fault.

The water abstracted from volcanic aquifers for municipal supply is around 40,000 m^3/day (out of which 30,000 m^3/day is from the Akaki wellfield; Fig. 18.1). Other private and governmental institutions withdraw as much as 50,000 m^3/day with the overall total abstraction of 90,000 m^3/day. In some localities in the city, especially in the southern part, there are signs of over-abstraction (Tamiru, 2005).

The older Miocene rocks have been intensively weathered and are very good recharging sites for the groundwater system. From the productivity point of view, these rocks contain a small amount of water where the productivity of wells drilled in this formation do not exceed 5 l/s. The younger scoriaceous basalts located in the southern part of the city contain a high primary permeability, where some wells produce around 30 l/s.

The 700 m elevation difference between the northern and southern parts of the city drives groundwater to move southward along with the contaminants collected elsewhere in the city. The groundwater movement direction is dominated by a north-south and northwest-southeast flow. The flow lines converge towards the southern parts of the investigated area, around the Akaki wellfield. In some localities, however, the groundwater flow direction changes due to the occurrence of fault lines. In general, the groundwater movement is sub-parallel to the surface water flow direction and more or less controlled by the topography of the area.

3 RESULTS AND DISCUSSION

Water entering the subsurface from different sources could contain different chemical constituents that can alter the chemistry of water-bearing zones in the city. The resulting physical and chemical properties of groundwater are most importantly related to the encountered geological media and its residence time. In addition to the natural factors, a major change in the constituents of groundwater in the study area has resulted from human activities. The most important parameter that can indicate anthropogenic impact in the urban area is bacteria. The results are presented in Tables 18.1 and 18.2. The pH-values of river water range between 6.08 and 7.50 (Tamiru *et al.*, 2005). The pH of the water in the city is entirely influenced by the sewage input. Calcium and sodium are the dominant cations, while bicarbonate and chloride are the major anions. The main water groups are Na-Ca-HCO_3, Ca-Mg-HCO_3 and Ca-Cl.

From the current survey it is noted that the NH_4 concentration in the rivers reaches as high as 74.9 mg/l, indicating fresh sewage input into the rivers, while the NO_3 is around 66 mg/l. In groundwater, the NH_4 concentration is low (0.06 mg/l), while NO_3 reaches as high as 120 mg/l.

Pathogen bacteria, virus or parasites from human and animal excrement or wastewater are the main biological contaminants present in water. All streams in the city contain large amount of E. coli (Tables 18.1 and 18.2). This indicates direct discharge of sewages into streams.

The groundwater contamination problems are related to poor well construction and defective sewerage lines or septic tanks located close to water supply wells. Moreover, groundwater becomes polluted as a consequence of its interaction with the contaminated surface water. The total coliform count at different times indicates rapid infiltration of bacterially contaminated water into the shallow groundwater system. Such a drastic fluctuation is the result of the nature of the aquifer that permits infiltration of coliform bacteria through fractures, indicating the highly vulnerable nature of the aquifer.

The Biological Oxygen Demand (BOD) content induced due to organic waste disposal within the rivers shows an increasing trend from upstream to downstream and is attributed to the continuous input of organic waste from domestic activities. The Lideta River is one of the most polluted rivers with respect to organic waste (150 mg/l) that has a very high BOD and low Dissolved Oxygen (DO) content. Figure 18.3 shows the BOD and nitrogen (N) concentration at different times, illustrating that the rivers are under high influence of urban activities.

More than 200 small and large-scale industries are located along the Big and Little Akaki Rivers. These industries discharge both organic and inorganic wastes into the rivers, which aggravate the

Table 18.1. Coliform bacteria in surface water.

Location of sample	Total coliform/100 ml	E. coli/100 ml
Little Akaki upstream	2.4×10^4	2.4×10^5
Little Akaki downstream	3.5×10^6	3.5×10^6
Big Akaki upstream	170	2
Big Akaki downstream	5.4×10^5	2.4×10^5
Lideta River	5.4×10^6	5.4×10^6

Table 18.2. Bacterial concentration in the groundwater at selected sites.

Location of sample	Total coliform/100 ml	E. coli/100 ml
Ras Mekonen Spring	160	160
Lideta Spring	46	9
Building College	10	1
Dewera Guda (Aba Samuel)	160	160
Legehar dug well	9	3
Addis tyre	3	1
Akaki 1	16	16
Fanta spring	16	16
Akaki 2	89	89

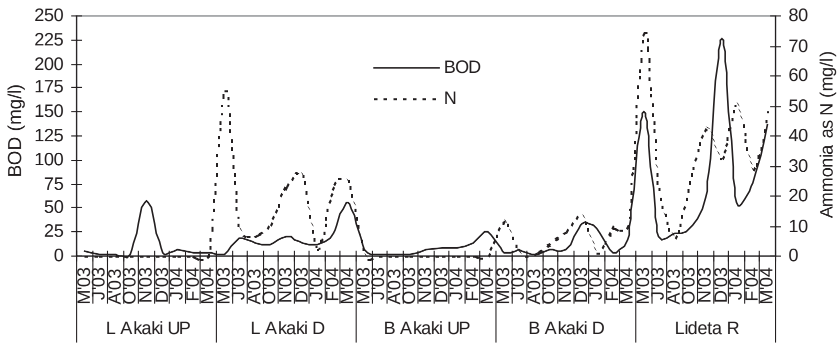

Figure 18.3. BOD variations in comparison with ammonia as N in rivers (2004).

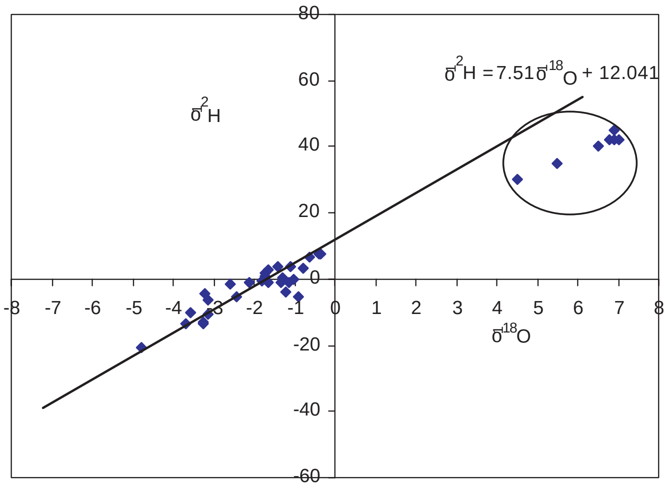

Figure 18.4. Stable isotope plot for Addis Ababa 1 = Rain, 2 = Borehole, 3 = Aba Samuel Lake (located at the southern tip of the project area), 4 = Hand-dug wells and spring, 5 = River.

degree of pollution of water bodies. Most of the large-scale industries are located in the southern part of the city, where there is a productive aquifer that currently supplies the city.

An attempt has been made to see the interaction of the different water bodies on the basis of environmental isotopes from previous studies and the current work. The isotopes used are ^{2}H, ^{18}O and ^{3}H. The isotopic signature for the different water bodies is given in Figure 18.4. The equation for the Local Meteoric Water Line (LMWL) plotted from rainfall data for Addis Ababa is $\delta^2H = 7.51\ \delta^{18}O + 12.12$. The result indicates that some groundwater samples are isotopically lighter than the average rainfall isotopic signature. This indicates the large groundwater recharge from intensive summer rains that travels all the way from the plateau to low-lying areas at relatively deeper levels.

The river water, which is relatively enriched with heavy isotopes, can mix with the direct infiltrating rainwater in the unsaturated zone. The process of mixing can also result from anthropogenic activities, which is the most probable process in addition to the evaporation at the surface or in the unsaturated zone during infiltration. These waters underwent evaporation in surface reservoirs where they are collected and stored. The wastewaters will mix with non-evaporated groundwater from the shallow system after use. The shallow groundwater found under the influences of local infiltration shows enriched isotope values (Fig. 18.4), which is correlated with the aquifer's high vulnerability to pollution. Generally, the shallow aquifer of Addis Ababa acts as an open system that allows mixing of groundwater with polluted surface water.

Water pollutants do not always have the ability to enter the groundwater system; instead the pollutant tends to be removed or reduced in concentration with time and distance travelled. The rate of pollution attenuation depends on the type of pollutants and on the local hydrogeological situations. Moreover, the mechanism of pollution attenuation includes filtration, sorption, chemical processes, microbiological decomposition and dilution (Vrba and Zoporozec, 1994; Civita and De Maio, 1998). Natural attenuation is not evident from the results in this study.

After exhaustive conventional mapping, an aquifer vulnerability map has been prepared. The preparation of an aquifer vulnerability map is a key consideration and becomes a forecasting tool. Via the planning processes, a prevention tool. A valid point count system model (DRASTIC) was used by the United States Environmental Protection Authority to assess the aquifer vulnerability (Aller *et al.*, 1987). The intrinsic vulnerability map is based on the assessment of various natural factors or attributes, such as soil, unsaturated zone, aquifer properties, recharge rate, slope, etc. into the determination of the vulnerability of groundwater to pollution. The application will be acquired when the intrinsic vulnerability of a certain area is associated with danger sources. In this case we are talking about specific vulnerability, which is defined by the interaction of intrinsic vulnerability of the hydrogeological system and danger sources. The hydrogeological factors defined in the DRASTIC system are many. DRASTIC stands for: D = Depth to water, R = net Recharge, A = Aquifer media, S = Soil media, T = Topography/slope, I = Impact of vadose zone media, C = hydraulic Conductivity. For each, parameter rating (r) and relative weights (w) are assigned with respect to their importance in aquifer vulnerability to pollution. DRASTIC Index (DI) is computed using the assigned rates (r) and weights (w) for each DRASTIC factors or parameters as follows:

$$DI = D_rD_w + R_rR_w + A_rA_w + S_rS_w + T_rT_w + I_rI_w + C_rC_w$$

Using Arcview GIS, the data were analysed and shape files constructed, followed by a girding procedure that resulted in raster files According to the ratings given, raster overlay operations were done using the different maps under the GIS environment to come up with final aquifer vulnerability maps. The final map showing DI is given in Figure 18.5.

Specific vulnerability is assessed in terms of the danger of the groundwater system becoming exposed to a certain contaminant load. The major attributes involved in assessing specific vulnerability include land use and population density. In this regard, the main land use considered is the built-up area for residential and business activities. The calculated population density was grid and superimposed on the intrinsic vulnerability map (Fig. 18.5). The result indicates that the more densely an area is populated, the greater the potential and real contaminant load on the groundwater system (Fig. 18.6). The densely populated area of Addis Ketema, Lideta, and Arada fall in a very high vulnerability zone. Due to low population density and agricultural land use, the peripheral part of the city falls in a low vulnerability class.

4 CONCLUSIONS AND RECOMMENDATIONS

The study indicated that the volcanic aquifers of Addis Ababa are highly vulnerable to contamination, owing to their high degree of fracturing and weathering that allow the fast movement of contaminants into the groundwater zone.

The main sources of these contaminants that deteriorate the quality of water in Addis Ababa are wastes generated from domestic activities followed by industries, garages, health centers and fuel stations. Moreover, rivers are characterised by objectionable physical and chemical properties. There is a temporal variation in EC, TDS and pH more rapidly both in surface and shallow

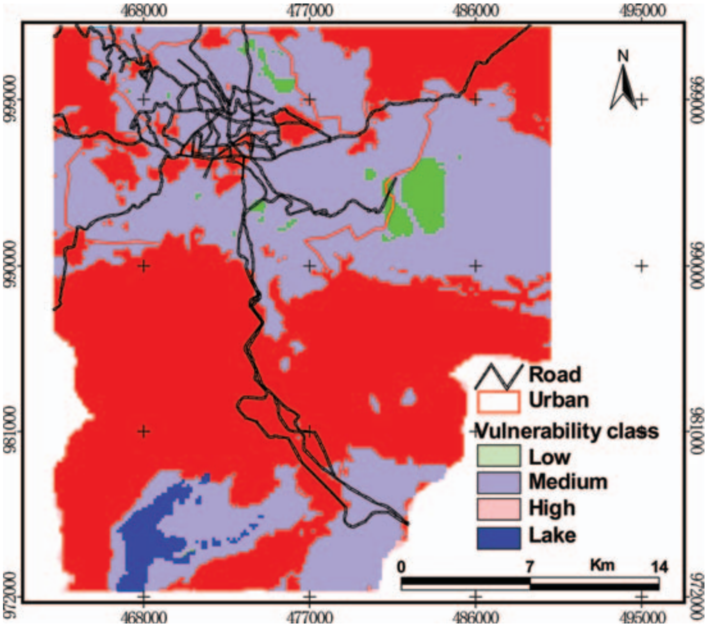

Figure 18.5. Intrinsic vulnerability (DI).

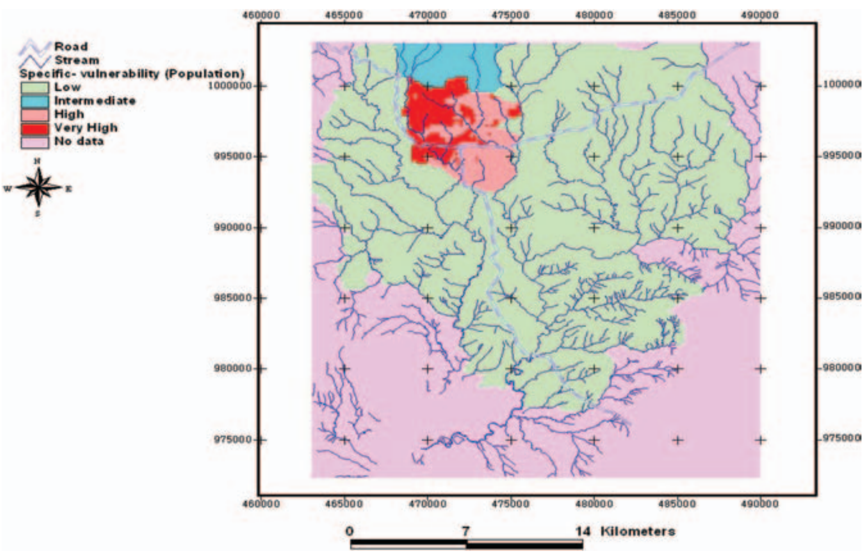

Figure 18.6. Specific vulnerability map with respect to population density.

groundwaters, attributed to the injection of different types of wastes. Therefore, the quality of groundwater depends on the quality of surface water and it is wise to control the quality of surface water to maintain the quality of groundwater. Intrinsic vulnerability mapping for the water supply aquifers of Addis Ababa revealed that a major part of the city lies in a medium risk area, while the southern aquifer is highly vulnerable to pollution. Most of the industries are situated in a high vulnerability zone. An intrinsic vulnerability map represents the distribution and the extent of groundwater potentially sensitive to pollution. It is believed that such an assessment and mapping activity is useful in land-use planning and decisions, policy analysis and development processes to identify the potential for groundwater pollution, and improving the general awareness of the population.

In a struggle to fulfill the demand gap (25 per cent shortfall) of Addis Ababa and meet future projected demands, the city is increasingly relying on supplies from groundwater sources such as the Akaki Wellfield, which requires attention to protect it from localised pollution. There is also a continuous effort by the municipality to establish new wellfields to fill the existing gap. An extensive geophysical survey has been carried out in different parts of the city periphery and drilling is in progress for potential studies.

It is prudent to continue monitoring the groundwater and surface waters and to trace and map the temporal and spatial variations of pollutants. Integrated environmental protection approaches and political commitment is essential for the effective protection of water supply aquifers.

ACKNOWLEDGEMENT

We gratefully acknowledge the UNEP and UNESCO Nairobi offices for their initiatives to bring together African researchers in the field of groundwater vulnerability mapping. The research is fully financed by UNEP and implemented by UNESCO. We thank Prof. Salif DIOP, Mr. Patrick Mmayi, Dr. Emmanuel NAAH, Prof. Yongxin Xu, Dr Brent Usher and others for their invaluable assistance.

REFERENCES

Aller L., Bernnett T., Lehr J.H., Petty R.J. & Hackett G. 1987. DRASTIC: A standardized system for evaluating groundwater pollution potential using hydrogeological settings. USA.

Civita and De Maio 1998. Mapping Groundwater Vulnerability By the Point Count System. Pitagora Editrice, Italy.

Central Statistical Authority, CSA 1994. Population and Housing Census of Addis Ababa. Analytical Report. V.II. Addis Ababa.

Daniel G. 1977. Aspects of Climate and Water Budget in Ethiopia. A Technical Monograph Published For Addis Ababa University. Addis Ababa University Press.

Haile Sellasie G. & Getaneh A. 1989. The Addis Ababa-Nazareth Volcanics: A Miocene-Pleistocene Volcanic Succession in Ethiopian Rift. SINET, Ethiopian Journal of Science 12 (1):1–24.

Morton W.H., Rex D.C., Mitchell J.G. & Mohr P.A. (1979). Rift ward Younging of Volcanic Rocks in the Addis Ababa Region, Ethiopian Rift Valley. Nature V. 280:284–288.

Solomon T. 2000. The extent of water pollution in Addis Ababa. Unpublished MSc Thesis, Addis Ababa University.

Tahal Consulting Engineers 1992. Akaki Water Supply Project. Feasibility Study. Vol. I, Main Report. AAWSA, Addis Ababa.

Tamiru A. 2001. The impact of uncontrolled waste disposal on surface water quality in Addis Ababa. SINET: Ethiopian Journal of Science 24(1):93–104.

Tamiru A. 2005. Uncontrolled groundwater exploitation from vulnerable aquifer of Addis Ababa, Ethiopia. International Workshop on groundwater management in arid and semi arid countries. 4–7 April 2005. Cairo, Egypt.

Tamiru A., Dagnachew L., Tenalem A., Yirga T., Solomon W.N. & Nuri M. 2005. Water quality assessment and groundwater vulnerability mapping of Addis Ababa water supply aquifers. UNEP reprt.

Vrba J. & Zoporozec A. (Eds) 1994. Guidebook On Mapping Groundwater Vulnerability. Vol. 16/1994. IAH Verlag Heinz Heise.

Zanettin B., Justin Visentin & Piccirillo 1978. Volcanic Succession, tectonics and Magmatology in Central Ethiopia. Padova, Italy.

19

Vulnerability and pollution of groundwater in Kisauni, Mombasa, Kenya

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ABSTRACT: Rapid urbanisation in the Mombasa District, and in particular the Kisauni area, has increased the demand for essential services, notably water supply and waste management infrastructure. This is manifested in inadequate clean drinking water from the reticulated supply, leaving the inhabitants with groundwater to supplement their resources, or in most cases as the sole option. An assessment of the intrinsic aquifer vulnerability to contamination was carried out by applying the DRASTIC model coupled with GIS analytical tools. Monitoring data on physico-chemical characteristics showed raised concentrations of nitrates in groundwater, in particular, in the more densely populated Kisauni areas, attributed to contamination from on-site sanitation systems dominated by pit latrines and septic tank-soak pit systems and uncollected municipal refuse. Concentrations of NO_3^{-1}/NO_2^{-1}-N ranged from 0.4 to 44.4 mg l^{-1}, with an indication of seasonal variations. About 50% and 70% of the water samples tested in June/July and November, respectively, did not exceed the 10 mg l^{-1} NO_3^{-1}/NO_2^{-1}-N guideline level set for potable water by WHO. The Kisauni area is indicated as experiencing a high degree of groundwater contamination by microbial contaminants, especially in the high-density housing settlements, attributed to on-site sanitation. The contamination levels are more severe during the rainy season, when aquifer recharge is high. A suggested strategy for intervention includes the control of pollution sources, education and awareness creation, and the implementation of existing laws and regulations to protect and manage groundwater resources.

1 INTRODUCTION

The rapid growth of the population in Mombasa City has exerted relentless pressure on limited resources and services such as housing, water supply and sanitation, education and health facilities. The increased demand in housing has resulted in mushrooming unplanned settlements and slums, with inadequate or lack of water supply and sanitation services. Consequently, inhabitants have had to increasingly rely on groundwater to supplement their sources, or as the sole source of

Table 19.1. Out-patient morbidity annual averages in Mombasa District, 1998–2000.

Disease	Island	Kisauni	Changamwe	Likoni	Total	%
Diarrhoea	1846	2019	1714	157	5736	5.1
Malaria	12,091	14,931	11,380	1456	39,858	35.5
Worms	1082	1089	713	72	2955	2.6
Eye inf.	822	517	367	51	1758	1.6
Skin inf.	3530	4642	3664	506	12,343	11.0
Others	17,905	16,141	14,071	1457	49,574	44.2

Source: Mwaguni, (2002).

potable water supply in most parts of the city. However, groundwater in the area is under threat of contamination due to the utilisation of on-site sanitation facilities, dominated by pit latrines and septic tank-soak pit systems. Inadequate solid waste collection and disposal services have resulted in mounds of uncollected domestic refuse that are sources of groundwater contamination through leaching. With Mombasa being a coastal city, the increasingly uncontrolled abstraction of groundwater may eventually reverse the natural hydraulic gradient and cause seawater encroachment.

In formulating the national water policy and management strategy, it was recognised that the major causes of morbidity are due to diseases or conditions arising from the low level of safe drinking water, lack of hygienic sanitation and poor environmental conditions (GOK, 2002). Mwaguni (2002) documented that over 50 per cent of all reported diseases in Mombasa from 1998–2000 were water-borne and associated with inadequate wastewater management (Table 19.1). This raises the need to address the problem of groundwater contamination with the view of monitoring the situation and formulating possible mitigation measures.

The aim of this study was to establish the pollution status of the water supply aquifer in the Mombasa district, focusing on the Kisauni area of the north mainland, with the following specific objectives.

- Analysis of the hydrogeological set-up of the area, and preparation of a hydrographic model of the aquifer and a pollution vulnerability map.
- Assessment of the pollution status of groundwater in Kisauni.

It was envisaged that the study would reveal information on the vulnerability of groundwater and provide an indication of the pollution status of this key resource in the area. The vulnerability assessment is expected to highlight the critical areas and anthropogenic activities that contribute most to groundwater pollution and thus provide a knowledge base for taking appropriate action to protect the resource. The information will be useful in raising the awareness of decision makers and the public on the vulnerability of groundwater and their responsibilities for protecting the resource.

1.1 *Description of the study area*

The Mombasa district lies between latitudes 3° 80′ and 4° 10′ S and longitudes 39° 60′ and 39° 80′ E, with a total land mass of 229.6 km^2 and inshore waters covering 65 km^2. The administrative boundaries comprise the Island Division, Changamwe in the west, Kisauni in the north and the Likoni Division in the south. The Island Division is the smallest and most developed, while the three other suburban divisions are predominantly rural. The thrust of this study was the Kisauni area, or Mombasa north mainland (Fig. 19.1).

Climatic condition variations in the district are attributed to SE Monsoon winds (blowing between April and September) and the NE Monsoons (October to March) and oceanic influence. The rains occur during the inter-monsoonal period, with the long rains starting from March to June, while the short rains occur from October to November/December. The mean annual rainfall in the period from 1999 to 2004 was 956 mm, peaking in May and October.

The Mombasa district is situated on the coastal lowland with extensive flat areas rising gently from 8 m above sea level to 100 m above sea level in the west. There are three main physiographic belts,

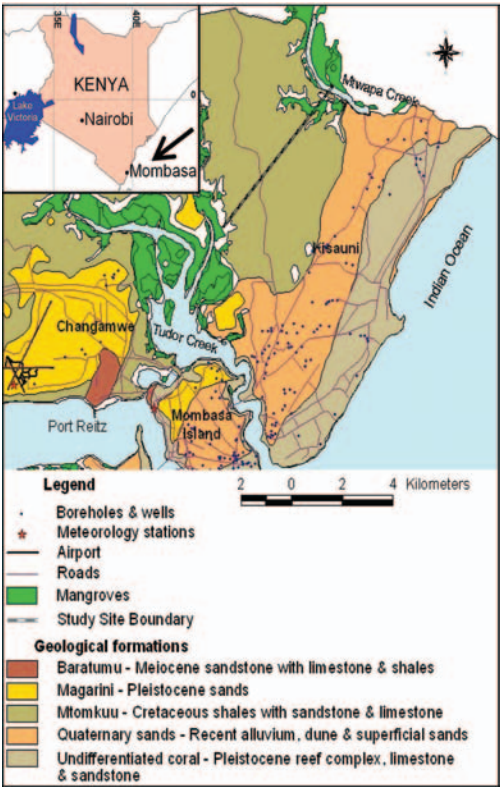

Figure 19.1. Map of the study site.

namely the flat coastal plain, which is 6 km wide, including the Island division, Kisauni on the north mainland and Likoni to the south. After this occurs the broken, severely dissected and eroded belt consisting of Jurassic shale overlain in places by residual sandy plateau found in the Changamwe division. Finally, there is the undulating plateau of sandstone that is divided from the Jurassic belt by a scarp fault. Closer to the sea, the land is composed of coral reef of Pleistocene age that offers excellent drainage (Fig. 19.1). The coral limestone and lagoonal deposit reach a thickness of 100 m.

The soil types are broadly associated with the geological formations along the physiographic zones in the district, as detailed by GOK Ministry of Agriculture (1988). Along the coastal lowlands, four soil types predominate. Overlying the raised reefs along the shore, well-drained, shallow (<10 cm) to moderately deep loamy to sandy soils predominate. The unconsolidated deposits in the quaternary sand zone are well drained moderately deep to deep sandy clay loam to sandy clay, underlying 20 to 40 cm loamy medium sand. On the quaternary sands there are also areas with very deep soils of varying drainage conditions and colour, variable consistency, texture and salinity. Also found on the quaternary sands are well-drained, very deep, dark red to strong brown, firm sandy clay loam to sandy clay, underlying 30 to 60 cm medium sand to loamy sand soils.

On the coastal uplands, consisting of the raised areas in Changamwe and the western parts of Kisauni, two soil types are dominant. Sandy to loamy soils developed on unconsolidated sandy deposits in the Magarini formation consists of well drained, very deep, sandy clay loam to sandy clay, with a topsoil of fine sand to sandy loam. Heavy textured soils developed on shales dominated by

Table 19.2. Population distribution in the Mombasa district.

Administrative division	Size: Area km^2	Population 1979	1989	1999	% Population increase since last census	Population density/ km^2 1999
Island	14.1	136,140	127,720	146,334	14.57	10,379
Kisauni	109.7	79,995	153,324	249,861	63.00	2,278
Likoni	51.3	39,665	67,240	94,883	41.11	1,850
Changamwe	54.5	81,348	113,469	173,930	53.28	3,191
Total	229.6	336,148	461,753	665,018	44.02*	2,896*

* Average values.
Source: GOK (1999, 1989 & 1979), Mwaguni (2002).

well drained to imperfectly drained, shallow to moderately deep, firm to very firm clay, and imperfectly drained deep, very firm clay, with a humic topsoil and a sodic deeper subsoil.

The Mombasa district has no permanent rivers, and the unconfined shallow aquifer on the island and coastal lowland areas of Kisauni and Likoni mostly depend on local recharge primarily through precipitation. Thus the sinking of boreholes and wells for the abstraction of groundwater to supplement the reticulated supply of fresh water has targeted the shallow unconfined aquifer (Fig. 19.1).

1.2 *Population and land use*

From the 1999 Population and Housing Census (GOK, 1999), the population of the Mombasa district stood at 665,000 persons distributed in the four divisions of the district as indicated in Table 19.2.

The rapid increase in population in the period 1979 to 1999 was attributed to natural growth and in-migration, mostly of the labour force from other parts of the country. The high population has proved to be a serious challenge in the provision of housing and essential services such as water, sanitation and health care.

A land-use classification study (UNEP/FAO/PAP/CDA, 1999) indicated that only 31.2% of the total land area in the Mombasa district fell under residential settlements. The direction of growth in human settlements is northwards, concentrated in the Kisauni division. This has resulted in the rapid growth of unplanned crowded settlements with very poor sanitation and generally poor infrastructural facilities (Gatabaki-Kamau *et al.*, 2000). Other significant socio-economic activities include beef and dairy farms, tourist hotels, the Shimo La Tewa School and Government Prison, the Kongowea wholesale market and Bamburi Cement Factory, which occupy large tracts of land.

1.3 *Water supply and waste management practices*

The main sources of freshwater supply for the Mombasa district are the Mzima Springs, located about 200 km west, the Baricho Water works, located about 150 km north and the Marere Springs and Tiwi Boreholes found about 40 km and 20 km, respectively, south of Mombasa, mainly supplying the Likoni area. The daily water demand for the district is approximately 200,000 m^3 against the available supply of 130,000 m^3. The water supply deficit of 70,000 m^3, about 35% of the demand, is met by exploiting groundwater sources (NWCPC, 2000). The shortfall in the water supply is further aggravated by the diminished capacity of the old and leaking reticulated supply network. With the current rapid increase in the urban population, the water supply deficit is expected to increase, thereby increasing the dependence on groundwater.

The shortage of water in Mombasa and lack of funds to undertake capital investment projects have constrained extensions of water-borne sewerage, compelling the residents to rely on on-site systems for sewage management. About 17% of households, as well as hotels and most public buildings, have septic tank and soakage pit systems. Most of the 13,000 septic tanks in use are found in high-income residential areas. A great majority of households (about 70%) use pit latrines.

Of the 34,000 latrines in the district, 55% are found in the Kisauni division, where the study area is located. It is a common practice to dig pit latrines to the water table to avoid filling up the pit in a short time. One housing estate in Kisauni discharges sewage and wastewater into an open area, which has evolved into a wetland with a poor capacity to treat the waste. The lack of adequate services for solid waste collection and disposal has resulted in a build-up of mounds of refuse in the high density housing settlements of Kisauni, posing a threat to public health. Less than 50% of the solid waste generated is collected and finally disposed of at a crude or uncontrolled dumpsite to the west of the Nguu Tatu hills in Kisauni. On-site disposal of both solid and liquid waste and the lack of appropriate sewage treatment are major sources of pollution of groundwater due to human waste through aquifer recharge.

2 GROUNDWATER FLOW CONDITIONS

The groundwater flow in the Kisauni area was assessed using the numerical model MODFLOW (Version 5.3.0) with PMPATH (Version 6.1.0) (Chiang & Kinzelbach, 1993; Pollock, 1988, 1989). The model boundaries were determined by considering physiological and hydrogeological features in the area. The model was bounded in the east by the Indian Ocean, in the north and south by the Mtwapa and Tudor Creek, respectively. Towards the west of Kisauni the land rises to form a ridge with three prominent peaks at over 120 m, locally known as the *Nguu Tatu* Hills. Beyond the ridge the land drops into undulating hills and valleys, rising gradually westwards. This physiographic feature, that is the *Nguu Tatu* Ridge, was considered a natural hydrologic boundary on the western side of Kisauni (Fig. 19.1). It is in the coastal lowland where housing settlements are concentrated and massive abstraction of groundwater is carried out from the aquifers in the quaternary sands and Pleistocene coral reefs.

The parameters used for the model simulation consisted of the topography, areal net groundwater recharge, and aquifer properties including hydraulic conductivity and thickness. The model was developed under steady-state conditions, with the relevant parameters input into the model, consisting of the following:

- Initial hydraulic head – actual heads for specific wells under dynamic conditions were input with the rest of the area remaining at zero.
- Aquifer topography – the top of the aquifer was kept at 30 m and bottom topography kept at − 100 m.
- Horizontal hydraulic conductivity was averaged at 2.31E−5 m s^{-1}.
- Aquifer recharge flux – there were two recharge zones in the area with the quaternary sand zone bearing the highest recharge rate at 6.7E−9 m s^{-1} and the coral reefs and shale areas in the west at 7.93E−10 m s^{-1}.

The output of the MODFLOW with PMPATH is presented in Figure 19.2.

The model indicates that, in the Kisauni area the dominant groundwater flow direction is towards the Mtwapa Creek along the northern boundary and Tudor Creek along the southern boundary of the study site, and relatively less intense flow eastwards towards the Indian Ocean (Fig. 19.2). The groundwater flow contributes significantly to maintaining the mangrove habitats, especially during the dry season when surface discharges are low. The model broadly agrees with the findings of Kitheka (1996), who reported a significant contribution of fresh water into the Nyali Beach Lagoon through groundwater flow (estimated groundwater flow of 186E+4 m^3 day^{-1} or about 2% of the total water volume in the lagoon) along the Kisauni shoreline.

3 AQUIFER VULNERABILITY

The intrinsic vulnerability to pollution of the water supply aquifer in, Kisauni particularly was assessed using the DRASTIC empirical model (Aller *et al*., 1985, 1987) coupled with GIS analytical tools (ESRI's ArcView 3.2 and Spatial Analyst 2.0) (ESRI, 1996a, b). The DRASTIC factors, namely **D**epth

Figure 19.2. Groundwater flow in Kisauni.

to the water table (**D**), net **R**echarge (**R**), **A**quifer media (**A**), **S**oil media (**S**), **T**opography (**T**), **I**mpact of the vadose zone (**I**) and hydraulic **C**onductivity (**C**), were assigned a rating according to their influence on the pollution of groundwater by a contaminant introduced on the surface or sub-surface. Typical ratings range from 1 to 10. The DRASTIC factors were assigned weights ranging from 1 to 5 relative to their significance in influencing groundwater contamination. The groundwater vulnerability to pollution is expressed by the DRASTIC Index (DI), which is the sum of the products of the ratings and weights of each factor. The higher the value of DI, the more vulnerable the area is to groundwater pollution. Thus DI can be represented by the following expression:

$$DI = D_rD_w + R_rR_w + A_rA_w + S_rS_w + T_rT_w + I_rI_w + C_rC_w$$

where the subscripts r and w denote rating and corresponding weight of the factor. The range of values of DI were then converted into qualitative values of low, moderate and high vulnerability. The GIS tools (ArcView 3.2 and Spatial Analyst 2.0) allow the computation of DRASTIC Indices, overlaying the factor DI values spatially and providing a spatial display of the intrinsic vulnerability of the area. The scheme for the analytical procedure, coupling the DRASTIC model with the GIS tools, is presented in the flow diagram (Fig. 19.3).

3.1 *Depth to the water table*

The piezometric data covered the Kisauni and Mombasa island area (Njue *et al*., 1994). The depth to the groundwater level ranged from 11.0 to 27.0 m. The interpolated depth to the aquifer indicates that the water table is shallowest in the south-eastern and towards the north of Kisauni and the south-western side of the island. This is reflected in the rating of the relative vulnerability of the aquifer due to depth. These shallow areas are indicated as the most vulnerable to pollution originating from the surface and sub-surface, with respect to the depth to the water table.

3.2 *Recharge of the aquifer*

The aquifer recharge was assessed by considering the mean annual rainfall distribution in the district (956 mm p.a.) and the relative permeability of the underlying geological formations. The following recharge rates were adopted for the five geological formations in the region (Table 19.3).

The higher the recharge rate, the more vulnerable the underlying aquifer.

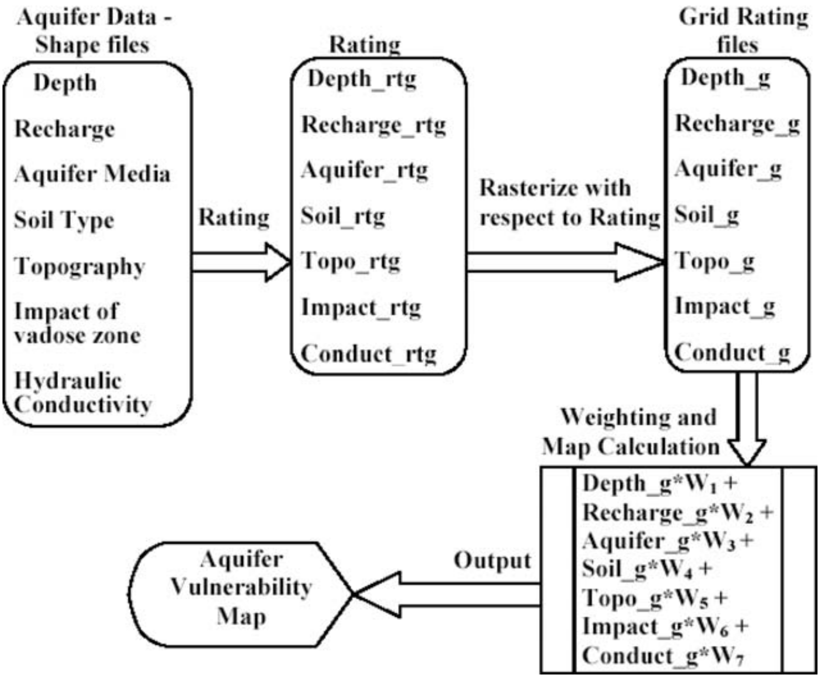

Figure 19.3. Schematic presentation of the analytical procedure.

Table 19.3. Water recharge rates and geological formations in Mombasa.

Geological formations			
Name	Formation	Recharge rate (%)	Recharge (mm p.a.)
Baratumu	Miocene sandstone with subordinate limestone and shales	3	28.7
Magarini sands	Pleistocene sands	8	76.5
Mtomkuu	Cretaceous shales with subordinate sandstones and limestone	2	19.1
Quaternary sands	Recent alluvium beach sands, dune sands and superficial sands	10	95.6
Undifferentiated corals	Pleistocene reef complex, limestone and sandstone	6	57.4

3.3 *Aquifer media*

The aquifer media determine its attenuation capacity to contaminants, influenced by, among other factors, the grain and pore sizes of rock material. In the saturated zone, contaminant attenuation is largely determined by dilution and natural die-off (in the case of microbial contamination). In the Mombasa district the aquifer media is broadly determined by geological formations. In particular, in the coastal lowlands, the geological formations are reported to extend to depths of 100 m, whereas the depth to the water level is as shallow as 11 m in Kisauni. The dominant aquifer media in the district include limestone, sandstone and shale. The ratings of the aquifer media to pollution

Table 19.4. Rating of aquifer media to aquifer vulnerability.

Aquifer media	Rating
Shale	2
Bedded sandstone, limestone and shell	6
Massive sandstone	6
Massive limestone	6
Sand and gravel	8
Karst limestone	10

Table 19.5. Rating of the hydraulic conductivity to aquifer vulnerability.

Range (m day^{-1})	Rating
<4	1
4–12	2
12–29	4
29–41	6

vulnerability in the district is presented in Table 19.4. The results indicate that the unconfined aquifer on the island and in the low-lying areas in Kisauni are the most vulnerable.

3.4 *Soil media*

The soil is the most biologically active layer and the first line of defense against groundwater contamination. It contributes significantly to the attenuation of contaminants introduced on the surface. The soil type, grain size and thickness play a limiting role in attenuation processes of contaminants, namely filtration, biodegradation, sorption and volatilisation.

3.5 *Topography*

The low-lying coastal zone is characterised by an even terrain, with cliffs sloping to the shoreline at certain places. Steep slopes are found in the raised areas towards the western part of Kisauni, especially along the Nguu Tatu Ridge, with peaks rising over 120 m. The slope influences run-off. The steeper the slope, the faster the run-off and reduced potential for groundwater contamination.

3.6 *Impact of the vadose zone*

The unsaturated layer or vadose zone has an impact on the attenuation of the contaminants in the aquifer. The material in the vadose zone is closely related to the geological formations. Thus, the zone is dominated by limestone, sandstone, sand and shale.

3.7 *Hydraulic conductivity*

The hydraulic conductivity determines the rate at which a contaminant moves, which depends on the inter-connectivity of voids within the aquifer. The higher the conductivity, the higher the vulnerability of the aquifer to pollution. The hydraulic conductivity of the aquifer in the district was estimated with reference to literature, because of the paucity of data from pumping tests. The ratings for hydraulic conductivity are presented in Table 19.5.

3.8 *Aquifer vulnerability*

The DRASTIC model factors, namely depth, recharge, aquifer media, soil media, topography, impact of the vadose zone and hydraulic conductivity, were weighted (Table 19.6) and overlayed.

Table 19.6. Weights for DRASTIC factors.

Factor	Weight
Depth to water table	5
Aquifer recharge	4
Aquifer media	3
Soil media	2
Topography	1
Impact of vadose zone	5
Hydraulic conductivity	3

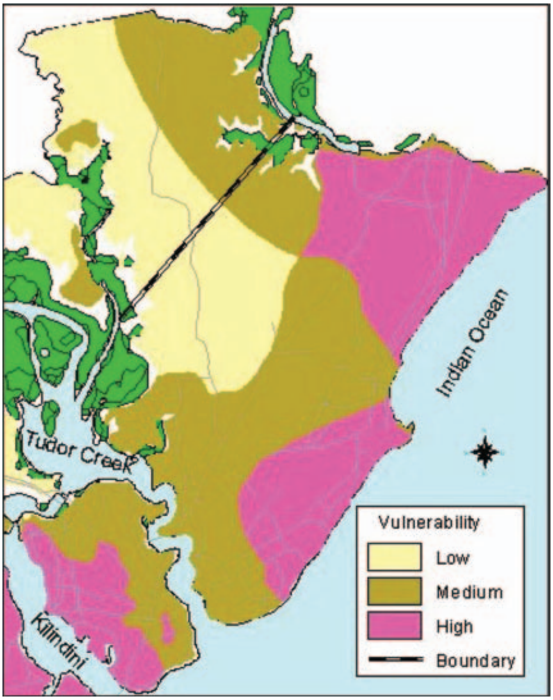

Figure 19.4. Vulnerability to pollution of water supply aquifer.

The resultant aquifer vulnerability map, primarily for Kisauni and the Mombasa Island, is presented in Figure 19.4.

The results of the vulnerability assessment of the water-supply aquifers indicate that the northern and south-eastern parts of Kisauni and the south-western part of the Mombasa Island are the most vulnerable to pollution.

4 ASSESSMENT OF GROUNDWATER QUALITY AND POLLUTION STATUS

4.1 *Methodology*

The study area covers Kisauni from the northern boundary of the Mombasa district along Mtwapa Creek to the Tudor Creek in the south (Fig. 19.5). The predominant geological formations in the

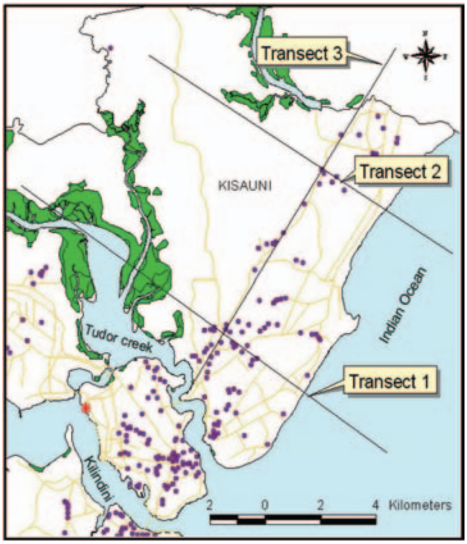

Figure 19.5. Transects of sampling points in Kisauni.

area comprise the pleistocene corals and limestone zone along the shorefront, the quaternary sands and cretaceous shales further west. Population settlements and housing developments are mostly concentrated in the coral and sand zones, which form the primary water-supply aquifer.

Most groundwater abstraction facilities in the area are wells, with some of the older facilities found in the coral zone being partially protected. Three transects were constructed arbitrarily, with Transects 1 and 2 approximately perpendicular to the shoreline and Transect 3 parallel to the shoreline (Fig. 19.6). Eventually, a total of 24 sampling points were identified, comprising 4 boreholes and 20 wells.

Water samples were collected on 16 and 29 June, 13 July 2004 and 9–10 and 17 November 2004. The samples were stored in a coolbox with ice before analysis within 24 hours of sampling. Samples were analysed for physical and chemical parameters, namely pH, electrical conductivity (EC), salinity, total dissolved solids (TDS), total alkalinity, sodium (Na^+), potassium (K^+), magnesium (Mg^{2+}), chloride (Cl^-), total hardness, ammonia, nitrates and phosphates. Using the colorimetric methods described by Parsons *et al.* (1984), the samples were analysed for Nitrite + Nitrate $\{(NO_2^- + NO_3^-)\text{-N}\}$ and orthophosphate (PO_4^{3-}-P). All chemicals used in these analyses were of analytical grade and glassware acid-washed.

As indicators of microbial contamination of water, faecal coliforms and *E. coli* were enumerated in samples by the multiple tube method of the most probable number (MPN). The 5-tube 3-dilution technique was used for water samples (FAO, 1979; UNEP/WHO/IAEA, 1985). MacConkey broth was used to enumerate total coliforms at 37°C incubation. Tubes found positive for total coliforms were used for the inoculation of fresh tubes of MacConkey broth and incubated at 44 – 45°C for faecal coliform estimation. *E. coli* was biochemically determined by indole production using the Kovacs reagent.

4.2 *Physical and chemical parameters*

Presented in Figure 19.6 are the variations of EC, salinity, TDS, total alkalinity and pH in groundwater along the three transects. The results generally show an increase in EC and corresponding salinity and TDS along Transects 1 and 2 as the sampling points approach the sea. Thus, the highest value

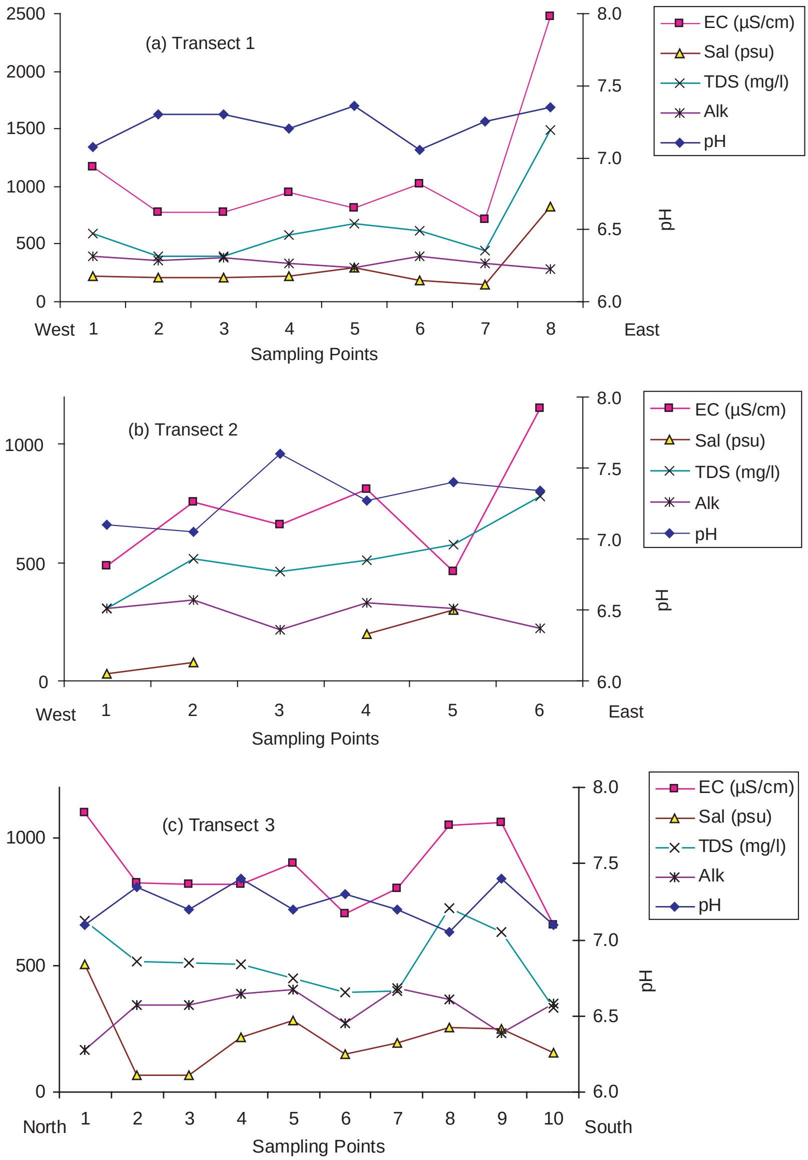

Figure 19.6. Variations of physiochemical parameters in groundwater in Kisauni, Mombasa in June 2004 along (a) Transect 1 (b) Transect 2 (c) Transect 3.

of EC was indicated in samples from the eastern-most sampling point along Transect 1 in the coral and limestone geological zone. Along Transect 3 an elevation of EC, salinity and TDS was indicated in sampling points approaching the Tudor Creek in the southern and Mtwapa Creek in the northern boundary of the study area.

Variations of the major ions Na, K, Mg, Ca and chloride and total hardness in groundwater were investigated. Chloride content showed an increase towards the sea as indicated in Transects 1 and 2. This is reflected by the general increase of Na ions as sampling points approach the sea. Along Transect 1, relatively high concentrations of Mg and Ca ions (compared to Na) result in raised levels of water hardness. Along Transect 3, a rise in chloride ions is indicated in the northern-most sampling point (towards Mtwapa Creek). A peak in hardness is indicated at Point 8, which is in the vicinity of Point 2 along Transect 1.

The concentrations of ammonia and inorganic nitrates/nitrites and phosphates in groundwater in Kisauni during the rainy season in June/July and the short rains in November are presented in Table 19.7. Whereas phosphate levels were moderate, ammonia and nitrate/nitrite concentrations were elevated in groundwater from most of the sampling points.

Relatively higher concentrations of nitrate/nitrite were recorded in June/July (range 2.1 to 44.4 $mg\ l^{-1}$) than in November (range 0.4 to 19.6 $mg\ 1^{-1}$). The concentrations were significantly

Table 19.7. Concentrations of ammonia-N ($mg\,l^{-1}$), nitrate + nitrite-N ($mg\,l^{-1}$) and phosphate-P ($\mu g\,l^{-1}$) in groundwater in Kisauni.

					Ammonia		Nitrate + Nitrite		Phosphate	
ID No.	Name	B/W*	East	South	Jul	Nov	Jul	Nov	Jul	Nov
20.1	Coast Hauliers	W	39.688	4.050	0.8		13.0		97.6	
20.2	Coast Hauliers B4	W	39.692	4.047	1.0		9.8		2.8	
20.3	Swafaa Mosque	W	39.697	4.028	2.7	2.0	13.1	8.2	33.8	12.0
21	Umoja Residence	W	39.681	4.040	1.1		14.6		114.1	
26	Snake Valley	B	39.683	4.030	1.2	0.9	36.4	10.6	51.5	8.8
14	Voyager Hotel	B	39.714	4.033	7.6	0.5	44.4	3.0	100.9	75.4
72	Freretown-Nyamu	W	39.698	4.028	0.6	1.3	10.9	18.5	61.0	55.0
74	Freretown-Jared	W	39.691	4.024	1.4	0.0	16.4	14.5	112.9	227.1
75	Freretown-Mterere	W	39.691	4.025	0.4	3.7	18.6	9.1	26.5	
76	Kisimani stage	W	39.695	4.024	5.4	0.0	8.1	5.7	76.3	151.5
32	Islam Ali 1	B	39.684	4.020	0.5		43.3		41.2	
33	Islam Ali 2	B	39.684	4.019	0.4	4.9	21.8	19.6	80.5	34.1
34	Abdalla Adam	B	39.683	4.019	1.3	1.7	12.7	8.0	78.6	
36	Masjid Bidalla	W	39.681	4.020		0.4		14.8		
106	Mgongeni Mosque	W	39.693	4.020	8.1	0.3	3.4	2.9	47.9	35.8
109	Mwandoni Katisha	B	39.692	4.017	0.7	2.6	19.4	5.1	33.6	4.5
122	Masjid Hussein	W	39.698	4.002	2.4	2.9	7.4	2.4	155.2	22.0
100	Masjid Noor	W	39.704	3.993	0.1	0.2	11.9	4.5	90.2	67.4
98	Utange R.C.	W	39.713	3.981	0.1	0.1	9.4	0.9	31.2	109.7
91	Utange-Anwaralli	W	39.719	3.971	2.1	0.6	4.1	0.4	56.8	7.3
90	Shimo Annex	B	39.731	3.961	0.1	3.1	19.9	13.4	21.7	
90.1	Masjid Dar al Kam	W	39.726	3.979	1.2		3.0		10.7	
90.2	Utange Pendua Viungani	W	39.725	3.976	2.1		4.7		16.7	
90.7	Masjid Radhaa	W	39.727	3.980	1.0		2.3		90.4	
97	Utange Pri Sch	W	39.718	3.976	0.0	0.4	2.5	0.8	28.8	25.0
94	Utange Pendua 1	W	39.724	3.973	0.7	1.9	6.6	4.6	39.0	31.5
95	Utange Pendua 2	W	39.722	3.972	0.5	0.0	3.1	0.9	276.7	166.5
93	Utange-Maingi	W	39.720	3.972	0.6	0.6	2.1	1.2	86.6	27.2
				Mean	1.6	1.3	13.4	7.1	69.0	62.4
				Max	8.1	4.9	44.4	19.6	276.7	227.1
				Min	0.0	0.0	2.1	0.4	2.8	4.5
				Std	2.1	1.4	11.7	6.0	55.7	64.7

* Borehole (B) or well (W)2.

different (t = 2.43, $p = 0.05$, df = 41). The results indicated relatively higher nitrate/nitrite concentrations occurring in the southern parts of Kisauni towards the Tudor Creek and along the Indian Ocean. It is instructive that some of the nitrate/nitrite hotspots were located in the medium vulnerability areas towards the Tudor Creek. This indicates that there was/were other factor(s) that influenced the contamination levels and this was attributed to proximate sources of the contaminant in the form of housing settlements. The area towards the Tudor Creek is occupied by high-density housing settlements, mostly unplanned, where the majority of the inhabitants use pit latrines for sewage management and disposal.

Towards the Indian Ocean shores, tourist beach hotels and low-density housing estates dominate, which mostly use septic tanks and soakage pits for sewage management. On the other hand, the northern parts with relatively low nitrate/nitrite concentrations have less dense housing settlements. However, a hot spot exists adjacent to the Mtwapa Creek in the north, representing the Shimo la Tewa Prison. The distribution of nutrients is approximately reflected by the groundwater flow model (Fig. 19.6). Thus, the contamination tends to move and concentrate towards the Tudor and Mtwapa Creeks and the Indian Ocean. The results are comparable to findings by Mwashote *et al*. (1996), who reported nitrate/nitrite concentrations in groundwater from one borehole and ten wells in the Kisauni ranging from 1.8 to 37.9 mg l^{-1}. In the present study, the nitrate concentration levels encountered in about 50% and 70% of the water samples tested in June/July and November, respectively, were within the WHO recommended potability limit of 10 mg l^{-1} NO_3^{-1}/NO_2^{-1}-N (Lawrence *et al*., 2001).

5 MICROBIAL CONTAMINATION

An indication of the contamination of groundwater with potentially harmful microbial organisms is given in Figure 19.7. Out of thirteen facilities sampled (five boreholes and seven wells) only two wells and one borehole produced water of acceptable potable quality in June 2004. The national (Kenya Bureau of Standards) and WHO drinking water quality standards are faecal coliform counts = nil and *E. coli* counts = nil. Analysis of the water quality in July gave an indication of an improved situation. Thus all three wells and two boreholes sampled produced water of acceptable potability. It is noted that June was a relatively wet period, whereas July was essentially dry. Thus, the dry conditions in July probably lowered the extent of contamination of the groundwater through recharge. In addition, it was noted that in some wells, chlorine balls had been suspended

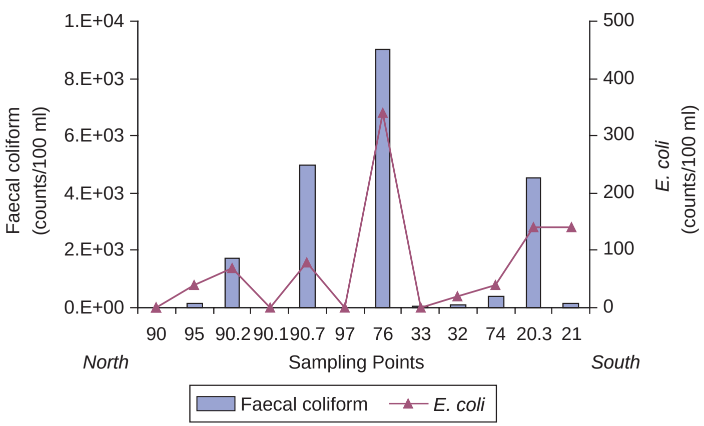

Figure 19.7. Microbial contamination in Kisauni groundwater – June 2004.

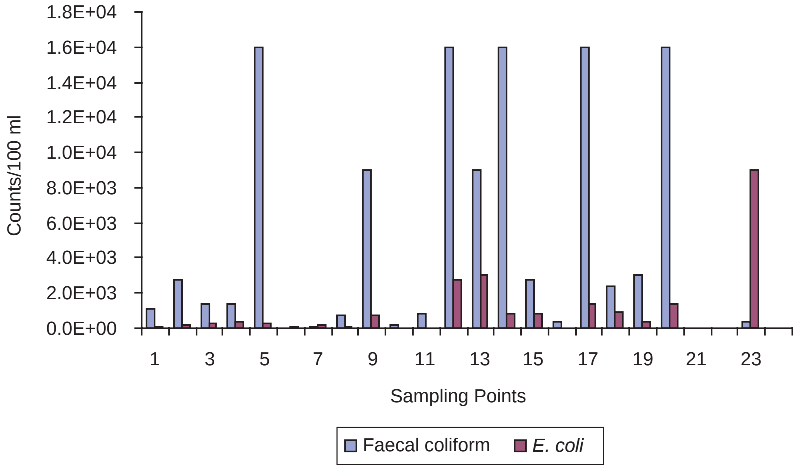

Figure 19.8. Microbial contamination in Kisauni groundwater – November 2004.

into the water to effect disinfection, which could have contributed to the drastic reduction in potentially harmful microorganisms. The water quality situation was worse in November, as all 25 boreholes and wells sampled were contaminated with unacceptable levels of faecal coliform and *E. coli* (Fig. 19.8). The presence of *E. coli* indicated that the primary source of contamination is human waste as a result of on-site disposal of domestic sewage. These findings are comparable to data obtained from the Government of Kenya Ministry of Water Resources Development and Management (MWRDM) including measurements carried out in July 2003 in the present study, which indicated that most of the wells and boreholes examined produced poor quality water with levels of microbial contamination exceeding the acceptable standards. Thus only one out of sixteen wells sampled gave water of acceptable quality, and none of the five boreholes samples met the standards. In comparison, Mwashote *et al*. (1996) and Mwaguni (2002) found 8% and less than 10%, respectively, of the groundwater facilities examined produced water of acceptable standard.

6 STRATEGY FOR GROUNDWATER PROTECTION IN KISAUNI

It is evident that groundwater in Kisauni is contaminated and the water quality is expected to deteriorate further in view of the rising demands for fresh water with increasing urbanisation. The primary source of groundwater pollution in the area is on-site waste management and disposal practices. During precipitation, contaminants are leached from waste matter deposited on the ground surface or sub-surface, such as uncollected municipal refuse and uncontrolled dumpsites. Direct recharge also occurs from soakage pits and wet pit latrines. It is realised that the strategy to effectively control groundwater contamination in Kisauni has to address the pollution sources, in this case on-site waste management.

Thus, there is a need for improved pit latrines properly constructed to minimise leakage of faecal matter into groundwater. This would entail the construction of protected pits that do not reach the water table, unlike the case in Kisauni. Soakage pits should be designed taking into consideration the depth of the water table so that maximum attenuation of contaminants as the wastewater sinks into the aquifer is attained. Uncontrolled disposal of sewage in wetlands should be avoided and instead septic tanks be utilised. Wastewater may be disposed of in a controlled wetland. Alternatively proper sewage treatment facilities (e.g. oxidation ponds or lagoons) would be required to minimise groundwater contamination. There is a need for appropriate regulations to guide the construction of waste management and disposal facilities and the authority to enforce compliance.

The effectiveness of measures to control sources of pollution can be enhanced by raising awareness and educating the community on the vulnerability of groundwater due to anthropogenic activities and the need to protect this valuable resource. This requires the generation of pertinent information on the state of the aquifer and regular monitoring of the groundwater to ascertain its pollution status.

Presently Kenya has in place a comprehensive policy framework and the necessary legislation and regulations guiding the management of water resources (GOK, 2002a, b). The law includes specific regulations on the exploitation of groundwater that are hardly enforced. Most water supply boreholes and wells, for example, were sunk without prerequisite permits and hence supervision by the water authority. There is a need to link the exploitation and management of groundwater resources with sanitation.

7 CONCLUSION

The output of the DRASTIC model indicates that the water supply aquifer in the northern and south-eastern parts of Kisauni and the south-western part of the Mombasa Island are the most vulnerable to pollution. The groundwater flow model gives an indication of the most probable direction of flow of contamination, which is useful for groundwater protection strategies.

The study has provided information on the general water quality in Kisauni with reference to physico-chemical characteristics. It is generally the case that water obtained from abstraction facilities located in the limestone geological zone is brackish and unsuitable for drinking. Within the sand geological zone, on the other hand, groundwater of acceptable potable standard is obtainable. The study does not reveal sufficient evidence of saline water intrusion into the aquifer. It is, however, realised that groundwater in particularly the high-population Kisauni areas, has raised concentrations of nitrates, which is an indication of contamination from on-site waste disposal systems, dominated by pit latrines and septic tank-soak pit systems as the mode of sewage disposal. Other sources of groundwater contamination in the area are uncollected municipal refuse. The nitrate concentrations encountered in 50–70% of the water samples analysed were, however, within the WHO recommended $10\,mg\,l^{-1}$ NO_3^{-1}/NO_2^{-1}-N limit for potable water.

The Kisauni area is indicated as experiencing a high degree of groundwater contamination by microbial contaminants, especially in the high-density housing settlements. This is primarily attributed to the sewage disposal method dominated by pit latrines and septic tank/soak pit systems. The contamination levels are more severe during the rainy season when aquifer recharge is enhanced. The Mombasa City local authority in conjunction with the Ministry of Water and Irrigation have put in place measures to ensure the availability of contamination-free water to the inhabitants by providing chlorinating agents free of charge, especially during the wet season. This direct intervention by the concerned authorities helps to control outbreaks of water-borne diseases such as cholera and typhoid.

In view of the findings, a comprehensive strategy to control groundwater deterioration in Kisauni should include the adoption of measures to control pollution sources, mainly on-site sewage management facilities, involving the community in groundwater protection initiatives and effective implementation of existing regulatory provisions.

ACKNOWLEDGEMENT

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REFERENCES

Aller, L. 1987. DRASTIC: A standardized system for evaluating groundwater pollution potential using hydrogeological settings. EPA 600/2-87-035. US EPA, Ada, Oklohoma.

Aller, L., Bennett, T., Lehr, J.H. & Petty, R.J. 1985. DRASTIC: A standardized system for evaluating groundwater pollution potential using hydrogeological settings. EPA 600/2-85-018. US EPA, Ada, Oklohoma.

Chiang, W-H. & Kinzelbach, W. 1993. Processing Modflow. Simulation system for modelling groundwater flow and pollution.

ESRI 1996a. Using ArcView GIS. Environmental Systems Research Institute, Inc. 350p.

ESRI 1996b. Using the ArcView Spatial Analyst. Environmental Systems Research Institute, Inc. 147p.

Gatabaki-Kamau et al. 2000. Urban governance, partnership and poverty in Mombasa. Unpublished report.

GOK 1979. National population and housing census. Vol 1. Ministry of Planning and National Development, Central Bureau of Statistics.

GOK 1989. National population and housing census. Vol 1. Ministry of Planning and National Development, Central Bureau of Statistics.

GOK 1999. National population and housing census. Vol 1. Ministry of Planning and National Development, Central Bureau of Statistics.

GOK 1988. Fertilizer use recommendation project (Phase I), Final report Annex III, Description of the first priority sites in the various districts. Vol 29. Kilifi District. National Agricultural Laboratories and GTZ German Agricultural Team.

GOK 2002a. Country strategy on the integrated water resources management. Ministry of Environment and Natural Resources, Department of Water Development.

GOK 2002b. Water Act 2002. Ministry of Water and Irrigation.

Kitheka, J.U. 1996. Groundwater outflow dynamics and circulation at Diani and Nyali mesitidal beaches in Kenya. In: Anthropogenically induced changes in groundwater outflow and quality, and the functioning of Eastern African nearshore ecosystems. GROFLO Final Report Part 2. pp. 5–30.

Lawrence, A.R., Macdonald, D.M.J., Howard, A.G., Barrett, M.H., Pedley, S., Ahmed, K.M. & Nalubega, M. 2001. Guidelines for assessing the risk to groundwater from on-site sanitation. British Geological Survey. 97p.

Mwaguni, S.M. 2002. Public health problems in Mombasa District. A case study on sewage management. MSc. Thesis, University of Nairobi. 86p.

Mwashote, B.M., Mwangi, S.N. & Kazungu, J.M. 1996. Groundwater-associated anthropogenic influence on coastal lagoons: Diani and Nyali Beach, Kenya. In: Anthropogenically induced changes in groundwater outflow and quality, and the functioning of Eastern African nearshore ecosystems. GROFLO Final Report Part 2. pp. 31–45.

Njue, F.J., Ngecu, W.M. & Ocharo, M.O. 1994. Investigations of the freshwater-saltwater interface within Mombasa Island and implications to groundwater management. Unpublished report.

NWCPC (National Water Conservation & Pipeline Corporation) 2000. Water situation in Mombasa and coastal area. Unpublished report.

Parsons, T.R., Maita, Y. & Lalli, C.M. 1984. A manual of chemical and biological methods for seawater analysis. Pergamon Press. 173p.

Pollock, D.W. 1988. Semianalytical computation of path lines for finite difference models. *Ground Water* **26(**6): 743–750.

Pollock, D.W. 1989. MODPATH (version 1.x), Documentation of computer programs to compute and display pathlines using results from the U.S. Geological Survey modular three-dimensional finite-difference ground-water model. U.S. Geological Survey Open-file report 89-381.

UNEP/WHO/IAEA 1985. The Determination of Faecal Coliforms in Sea Water by the Multiple Test Tube (MPN) Method. Reference Methods for Marine Pollution Studies No. 22 (draft). 23p.

UNEP/FAO/PAP/CDA 2000. Progress in integrated coastal management for development of Kenya's coast. The case of Nyali-Bamburi-Shanzu area. East African Regional Seas. Technical Report Series No. 6.

20

A chemical study of surface and groundwater in the Lake Chilwa Basin, Malawi

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ABSTRACT: The impact on water quality related to the kind and quantity of agrochemicals used in the Lake Chilwa catchment area was determined. The major sources of water pollution in the Lake Chilwa Basin are agricultural and horticultural materials and residuals, use and disposal of metal commodities including electronic appliances, fossil fuel combustion, industrial and household waste disposal and military training at Zomba Barracks. The most widely used fertilisers in the catchment area include urea, calcium ammonium nitrate (CAN), 23:21:0+4 S, D-Compound, diammonium phosphate (DAP), S-mixture and sulphate of ammonia (SA). The total quantity of fertilisers for the seasons 1994/95–1998/98 was 46,000 metric tonnes. The estate sector in the Lake Chilwa Basin contributes significantly to the total tonnage of fertiliser with the D-Compound being the major source of plant nutrients. Water quality data and results on soil sediments from the catchment reveal that the lake and the groundwaters are sinks of the residues of the industrial and agricultural activities. The lake is fairly alkaline due to high levels of carbonates and bicarbonates. Soil sediments gave higher levels of phosphates, nitrates and metals than fresh water from rivers, land reservoirs and the lake itself. These are released into both the ground- and lake waters and thereby raise their values. The accumulation seems however to be high in the southern end of the lake at Swang'oma.

Keywords: Lake Chilwa Basin, groundwater, surface water, quality, Malawi.

1 INTRODUCTION

There are several processes occurring in the Lake Chilwa catchment which are of environmental concern, the major ones being siltation due to erosion and chemical loading from domestic and industrial discharges (Venema, 1991; DREA, 1994). This is exacerbated by intensive cultivation and almost complete disregard of the natural catchment by the smallholder farming communities and the estate sector, leading to high run-off.

Since the country attained independence in 1964, the rate of agricultural activities in the catchment area has increased (MOALD, 1994). Expansion of agricultural enterprises, including rice, tobacco and maize growing, has been due to the increased use of agrochemicals such as fertilisers and pesticides in the catchment, most of which have been washed into the lake due to increased soil erosion (Ambali and Maluwa, 1998). These have brought about eutrophication in the lake, so that during the 1995 recession there was a heavy phytoplankton bloom and continuous green scum on the water surface (Ambali, pers. comm.).

Lake Chilwa, the second largest in Malawi after Lake Malawi, is found in southern Malawi, 50 km east of the southern end of the Great Rift Valley, about 120 km from the tip of Lake Malawi and is 622 m above sea level (Lancaster, 1979). The lake is 40 km long from north to south and 30 km wide from east to west and is very shallow with depth varying from 3 to 5 m. Its catchment occupies an area of 2000 km^2 and about 70% of the water inflows come from the Shire Highlands including the

Zomba Mountain. The five main rivers are the Sombani, Phalombe, Likangala, Thondwe, and Domasi Rivers and the minor rivers include the Naizi and Songani. The rivers on the Mozambique side are the Mnembo, Mbungwe and Nchimadzi. The lake has suffered moderate recessions in the years 1900, 1923, 1931/33, 1943, 1949, 1953/55, and 1960/61, with severe ones in 1914/15, 1966/67 (Morgan, 1968) and more recently, 1997/98 (Ambali and Maluwa, 1998), when the lake dried completely. The lake is divided into three equal sections: open water, swamp and flood plains, which is surrounded by a dense *Typha domingensis* (bulrush or reedmace) swamp (Lancaster, 1979). Unlike Lakes Malawi and Malombe, Lake Chilwa has no surface outlets so that its waters are therefore salt-like (McLachlan *et al*., 1972).

The Lake Chilwa catchment has eight traditional authorities: Mkumbira, Kuntumanji and Mwambo in Zomba, Kawinga, Mlomba and Mposa in Machinga, Mkhumba and Nazombe in Phalombe district (NSO, 1998). The population of the catchment, including the Municipality of Zomba, increased by 13.3% over a ten-year period (1987–1998). The remarkable population growth has been accompanied by an increase in agricultural activities and higher exploitation of cultivable land. The increased urbanisation and industrial activities around the Zomba Municipality has also resulted in a higher quantity of discharges and diversification in the types of pollutants reaching river waters and finally Lake Chilwa.

The unprecedented population growth has exacerbated the siltation due to erosion and thus constitutes an important factor in the transfer of pollutants into the lake. The catchment has been subjected to intensive cultivation, leading to high run-offs. Studies have shown that soil erosion in the Phalombe, Likangala and Mulungizi river catchments are 7.8, 13.8 and 0.06 tonnes per hectare per year respectively (Chimphamba, 1993). These catchments suffer high nutrient losses of nitrogen, phosphates and potassium, which end in Lake Chilwa (Chimphamba, 1993). Thus, sources of environmental pollution include natural and anthropogenic, direct and indirect pathways that are classified into three groups: organic loads with high oxygen demand, metals and pesticides. The most important sources of pollutants in the Lake Chilwa catchment are therefore domestic and urban discharges, leaching of metals from garbage and solid waste dump, metals contained in pesticides, battery wastes, pigments/paints, biocides (pesticides, herbicides and preservations), fertilisers, dental and cosmetics, and combustion of leaded petrol in motor cars (DREA, 1994; Calamari and Naeve, 1994).

2 GROUNDWATER AND SURFACE WATERS IN MALAWI

Malawi has a rich resource of both ground- and surface water, which constitutes 20 per cent of the country's area (MOWSWD, 1994). The water systems include Lake Malawi (28,750 km^2 and volume of 7725 km^3), Africa's third largest fresh water lake, Lake Malombe (303 km^2), an inflation of River Shire and Lake Chilwa (683 km^2) and a network of perennial rivers and groundwater (Fig. 20.1).

Malawi has three major groundwater zones, which are distinguished by the major physiographic units:

(i) The Rift Valley zone, which includes the Lake Malawi Rift and the upper and the lower sections of the Shire Valley, and is separated by the Rift Escarpment zone of tectonic origin from the High Plateau.
(ii) The Lower Shire Valley.
(iii) The High Plateau zone weathered rock products and fractured rock.

Groundwater in Malawi is recharged by seasonal rainfall, especially in the Rift Valley and High Plateau zones and unconsolidated sediments and fractured crystalline rocks. Supplementary contribution comes from perennial water courses in dry periods. Lake Malawi lakeshore aquifers occur in sands and gravel in unconsolidated sediments; the deeper aquifers provide a better supply while the fractured quartzitic rocks give better yields than other types of crystalline rock. In the

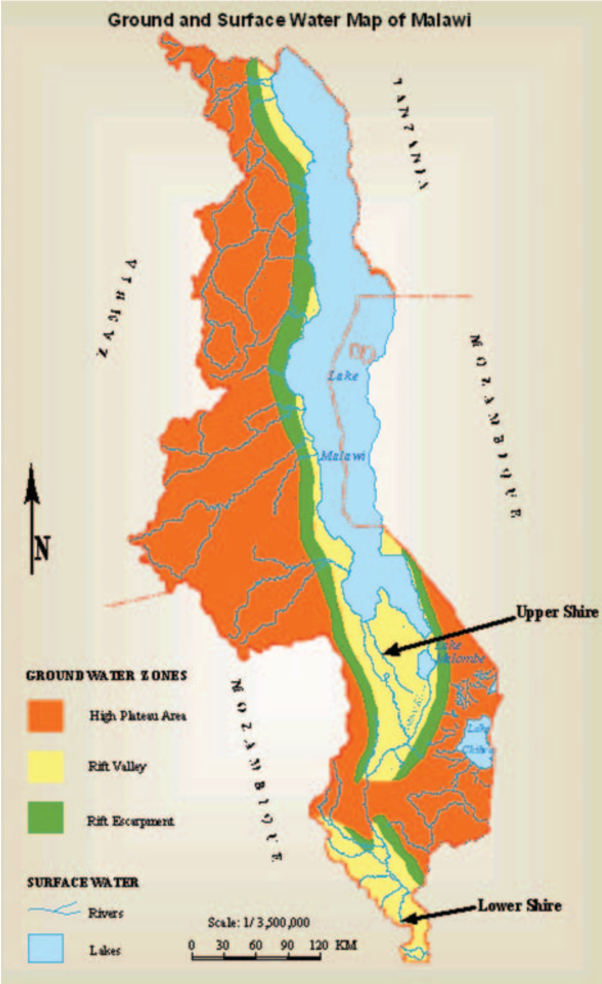

Figure 20.1. Distribution of ground- and surface water in Malawi.

High Plateau zone, which includes the Lake Chilwa plains, groundwater is generally under non-flowing, confined conditions.

Chemical analyses of borehole groundwater show that dissolved solids of bicarbonate composition predominate in all aquifer types (MOWSWD, 1994). While calcium and sodium bicarbonates are the major dissolved solids in the Rift Valley zone, sodium bicarbonate in the Lower Shire Valley, dissolved sodium and chlorine are prevalent along the west bank of the Lower Shire. The quality of groundwater is affected by surface water and therefore the problem is exacerbated during the rainy season.

In Malawi surface waters are generally unpotable or unavailable for considerable parts of the year. Consequently, aquifers provides the most reliable potable water supplies for the majority of rural communities and facilitates agricultural development of dry areas.

Replenishment of surface water resources in Malawi is dependent on the seasonal rainfall. Most of the rivers and lakes therefore display seasonal flow and level patterns and dry up during the months of July to October. This exacerbates the distribution and availability of the surface water resources to 50% of Malawi's surface area. To combat this problem, Malawi has storage dams for water supply and conservation purposes.

2.1 *Physico-chemical characteristics of Lake Chilwa and its environment*

Lake Chilwa is characterised by marked fluctuations in water level due to periodic recessions of the lake; the various degrees of evaporation. Dilution due to rain and river inflows affect the ionic content and physical properties of the lake water (McLachlan, 1979; Howard-Williams, 1979). Conductivity has been used successfully to determine the levels of dissolved salts; conductance of water depends on numbers and composition of ions (McLachlan, 1979; Howard-Williams, 1979; Ambali and Maluwa, 1998; Howard-Williams, 1973).

Earlier studies have revealed that conductivity is strongly correlated with sodium ions and the major ions in the lake are sodium, chloride and carbonates. The latter is responsible for the alkaline pH of the water (8–9). Large precipitation of chlorides in Lake Chilwa is due to the presence of three underground hot springs, which are the main sources of feldspars and nepheline syenites and volcanic intrusions, and alluvia, over which flow occurs (Lancaster, 1979). During normal years, for example in 1970, conductivity was low in February when the water level was maximum and highest in November/December in the same year. Ambali and Maluwa (1998) have also observed this trend, which reflects the effect of dilution and evaporation respectively during the two periods. Evaporation leads to concentration of salts making the water more salty (Morgan, 1968). These earlier studies (McLachlan, 1979; Howard-Williams, 1979; Ambali and Maluwa, 1998; Howard-Williams, 1973; Moss and Moss, 1969; Kalindekafe, 1996) have established that indeed sodium is the most important metal cation and that during recession its values increase (321.0–474 $mg\,l^{-1}$). This is also the case for conductivity (1524.5–1970 $\mu mho\,cm^{-1}$), alkalinity (450.5–482 $mg\,l^{-1}$) and chloride ions (298–461.0 $mg\,l^{-1}$). The sulphate levels (15.0–326.0 $mg\,l^{-1}$) reflect the accumulation of this constituent following the drying of the lake. The oxygen levels (6.7–7.4 $mg\,l^{-1}$) indicate that, over the years, there have been no significant changes and fish survival has not been adversely affected.

An assessment of water contamination in the Lake Chilwa Basin and factors responsible for the degradation have not been established. The objective of the study was to investigate the chemical pollution of Lake Chilwa by agrochemicals and other wastes. The specific objectives were to (i) identify sources of industrial and domestic wastes discharged in the Lake Chilwa Basin, and (ii) determine the level of loading, accumulation (concentration) of agrochemicals in the water and bottom sediment, at various sites, in the Lake Chilwa Basin. This information was required to mitigate environmental degradation and the development of a plan for sustainable utilisation by the local communities of the natural resources found in the Lake Chilwa Basin.

3 MATERIAL AND METHODS

3.1 *Sampling sites and collection*

Water and sediment samples were collected from seven rivers: Phalombe, Sombani, Namadzi, Thondwe, Likangala, Domasi and Sumulu and two rice schemes: Domasi (Mpheta) and Sumulu as well as at the Makoka Research Station, Thondwe. Water was also collected from three boreholes: Kachulu, Makhanga at Swang'oma and Tchuka on Chisi Island. Duplicate water samples were collected; one of which was acidified with concentrated nitric acid for metal determination. Sampling was done from February–March, 1999.

3.2 *Sampling preparation*

Soil sediments were air dried in the laboratory until the moisture content was about 10%. Soils were ground and sieved through a 2.0 mm sieve.

3.3 *Water analyses*

Water was analysed for various parameters using standard procedures outlined elsewhere (AOAC, 1990). The following variables were determined: turbidity, temperature, total alkalinity, salinity, conductivity and pH. Conductivity, pH and temperature of water were determined in the field. The conductivity of soil sediments in 0.1 M KCl was obtained in the laboratory. The metal content of water samples was obtained according to literature methods (AOAC, 1990).

3.4 *Soil analysis*

Phosphates, nitrates and metal elements in soils were determined according to known methods (Chilimba, 1996). To ground soil (2.5 g) was added Mehlich 3 extractant (25 ml). The Mehlich 3 extractant was prepared by adding acetic acid (23 ml), conc. HNO_3 (1,63 ml) and NH_4F-EDTA (16 ml) to 0.25 M NH_4NO_3 (1500 ml) and diluting to mark in a 2.0 litre volumetric flask. The NH_4F-EDTA mixture was made by dissolving ammonium fluoride (13.9 g) and EDTA (7.4 g) in water and diluting to mark in a volumetric flask (100 ml). The soil extract composite was shaken on a mechanical shaker for 25 min, filtered and the filtrate collected for nitrate, phosphate and metal content determination. Metals analysed included calcium, magnesium, potassium and sodium (Saka and Ambali, 1999).

3.5 *Data analysis*

Sample mean and standard deviation for all the variables were calculated and compared with literature maximum allowed levels of toxic elements in water.

4 RESULTS AND DISCUSSION

4.1 *Sources of contaminants*

Interviews and a desk study revealed that the various sources of contaminants in the Lake Chilwa Basin are due to human and industrial activities (Alloway, 1995; Calamari and Naeve, 1994). The major sources of pollution in the Lake Chilwa Basin are therefore mainly,

(i) agricultural and horticultural materials
(ii) use and disposal of metal commodities
(iii) fossil fuel combustion
(iv) use and disposal of electronic appliances
(v) waste disposal
(vi) military training.

Agricultural activities seem to constitute the most important sources of metals accumulating in water and soils (DREA, 1994). Important sources of metals include fertilisers, sewage sludge, manures, pesticides, wood preservatives and corrosion of metal structures such as galvanised roofs or wire fences.

Combustion of petrol that contains lead additive constitutes a significant source of this heavy metal. Lead is emitted into the environment from motor vehicle exhausts. The lead aerosols concentrate in motorways, rural and urban areas. Accumulation in the Lake Chilwa waters is due to run-off, which transports a large amount of soil. Erosion is therefore an important route for metal pollution. Low and Kumpalume (personal communication) have indicated that some accumulation of lead occurs along the roadsides in the Municipality of Zomba and River Likangala; the values in the Lake Chilwa waters are <5.0 parts per million.

The waste dumps in the Municipality of Zomba and around households contain metals leached into the basin through run-off and during floods. Fabrication of metal products for machines and vehicles and their disposal or recycling of scrap metal also leads to environmental pollution by metals. Landfills of solid wastes in the Zomba Municipality facilitate contamination of the environment by metals such as cadmium, copper, lead, tin and zinc. Burning these dumps or landfills contributes to significant metal accumulation in soils and finally in water bodies (Zembere, Maruwo and Ngwangwa, 1999; Chinyama and Maldhopa, 1999).

The Municipality of Zomba has two army bases, Cobbe Barracks and the Airwing. The use of cartridges and shells during training releases lead, copper and zinc into the soils, which are finally leached into the lake through run-off (Alloway, 1995). Other important sources of contaminants in the Lake Chilwa Basin include batteries, pigments and paints, soaps, incineration of plastics, printing and graphics, and medicines. Biotic sources of contamination include the decomposition of dead plants and animals, which release metals and anions into the lake water.

4.2 *Types and quantities of fertilisers*

In the Lake Chilwa attachment, fertilisers (tonnes) used by both smallholder farmers and estates are 23:21:0+4 S (4847), diammonium phosphate (1721), gypsum (1.35), D-Compound (3239), S-Mixture (76.8), Sulphate of Ammonia (1515.6), Urea (7.798), calcium ammonium sulphate (CAN) (26,254), Super D (397.1), Double super phosphate (0.1) and Solubor (0.002). These fertilisers are largely sources of nitrogen and phosphorus. Estates in the catchment area use mostly D-compound and SA while the smallholder-farming community largely applies CAN to their crops. The total amount of fertilisers used in the area from 1994/95 to the present for those institutions involved is 45,851 tonnes. The total tonnage of fertilisers (tonnes) were Zomba RDP (38,530), Kawinga RDP (6215), Kachulu Estate (170), Mgodi Estate (901) and Makoka Research Station (35). The four major fertilisers are thus CAN, urea, 23:21:0+4 S and D-Compound. The most important fertilisers in the catchment are phosphates and nitrogen based.

4.3 *Water quality*

The physico-chemical characteristics of water from various sites in the Lake Chilwa basin are provided in Figures 20.2 and 20.3. From the results it seems that the turbidity of the lake water is much higher than that of fresh water in rivers, rice schemes, boreholes, dams and the lake shoreline. This is because the lake is a sink without a surface outlet for all sediments from the catchment rivers and run-offs. Similar trends have been obtained for inland reservoirs in Malawi (Mumba, Banda and Kaunda, 1999). The continuous mixing of the water and high rate of eutrophication is also responsible for this trend (Howard-Williams, 1973; Ambali and Maluwa, 1998). The composition of anions seems to show that the predominant ions are carbonates, chlorides and sulphates. This is consistent with earlier findings (Ambali and Maluwa, 1998). Phosphates and nitrates are generally much lower and less than $10\,\mathrm{mg\,l^{-1}}$. A comparison of these anions for the fresh river waters, wells, boreholes and the lake waters reveals that the lake is a sink without a surface outlet. The high conductivity of water samples from the lake and ground sources (wells and boreholes) is due to this feature (Mumba, Banda and Kaunda, 1999). When compared with WHO guidelines for potable water; the turbidity, total alkalinity, phosphate, nitrates, sulphates and chlorides are 5 NTU, 500, 45, 10, 400 and $250\,\mathrm{mg\,l^{-1}}$ respectively. (WHO, 1984), the catchment waters give higher turbidity. The alkalinity values for water from wells, boreholes dams, shorelines and the lake are higher than the recommended limit; thus the water is more basic. Although, the nitrate and sulphate values are below the maximum permitted, the chloride levels from the lake and some boreholes are much higher than the maximum allowed. Thus, the Lake Chilwa and groundwaters contain higher chlorides, carbonates and are more turbid than the maximum WHO requirement. The waters are therefore not suitable for human consumption.

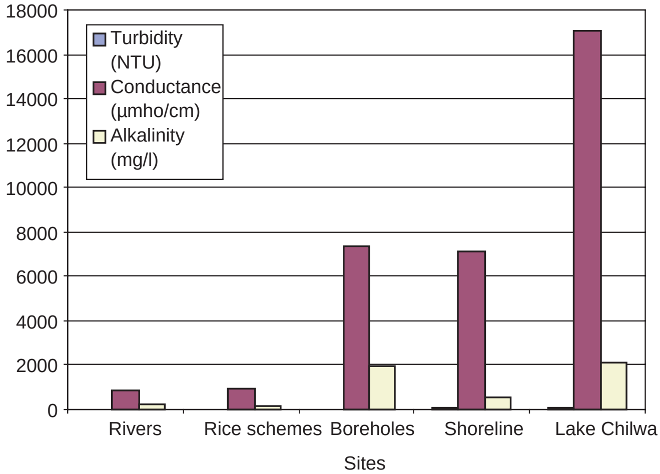

Figure 20.2. Turbidity, conductance and alkalinity of water from various sites.

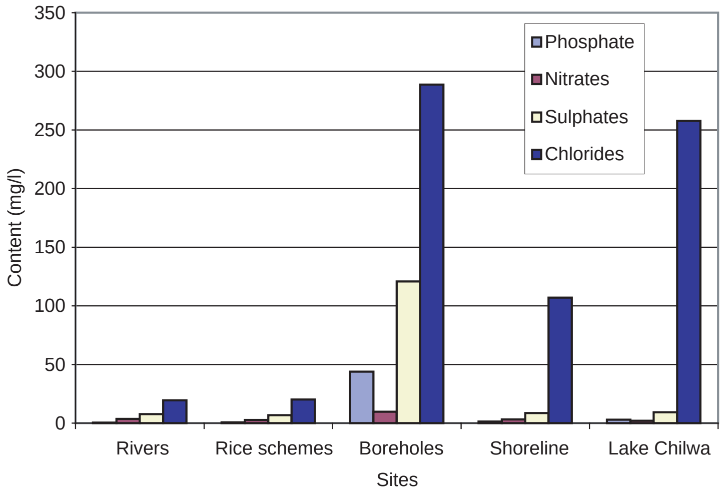

Figure 20.3. Variation of anion concentrations at various sites around the Lake Chilwa catchment.

The major cations in waters are sodium and potassium; the former is predominant. Calcium and magnesium levels are generally $<10\,mg\,l^{-1}$. The conductivity is thus due to carbonates, chlorides, sulphates, sodium and potassium ions (Howard-Williams, 1973). The values obtained in this study are lower than those obtained by Ambali and Maluwa (1998); this is because samples used in this study were collected after the lake had significantly refilled following a recession in 1995, which resulted in substantial dilution. A continuous monitoring of the composition over a number of seasons and associated water depth is therefore necessary to ascertain the level of accumulation. The water depth of the lake depends on the quantities of sediments from rivers and run-offs in the catchment area.

Table 20.1. Physico-chemical characteristics of soil sediments from the Lake Chilwa Basin (mg kg^{-1}) (mean ± sd, n = 3; dl = below detection limit; blank undetermined).

Catchment site	pH (0.1 M KCl)	NO_3^-	PO_4^{3-}	Ca	Mg	K
Sumulu river	4.38 ± 0.03	10.84 ± 1.45	6.75 ± 0.79	1643.2 ± 185.2	368.2 ± 45.8	6.0 ± 1.4
Phalombe river	5.92 ± 0.08	4.89 ± 0.00	8.36 ± 0.00	1149.5 ± 149.1	<dl	42.0 ± 4.2
Sombani river		9.72 ± 0.18	<dl	1648.1 ± 104.7	812.1 ± 52.0	12.0 ± 0.0
Likangala river		16.23 ± 0.87	<dl	729.8 ± 141.2	124.2 ± 28.5	7.5 ± 0.7
Sumulu rice scheme canal		8.97 ± 0.83	<dl	1268.1 ± 85.6	260.3 ± 51.5	10.0 ± 1.4
Sumulu rice scheme paddy		20.91 ± 1.90	<dl	1643.2 ± 185.2	368.2 ± 45.8	9.0 ± 1.4
Mpheta rice scheme	5.12 ± 0.21	9.46 ± 0.65	<dl	2501.5 ± 419.5	299.2 ± 6.9	45.0 ± 4.2
Mgodi estate dam		9.87 ± 1.15	<dl	2324.1 ± 173.4	842.3 ± 5.8	11.5 ± 0.7
Mgodi estate – top soil		21.90 ± 0.54	15.38 ± 0.45	2944.7 ± 28.5	1210.8 ± 10.5	17.5 ± 2.1
Mgodi estate – sub soil		9.66 ± 0.31	14.75 ± 1.74	2577.1 ± 15.9	859.9 ± 4.0	9.0 ± 1.4
Swang'oma fishing site	6.55 ± 0.13	6.47 ± 0.30	91.34 ± 5.23	3859.1 ± 1184.2	968.7 ± 52.4	66.0 ± 3.4
Swang'oma Lake Chilwa 500 m from harbour	7.95 ± 0.03	5.61 ± 0.72	<dl	57,749.7 ± 2076.3	1910.2 ± 249.8	37.0 ± 2.7
Tchuka swamp	7.65 ± 0.01	8.87 ± 0.26	20.71 ± 1.65	3073.7 ± 186.6	1232.7 ± 72.5	7.0 ± 1.9
Tchuka well	7.70 ± 0.01	9.27 ± 1.47	40.83 ± 1.10	2770.2 ± 209.6	1004.6 ± 175.4	36.0 ± 0.0
Khumali harbour	7.48 ± 0.04	10.56 ± 1.36	56.50 ± 3.34	2228.6 ± 161.3	657.1 ± 23.3	17.0 ± 2.8
Kachulu harbour	7.33 ± 0.08	3.39 ± 0.31	41.89 ± 3.93	1355.7 ± 120.5		51.2 ± 4.2

The chemical data indicates that calcium and magnesium levels are much lower that the WHO maximum recommended values for potable water. The Lake Chilwa waters generally contain much higher sodium levels. The values seem however to be lower in swamps, probably due to filtration and accompanied sedimentation.

4.4 *Accumulation in soil sediments*

The concentration of nitrates, phosphates, calcium, magnesium and potassium in soil sediments are given in Table 20.1. The data show that soils contain higher levels of these elements than water (Table 20.1); this is consistent with findings reported for the River Shire (Saka and Ambali, 1999). The levels of these elements in the soil sediments seem to constitute an indicator of accumulation in the lake basin. The sources of nitrates and phosphates are largely from fertilisers used in the catchment. High phosphate levels ($>50\,mg\,l^{-1}$) are found around the Chisi Island and Swang'oma. The higher values at the latter site are probably due to accumulation of phosphates at the southern end of the lake. This trend is also obtained with calcium and magnesium. Further, the southern end is a complex of alkaline silicate rocks, which are rich in sodium and calcium carbonate and agglomeration (Lancaster, 1979).

The sediments from Swang'oma contain much higher calcium and magnesium, but lower potassium levels than the Lower Shire sediments. The soil sediments from Chiromo Bridge contain 6149, 1127 and 223 $mg\,kg^{-1}$ dry weight of Ca, Mg and K respectively (Saka and Ambali, 1999). Thus, the greater accumulation of the latter two elements occurs at the southern end of Lake Chilwa. This is also the case for phosphates; nitrates are lower probably because they are deposited over the catchment areas of the lake.

5 CONCLUSIONS

This study has revealed that groundwaters found in wells and boreholes contain higher nutrient loads than fresh water in rivers and Lake Chilwa. The lake water affords higher sodium and chloride levels and thus accounts for the saltiness of the lake. The minerals and anions accumulate in the soil sediments. The present findings form a basis for a planned multi-disciplinary study that aims to:

(i) determine the seasonal variation of the major elements in groundwater in the basin
(ii) assessing a water budget of the lake (inflows, evaporation, leaks)
(iii) analysis of sediment budget (erosion/sedimentation) and a chemical budget of the lake, including leaks and losses of chemical elements by sedimentation (or even remobilisation of the sediment content).

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REFERENCES

Alloway, B.J. 1995. The origin of heavy metals in soils. p. 38–57. In Heavy Metals in Soils, 2nd edn. London: Blackie Academic & Professional, Chapman & Hall, pp. 362 + Appendix.

Ambali, A.J.D. and Maluwa, A.O. 1998. Determination of Recolinization Potential of the *Oreochromis shiranus chilwae* Fishery after Recovery from Severe Recession of 1995. Final Report submitted to Environmental capacity Enhancement Project (ECEP), 40 pp.

AOAC. 1990. *Official Methods of Analysis*. Association of Official Analytical Chemists, Inc., 15th Edition, Arlington, United States of America.

Calamari, D. and Naeve, H. (Eds). 1994. Review of Pollution in the African Aquatic Environment. CIFA Technical paper 25. Rome: Food and Agriculture Organization, 118 pp.

Chinyama, M.P.M. and Madhlopa, A. 1999. An assessment of municipal waste management in the city of Mzuzu, Malawi. Malw. J. Sci. and Techno. 5: 40–49.

Chilimba, A.D.C. 1996. Effectiveness of Universal Extractant in Detecting Nutrient Deficiencies in Selected Malawi Soils. Mac Thesis, University of Malawi, 121 pp.

Chimphamba, J.B. 1993. Effect of Different Land Use Management Practices: Lessons from Bumble Soil Erosion Research Project. Land Husbandry Branch Conference, Mzuzu, 25–39 June, 1993, 25 pp.

DREA. 1994. National Environmental Action Plan. Vol. 1: The Action Plan, Vol. 2: Initial Districts Environmental Action Suggestions, June 1994.

Howard-Williams, C. 1972. Limnological studies in African swamp: seasonal and spatial changes in the swamps of lake Chilwa, Malawi, and Arch. Hydrobio. 70(3): 379–391.

Howard-Williams, C. 1973. Vegetation and environment in the marginal areas of a tropical African lake (L. Chilwa, Malawi). PhD. Thesis, University of London.

Howard-Williams, C. 1979. The aquatic environment: ii. Chemical and physical characteristics of Lake Chilwa, p. 79–90. In Lake Chilwa: Studies in a Tropical Ecosystem. Monographiae Biologicae. Kalk, M., McLachlan, A.J. and Howard-Williams, C. Eds. 35, 462 pp.

Kalindekafe, M.P. 1996. Ecological status of Lake Chilwa environments during part of the filling phase (April 1996) after drying. p. 17–37. In The Lake Chilwa Environment: A Report of the 1996 Ramsar Site Study. Zegeren, K. van and Munyenyembe, M.P., Eds. 162 pp.

Lancaster, N. 1979. The environmental setting: the changes in the lake level, p. 41–58. In Lake Chilwa: Studies in a Tropical Ecosystem. Monographiae Biologicae. Kalk, M., McLachlan, A.J. and Howard-Williams, C. Eds. 35, 462 pp.

McLachlan, A.J. 1979. The aquatic environment: i. Chemical and physical characteristics of Lake Chilwa, p. 59–78. In Lake Chilwa: Studies in a Tropical Ecosystem. Monographiae Biologicae. Kalk, M., McLachlan, A.J. and Howard-Williams, C. Eds. 35, 462 pp.

McLachlan, A.J., Morgan, P.R., Howard-Williams, C., McLachlan, S.M. and Brown, D. 1972. Aspects of the recovery of a saline African lake following a dry period. Arc. Hydrobiol. 70(3): 325–340.

MOALD. 1994. Guide to Agricultural Production in Malawi. Lilongwe: Agricultural Communication Branch, Ministry of Agriculture and Livestock Development, 227 pp.

Moss, B. and Moss, J. 1969. Aspects of the limnology of an endorheic African lake (Lake Chilwa, Malawi). Ecology, 50(1): 109–118.

MOWSWD. 1994. Water Resources Management Policy and Strategies. Ministry of Works, Supplies and Water Development, May 1994, 47 pp.

Morgan, A. 1968. A Physicochemical study of lake Chilwa. MSc Thesis, University of Malawi.

Mumba, P.P., Banda, J.W. and Kaunda, E. 1999. Chemical pollution in selected reservoirs and rivers in Lilongwe district, Malawi. Malw. J. Sci. and Techn. 5: 74–86.

NSO. 1998. Population and Housing Census. Report of Preliminary Results. National Statistics Office, Zomba, Malawi, 15 pp + Appendix.

Saka, J.D.K and Ambali, A.J.D. 1999. A biochemical study of pollution in the middle and lower Shire river, Malawi. Malw. J. Sci. and Techn. 5: 63–73.

Venema, J.H.1991. Land Resources Appraisal of Liwonde Agricultural Development Division. Malawi Government Ministry of Agriculture, Land Husbandry Branch, United Nations Development Programme, Food and Agriculture Organisation of the United Nations, 258 pp.

WHO (World Health Organization). 1984. Guidelines for Drinking Water Quality. *Vol.1. Recommendations*. Geneva, World Health Organization.

Zembere, S.N., Maruwo, W.B. and Ngwangwa, M. 1999. Industrial solid wastes and methods of disposal by some manufacturing companies. Malw. J. Sci. and Techn. 5: 56–62.

21

Groundwater quality assessments in the John Laing and Misisi areas of Lusaka

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ABSTRACT: Groundwater is becoming a more accessible and comparatively cheap source of water for drinking, agriculture and industry in Zambia than surface water. However, in Lusaka, the country's capital city, increased rates of urbanisation have caused large numbers of people who cannot readily obtain water supply services by self-provision to exploit any other available sources of groundwater supply, thereby exerting enormous pressure on the Lusaka aquifer through construction of private boreholes or hand-dug wells. Consequently, contamination in the city aquifer appears to be increasing and waterborne diseases have increased to endemic levels as well.

Keywords: health, Lusaka, pollution, urbanisation, vulnerability

1 INTRODUCTION

Groundwater use in Lusaka has been increasing rapidly in recent years. Groundwater is relied on for drinking water and as an essential prerequisite for many socio-economic activities, development and growth. In Lusaka alternative sources of water supply are becoming increasingly scarce. The major advantages in the exploitation of this resource are:

(i) In many cases, particularly for drinking purposes and in areas that are remote from intense human activities, treatment is usually simpler and less costly than for surface water.
(ii) Competition and conflict over the use of groundwater resources are typically less than for surface water.
(iii) It usually does not need to be transported over long distances to points of use. As such, it generally provides reliable supplies at comparatively low capital investment.

However, the rapid growth of population in the city initiated by the rural-urban migration has increased rates of urbanisation. Currently, of the estimated two million inhabitants of the city, about 75% live in high-density and poor/low-income settlements. Of these, only about 55% have access to a safe water supply, while more than 80% have no access to satisfactory sanitation facilities. Since these mushrooming low-income settlements are located over the aquifer recharge areas, the use of pit latrines or septic tanks to dispose of excreta threatens to contaminate the groundwater.

2 LOCATION

The city of Lusaka is the capital of the Republic of Zambia. Zambia lies in southern Africa between latitudes 22°–34° E and longitudes 9°–18° S.

3 POPULATION

The settlement and development patterns in Lusaka have been greatly influenced by the growth of the population. With a population of only 195,753 at independence in 1964, the city has experienced a progressive increase in population over the years, rising to 535,850 in 1980 and 769,353 in 1990 (Central Statistics Office, CSO, 1990). According to the 2000 census, the population of Lusaka reached one million two hundred thousand (although the author feels this is understated).

4 CLIMATE

The city experiences a sub-tropical climate that is strongly seasonal. It has three distinct seasons, namely:

a) A cool, dry season from mid-April to mid-August with mean day temperatures varying between 15°C and 23°C. Minimum temperatures may sometimes fall below 10°C in June and July.
b) A hot, dry season lasting from mid-August to mid-November. During this period, day temperatures may vary between 27°C and 38°C.
c) A warm, wet season from mid-November to mid-April, during which 95 per cent of the annual rainfall takes place. The annual rainfall averages about 800 mm/a.

5 TOPOGRAPHY AND GEOLOGY

The topography of Lusaka is characterised by a plateau of the mid-Tertiary (Miocene) peneplain, which stands at an elevation of 1200 m. There are flat-topped hills to the north and east of the city, which stand at an elevation of about 1300 m above sea level, and are assumed to be remnants of an earlier peneplain of Cretaceous age (Dixey, 1960).

The drainage pattern of the Lusaka area is in an essentially radial pattern (Fig. 21.1), which appears consistent with the domical-type relief that conforms with the basin-and-swell concept applied by Holmes (1965) to explain the relief of Africa in which the Lusaka plateau forms a minor swell.

Rocks underlying the city of Lusaka consist of schists interbedded with quartzites and dominated by thick and extensive sequences of marbles (Fig. 21.1), with the latter generally referred to as the Lusaka Dolomites/Lusaka Limestones. The underlying marble has suffered extreme differential dissolution, resulting in the development of a system of subterranean conduits and solution channels.

Available borehole drilling data indicate that carbonate rocks extend to depths in excess of 100 metres, but showing variations in the fracturing intensities (Nkhuwa, 1996). Some of the solution features have been intersected at depths in excess of 60 m below ground surface, and form what are usually referred to as underground rivers (as depicted in Fig. 21.2). From the borehole logs shown in Figure 21.2, no borehole in Lusaka has completely intersected the whole thickness of the aquifer. So it can safely be concluded that the Lusaka aquifer is more than 100 m, which is the deepest drilling that has so far been done.

As a result of the karstification in the Lusaka marbles, one of the most prominent features of the Lusaka plateau is the scarcity and/or complete lack of surface drainage, particularly in its central part, because rainwater drains into the fissures and/or infiltrates through the overburden to enter the groundwater. Only surface water in excess of the infiltration capacity is drained into minor seasonal streams.

Therefore, solution (karstic) features in the marbles are the major factors controlling groundwater flow in the Lusaka aquifer and have transformed the underlying rocks into a favourable and comparatively cheap source of water supply to the residents.

6 SOCIO-ENVIRONMENTAL PROBLEMS

The development of infrastructure for the provision of basic needs and essential social services, including water supply and adequate and safe methods of liquid and solid waste disposal in

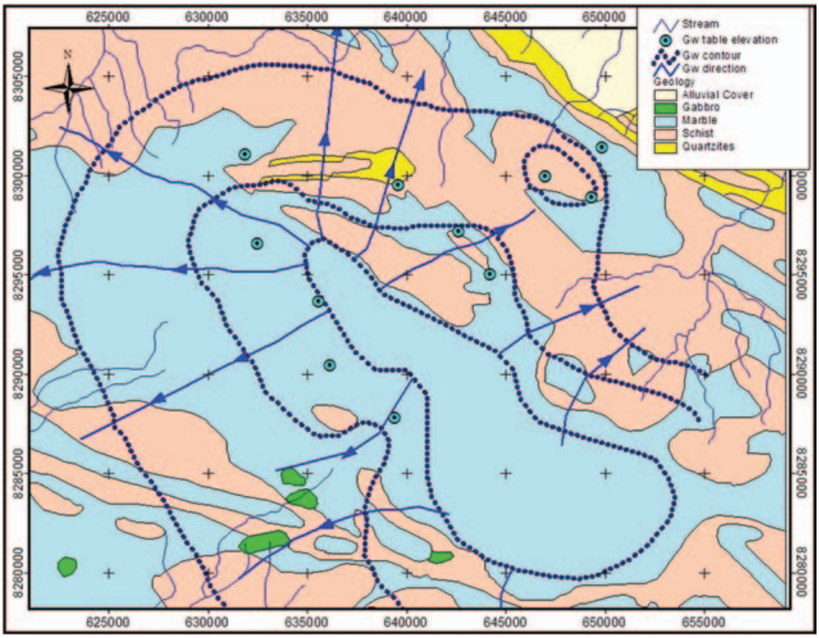

Figure 21.1. Map of the Lusaka plateau showing the geology and surface as well as underground drainage *(Modified from von Hoyer et al., 1978).*

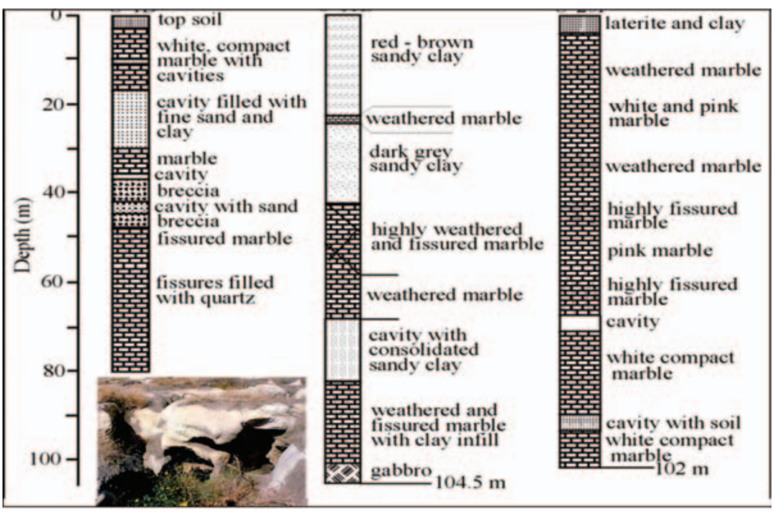

Figure 21.2. Solution features in the Lusaka marbles in John Laing.

Lusaka, has not kept pace with the rate of the population growth. As a result, increasing amounts of waste are being disposed indiscriminately.

Currently, of the estimated 1.2 million inhabitants of Lusaka, about 25% are serviced by a sewer system, about 20% by septic tanks, while 55% rely on pit latrines to dispose of their sewage and waste water. Areas using pit latrines are high-density residential townships that have generally developed in close proximity to areas of natural groundwater discharge or springs, where the groundwater table is very shallow. However, karstic aquifers, such as the one underlying Lusaka, are known to be notorious for being easily contaminated by surface activities, including faecal contamination.

This article discusses the water quality results of the UNEP – UNESCO – UN-Habitat sponsored research carried out in two high-density settlements – John Laing and Misisi Compounds – in Lusaka to examine what impact current methods of excreta disposal have on groundwater quality. The paper also examines public health issues associated with the consumption of water drawn from the aquifer underlying these settlements.

7 JOHN LAING AND MISISI AREAS – A CASE STUDY

The John Laing and Misisi areas are located south of the Central Business District (CBD) of Lusaka and bounded by Eastings $^{6}35500$ to $^{6}38000$ and Northings $^{82}92000$ to $^{82}93500$. The two areas lie in close proximity to one of the major discharge surface excavations, but may also be a recharge facility for some water points that lie downstream as indicated by the general ground water flow direction (Fig. 21.3).

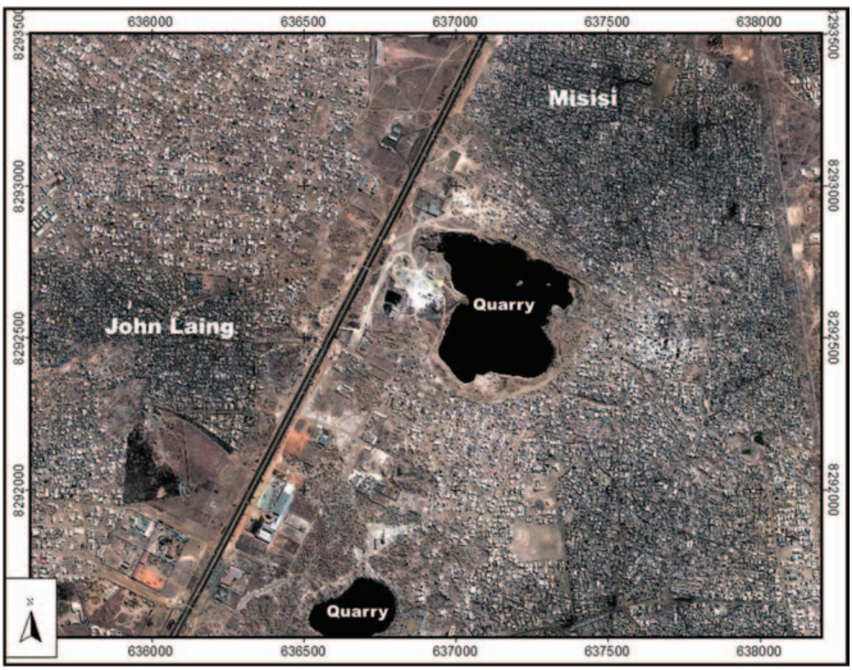

Figure 21.3. Location of John Laing and Misisi townships south of the Central Business District (CBD) and the quarries, which probably act as recharge and discharge points for the aquifer.

The selection of these project areas was based on five major criteria, of which the settlements of John Laing and Misisi satisfied most. The areas needed to:

a) Have a self-contained aquifer, where it would be possible to study source(s) of contamination.
b) Have a predominance of on-site sanitation for the disposal of its excreta.
c) Be in a location where contamination was identified to be a problem (area lying directly over the aquifer).
d) Be where the quality of the aquifer water could be correlated with health status, and where health data could be monitored.
e) Preferably be in a location where there were some water projects of one kind or another taking place or had taken place so that there were some data on which to build.

8 METHODS OF EXCRETA DISPOSAL

Disposal of excreta in the John Laing and Misisi compounds is mainly through pit latrines with minor systems of open defecation, pour flush latrines and limited use of flushing toilets or septic tanks. A number of families that are unable to afford their own latrine share with other families.

Two types of pit latrines are found in these compounds, namely ground-level and raised latrines. The ground-level pit latrines are often constructed in sinkholes over which residents build some form of structure made of sacks, plastics, grass or any other available materials (Fig. 21.4). These pit latrines are not built for structural strength, but only for privacy.

The superstructures for raised pit latrines (Fig. 21.4) are usually made of blocks, and when they are filled, they are either abandoned or the excreta removed and the pit re-worked for re-use. The excreta removed from these pit latrines is sometimes drained into primary discharge points, which may also act as recharge points in the dry season, bringing the excreta into direct contact with the water table.

An increasingly common practice has been noted in these compounds, where dried-up shallow wells are converted into pit latrines during the dry season. Unfortunately, these points have the potential for flooding during the wet season. Probably because of the danger of falling into pits, children in these townships are usually not encouraged to use pit latrines until about five years of age. Before that, the most common method noted is open defecation.

Most of these practices have greatly increased the potential for groundwater contamination. As many of these (low- and high-income) settlements that rely either on pit latrines or septic tanks to

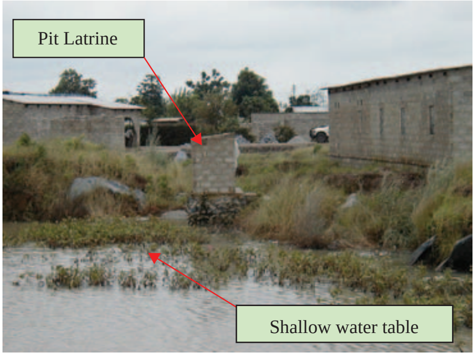

Figure 21.4. Raised pit latrines in the project areas.

dispose of their sewage also depend heavily on groundwater obtained from their private sources, consumption of such water, which is usually of unfavourable quality, has undoubtedly heightened outbreaks of water-borne diseases such as cholera and dysentery experienced in many areas of the city. In most parts of the city, these outbreaks are turning into almost perennial problems. Outbreaks have led to:

(i) An increase in costs for health-care services, posing a special burden, particularly for those members of society that are poor.
(ii) Loss of productive time due to illnesses that could otherwise have been avoided.

While this has been a recurrent problem in the city, there has been no detailed study attempting to link the outbreaks to groundwater quality. Although morbidity and mortality figures highlight the magnitude of the problem, a perspective of annual expenditures to treat these diseases will undoubtedly show the problem from another viewpoint.

Given this scenario, a critical element in the assessment, protection and management of groundwater resources in these areas involves the identification of contamination sources.

9 MAPPING OF WATER POINTS AND EXCRETA DISPOSAL FACILITIES

Most of the residents in the John Laing and Misisi compounds rely exclusively on groundwater from shallow hand-dug wells, which have different levels of protection (Fig. 21.5). Mapping of excreta disposal facilities, water points and dry wells, the latter representing the major sites for solid waste disposal, was done using a Garmin E-Trex hand-held GPS. Results of the mapping campaign were entered in spreadsheets and imported into the ArcView GIS programme (Fig. 21.5).

10 SAMPLING

Since the water table is very shallow in the project areas (up to 0.5 m below ground surface during the rainy season), human activities in these areas pose great risks of pollution to groundwater, which is envisaged to be the primary vector by which epidemic faecal-oral diseases (cholera, typhoid, hepatitis A and many diarrhoeal diseases) are passed from excreta into the water supply system and transmitted to large numbers of people very rapidly.

(a) (b)

Figure 21.5. Typical arrangement of water points (and excreta disposal systems) in the John Laing and Misisi compounds.

Three sampling campaigns were undertaken in mid-November 2003 (just before the onset of the rainy season) March 2004 (during the rainy season), and October 2004 (at the peak of the dry season). This arrangement was meant to compare the variability of pollutant loading with the varying levels of the water table. The second and third sampling campaigns targeted those points that proved qualitatively problematic during the first sampling campaign.

The following parameters were analysed – *pH, conductivity, chloride, sulphate, nitrite and nitrate, total coliforms and faecal coliforms.* These parameters were selected to determine the potability of the water. The values of the parameters in domestic water must conform to the WHO Guidelines for Drinking Water.

11 RESULTS

A summary of the results for the locations sampled during all three campaigns are given in Table 21.1. The most important water quality problem in the project areas of John Laing and Misisi is faecal pollution together with the associated disease-causing organisms.

Water quality problems are particularly serious during the rainy season, when faecal contamination appears to be flushed into the groundwater system. A plot of faecal coliform counts at different sampling points (Fig. 21.7) shows increasing contamination with rising water table levels. Minor spikes during the dry season are probably a result of local through-flow, since these locations are in an abandoned quarry, which places them at a lower elevation than the surrounding area. Thus, they are probably recipients of recharge from most of the (contaminated) surrounding area.

Chemical loading to the aquifer appears to behave inversely with regard to bacteriological contamination and groundwater levels. For example, conductivity (a proxy for TDS), chloride, and

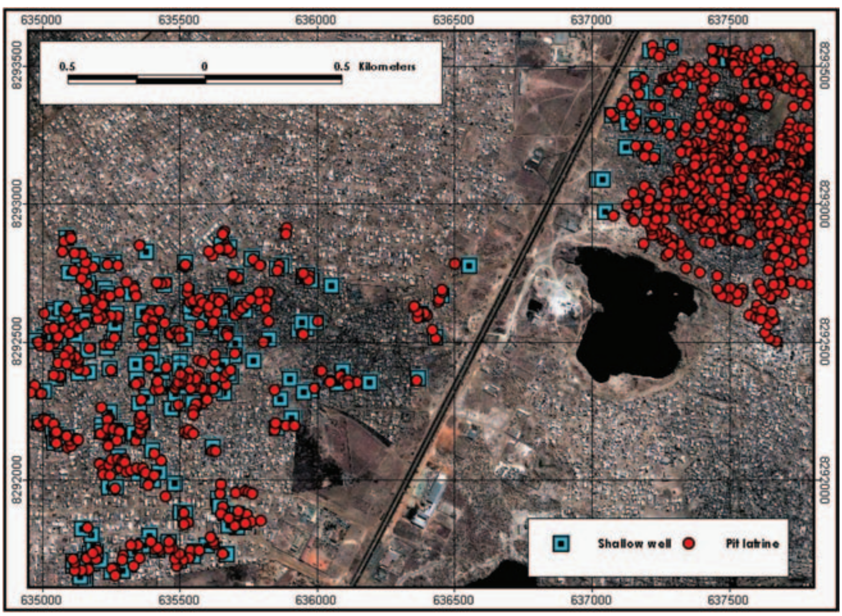

Figure 21.6. The distribution of excreta disposal points (pit latrines in red) and shallow wells in the John Laing and Misisi Compounds *(Image courtesy of the Surveyor General's Office).*

Table 21.1. A summary of results for all the locations that were subjected to at least two sampling campaigns in the John Laing and Misisi project areas.

	Conductivity (μS/cm)			pH			Nitrate (as N, mg/l)			Chloride (mg/l)			TC (counts/100 ml)			FC (counts/100 ml)		
Location	Nov ’03	Mar ’04	Oct ’04	Nov ’03	Mar ’04	Oct ’04	Nov ’03	Mar ’04	Oct ’04	Nov ’03	Mar ’04	Oct ’04	Nov ’03	Mar ’04	Oct ’04	Nov ’03	Mar ’04	Oct ’04
Loc-1	1228	468	1278	8	8.4	6.9	17	28.7	21.4	84	55	95	1,500	285	400	500	240	275
Loc-2	1446	478	1345	7	8.3	8	12	7	19.1	6	75	110	500	35	460	650	20	300
Loc-3	1177	387	ND	8	8.2	ND	19	2.9	ND	81	33	ND	66	100	ND	66	60	ND
Loc-4	1063	442	1079	8	9.1	7	16	2.1	23.9	80	61	85	800	150	21,600	4200	90	11,400
Loc-5	1413	392	1585	7	8.3	7.1	39	7.5	23	157	21	121	550	88	450	500	30	420
Loc-6	1044	754	1038	8	7.6	7.4	19	6.2	27.9	70	80	72	500	120	355	510	95	290
Loc-7	765	447	823	8	7.5	7.6	12	6.8	24.7	39	37	45	560	50	310	550	25	100
Loc-8	652	547	ND	8	8.2	ND	11	11.6	ND	44	55	ND	160	65	ND	160	30	ND
Loc-9	1371	460	1342	8	7.6	7.3	18	10.3	18.3	110	42	108	900	92	13,300	4500	40	9900
Loc-10	1408	724	1375	7	8.1	7.4	25	4.51	23.1	100	119	93	700	18,000	625	500	12,000	500
Loc-11	1056	656	1369	7	8	7.1	40	6.2	19.7	103	115	105	890	14,000	500	500	11,000	385
Loc-12	610	659	786	6	7.3	7.1	16	20	22	28	52	31	5000	5300	9000	500	3000	3200
Loc-13	1043	ND	1242	7	ND	7.2	39	ND	20	92	ND	85	ND	ND	300	ND	ND	210
Loc-14	1340	1233	ND	7	7.3	ND	15	2.92	ND	107	83	ND	ND	8000	ND	ND	6000	ND

ND – Not Determined.

[1] Sampling campaigns in November 2003 and October 2004 were during the dry season, while the March 2004 campaign was done during the rainy season.

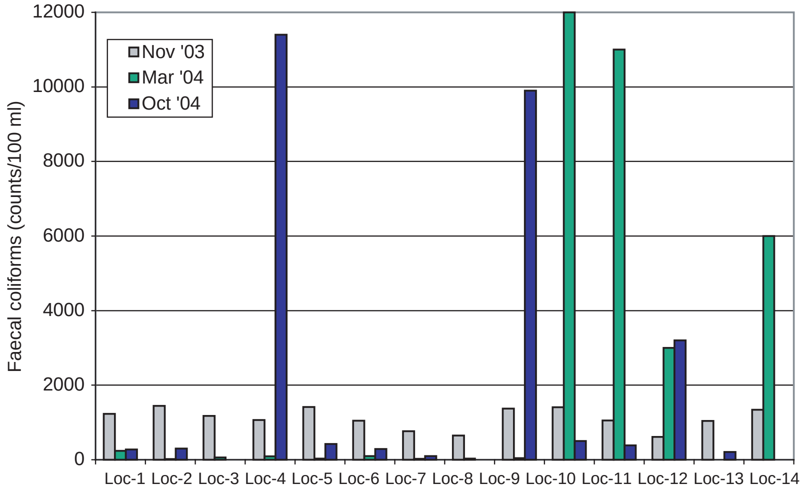

Figure 21.7. Variability of coliform contamination with varying levels of the water table in the aquifer.

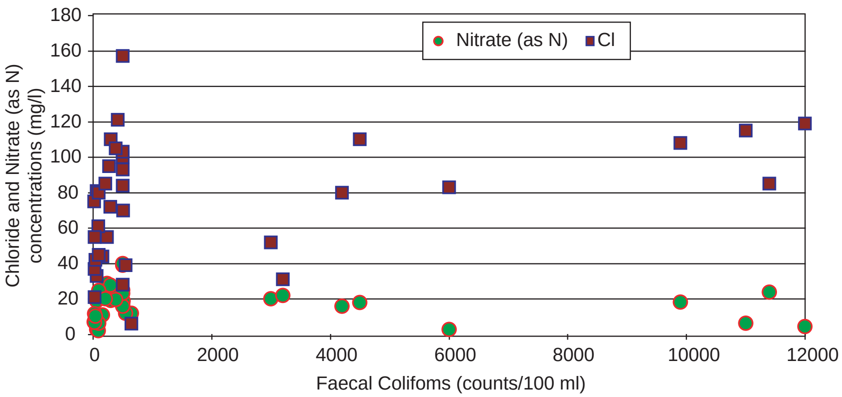

Figure 21.8. Regression plots to show the relationships between chloride and nitrate with faecal coliforms as influenced by varying levels of the water table in the aquifer in the John Laing and Misisi compounds.

nitrate were generally elevated during the dry season (Fig. 21.8). Almost all chemical elements appear to conform to this trend, which can be explained as resulting from lower recharge rates (due to reduced flows) and thus less dilution and greater dissolution of aquifer material.

During the rainy season, the water appears to become more alkaline, probably indicating the predominance of the HCO_3^- (and some CO_3^{2-}) ions in solution. In other words, the alkalinisation process may result from the exposure of carbonate rocks to a lot of water, thereby raising the pH-levels during this period. Conductivity (arising from high concentrations of chloride, nitrate and sulphate), on the other hand, appears to decrease during the rainy season – also probably as a result of high flows that keep the system flushed and deprived of mineralization.

Water with high conductivity (high TDS) also tends to have high chloride (and sulphate) concentrations. Sulphate removal is expensive (desalination or ion exchange) and normally not considered viable.

12 SAFETY OF WATER FOR DOMESTIC USE

Decision making on the suitability of water for domestic use is largely determined by the health problems related to drinking the water. One way to determine the safety of the water for domestic use is based on faecal coliform counts. On the basis of the South African Standards, most of the water points in the project areas pose serious effects on all user groups. Based on the WHO Guidelines most of the water sources may be classified as needing very urgent action. This information has not been available previously, and as a result there has been no prioritisation of water supplies to ensure that those sources presenting the greatest risk to public health were improved first.

13 THE RELATIONSHIP BETWEEN WATER QUALITY AND WATERBORNE DISEASES

Annual figures for Lusaka in general and the project areas in particular for the period 1996 to March 2004 are given in Table 21.2. The reason for the lack of and very low cases between 1998 and 2002 could be attributed a heightened campaign for home chlorination, which excited many residents quite regarding this new remedy for cholera outbreaks. In addition, this period does not appear to have been particularly wet. This trend appears to have made the authorities relax their campaign for chlorine use.

However, the outbreak of cholera cases even before the onset of the rainy season in November 2003 took the authorities by surprise. On 3 March 2004, a cumulative total of 4734 cholera cases were reported for the entire city of Lusaka, with 157 deaths and 55 brought-in-dead (BID). For the project areas, John Laing shows a higher disease outbreak than Misisi. This is likely, because there are more shallow wells in use in John Laing.

14 POSSIBLE IMPLICATIONS OF HIGH LEVELS OF FAECAL COLIFORMS ON PUBLIC HEALTH

The presence of faecal coliforms in the groundwater shows the presence of disease-causing microorganisms – *faecal coliforms* – which may be responsible for gastro-intestinal diseases, such

Table 21.2. Annual cholera figures for Misisi and John Laing (1966–2004).

	Cases of Cholera		
Year	Misisi	John Laing	Total ases in the city of Lusaka
1996	109	28	2469
1997	140	26	2492
1998	0	0	0
1999	–	–	6485
2000	0	0	0
2001	46	30	887
2002	–	–	2
2003/2004	96	250	4734*

* Figure for the period 28/11/03–03/03/04; Source: LDHMB cholera reports.

as cholera, typhoid, dysentery, and sometimes including fever and other secondary complications. In this regard, the dependence of residents in the John Laing and Misisi compounds on groundwater resources obtained from private boreholes and shallow had-dug wells is undoubtedly the cause for heightened outbreaks of water-borne diseases, which have also been experienced in other areas of the city with similar sanitary and water-source arrangements.

15 METHODS OF WATER TREATMENT AT HOUSEHOLD LEVEL

Decision making on the suitability of water for domestic use must be determined through the health problems related to drinking the water. Since most of the water sources, particularly in the densely settlement areas are of unsatisfactory quality, a number of methods may need to be employed to purify the water and make it safe for household use. Some of these have been and are still being used by some communities. However, some of the methods have serious drawbacks.

15.1 *Boiling*

This is the simplest way to kill pathogens. Some of its major disadvantages include:

a) Uses a lot of fuel-wood – about 1 kg of wood is needed to boil one litre of water. Because wood is scarce in Lusaka, its use may be limited by its availability and cost.
b) It sometimes gives an unpleasant taste to water.
c) Water may be contaminated again when it cools down.
d) Hot water may cause serious accidents in homes.

15.2 *Chlorination*

This process inactivates all types of microorganisms – protozoa, bacteria and viruses. However, its efficiency to inactivate microbes is affected by pH, contact time and its reactions with water. Microbes may be protected from chlorine if they are attached to or within particles in the water.

There are two types of reactions that can occur when chlorine is added to water that may affect its availability and efficiency as a disinfectant:

(i) Iron (Fe), Manganese (Mn) and hydrogen sulphide (H_2S) react irreversibly with chlorine. As a result, chlorine is removed from the water without contributing to the disinfection process.
(ii) Chlorine may react reversibly with organic matter and ammonia to form weak disinfectants. It has also been recognised that chlorine may react with organic substances in some waters to form trihalomethanes (THMs). There is some evidence that THMs may be carcinogenic.

16 POSSIBLE MEASURES FOR RESOURCE PROTECTION

Aquifer protection in Lusaka is very difficult, because the aquifer is generally characterised by shallow water tables, a thin soil cover, coarse soils with low clay contents, unconfined conditions, a flat topography that generally facilitates increased recharge, and numerous pollution sources located in aquifer recharge areas. Aquifer protection in Lusaka has generally been reactive, usually only in response to aquifer contamination resulting from human activities already established in areas that would, under normal circumstances, have been candidate zones for aquifer protection.

All efforts to protect the aquifer should have started with prohibiting any potentially contaminating development within each borehole's capture zone. Any future groundwater protection measures adopted in Lusaka must accept the fact that existing infrastructure and anthropomorphic activities cannot be moved, but must not be ignored.

Two options for aquifer protection in Lusaka are available – proactive/preventive and retroactive. Clearly, it would be preferable and to the benefit of public health to adopt the former, since a source that can be maintained contaminant-free would obviously be cheaper to use because of low treatment costs.

Retroactive measures in Lusaka, on the other hand, will generally consist of reducing and/or preventing further contaminant entry into the aquifer, the prevention of deleterious land uses in sensitive areas and an element of emergency clean-up.

However, high-density settlements like John Laing and Misisi may render certain forms of protection ineffective. Further, due to reasons of economics and inadequate public awareness, it might be very difficult, if not impossible, to force changes in land use practices in pursuit of aquifer-wide protection strategies.

In order to assess the options and effectiveness of the protection measures, it is strongly recommended that a monitoring system established under this pilot project be up-scaled to cover the rest of the aquifer. Water samples from supply points must be taken on a regular basis for a range of physico-chemical and microbial parameters that will give an early warning system for any contaminant event likely to occur.

17 CONCLUDING REMARKS

- The presence of a well-developed system of conduits, solution channels and subterranean cavities in the Lusaka aquifer(s) reduces and/or completely eliminates the natural attenuation of pollutants through natural filtration.
- Most physico-chemical parameters show a general increase in concentration from the wet season to the dry season, which is caused by the less recharge, dilution and an increase in dissolution of aquifer material. On the other hand, bacteriological loading appears to be greatest during the rainy season, when faecal contamination appears to be flushed into the groundwater system. Regular water quality monitoring from supply points for physico-chemical and microbial parameters will give an early warning of any contaminant event likely to occur.
- There is a need to formulate adequate community-based sensitisation and educational awareness campaign programmes on, for instance, pathways of pathogens into water supply sources and what methods, at household level, may be used to treat and safeguard the water.

The long-term deterioration of water quality, leading to progressively more costly water treatment, is the inevitable result of the current ad-hoc development in a thriving city such as Lusaka, especially as it is located largely on a karstic aquifer. In the long term, groundwater beneath Lusaka may become unfit for human consumption even with expensive treatment. The result is that new sources of water supply away from the current sources will need to be identified, or at least a consideration of whether or not a new site should be sought for the city.

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REFERENCES

Central Statistical Office 1990. 1990 Census of population, Housing and Agriculture – Preliminary report, 17pp, Lusaka.

Chapman, D. (Editor) 1992. Water quality assessments – A guide to the use of biota, sediments and water in environmental monitoring. *2nd Edition. Published on behalf of UNESCO, WHO & UNEP. ISBN 0-419-21600-6.*

Drysdall, A.R. 1960. *The Geology of Lusaka.* Rec. Geol. Surv. Northern Rhodesia, pp. 6–25.

Gibb Eastern Africa 1999. Final Report on Hydrogeological Investigations and Borehole inspections, Rehabilitation and test pumping. North-West Lusaka Water Project.

GRZ/Ministry of Health 1991. National Health Policies and Strategies (Health Reforms). Planning Unit.

Headworth, H.G. 1983. The influence of urban development on groundwater quality. In; Groundwater in Water Resources Planning, *vol.1, pp. 233–243, Koblenz, FRG.*

Nkhuwa, D.C.W. 2001. Relevance of groundwater quality assessment in the Lusaka aquifer. In Scott, R (Editor); People and Systems for Water, Sanitation and Health. *Proceedings of the 27th WEDC Conference, Lusaka, Zambia,* pp. 472–474.

Nkhuwa, D.C.W. 2000. Management of groundwater resources in Lusaka, Zambia, and expectations for the future. In Sililo, O.T.N. *et al.* (Eds.); *Groundwater – Past Achievements and Future Challenges. Proc. XXX. Conf. of the Int. Ass. of Hyrogeol.,* pp. 993–998*, Balkema, Rotterdam.*

Nkhuwa, D.C.W. 1999. Is groundwater management still an achievable task in the Lusaka aquifer? *Proc. XXIX. Conf. of the Int.Ass.of Hyrogeol., Bratislava, Slovak Republic,* pp. 209–213.

Water Research Commission 1998. Quality of domestic water supplies. Vol. 1; Assessment Guide. Water Research Commission No. TT 101/98.

WHO 1996. Cholera and other Diarrhoeal Disease; *Fact Sheets on Environmental Sanitation, Geneva.*

22

Groundwater quality case studies in Botswana

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ABSTRACT: The Botswana Department of Geological Survey (DGS), in collaboration with the German Federal Institute for Geosciences and Natural Resources (BGR), has established an Environmental Geology Division. The overall objective of this bilateral technical co-operation project was to enhance the capacity of the Department of Geological Survey to contribute to the protection of the environment in Botswana. During her first five years of operation (2000–2005), the division emphasised groundwater quality studies mainly carried out in the eastern part of the country. The results of the studies show that nitrate pollution is prevalent in many aquifers underlying built-up areas, mainly due to discharge from pit latrines. Even in rural areas groundwater nitrate pollution is a problem, although in a more haphazard fashion and due to anthropogenic and natural sources. In addition to nitrate pollution, the high levels of salinity are the most common groundwater quality concern in Botswana; especially the chloride, sodium, and sulphate levels. Elevated salinity levels render groundwater unsuitable for human and livestock consumption in many places, in particular the semi-arid Kalahari region. There are also a few cases where concentrations of trace elements (heavy metals) are reason for concern. Although most of the affected areas are associated with mining (gold, base metals), there are also instances where elevated concentrations appear to stem from geological formations.

1 INTRODUCTION

The desire for setting up an Environmental Geology Division emanated from an increasing awareness of the impact that mining and other forms of utilising geologic materials have on the environment in Botswana. The prime concern is with the various geologic resources, how man makes use of them, and the environmental consequences of this exploitation.

Pivotal activities concentrate on groundwater and mining, since these two geologic resources are the key to the economic development of Botswana. The division carries out studies into these two strategic resources both in urban and rural areas.

Figure 22.1. Botswana overview map and study locations (Source: Encyclopaedia Britannica, 2001, modified).

So far, priority areas are settlements and mining sites along the eastern population axis from Lobatse in the south to Francistown in the north, and the diamond mines of Orapa and Letlhakane (Fig. 22.1). In Lobatse, Francistown, Orapa and Letlhakane, the division carried out one-off studies. In Selebi-Phikwe, Mochudi, and Ramotswa, groundwater-monitoring programmes are in progress that will last for several years to establish long-term data series.

A regional study in the Central District looked into the problem of high groundwater nitrate and salinity levels. This study was initiated at the diamond mine site of Orapa, but subsequently grew in scope to cover the entire district.

2 GROUNDWATER QUALITY STUDIES

Groundwater quality studies enjoy top priority within the Environmental Geology Division, because Botswana is a severely water-stressed country. The majority of people in rural areas, as well as livestock, rely on groundwater for their drinking water supply.

Research work by the Environmental Geology Division has focussed on inorganic indicators such as total dissolved solids (TDS) and individual salinity parameters such as sulphate (SO_4^{2-}), chloride (Cl^-) and sodium (Na^+), nitrate-nitrogen (NO_3^-), and trace elements (heavy metals). Groundwater salinity is of major concern in many parts of Botswana, in particular the vast

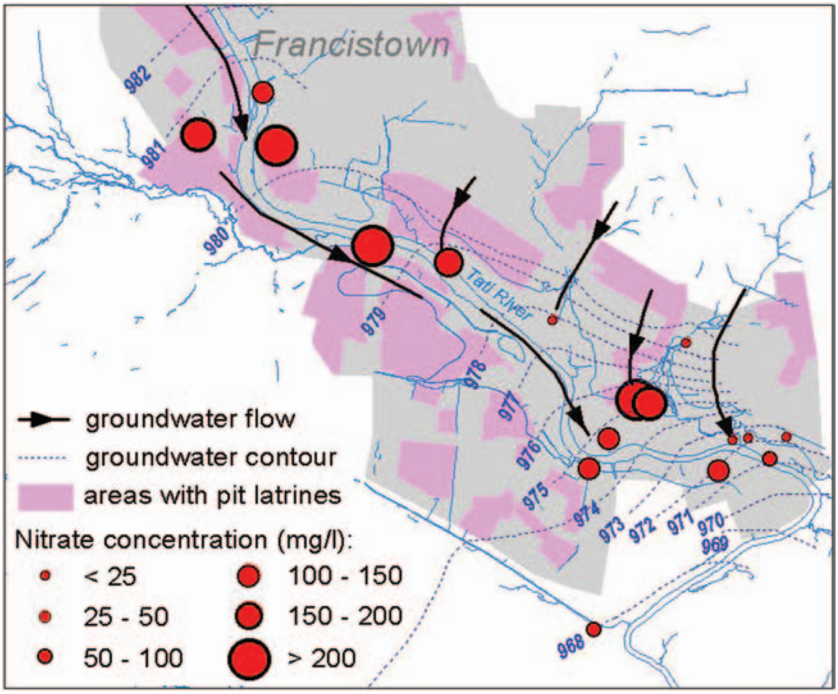

Figure 22.2. Spatial distribution of areas with pit latrines and boreholes with high nitrate levels (Artwork: N. Martin, BGR HQ, Hanover, Germany).

Kalahari region. Nitrate and trace elements are prime indicators of groundwater quality because of the associated health risk in humans and livestock.

An assessment of the levels of pollution is done in accordance with the Botswana Bureau of Standards drinking water specifications (BOS, 2000). These comprise three classes, namely Class 1 (ideal), Class 2 (acceptable for whole lifetime consumption), and Class 3 (maximum allowable for short-term consumption). Following is an area-by-area account of groundwater and environmental quality in Botswana.

2.1 *Francistown*

The division carried out its first groundwater quality study in Francistown, the second largest city after the capital Gaborone (Fig. 22.1). Francistown is a historic gold mining city. Forty-eight boreholes were sampled in mid-2000 (Mafa & Vogel, 2004). Most of the boreholes were aligned along the Ntshe and Tati rivers that flow through the city.

The study showed that groundwater in built-up areas suffers from nitrate pollution because of human sanitation (Fig. 22.2). Two other pollution sources were identified as abandoned gold mine tailings (heavy metals) and landfills (chloride, sulphate), although their influence was not as pronounced due to their distance from the rivers and thus the sampled boreholes.

Contamination from these three sources was limited to those zones within which pit latrines, mine waste dumps, and waste disposal sites are located. Groundwater from boreholes within these zones was not suitable for human consumption, because it exceeded certain World Health Organisation

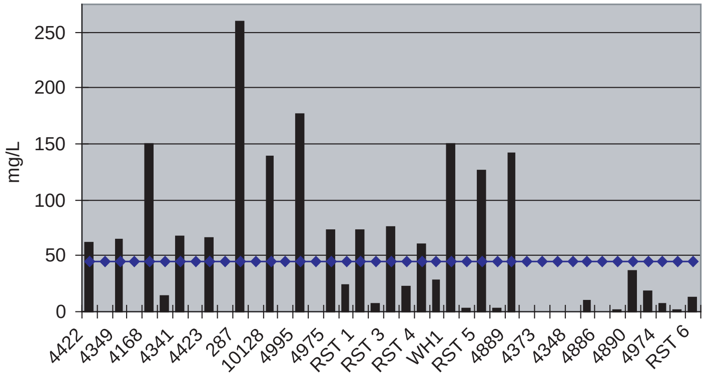

Figure 22.3. Nitrate levels in 40 boreholes in Ramotswa in April 2004.

(WHO, 1998) and Botswana Bureau of Standards (BOS, 2000) recommendations for drinking water.

Because of the observed groundwater contamination, the Environmental Geology Division advised authorities in the affected areas to determine which boreholes were in use for human consumption in order to discontinue their use. The division also recommended phasing out the use of pit latrines in favour of the newly built municipal sewerage. A complementary recommendation was not to embed infrastructures such as the new waste pipes that carry away the sewage into the riverbanks as in the year 2000. Such disturbance compromises the natural flood barrier and may lead to serious pollution in the case of leakage.

2.2 *Ramotswa*

The village of Ramotswa (Fig. 22.1) experienced one of Botswana's worst cases of groundwater pollution during the 1990s (Norwebb, 1996). The successful promotion of pit latrines and the location of Ramotswa on top of a highly productive yet shallow karst aquifer spelled disaster for groundwater quality. As a result, authorities had to abandon the entire wellfield in favour of surface water from the dam in nearby Gaborone.

In late 2001, the Environmental Geology Division carried out its first groundwater sampling in Ramotswa (Staudt & Vogel, 2004). The division considered the abandoned aquifer in Ramotswa indispensable for future development. Of special interest was the extent of nitrate pollution.

Of the 31 boreholes sampled in October and November 2001, 11 revealed nitrate levels that exceeded the Botswana standard of 45 mg L^{-1} for drinking water (BOS, 2000). The maximum value was a staggering 442 mg L^{-1}.

Two years later, the division started a groundwater-monitoring programme and re-sampled 24 boreholes in August/September 2003. The results confirmed the pollution levels established in 2001. Out of 24 boreholes, nine featured nitrate levels above 45 mg L^{-1}.

In April 2004, the division sampled 40 boreholes (including rehabilitated and new ones) and the results confirmed the high levels of nitrate pollution in Ramotswa (Fig. 22.3). In fact, out of the 40 boreholes, 15 featured nitrate levels in excess of 45 mg L^{-1}.

With the monitoring programme now firmly in place, the same boreholes were re-sampled in December 2004, except for three boreholes that were inaccessible because of crop growth. Unfortunately, amongst them was one (WH1) that previously had high concentrations. However, once again, a similar picture emerged (Fig. 22.4). It was also interesting to note that more or less the same boreholes featured high nitrate levels. The affected boreholes and their environments will have to receive particular attention in future.

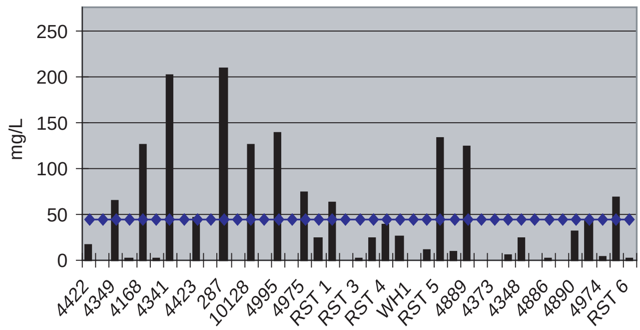

Figure 22.4. Nitrate levels in 37 boreholes in Ramotswa in December 2004.

So far, the division has not done complementary analyses for bacteria, because sampling requires the installation of a pump, which almost certainly introduces bacterial contamination to the samples. However, comparable to the case in Francistown, the spatial distribution of polluted boreholes made it clear that they are under the influence of pit latrines from within the village. While unpolluted groundwater enters the built-up areas from the south, boreholes in the village are contaminated. Previous bacteriologic studies support this finding (Lewis *et al*., 1980, Lagerstedt *et al*., 1994; Jacks *et al*., 1999).

2.3 *Lobatse*

In October and November 2000, the division sampled groundwater from 55 boreholes in and around the town of Lobatse (Fig. 22.1). Most of the analysed groundwater samples were of the Ca-Mg-HCO_3 type (Beger, 2001), which came as no surprise, since the Ramotswa Dolomite Formation is the most important aquifer in the Lobatse area (Key, 1983).

What came as a surprise, however, was the finding that very few boreholes displayed any sign of contamination. Except for a few cases, the overall groundwater quality in Lobatse was good in late 2000. The main exceptions were the boreholes on the premises of the huge abattoir of the Botswana Meat Commission (BMC) and one borehole on the premises of the brewery. In the boreholes at BMC, nitrate, sodium, and chloride concentrations in excess of the maximum allowable limits stipulated by the Botswana Bureau of Standards (BOS, 2000) were recorded. This was clearly indicative of human influence, possibly caused by wastewater discharge to the open environment with little pre-treatment. The borehole at the premises of the brewery showed elevated nitrate levels.

In the face of similar sanitation in Lobatse compared to Francistown and Ramotswa, and the presence of a predominantly similar karst aquifer in Lobatse compared to Ramotswa, it was not clear whether the situation encountered in late 2000 was representative of generally better groundwater in Lobatse as compared to the other settlements. The time of sampling prior to the onset of rains may have played a moderating role. Whatever the reason, the findings underlined the necessity for long-term groundwater monitoring to produce reliable groundwater quality trends.

2.4 *Orapa*

At Orapa (Fig. 22.1), the Debswana Diamond Company (Pty) Ltd. operates the second largest diamond mine in the world in terms of area, covering about 120 ha. Groundwater sampling in Orapa lasted from early December 2000 to early February 2001. Groundwater samples from 65 sites were taken (Matthes, 2002). They were located in and around the Orapa diamond mine area,

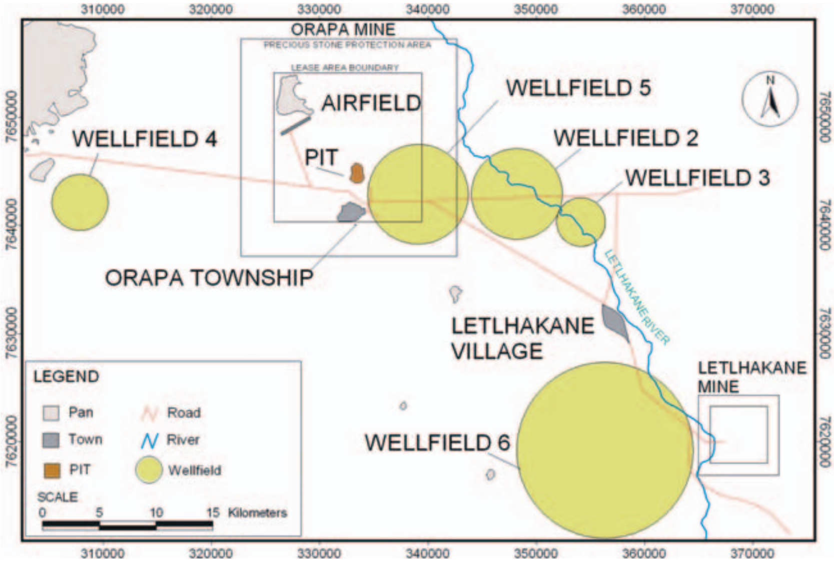

Figure 22.5. Location of Orapa and Letlhakane mines and of the wellfields.

around the Orapa landfill sites, and in wellfields close to Orapa and the neighbouring smaller satellite mine at Letlhakane (Fig. 22.5).

The reason for this study was the observed presence of consistently high nitrate concentrations in groundwater in the principal Ntane sandstone aquifer, which supplies the Orapa diamond mine (Mokokwe, 1999). The initial study started in April 1997 at the request of the Water Apportionment Board (WAB). The aim was to establish the order of magnitude and spatial extent of high nitrate levels and to inform residents in the area accordingly. Staff from the Hydrogeology Division of the Geological Survey completed the study in mid 1999, but the results were inconclusive.

The new study, carried out by the Environmental Geology Division, showed that groundwater conditions in the Orapa area are highly complex (Matthes, 2002). For example, ion concentrations proved to be highly variable in space and time. Results also revealed that the majority of the groundwater samples had a surplus of bicarbonate (HCO_3^-) over the sum of calcium (Ca^{2+}) and magnesium (Mg^{2+}), which was indicative of active ion exchange processes, because in the absence of ion exchange processes, total hardness ($Ca^{2+} + Mg^{2+}$) exceeds carbonate (HCO_3^-) hardness.

With regard to the origin of elevated nitrate concentrations, the study could not come up with concrete conclusions. However, there were strong indications that highly saline artesian water that rises along zones of structural weakness (contact zones of dolerite dykes and Kimberlitic pipes, or fissures and fractures) could be the source of the observed increases of nitrate levels in the Ntane sandstone aquifer. It is known from this area that, apart from highly saline waters, there are sedimentary layers containing coal in deep strata that will release ammonium (NH_4^+) under reducing conditions.

The study also put forward that, in principle, nitrate input from the surface was possible both as far as the topmost Kalahari beds and the below Stormberg basalt aquifer were concerned, due to their unconfined condition. Even leakage from the Stormberg into the below Ntane and the beneath Mosolotsane (mudstone) aquifer may be possible along fissures and fractured zones once the confined condition in the Ntane sandstone is removed. The latter may occur because of excessive groundwater pumping to allow for diamond mining, which in turn causes massive drawdown and thus a reversal of the hydraulic gradient.

Concentrations of total dissolved solids (TDS) highlighted a general increase in groundwater salinity from Letlhakane towards Orapa, that is from the south-east to the north-west (Fig. 22.5). The highest TDS levels were identified in Wellfields 4 and 5, and in the boreholes dewatering the open mine pit. The recorded concentrations were in the range of 1800 to 3800 mg L^{-1}. Data obtained from Debswana Diamond Company (Pty) Ltd. supported this spatial trend line.

2.5 *Letlhakane*

The water demand for both the Letlhakane village and diamond mine (Fig. 22.5) is met entirely from groundwater resources. Groundwater sampling in and around the diamond mine and village respectively took place in June and July 2002 (Keipeile, 2004).

Boreholes within the Letlhakane village generally complied with the BOS (2000) drinking water standard. However, in some boreholes, determinants such as chloride, sodium, nitrate, and TDS were found to be above the maximum allowable limit. A number of hand-dug wells in the village also violated the Botswana drinking water standards in terms of certain trace elements.

Diamond mining also displayed some negative impacts on groundwater quality, in particular around the slimes and slurry dams, but also due to groundwater mining. There were clear indications that groundwater mining has altered the natural baseline chemistry in a few places in the Letlhakane mine wellfield and in Wellfield 6, between the time of their initial development and mid 2002.

An unexpected finding was concentrations of some heavy metals above the drinking water standard in Wellfield 6 (Fig. 22.4), in particular selenium. Arsenic and lead were also elevated, but in a smaller number of boreholes, with cadmium and strontium trailing a distant third. In the absence of other obvious sources such as agricultural fertilisers, sewage sludge, hazardous waste landfills, or base-metal mining and smelting industries, we assumed a natural origin (coal layers in the deep Tlapana formation). No other explanation offered itself at the time. Clearly, authorities need to monitor the affected wellfield regularly in future. Once-off studies can only provide for wake-up calls.

2.6 *Selebi-Phikwe*

In Selebi-Phikwe, BCL (Pty) Ltd. mines and smelts copper, nickel and cobalt. During the course of 2004, the Environmental Geology Division set up a groundwater quality-monitoring programme in collaboration with BCL.

In a first step taken in March 2004, groundwater was sampled from nine private boreholes and eight hand-dug wells. All sampling sites were located outside Selebi-Phikwe in a radius of approximately 5 to 20 km away from the mine and town respectively.

Out of the 17 sites, 10 featured elevated nitrate concentrations (Fig. 22.6). Amongst them were four boreholes and six hand-dug wells. The highest concentration recorded was 298 mg L^{-1}, which came from a hand-dug well located about 11 km SW of Selebi-Phikwe.

In July 2004, while the drilling unit was engaged in the drilling and/or rehabilitation of 17 observation bores, fissure water was collected from three separate shafts and assayed for nitrates. The results revealed that the water-bearing fissure systems were high in nitrates.

Water pumped from various depths into a surface reservoir at Selebi Mine, located approximately 10 km to the SW of the town and plant respectively, showed nitrate concentrations of 235 to 641 mg L^{-1}. At Selebi Mine, fissure water occurs at several depths due to a major fault. However, it poses minor problems to mining. Where necessary, the mine grouted the fissures.

The highest concentration was recorded in Shaft 4, located approximately 4 km SW of the town centre. Here the nitrate concentration in the fissure groundwater was a staggering 888 mg L^{-1}. We assume that the nitrate in the shafts stems from nitrate-based explosives.

In November/December 2004, groundwater was sampled from the 17 new and/or rehabilitated observation bores, which are all located to the NE of the mine that is downstream of the plant along the prevailing groundwater flow direction. These boreholes are arranged in adjacent pairs, one of which is approximately 20 and the other about 30 m deep.

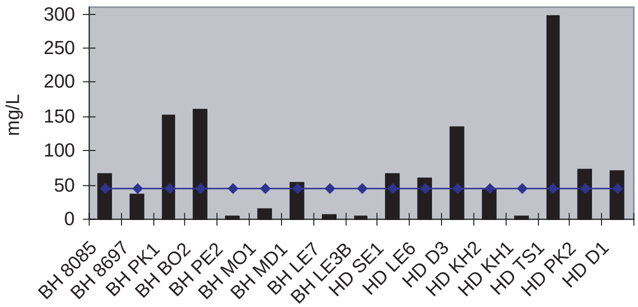

Figure 22.6. Nitrate concentrations in boreholes and hand-dug wells around Selebi-Phikwe in March/April 2004.

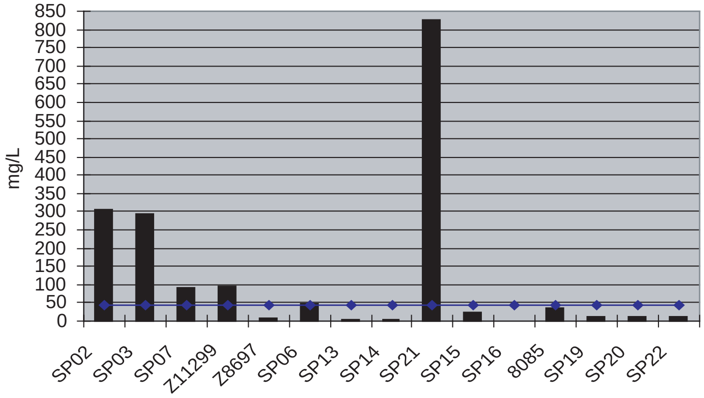

Figure 22.7. Nitrate concentrations in 15 boreholes in the wider Selebi-Phikwe area in April 2005.

The most prominent pollutant was nickel. Nine boreholes revealed nickel levels that were above the upper limit of 20 μg L^{-1} as stipulated by the Botswana Bureau of Standards (BOS, 2000). Manganese, iron and lead also exceeded their stipulated upper limits of 500, 2000, and 10 μg L^{-1} respectively in a number of boreholes. In contrast, copper and cobalt levels were below their defined maximum allowable limit of 1000 μg L^{-1} in all observation bores. Similarly, none of the observation bores showed elevated nitrate levels. However, they exceeded the maximum allowable TDS limit of 2000 mg L^{-1} in all but three observation boreholes. The prominent elevated dissolved solids were sulphate, calcium, and magnesium, and, to a small extent, chloride.

In preparation for the first re-sampling of the mine observation bores, efforts were made to identify more existing boreholes in the wider Selebi-Phikwe area. In April 2005, 15 accessible boreholes were sampled and the water assayed for major cations and anions. The results revealed frighteningly high levels of nitrate pollution at three chicken farms (Boreholes SP02, SP03, and SP21) of between 300 to above 800 mg L^{-1} (Fig. 22.7). All three boreholes are located on the premises of the chicken farms right next to production facilities.

Clearly, nitrate concentrations of 300 mg L^{-1} are unsafe for humans and risky for livestock if feed is also high in nitrates (Vogel *et al*., 2004). A concentration of 826 mg L^{-1} is outright dangerous from a health point of view and the water should not be used for human or livestock consumption.

2.7 *Mochudi*

Groundwater sampling in Mochudi (Fig. 22.1) commenced in January 2005. It involved 37 boreholes that already exist in a radius of approximately 4 to 15 km around the village. No borehole within this major village was accessible and no new monitoring bores had been drilled.

The first results for the rural boreholes showed that the vast majority contained water of acceptable drinking quality. Nitrate levels were mostly below the stipulated upper limit of 45 mg L^{-1}, except for three boreholes that had concentration levels of 51, 86 and 402 mg L^{-1} respectively. Interesting to note was that the outlier of 402 mg L^{-1} applied to a borehole sampled in the centre of the hamlet of Bokaa, which is located approximately 13 km SW of Mochudi. Again, we consider this level unsafe for humans and risky for livestock if feed is also high in nitrates.

Only two boreholes exceeded the maximum allowable TDS limit of 2000 mg L^{-1} (Class 3), while seven boreholes exceeded the acceptable limit of 1000 mg L^{-1} (Class 2). Twenty-six out of 37 sampled boreholes exceeded the upper limit of 450 mg L^{-1} for Class 1 (ideal).

2.8 *Central district*

Apart from the above local studies, one study covered an entire administrative district, namely the Central District, which is the economic hub of Botswana due to mining. The Central District stretches from the Makgadikgadi Pans in the north-west to the South African border in the south-east encompassing the diamond mines of Orapa and Letlhakane and the major settlements in the east that is Selebi-Phikwe, Serowe, Palapye and Mahalapye (cf. Fig. 22.1).

The parameters considered were nitrate and salinity (Vogel *et al.*, 2004). A number of methods, namely total dissolved solids (TDS), electrical conductivity (EC), sulphate (SO_4^{2-}), chloride (Cl^-), and sodium (Na^+) appraised salinity. Data were collated from various internal and external sources. They covered the period 1970 to 2003, but with emphasis on the years 1990 to 2003.

The study confirmed that there is reason for concern because elevated nitrate and salinity levels render groundwater unsuitable for human and livestock consumption in many places. Concerning nitrates, there was reason to believe that in rural areas the hazard was more of a sporadic nature, although of an alarmingly high magnitude at times. Twenty-eight boreholes recorded between 300 to 945 mg of nitrate (NO_3^-) per litre of water at one point.

The data also indicated that nitrate distribution and occurrence was highly variable and that concentrations may fluctuate significantly even over short periods, possibly due to downpour recharge. The irregular distribution of boreholes that distinguished themselves as nitrate hotspots suggested that nitrate mainly derived from point sources.

In the latter context, it is interesting to note that complementary nitrate research carried out in the Central District found that cattle posts, termite mounds, and indigenous acacia trees and shrubs, are sites of substantial nitrate enrichment in Kalahari soils (Stadler *et al.*, 2004; Schwiede *et al.*, 2005). Their hydrochemical results also imply a natural origin of groundwater nitrate from accumulation processes in the soil zone. Nevertheless, contrary to expectation, their on-site nitrogen species analysis and stable nitrate isotopes showed no evidence that cattle farming is yet a source of present-day nitrate concentrations in groundwater.

Field observations raise the question whether pans may provide for entry points, in particular since there is reason to believe that the distribution of pans in Botswana is structurally controlled (Wormald *et al.*, 2003). Future research is aimed at establishing whether preferential flow paths exist (in fractured rock) along the edges of pans, what kind of nitrate is present in pans, and what nitrate concentrations are to be found in the unsaturated (i.e., vadose) zone below the pans.

As expected, groundwater salinity proved to be of gravest magnitude around the Makgadikgadi Pan complex, one of the largest saltpans in the world. Data showed that groundwater was highly saline (TDS $>$ 10,000 mg/L) and hyper-saline (TDS $>$ 45,000 mg/L) in places. A linear correlation analysis suggested that halite (NaCl) dissolution was the mineral source of both sodium and chloride for the vast majority of groundwaters. A considerable amount may originate from wind-blown salts from the evaporative deposits of the Makgadikgadi Pan complex (Wood *et al.*, 2004).

Major settlements in the east also had a fair number of boreholes with high salinities. In these cases, we assumed that human activities such as sewage disposal sites, landfills, and livestock production were mainly responsible for groundwater salinisation.

3 CONCLUSIONS

The various groundwater quality studies carried out between mid-2000 and early 2005 highlight that human activities frequently compromise groundwater quality. Since a severely water-stressed country such as Botswana cannot afford to forego existing water resources, she needs political will and action to stop pollution from occurring or developing further. To kick-start and support concrete action, the Environmental Geology Division has set in motion briefings at the political level and called for media coverage.

High nitrate concentrations emerged as the single most important pollutant. Nitrate concentrations encountered in settlements often ranged between 100 and 200 mg L^{-1}, and sometimes they were as high as 400 to 600 mg L^{-1}. Even in sparsely populated rural areas, nitrates pose a threat at times, especially in years that experience prolonged heavy rainfall, as was the case in the year 2000. In such years, accumulated soil nitrate deposits as a result of natural processes in the soil appear to flush into the groundwater in places.

Elevated concentrations of toxic trace elements surfaced in a few locations. As expected, they were mainly associated with sulphidic ore material at mine sites. However, in the absence of supportive previous studies and/or historical hydrochemical records on these elements, current data are insufficient to appraise their magnitude and their health and environmental impacts. Research also still needs to be done to establish whether mining activities raised concentrations above natural background levels.

As expected, salinity parameters showed up at many sites. Enhanced concentrations of total dissolved solids, sulphate, chloride and sodium in groundwaters below settlements were clear signs of anthropogenic pollution. In the case of Francistown and Orapa, elevated salinity parameters suggested that waste disposal sites (landfills) were sources of groundwater pollution. The same applied to the industrial complex in Ramotswa (mainly steel and textile industries) and the huge abattoir in Lobatse. In the rural areas of the Kalahari, it is known that widespread groundwater salinity is due to natural processes.

The studies also spotlighted the need for regular long-term monitoring of groundwater quality, including constant control of laboratory results. That is why in the year 2003, the Environmental Geology Division successfully requested approval and support for three such projects (Ramotswa, Mochudi, Selebi-Phikwe) within the framework of Botswana's current National Development Plan 9 (2004 to 2009).

REFERENCES

Beger, K. 2001. Environmental hydrogeology of Lobatse. South-East District, Republic of Botswana. Report by the Environmental Geology Division, 45 p. incl. attachments, Department of Geological Survey (DGS), Lobatse, Botswana.

BOS 2000. Water quality – Drinking water – Specification. BOS 32, Botswana Bureau of Standards, Gaborone, Botswana.

Jacks, G., Sefe, F., Hammar, M. & P. Letsamao 1999. Tentative nitrogen budget for pit latrines, Eastern Botswana. *Environmental Geology* 38(3): 199–203.

Keipeile, K. 2004. Environmental hydrogeology of Letlhakane. Report by the Environmental Geology Division, 52 p. incl. attachments, Department of Geological Survey (DGS), Lobatse, Botswana.

Key, R. 1983. The geology of the area around Gaborone and Lobatse, Kweneng, Kgatleng, Southern and Southeast Districts. Department of Geological Survey (DGS), Ministry of Minerals, Energy and Water Resources (MMEWR), Botswana.

Lagerstedt, E., Jacks, G. & F. Sefe 1994. Nitrate in groundwater and N circulation in Eastern Botswana. *Environmental Geology* 23: 60–64.

Lewis, W., Farr, J. & S. Foster 1980. The pollution hazard to village water supplies in eastern Botswana. *Proc Instn Civ Engrs* 2(69): 281–293.

Mafa, B. & H. Vogel 2004. Effects of mining and urban expansion on groundwater quality in Francistown, Botswana. In: Stephenson, D., Shemang, E.M. & T.R. Chaoka (Eds.): Water Resources of Arid Areas. Proceedings of the International Conference on Water Resources of Arid and Semi Arid Regions of Africa (WRASRA), 3–6 August 2004, Gaborone, Botswana, pp. 133–141. A.A. Balkema Publishers.

Matthes, L. 2002. Environmental hydrogeology of Orapa. Report by the Environmental Geology Division, 67 p., Department of Geological Survey (DGS), Lobatse, Botswana.

Mokokwe, K. 1999. Occurrence of groundwater with high nitrate content, Orapa wellfields. Inception report, Department of Geological Survey (DGS), Lobatse, Botswana (unpublished).

Norwebb, B. 1996. Nitrate levels, Ramotswa Wellfield. Department of Water Affairs (DWA), Gaborone, Botswana.

Schwiede, M., Duijnisveld, W. & J. Böttcher 2005. Investigation of processes leading to nitrate enrichment in soils in the Kalahari region, Botswana. In: Vogel, H. & C. Chilume (Eds.): Environmental Geology in Semi-Arid Environments. Proceedings of the International Workshop on Environmental Geology in Semi-Arid Environments, 11–12 November 2004, Selebi-Phikwe, Botswana, pp. 103–112, Department of Geological Survey (DGS), Lobatse, Botswana.

Stadler, S., Hoyer, M., von, Duijnisveld, W., Himmelsbach, T., Schwiede, M., Böttcher, J. & H. Hötzl 2004. Investigation of natural enrichment processes of nitrate in soil and groundwater of semi-arid regions: Case-study Botswana. In: Stephenson, D., Shemang, E.M. and T.R. Choaka (Eds.): Water Resources of Arid Areas. Proceedings of the International Conference on Water Resources of Arid and Semi Arid Regions of Africa (WRASRA), 3–6 August 2004, Gaborone, Botswana, pp. 379–388. A.A. Balkema Publishers.

Staudt, M. & H. Vogel 2004. Environmental hydrogeology of the dolomite aquifer in Ramotswa, Botswana. In: Stephenson, D., Shemang, E.M. & T.R. Chaoka (Eds.): Water Resources of Arid Areas. Proceedings of the International Conference on Water Resources of Arid and Semi Arid Regions of Africa (WRASRA), 3–6 August 2004, Gaborone, Botswana, pp. 371–377. A.A. Balkema Publishers.

Vogel, H., Mokokwe, K. & T. Setloboko 2004. Nitrate hotspots and salinity levels in groundwater in the Central District of Botswana. Report by the Environmental Geology Division, 34 p. including appendices (maps), Department of Geological Survey (DGS), Lobatse, Botswana.

WHO 1998. Guidelines for drinking water quality. Second edition, Volumes 1 and 2. World Health Organization (WHO), Geneva, Switzerland.

Wood, W., Eckardt, F., Kraemer, T. & K. Eng 2004. Quantitative eolian transport of evaporate salts from the Makgadikgadi Depression (Ntwetwe and Sua Pans) in northeastern Botswana. Implications for regional ground-water quality. In: Barth, H.-J. & B. Böer (Eds.): Sabka Ecosystems. Volume 3, Africa. Kluwer (in press).

Wormald, R., Eckardt, F., and J. and S. Vearncombe 2003. Spatial distribution analysis of pans in Botswana: The importance of structural control. *South African Journal of Geology* 106: 287–290.

23

Contamination and protection of the Cape Flats Aquifer, South Africa

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ABSTRACT: In the last decade the water supply needs for drinking and industrial purposes have increased remarkably in the Cape Municipality, requiring the exploitation of groundwater as a consequence of the reduced quantity of surface water resources. Monitoring has detected elevated levels of chlorides, nitrates, fluorides, and metals. This paper focuses on conceptualising the dominant spatial trends of essentially non-point pollution and identifying the main controlling factors within the Cape Flats. The groundwater pollution trends and source identification are all emphasised. In considering the Cape Flats aquifer as a water resource, these factors are important. Subsequent to point source identification and the various non-point or diffuse sources of pollution, the need for protection zoning in the wellfields of the Cape Flats aquifer has also been highlighted.

1 INTRODUCTION

The Cape Flats represents a region of broad coastal sand between the Cape Peninsula and the mainland. The sands, covering an area of approximately 630 km^2, extend in a northerly direction along the west coast (Fig. 23.1). The Cape Town metropolitan area contains both primary and secondary aquifers. Of these, the Cape Flats Aquifer is said to have the highest groundwater potential, yet remains the most under-utilised (Wright & Conrad, 1995). The Cape Flats area is characterised by both high productivity of groundwater resources and a dense human settlement, with industrial and agricultural activities particularly sensitive to the sources of pollutants. The assessment of the fate and behaviour of non-point source pollutants is a complex environmental problem, because of the heterogeneity of conditions in the subsurface system and spread over large areas relative to concentrations.

Due to the recent drought, water restrictions were imposed by the City of Cape Town, which triggered the public outcry that the potential of the Cape Flats Aquifer be reviewed. Based on present and previous investigations (Tredoux, 1984; Weaver & Tworeck, 1988; Ball *et al.*, 1995; McLear, 1995; Engelbrecht, 1998; Bertram, 1989; Saayman, 1999; Traut & Stow, 1999; Mehlomakulu, 2000; Parsons & Taljard, 2000; Saayman *et al.*, 2000; Ball & Associates, 2003), this paper reviews groundwater contamination in the area around the Cape Flats and presents current trend highlighting the potential sources of contamination, thereby yielding information useful for precautionary planning in water resource management.

2 CHARACTERISTICS OF THE CAPE FLATS

2.1 *Physiography and climate*

The Cape Flats are essentially lowlands with an average elevation of 30 metres above sea level (m.a.s.l.). Numerous sand dunes are present on the Cape Flats and have a prevalent south-easterly orientation and elevation up to 65 m. In several depressions; flat marshy sites, locally named 'vlei's', are present. These sometimes contain open water and are connected with the sea by a river. The drainage towards the south takes place by the Eersteriver and the Zeekoevlei into False Bay, whereas to the north the Salt River and Diep River flow into Table Bay.

The Cape Flats has a typical Mediterranean climate with cold, wet winters and warm, dry summers. The rainfall is largely controlled by the topography. To the north of the Western Cape, this climate regime grades into semi-desert, whereas to the east the climate becomes less seasonal and tends towards sub-tropical on the coast. Rainfall is highly variable and concentrated within the winter months. The mean annual rainfall ranges from 500 mm to 800 mm, with the maximum temperature sometimes approaching 40°C, while a minimum temperature below 8°C is unusual. The generally mountainous nature of the Cape Fold Belt results in sharp climate changes for the entire region. In the summer (December-March), the monthly mean of 10–50 mm of rainfall is recorded in the coastal plains and approximately 50 mm in the mountains while in winter (June–August) monthly means of 40–100 mm and over 200 mm are recorded, respectively.

2.2 *Geology and hydrogeology*

The Cape Flats Aquifer is essentially a sand unit (Sandveld Group) of Cenozoic age that was deposited on top of the impervious Malmesbury shales and Cape granites (Fig. 23.1). The bedrock comprises the Cape Granite Suite (in the extreme west) and metasediments of the Malmesbury Group overlain by Late-Tertiary to Recent sediments, up to 50 m thick. The bedrock topography shows that there is a Palaeo-valley reaching more than 40 m below mean sea level towards the north-eastern portion of the area. The sand body is generally stratified horizontally and several lithostratigraphic units can be recognized.

The aquifer sands are well sorted and rounded resulting in hydraulic conductivities of 30–40 m/d in the central area and 15–50 m/d in the eastern portion (Gerber, 1981). Transmissivity values range from 50–650 m^2/d, with typical values between 200 and 350 m^2/d. The effective porosity was of the order of 0.10 to 0.12, but values of 0.25 are found over large areas (Gerber, 1981). The net groundwater recharge of primary aquifers in the south-western Cape varies between 15% and 37% of annual precipitation. The Cape Flats Aquifer is not hydrogeologically linked to any other aquifer, except the talus/scree material along the foot of the mountains in the west (Wright & Conrad, 1995).

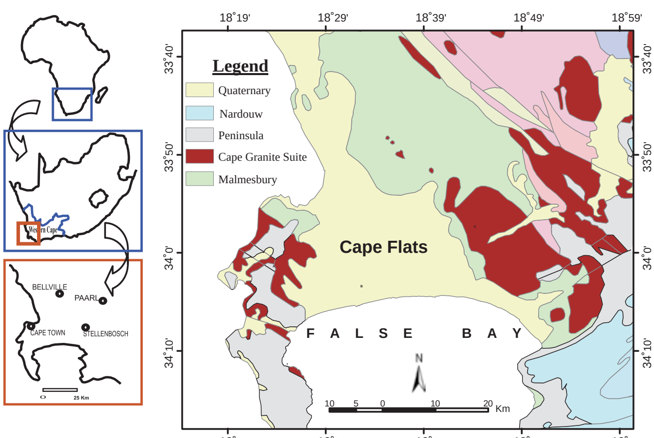

Figure 23.1. Location and geological map of the area around the Cape Flats.

3 APPROACH

Studies of urban impact on groundwater quality in this area have been investigated in detail. The various work related to groundwater contamination in the Cape Flats have been reviewed. Based on the available data and records so far, the current trend of groundwater quality in the Cape Flats, as well as the potential sources of contamination is to be highlighted, thereby yielding information useful for precautionary planning in water resources management.

This study addresses three areas: (1) Categorising urbanized–industrial, residential and green areas based on data obtained from municipal and environmental departments; (2) geological-hydrogeological characterisation based on available maps and information accumulated in various reports; (3) hydrochemical assessment of shallow groundwater based on previous publications and reports, as well as interpretation of data from existing databases.

Monitoring and observation borehole records were employed in the interpretation of the current groundwater quality in the Cape Flats. Access to several unpublished reports also yielded valuable information. The limitation in the use of the available data on groundwater chemistry lies in the fact that most of the boreholes were not consistently sampled over the same period.

4 CLASSIFICATION OF POLLUTION SOURCES

Various sources of pollution have been classed according to human activities ranging from agricultural practice, sanitation and mining. An idea of the more common types of activity capable of causing significant groundwater pollution hazards can be gained from Table 23.1 (Foster *et al.*, 2002).

Table 23.1. Common pollution sources and associated groundwater contaminants.

Pollution source	Type of contaminant	Potential impact
Agricultural activity	Nitrates; ammonium; pesticides; fecal organisms	Health risk to users (e.g. infant methemoglobinemia), toxic/carcinogenic
In-situ sanitation	Nitrates; faecal organisms; trace synthetic hydrocarbons	Health risk to users, eutrophication of water bodies
Gasoline filling stations & garages	Benzene; other aromatic hydrocarbons; phenols; some halogenated hydrocarbons	Carcinogens & toxic compounds, odour and taste
Solid waste disposal	Ammonium; salinity; some halogenated hydrocarbons; heavy metals	Health risk to users, eutrophication of water bodies, odour & taste
Metal industries	Trichloroethylene; tetrachloroethylene; other halogenated hydrocarbons; heavy metals; phenols; cyanide	Carcinogens & toxic elements (As, Cn)
Painting and enamel works	Alkyl benzene; tetrachloroethylene; other halogenated hydrocarbons; metals; some aromatic hydrocarbons	Carcinogens & toxic elements
Timber industry	Pentachlorophenol; some aromatic hydrocarbons	Carcinogens & toxic elements
Dry cleaning	Trichloroethylene; tetrachloroethylene	Carcinogens & toxic elements
Pesticide manufacture	Various halogenated hydrocarbons; phenols; arsenic	Toxic/carcinogenic compounds
Sewage sludge disposal	Nitrates; various halogenated hydrocarbons; lead; zinc	Health risk to users, eutrophication of water bodies, odour & tastes
Leather tanneries	Chromium; various halogenated hydrocarbons; phenols	
Oil and gas exploration/extraction	Salinity (sodium chloride); aromatic hydrocarbons	May increase concentrations of some compounds to toxic levels
Metalliferous and coal mining	Acidity; various heavy metals; iron; sulphates	Acidification of groundwater & toxic leached heavy metals

In the Cape Municipality contaminant sources related to human settlement activities are cemeteries, storm water and wastewater systems. Other significant sources of contamination in and around the city of Cape Town is from leakage of underground petrol and diesel storage tanks, nutrients and pathogens in human wastes (e.g. nitrate, phosphate, potassium), cyanide and trichloroethylene (TCE) from metal plates, chemicals used for cleaning and agrochemicals (fertilisers and pesticides). Potential sources of groundwater contaminants around the city have been identified and prioritised. A typical contaminant inventory in Cape Town resulted in an exhaustive listing of all potential sources of groundwater contamination reported by Usher *et al.* (2004).

There are several waste disposal sites in and around the Cape Flats, most of which have served as a place of refuse and sewage dumps for decades. Quite a good number of these are still actively receiving waste, although some have placed restrictions on the kind of waste, especially on toxic wastes under the guidelines of the Department of Water Affairs and Forestry (DWAF, 1994). Area pollution sources were associated with the waste disposal sites (for both solid and liquid wastes), mostly generated from industries and high-density residential areas. In the residential areas, the formal settlements have organised sewage and sanitation systems, but these are lacking in informal housing units. Irrigation, fertiliser and pesticide applications contribute significantly in the Philippi agricultural areas.

In the Cape Town area, several potential point-pollution sources were identified. These include chemical and pharmaceutical industries, the long existence of a major harbour, with reported contaminated waters, urban infrastructure, and particularly sanitary landfills and pipeline outlet disposal. Although a detailed vulnerability assessment is still underway, parameters such as depth to static water level, aquifer characteristics and thickness of the aquiferous sandy layer could control the vulnerability of the Cape Flats Aquifer to pollution.

In addition, saltwater intrusion from the sea also poses pollution threats to groundwater. This has not attracted much research attention in South Africa, especially in the Western Cape with its long coastline. Protection of the catchment area is a complex issue in this case and should involve analysis of both hazards and pathways.

5 RESULTS AND DISCUSSION

5.1 *Pollution issues identified in previous studies*

Tredoux (1984) identified the effluents from the sewage treatment works reaching the water table and active pollution source around the waste disposal sites. Results of groundwater sampled from both observation and monitoring boreholes indicated pollution in the Cape Flats with respect to NH_4, NO_3, K, total alkalinity and COD. Although potassium is not a common pollutant of groundwater, it was used to trace leachates from sanitary landfills and sewage effluents into the Cape Flats Aquifer. Groundwater quality monitoring indicated an increasing trend in pollution between 1979 and 1982; while prior to 1979 no definite sign of groundwater pollution could be discerned. Pollution reached its peak values between 1980 and 1981.

Similarly, monitoring the Coastal Park landfill area located adjacent to the outlet of the Zeekoevlei Sewage Works, some 400 m from the coast and separated from the groundwater by a 2 m natural unsaturated zone comprising the Cape Flats sand, gave indications of pollution. The results show that leachate from the landfill has entered the groundwater; migrating mainly eastward towards the Zeekoeivlei Outlet, although pollution migration in a southerly direction is now also being detected (Ball & Associates, 2003). Furthermore, differential depth sampling of boreholes shows a persistent trend of pollution with depth over a period of time. There is general organic pollution throughout the profile, while NH_4 gave indications of high concentrations associated with the top of the profile and attributable to leachate from the landfill. However, the long-term trends for COD and NH_4 provided new insight into the possibility of leachate attenuation (Ball & Associates, 2003).

In the study of the interaction between the Cape Flats Aquifer and False Bay, the risk of the aquifer becoming increasingly contaminated from low- to medium-risk pollution sources was identified (Giljam & Waldron, 2002). According to Giljam and Waldron, the Cape Flats Aquifer is vulnerable to many outside influences: the informal settlement of the Khayelitsha (where there is poor sanitation) and the Philippi agricultural area (where fertiliser application takes place regularly) and numerous nodal sources of pollution (e.g. Waste Water Treatment Works, WWTW and the waste disposal sites). High silicate, nitrate and phosphate concentrations, which confirmed the earlier report of Hartnady & Rogers (1990) and Grobicki (2000), were identified in the Philippi agricultural area. In the same vein, Traut & Stow (2001) identified high NH_4 concentrations at the Swartklip waste disposal site (between Mitchell's Plain and Khayelitsha). A marked increase in salinity was recorded at eastern boundary of the Cape Flats WWTW. The nitrate, phosphate and silicate concentrations in the berm and behind the surfzone were high in comparison to other sample sites south of the Philippi area and correspond with the generally high concentrations of these nutrients in the groundwater of the area.

Furthermore, Saayman (1999) investigated the Bellville waste disposal site in a study identifying the chemical characteristics of a pollution plume and determined its direction of movement. The results show high concentrations of potassium, sulphate and orthophosphate, with elevated concentrations of other ions (magnesium, chloride), COD, electrical conductivity and heavy metals (nickel and lead). In an attempt to establish which of the sources contribute significantly to groundwater pollution, the boreholes were sampled for isotopes of oxygen and hydrogen (Saayman *et al.*, 2000). The results of the stable isotopes define the level of influence of recharge from surface pollution on the groundwater of this area.

Another report of concern in the Cape Town area is pollution of groundwater by cemeteries. Engelbrecht (1998) reported the occurrence of groundwater pollution in the unconsolidated sands of the Bredasdorp Group by the influence of a cemetery. In a local municipal cemetery, 21 wellpoints were installed in the cemetery grounds and one outside the cemetery for sampling and quantifying the groundwater quality. The results showed an increase of colony-forming units (cfu) for all microbiological indicators, indicating that the groundwater within the cemetery area is extremely polluted compared with the expected regional groundwater quality. According to Engelbrecht, pathogenic bacteria, viruses, protozoa and helminths reached the groundwater, causing elevated concentrations above the regional groundwater quality (as represented by the municipal borehole). Thus, K, NH_4, NO_3 and NO_2 with dissolved organic carbon and electrical conductivity showed increased concentrations in all the wellpoints in addition to high levels of escherichia coli, faecal streptococci and staphylococcus aureus.

5.2 *Current status and trend of water quality in the Cape Flats*

Generally, the water supply in the Cape area is mostly derived from surface water sources, although records show that some municipal departments and private institutions do use groundwater and reclaimed sewage water for irrigation. However, the results of hydrological investigations to determine the potential role of the Cape Flats Aquifer in the alleviation of the water-supply problems showed that the phreatic aquifer represents a viable source of groundwater for the greater Cape Town area. Unfortunately, urban development has taken place over many parts of the aquifer and cities are known to always have a negative impact on the quality of shallow groundwater. This is the result of two reasons: (1) water-table aquifers are vulnerable to pollution, and (2) a city is often a source of concentrated pollution of various origins.

The overall groundwater chemistry from the DWAF database on the Cape Flats Aquifer did not show the quality as unacceptable, considering the individual constituents. However, high TDS reflects some pollution. This was traced to high metal and inorganic constituents in the waters. It was difficult to interpret the chemical data because of the inconsistency in the measuring periods. Where records seem to be consistent over a given period, not all the observation and monitoring boreholes were sampled. Therefore, the interpretation was done selectively to make sense of the enormous amount of data recorded from the mid-1960s. For example, selected boreholes consistently monitored for water quality from 1985–1993 in the Mitchells' Plain area showed a clear trend in nitrate content. Nitrate showed a consistent increase from 1987. With the patterns observed, it is logical to assume that the socio-economic activities of the particular land-use category exert a significant influence on the groundwater quality. Poor sanitation facilities in some of the densely populated residential areas and commercial land use result in a high occurrence of chlorides and nitrogen.

The sampling from 1985 to 1989 in the Philippi area showed (on average) NO_3 and PO_4 as <5 and 0.1 mg/L respectively. From these data, NH_4, B, and sometimes Pb are significant, but values do not exceed maximum permissible limits. Cr, Cd, and Ni are low in concentration while Co, As, Zr, Mo and Hg are all below detection limits. Cl was recorded to a maximum of 537 mg/L, sulphate 326 mg/L and TDS (1583 mg/L) indicating some pollution. Although chlorides are high, they present no adverse potability implications except for taste. There are no significantly high concentrations of phosphates to justify contamination due to the use of detergents and related activities in the residential and industrial sector. However, the application of chemical fertilisers in the agricultural areas is a possible indicator of increased phosphates in groundwater in the nearest future. Weathering is a hydrogeological factor that can be responsible for the high levels of some elements like silicates, fluorides, and metals in certain areas. This is subject to further research.

The data on chemistry for 21 boreholes (sampled between September 19 and October 11, 1990 and stored in the DWAF database) showed some interesting results and as such were interpreted separately. Cl, Na, and the total dissolved solids (TDS) were high in these wells, as illustrated in Figure 23.2. The wells are located in the north-western part of the Philippi agricultural area (around Ottery and Woodside). Six of these wells showed an exceptionally high content of Na, Cl and TDS (with TDS up to a maximum of 4170 mg/L and a mean value of 971 mg/L). Fluoride contents are also high and close to the upper limit acceptable for drinking water. These are obviously polluted, but it is not yet clear whether the wells are under the influence of seawater or the effects of farming activities. Other parameters often indicating agricultural pollution are, however, low in concentrations: NO_3 ($\leqslant 5$ mg/L except 13.1 mg/L in Borehole Number 21), NH_4-N (0.04–0.09 mg/L) and PO_4 (0.006–0.043 mg/L).

The more recent data on the Philippi farming area of Cape Town have been interpreted as showing pollution by heavy metals. The data obtained from Meerkotter (2003) in the winter and summer periods, indicate that Cd, Cr, Pb and Ni concentrations are high, illustrating pollution as a result of land use. There is no marked seasonal effect on the heavy metal pollution indicators except for the higher concentrations of Cr during summer sampling.

However, Ca, Mg, Na, and K all showed a slight increase from summer to winter. This is confirmed in the soil samples that showed an increasing trend and evidence of heavy metal pollution. Leaching is of most concern in groundwater systems as the contaminant moves through the soil

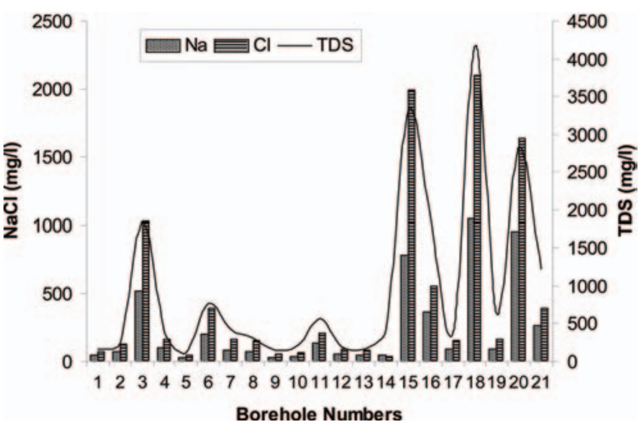

Figure 23.2. NaCl concentration and TDS in wells located in the north-western part of the Philippi area.

with the groundwater. The concern about this level of concentration in soils is related to heavy metals tending to accumulate in soils over time, increasing the chances of groundwater contamination. The minute concentration of heavy metals can adversely affect human health.

5.3 *Surface water pollution and contribution to groundwater quality*

The significance of surface water pollution in the Cape Flats with respect to groundwater cannot be over-emphasised. Saayman *et al.* (2000), through the use of stable isotopes, established the influence of surface water pollution on groundwater. From the monitoring records of the City of Cape Town Catchment, Stormwater and River Management (CSRM), there are indications that inland water quality has declined over the years, while coastal water quality only declined slightly, although there are areas that are not fully compliant with the DWAF public health guidelines. The most recent data for the record year from the City Council's Water Pollution Control range from October 2003 to September 2004. The number of incidence involving discharges of pollutants to stormwater was reported with the final effluent compliance of wastewater treatment estimated based on the nutrients for each recorded incident period (Table 23.2). Experience has shown that this introduces contaminants into groundwater.

In addition to the point sources tabulated above, various non-point or diffuse sources of pollution exist and have been tabulated as a summary of sanitation backlog based on the aerial photograph presented in CSRM Report (2004). The major non-point pollution source is run-off from informal settlements. The associated contributory factor has been the backlog in the provision of adequate sanitation services to informal settlements. Quite a number still use a bucket toilet system. The summary of sanitation backlog to informal settlements is presented in Table 23.3.

Table 23.2. Summary of data from point sources of pollution.*

	Recorded incidences				
Type of pollution	Oct–Dec 2003	Jan–Mar 2004	Apr–Jun 2004	Jul–Sep 2004	Total
Illegal discharges to stormwater	32	56	45	36	169
Gravity sewer blockages	17,397	19,503	21,368	20,620	78,888
Pump station & rising sewer incidence	15	10	6	12	43
Wastewater treatment final effluent compliance (%):					
Suspended solids	76	77	82	95	82
COD	70	67	86	90	78
NH_4-N	89	81	86	86	85
NO_3	99	99	99	99	99
Ortho phosphate	8	3	7	7	6
E. coli	54	58	51	58	55

*Source: CSRM Report (2004).

Table 23.3. Summary of sanitation backlog to informal settlement around the Cape Flats.*

			Toilet backlog (based on 3 levels of service)		
Date	Estimated number of dwelling units	Number of toilets	Full (1 toilet: 1 unit)	Rudimentary (1 toilet: 4 units)	Temporary (1 toilet: 5 units)
June 2003	98,410	4140	94,270	20,463	15,548
June 2004	110,000	7135	102,865	20,365	14,865

*Source: CSRM Report (2004)

5.4 *Land use and shallow groundwater quality*

Land use in the Cape Province is complex. Rapid population growth has overwhelmed the rate of water supply provision in the city area while basic services, particularly to the informal settlements, have resulted in poor sanitation and inadequate facilities for proper disposal of wastes. Description of the land use and water quality challenges have been compiled from the CSRM report (2003/2004), according to the catchment and management areas of the Cape Town Municipality (Table 23.4).

Table 23.4. Description of land use and water quality within the vicinity of the Cape Flats.

Catchment name (Area km^2)	Rivers in the catchment	Land use and water quality challenges
SOUTH		
Sand (85)	Sand, Keysers, Westlake	Land use: Primarily residential, also National Park, agriculture and light industrial Water quality: Generally acceptable
Zeekoe (96)	Big Lotus, Little Lotus	Land use: Horticultural, dense residential and light industrial Water quality: Generally unacceptable
Mitchells Plain (40)	Western Culvert, Eastern Culvert	Land use: Mostly dense residential with small commercial/industrial areas Water quality: Generally acceptable
Noordhoek (35)	Brookwood, Stream, Bokramspruit	Land use: Mostly sparse residential with small industrial areas Flooding: Negligible Water quality: Generally acceptable except Bokramspruit
Silvermine (23)	Silvermine	Land use: Mostly national Park, sparse residential with small commercial areas Water quality: Generally acceptable
Glencairn (10)		Land use: Mostly National Park with a sparse residential area Water quality: Generally acceptable
EAST		
Lourens (122)	Lourens, Geelsloot	Land use: Residential, agricultural and light industrial. Nature reserve in the upper catchment Water quality: Generally acceptable
Sir Lowry's Pass (51)	Sir Lowry's Pass	Land use: Residential, agricultural and natural Water quality: Generally unacceptable
Eerste/Kuils (668)	Kuils, Eerste, Moddergat	Land use: Mostly dense residential and small commercial/industrial areas Water quality: Generally unacceptable
Soet (16)	Soet	Land use: Mostly sparse residential Water quality: Unacceptable
CENTRAL		
Salt (214)	Liesbeek, Black, Kromboom, Bokmakierie, Blomvlei, Vygekaal, Jakkalsvlei, Elsieskraal, Black, Salt	Land use: Mixed land use includes less densely populated areas such as Welgemoed and Newlands as well as highly urbanised areas such as Athlone, Botheheuwel and Joe Slovo. Industrial areas include Epping, Ndabeni and Paarden Eiland. There is also Kirstenbosch Botanical Gardens and Tygerberg Nature Reserve

(*Continued*)

Table 23.4. (*Continued*)

		Water quality: River water quality is generally unacceptable apart from the upper reaches of the Liesbeek and Elsieskraal rivers
Hout Bay (37)	Disa, Baviaanskloof	Land use: Mostly low-density residential with horse paddocks. Also includes the high-density of Imizamu Yethu, a fishing harbour and the popular recreational bathing beach Water quality: Generally unacceptable in the lower reaches and estuary
City Bowl & Atlantic seabord	Platteklip, Silver, Camps bay, Kalsteelpoort	Land use: Include mountainous areas within the Cape Peninsula National park and various densities of residential development in the City Bowl, Camps Bay, Bakoven and Llanducho Water quality: Unacceptable along the Atlantic Seaboard at Green Point, Three Anchor Bay and Saunders Rock.

Areas of high- and low-density populations are juxtaposed and small-scale industrial as well as agricultural development is sometimes incorporated within residential areas. Animals are reared within the vicinity of residential areas, especially in Guguletu/Khayelitsha. Current disposal practices have increased the use of land to dispose of various forms of waste with no due consideration for the underlying geology.

Based on land use, the types of pollution that threaten the Cape Flats Aquifer are: (1) low- to medium-risk pollution sources, occupying large areas of the Cape Flats (Guguletu/Khayelitsha, Philippi farming areas) and (2) nodal sources (the Waste Water Treatment Works and the numerous waste disposal sites). However, in the vulnerability assessment of the Cape Flats, few areas of medium to high pollution risk have been delineated (Adelana & Xu, 2005). In addition to the point sources, various non-point or diffuse sources of pollution exists and have also been linked to land use. The major non-point pollution source identified is stormwater, as described in the previous section.

The chemical quality of groundwater in most parts of the Cape Flats Aquifer comprises fairly low salinity, as illustrated in Wright and Conrad (1995). The more saline waters generally occur in the periphery areas, for example the Pinelands and Parow areas. The Philippi area showed the highest salt concentration and this can be attributed to intense irrigation practices in the zone (Bertram, 1989). Calcium-bicarbonate was identified as the dominant type of groundwater in the Cape Flats and this is in agreement with the classification of Vandoolaeghe (1989), although a mixed-chloride type of water occurs in the eastern and northern central part of the area. The concentrations of SO_4 and HCO_3 are relatively higher. The high SO_4 content could be linked to the occurrence of peaty clay and peat lenses in parts of the Cape Flats. The reason for the high calcium bicarbonate level in water samples in this area is obviously the calcareous nature of the lithological units (except the Springfontein Member and the Elandsfontein Formation).

6 GROUNDWATER PROTECTION ZONING IN THE CAPE FLATS

The overall quality of groundwater in the Cape Flats is good enough to warrant its development for the water supply augmentation scheme of the City of Cape Town Municipality. However, the occurrence of pollution from present and previous studies, as highlighted in this paper, and the identification of point- and non-point sources call for groundwater protection in the Cape Flats. Since the point sources have been identified, protection zoning is proposed as the obvious solution for the full utilisation of the Cape Flats Aquifer. The establishment of site-specific protection zones, with the regulation of land use and the Resource Directed Measures (RDM) under the

South African Water Act of 1988 has been seen as a possible way forward in future groundwater management in South Africa. Therefore, the full involvement and usability of the RDM in defining protection zoning will be discussed here. In order to achieve the protection of the Cape Flats Aquifer, the RDM should be implemented as a whole (i.e. in terms of classification, resource quality objectives (RQOs) and reserve).

With a population of nearly 3 million in the Cape Town area, about 73 million litres will represent the groundwater component of the Reserve. The sustainable use of the Cape Flats Aquifer alone is estimated at 18 billion litres per annum (49.32 million L/d). This implies that more than two thirds of the basic water needs of the entire population in the Cape area can be met by the Cape Flats Aquifer. From the pilot abstraction scheme earlier conducted, a mean annual groundwater output from ten production boreholes was estimated at 5 billion litres (5,000 M L/a). This yield was maintained consistently for three years of pumping the boreholes in the period May 1985 to April 1988 under suboptimal operating conditions and favourable recharge conditions (Vandoolaeghe, 1989). Therefore, the reserve component of the aquifer would pass the test of sustainability and serve the augmentation scheme proposed for the municipal water supply.

The application of the classification system under the NWA provides guidelines for setting the appropriate levels of groundwater protection. Under this guideline, the Cape Flats Aquifer can be delineated into groundwater management units. The legal emphasis, therefore, is on protection for use and not necessarily the cost of use. The first stage in the protection process is the development of a system to classify the aquifer within the catchment. This classification system provides the framework for groundwater protection (Xu *et al.*, 2002). The procedures to satisfy the water quality requirements of water users relate to the protection of the aquifer from pollution. This is in accordance with the DWAF's programme on Aquifer Classification. Using the classification system prescribed under the NWA, therefore, the Cape Flats Aquifer (CFA) has been delineated to imply different levels of resource protection and impact acceptable to stakeholders, as follows:

(i) Protected
(ii) Good
(iii) Fair or
(iv) Severely Modified

The protected area of the CFA represents a zone in which, (a) anthropogenic activities have caused minimal or no changes to the hydrological characteristics or to the beds or banks and to the ecosystem; (b) chemical concentrations are not significantly different from the background concentration levels or ranges for naturally occurring substances; (c) concentration levels of artificial substances do not exceed the detection limits of advanced analytical methodologies.

The protected class has been used as a reference condition. Other classes are defined in terms of the degree of deviation from conditions of no or minimal impact. The area classified as 'Good' therefore will represent a protection level in which the aquifer is slightly to moderately altered from the protected reference conditions. The portions at the north-western and north-eastern ends of the Cape Flats, as well as the eastern portion of the pilot abstraction wellfield rightly belong in this class, based on vulnerability mapping (Adelana & Xu, 2005). The area classified as 'Fair' is one in which there is a significant degree of change from the reference conditions (Fig. 23.3). This has been designated to include the area around urban development, the location of the Swartklip Waste Disposal Site and the Mitchell's Plain WWTW with the greater portion around the sewage works. The locations of the Swartklip Waste Disposal Site and the Mitchell's Plain WWTW on areas of high transmissivity may have increased the possibility of polluting the aquifer.

Under resource classification, provision is made for some resources that may be worse than fair. In such cases the management class is set at 'Fair', i.e. in a condition that may be described as 'Poor' or 'Severely Modified'. On this level of protection management, efforts are required to rehabilitate the resource to an improved condition.

In the Cape Flats, this has been delineated at the south-east extreme corner towards Eerste River, a narrow area around the sewage works, the Swartklip Waste Disposal Site, the Mitchell's

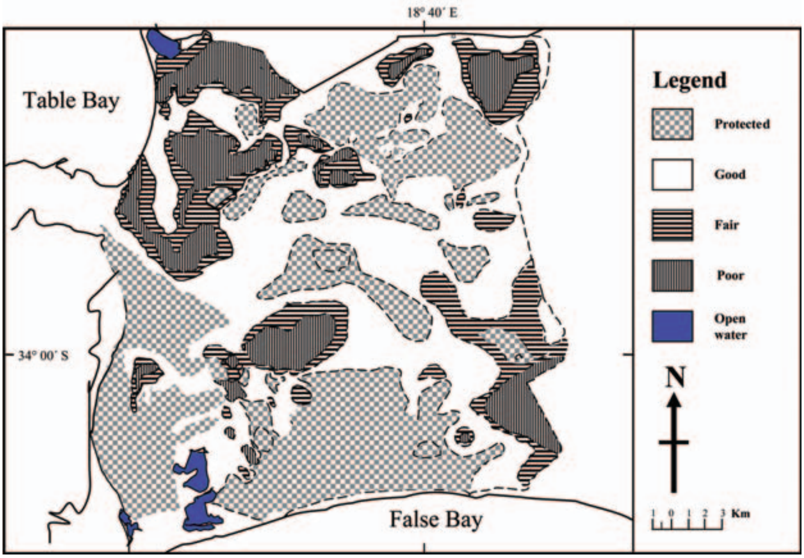

Figure 23.3. Cape Flats Aquifer protection zoning based on the groundwater quality and Resource Directed Measure concept.

Plain WWTW, and the north-western part of the Philippi agricultural area. It should be borne in mind that some of the areas under this classification are vulnerable to land-based activities that could be practically impossible to rehabilitate once polluted; hence the need for legislation. It is obvious then that the Cape Flats Aquifer needs to be protected as a whole.

Under the RDM concept, the area on the eastern side of Weltevreden Road (in the western part of Mitchell's Plain) and about 2 km north of False Bay, should be protected by legislation to restrict or regulate the release of high-priority contaminants into the ground or surface water. Those high-priority contaminants identified for the study area include 1,4-dioxane, diethyl ether, ethyl acetate (mostly associated with the solvent, cleaning, painting and textile industries) and inorganic compounds associated with a wide range of sources (Usher *et al.*, 2004).

For the good of both the drinking water supply and water environment, a multi-functional land-use change scheme could be introduced under the National Water Act to prevent further introduction of nitrogenous compounds into the subsurface. At a time of excessive production in the Philippi agricultural area, they offer a way forward towards generating the benefits of improved quality raw drinking water, human lifestyle and the water environment. This represents a more sustainable solution than allowing continued environmental degradation through excessive nutrient and pesticide leaching, and meeting drinking-water requirements through costly, energy-intensive treatment. The national government through the Water Act could acquire or allow the purchase of some hectare of land, in priority groundwater protection areas, for conversion to groundwater-friendly uses (e.g. 65% woodland and the remainder healthland and rough grassland). It should also promote farmer cooperative agreements (with payment of compensation for cultivation constraints in market gardening and arable cropping) in the high priority groundwater protection areas. At the province-wide level such agreements can be reached with the Cape farmers cultivating around the high priority areas to protect groundwater.

7 CONCLUSION AND RECOMMENDATIONS

While the groundwater quality of the Cape Flats Aquifer is good and can support the many different uses of this resource, aquifers across the Western Cape are showing measurable impacts from human activities. Pesticides, petroleum hydrocarbon compounds, volatile organic compounds, waste disposal sites and the Waste Water Treatment Works are making a considerable contribution to groundwater pollution. Improperly constructed and poorly maintained septic systems are believed to cause substantial and widespread nutrient and microbial contamination to groundwater.

Based on land use, the types of pollution that threaten the Cape Flats Aquifer are: (1) low- to medium-risk pollution sources, which occupy large areas of the Cape Flats (Guguletu/Khayelitsha, Philippi farming areas) and (2) nodal sources (the Waste Water Treatment Works and the numerous waste disposal sites). Physio-chemical analysis of groundwater in the study area revealed high levels of nitrates, chlorides, phosphates and, locally, fluorides. The factors responsible for groundwater pollution are identified and suitable preventive measures discussed. The provision of adequate sanitation to the numerous people living in informal settlements, who currently lack it, is prominent and fundamental to public health. In addition to the point sources, various non-point or diffuse sources of pollution exist.

In order to utilise the groundwater resource of the Cape Flats Aquifer, it is recommended that appropriate wellfield protection zones be delineated and implemented in line with the RDM. The application of the RDM concept to protect the aquifer would be a step in the right direction for water resource planning and the management of the Cape Flats Aquifer as a potential water supply option for the city of Cape Town. In this way the Cape Flats Aquifer can be abstracted continuously in the period of no rain (or low dam levels) to augment the city's water supply, alongside other primary aquifers, and allow to recharge in the raining months, when the dams are expected to be at their full capacity. The areas identified as polluted can be remediated and subsequently plunged into the supply scheme. The cost of remediation and treatment of groundwater to meet the drinking water quality standard is expected to pay off on the long term.

It is further recommended that in the augmentation schemes, all primary aquifers within the region be developed with other available options considered, so that they can be utilized for water supply, especially during periods of low dam levels, instead of imposing water restrictions. Water is basic to living and water has value equal to life itself. While wastage must be curtailed, restrictions cannot continue to be in place, but should be considered as very temporal, pending the full development and implementation of alternative sources.

REFERENCES

Adelana, S.M.A. & Xu, Y. 2005. Vulnerability assessment in the Cape Flats Aquifer, South Africa. *Proc. AVR05, 21–23 September.* Parma, Italy.

Ball, J. & Associates 2003. Coastal Park sanitary landfill. Water monitoring reports No. XXI submitted to the City of Cape Town.

Ball, J.M., Blight, G.E. & Vorster, K. 1995. Leachate pollution in seasonal water deficit area. *Proc. 5th International Landfill Symposium*, Cagliari, Sardinia.

Bertram, W.E. 1989. Geohidrologiese Opname in die Philippi-landbougebied Kaapse Vlakte. *Technical Report GH3595*, Directorate Geohydrology, Department of Water Affairs.

Catchment, Stormwater and River Management (CSRM) 2004. Annual Report for 2003/2004.

Department of Water Affairs and Forestry (DWAF). 1994. Minimum Requirements for the Establishment of a Waste Disposal Site, DWAF, Pretoria.

Engelbrecht, J.F.P. 1998. Groundwater pollution from cemeteries. *Proc. Biennial Conference, 4–7 May*. Cape Town: The Waste Institute of Southern Africa.

Foster, S., Hirata, R., Gomes, D., D'Elia, M. & Paris, M. 2002. Groudnwater quality protection: a guide for Water Utilities, Municipal Authorities and Environmental Agencies. World Bank Publication, Washington, D.C., USA.

Gerber, A. 1981. A digital model of groundwater flow in the Cape Flats. CSIR Contract Report C WAT 46, Pretoria.

Giljam, R. & Waldron, H. 2002. The effect of the Cape Flats aquifer on the water quality of the False Bay. *Proc. Western Cape Conference: Tales of a hidden treasure, September 16*. Sommerset West.

Grobicki, A. 2000. Integrated catchment management in an urban context of the Great and Little Lotus Rivers, Cape Town, WRC Report No. 864/1/01.

Hartnady, C.J.H. & Rogers, J. 1990. The scenery and geology of the Cape Peninsula. *Guidebook Geocongress '90,* Geological Society, South Africa.

McLear, L.G.A. 1995. A note on the potential of the Cape Flats aquifer unit to supply groundwater for domestic use in the Cape Town Metropolitan area. *Technical Report* No. GH 3868, DWAF, Cape Town.

Mehlomakulu, M. 2000. The influence of urban development on the water chemistry of the Cape Flats aquifer. Unpublished M.Sc. thesis, University of Cape Town, Cape Town.

Meerkotter, M. 2003. Heavy metals and vegetable farming in Cape Town. Unpublished M.Sc. Thesis, University of the Western Cape, 147p.

Parsons, R. & Taljard, M. 2000. Assessment of the impact of the Zandvlei Wastewater Treatment Works on groundwater. *Biennial Conference, 28 May–1 June*. Sun City: Water Institute of South Africa.

Saayman, I.C., Adams, S. & Harris, C. 2000. Examples of O- and H-isotopes to identify surface water pollution in groundwater. In Sililo *et al.* (eds), *Groundwater: Past Achievements and Future Challenges*: 599–603. Rotterdam: Balkema.

Saayman, I.C. 1999. Case study on the chemical characteristics of a pollution plume and a determination of its direction of movement at the Bellville Waste Site, Cape Town, South Africa. Unpublished M.Sc. Thesis, Royal Institute of Technology, (KTH), Stockholm.

Traut, M.J. & Stow, J.G. 2001. Groundwater quality monitoring: Coastal landfill Site. Report No.2, CMC Administration, Cape Town.

Traut, M.J. & Stow, J.G. 1999. Coastal Park landfill site groundwater quality monitoring – Report 1, Scientific Services Dept., Directorate of Water and Waste, Cape Town City Council.

Tredoux, G. 1984. The groundwater pollution hazard in the Cape Flats. J. Water Poll. Cont. Vol.83, No. 4, pp 473–483.

Usher, B.H., Pretorius, J.A., Dennis, I., Jovanovic, N., Clarke, S., Titus, R. & Xu, Y. 2004. Identification and prioritisation of groundwater contaminants and sources in South Africa's urban catchments. WRC Report No. 1326/1/04.

Vandoolaeghe, M.A.C. 1989. The Cape Flats groundwater development pilot abstraction scheme. Technical Report No. GH3655, Directorate Geohydrology, DWAF, Cape Town.

Weaver, J.M.C. & Tworeck, W. C. 1988. Hydrogeological investigation of leachate attenuation at the Bellville South Waste Disposal Site. Report Foundation for Research Development, CSIR.

Wright, W. & Conrad, J. 1995. The Cape Flats Aquifer: Current status. CSIR report 11/95, Stellenbosch.

Xu, Y., Colvin, C., van Tonder G.J., Hughes S., le Maitre D., Zhang J., Mafanya T. & Braune E. 2002. Towards the resource directed measures: groundwater component. WRC Report No. 1090-2/1/03.

24

An evaluation of groundwater pollution from pesticides in South Africa using the VLEACH model

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ABSTRACT: Modelling is an effective tool for investigating leaching potential and groundwater contamination from non-point source pollutants. The aim of this study was to evaluate the most important factors that determine the leaching of 10 pesticides using the VLEACH model. A sensitivity analysis was carried out by varying inputs of hydrogeological properties (groundwater depth and recharge rate), management (application rate and depth of pesticide incorporation into the soil), pesticide properties (organic carbon partitioning coefficient) and properties for two typical soils (organic carbon content, bulk density, porosity and volumetric water content) of the Western Cape, South Africa. The leaching rate peaks and time required to reach these peaks were simulated after once-off pesticide application to the soil surface at the beginning of the simulation. Based on the simulations with VLEACH, the major factors affecting pesticide leaching are depth to groundwater, recharge, application rates, pesticide properties, particularly the organic carbon-partitioning coefficient, and soil organic matter content. Although the VLEACH model does not simulate pesticide degradation, it could be used for a comparative assessment and preliminary studies of the fate of contaminants.

1 INTRODUCTION

Agricultural practices are becoming increasingly dependent on the widespread use of toxic pesticide chemicals for crop protection, growth regulation and seed treatment to enhance productivity (London and Myers, 1995a and b). The use of pesticides poses a serious threat to already limited water resources, as the amounts not taken up by plants are often washed away by run-off into surface waters or leached through the soil, causing groundwater pollution (Aharonson, 1987: in Weaver, 1993).

Modelling pesticide processes has proven to be an effective predictive tool for the management of agrichemical usage and is often associated internationally with screening new pesticides for registration (Boesten, 1999; Vanclooster *et al.*, 2000). In order to model the behaviour and fate of pesticides, it is necessary to identify a multitude of processes. Processes can be broadly grouped into transfer (mobility) or transformation (degradation). Transfer can occur through run-off (Gallagher *et al.*, 2001), volatilisation (Rüdel, 1997), adsorption/desorption (or simply sorption) (Hesterberg, 1998), plant uptake (Hantush *et al.*, 2000) and soil water fluxes (leaching, throughflow and preferential flow) (Muller *et al.*, 2003; Renaud *et al.*, 2004). Transformation includes chemical, microbial and photo-degradation (Martins and Mermoud, 1998; Burrows *et al.*, 2002). In general, the processes are highly sensitive to pesticide properties (volatilisation, sorption and half-life), site conditions (weather, soil, geohydrology etc.) and management practices (pesticide application, irrigation, soil tillage etc.).

A large number of models have been developed in order to predict pesticide behaviour and fate in the environment. Boesten (1999) and Vanclooster *et al.* (2000) provide a comprehensive review of pesticide models worldwide. The applications of models are numerous. In particular, they can be used to quantify the hazard of pesticide leaching to groundwater under specific hydrogeological conditions.

In previous work, the identification of priority pesticides in South Africa was based on their detection in surface and groundwaters as well as usage (Maharaj, 2005). The aim of this study was to evaluate the leaching potential of these priority pesticides and to determine the most relevant factors for modelling. The approach included a sensitivity analysis of inputs to the VLEACH model in order to determine the most critical hydrogeological factors, soil and pesticide properties in terms of potential pesticide leaching and groundwater contamination.

2 MATERIAL AND METHODS

Groundwater pollution from pesticides was evaluated by simulating different scenarios using the VLEACH model (Ravi and Johnson, 1997). It should be noted that the scenarios in this study are not necessarily true and that the analysis is predominantly a conceptual one.

VLEACH simulates the instantaneous equilibration of an organic contaminant between the aqueous, gaseous and solid phases of the vadose zone by means of advection, diffusion and adsorption. The model uses linear sorption isotherms and Henry's Law constants for volatilisation defined by the user to describe the partitioning mechanisms between phases. VLEACH simulates one-dimensional vertical transport by advection in the liquid phase and by gaseous diffusion in the vapor phase, based on concentration gradients. Contaminant mobility is simulated within homogeneous user-defined polygons that are vertically divided into a series of user-defined cells representing soil layers. During each user-defined time step, the transport of the contaminant between cells is calculated. The total mass in each cell and instantaneous equilibration between the different phases are then recalculated. Soil moisture conditions are assumed to be uniform within a polygon over the simulation period, whilst the contaminant concentration may vary between layered cells. These steady-state conditions are unlikely to occur in the field; however, estimates can be made by repeating simulations for a range of likely soil/hydrological conditions. Also, the fact that the model does not account for solute dispersion and degradation can lead to conservative estimates of contamination. At the end of the model simulation, the results from different polygons can be used to determine an overall area-weighted groundwater impact for larger areas.

Simulations with VLEACH were run for ten pesticides commonly used or detected in South African waters, namely 2,4-D, atrazine, simazine, carbofuran, carbaryl, dieldrin, lindane, dimethoate, fenthion and parathion (Maharaj, 2005), and for two soils with different properties commonly found in the Western Cape (South Africa). The pesticide properties are shown in Table 24.1, whilst the soil properties are presented in Table 24.2. The organic carbon partitioning coefficient (Koc) was calculated as a function of soil properties using the empirical equations determined by Weber *et al.* (2004), whilst water solubility and Henry's Law constants for volatilisation were obtained from international databases. Initial concentrations in the top soil match typical applications recommended by manufacturers (Table 24.1). The model was assessed for variations in pesticide leaching by varying inputs of hydrogeological properties (groundwater depth and recharge rate), management (application rate and depth of pesticide incorporation into the soil), pesticide properties (Koc) and soil properties (organic carbon content, bulk density, porosity and volumetric water content).

Baseline data included a water table depth of 6 m, a yearly recharge of 350 mm, recommended application rates as reported by manufacturers, incorporation of pesticides in the top 3 cm of soil, volumetric soil water content of 20% and a polygon area of 1 m^2. Simulations were run for once-off pesticide applications at the beginning of the simulation. The time step of the simulation was one year. Groundwater impact was expressed as either the leaching/contamination rate in g m^{-2} a^{-1} or cumulative mass that enters groundwater in g m^{-2}.

Table 24.1. Pesticide properties used as input data in the simulations with the VLEACH model.

Pesticide	Koc (mL g^{-1}) Soil A (Table 24.2)	Koc (mL g^{-1}) Soil B (Table 24.2)	Water solubility (mg L^{-1})[1]	Henry's Law constant[3]	Initial concentration in top soil (μg active ingredient kg^{-1})
2,4-D (2,4-DDD)	43	90	900	7.35E–11	6154
Atrazine	0	0	28	1.20E–07	5026
Simazine	40	9	5	3.96E–08	15,808
Carbofuran	21	10	320	2.10E–08	15,077
Carbaryl (mean)	155	360	40	5.30E–06[4]	9949
Dieldrin	12,919	22,102	193[2]	2.65E–05	5744
Lindane	669	0	7.3	7.42E–05	1699
Dimethoate	26	40	25,000	5.61E–10	1423
Fenthion	1429	448	2	1.00E–04[4]	2564
Parathion	1413	683	12.4	4.98E–06	0.615

[1]Extension Toxicology Network (EXTOXNET), Pesticide Information Profiles: www.ace.orst.edu/info/extoxnet/pips.htm
[2]Ingrid Dennis, Institute for Groundwater Studies, Bloemfontein, personal communication
[3]Illinois General Assembly: www.legis.state.il.us/commission/jcar/admincode/008/00800259ZZ9997dR.html
[4]International Programme on Chemical Safety, 1994: www.inchem.org/documents/ehc/ehc/ehc153.htm

Table 24.2. Summary of soil properties used in the simulations with the VLEACH model.

Soil properties	Soil A	Soil B
pH (H_2O)	7.2	7.5
% organic carbon	1.05	0.45
% sand	92.8	82.8
% silt	6.0	15.6
% clay	1.2	1.6
Dry bulk density (g cm^{-3})	1.62	1.42
Porosity (ratio)	0.39	0.42

3 SIMULATION RESULTS

For all pesticides and both soils, groundwater contamination was found to be inversely related to depth to groundwater (2 to 10 m), as this relates to the travel time of pesticides through the unsaturated zone. Leaching occurs sooner and at a higher rate for shallower groundwater depths compared to deeper groundwater tables. This is illustrated in Figures 24.1 and 24.2, using simazine and soil A as an example. The bulk of simazine (peak leaching rate) is expected to reach a groundwater table at 2 m after ~5 years, whilst breakthrough into a 10 m groundwater table occurs only after ~25 years (Fig. 24.1). The peak leaching ranged from 0.012 g m^{-1} a^{-1} for a 10 m groundwater depth up to 0.026 g m^{-1} a^{-1} for a 2 m groundwater depth (Fig. 24.1). All simazine applied to the soil surface at the beginning of the simulation will take between 20 (2 m groundwater depth) and 50 years (10 m groundwater depth) to completely leach from the unsaturated zone (Fig. 24.2). These results indicate that leaching will occur at steadier and lower rates, but over longer time periods for deep groundwaters compared to shallow ones.

Simulations of varying yearly recharge rates indicated that a larger recharge caused shorter leaching time with higher peak leaching rates compared to a lower recharge. No leaching was predicted in the absence of recharge. The leaching rate was also sensitive to initial contaminant concentrations in the top soil or application rate. Higher peaks were predicted for higher initial concentrations.

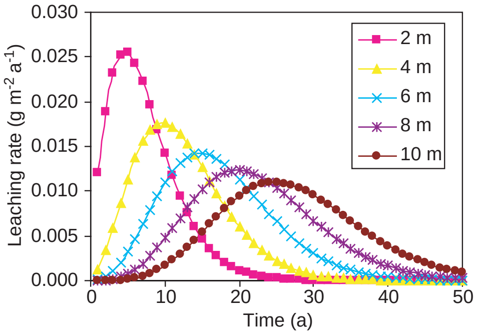

Figure 24.1. Leaching rate of simazine in soil A for varying groundwater depths.

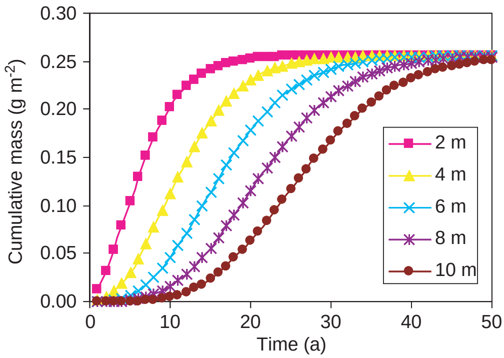

Figure 24.2. Cumulative leaching of simazine in soil A for varying groundwater depths.

The time of leaching was similar for different initial concentrations, if all the other variables were constant. The leaching rate also depends on pesticide properties. Simulations showed that, for example, leaching takes longer and occurs at lower rates for higher organic carbon partitioning coefficients, due to the retention of pesticide by soil organic matter. Slower and longer leaching was also predicted for higher soil organic matter contents compared to lower values.

Besides soil organic matter content, other soil properties play a role in pesticide leaching, but to a lesser extent, like bulk density and volumetric water content, which is assumed to be constant over time in VLEACH. Higher peaks and longer leaching were predicted for higher bulk densities, due to lower water retention in these soils. Simulations also showed that leaching takes place at a higher rate and displays a sharper breakthrough at higher volumetric soil water content values, due to a higher drainage and dilution capacity compared to that of drier soils. The VLEACH model output was almost insensitive to change in depth of soil incorporation of the pesticide and soil porosity.

Table 24.3 represents the peaks in leaching rates and the time taken to reach the peak for 10 pesticides applied to the top soil at the beginning of the simulation, using baseline input data. Both the peak leaching rate and the time required to reach the peak depended on the pesticide properties and application rates, given that the other conditions were constant. Atrazine recorded the highest peak in leaching rate (1.0216 g m^{-2} a^{-1}) and the breakthrough occurred in the shortest time (3.8 years) for both soils, as the Koc sorption coefficient calculated with the equation of Weber *et al.*

Table 24.3. Simulated peaks in the rate of leaching and time required to reach the peak for soils A and B, and half-lives for ten priority pesticides in South Africa.

Pesticide	Soil	Peak in leaching rate ($g\ m^{-2}\ a^{-1}$)	Time to reach peak (a)	Half-life (d)
2,4-D	A	0.0163	15–16	10[1]
	B	0.0177	13	
Atrazine	A	1.0216	3.8	60[1]
	B	1.0216	3.8	
Simazine	A	0.0142	15	60[1]
	B	0.0665	5	
Carbofuran	A	0.0736	10	50[1]
	B	0.194	6	
Carbaryl	A	0.00434	32–36	38[2]
	B	0.00433	32	
Dieldrin	A	0.0000274	600	1000[1]
	B	0.0000394	600	
Lindane	A	0.000178	116–144	690[3]
	B	0.0966	3.8	
Dimethoate	A	0.00583	11	7[1]
	B	0.00791	8	
Fenthion	A	0.000127	252–280	34[1]
	B	0.000906	36	
Parathion	A	0.0000000307	240–288	14[1]
	B	0.000000145	52–56	

[1]Oregon State University Extension. Pesticide Properties Database, 1994: npic.orst.edu/ppdmove.htm
[2]US EPA, National Pesticide Information Centre: www.epa.gov/pesticides/factsheets/npic.htm
[3]US EPA, Office of Prevention of Pesticides and Toxic Substances, Re-registration Eligibility Decision for Lindane, Case 315: www.epa.gov/oppsrrd1/REDs/lindane_red.pdf

(2004) was 0 (Table 24.1). The same conclusion was drawn for lindane in soil B. Dieldrin, on the other hand, showed relatively low peaks in leaching rate and by far the longest time of leaching of all pesticides, due to its highly sorptive characteristics (Table 24.1). Parathion displayed the lowest peak in leaching due to the lowest initial pesticide concentration in the top soil or application rate (Table 24.1).

In most cases, the leaching rate was higher and faster for soil B compared to soil A, due to the higher organic matter content in the later, which caused higher organic sorption. For comparative purposes, the half-life values found in the literature are reported in Table 24.3. VLEACH does not simulate the degradation of chemicals; however, the half-life values in the literature were generally much lower than the simulated time of breakthrough.

In order to assess the reliability of model predictions, measured data are required for comparison with simulated values. Most pesticide data in the literature are aimed at the detection of pesticides (e.g. Weaver, 1993) and multi-residue analysis, rather than long-term monitoring. Other studies are aimed at understanding the transport and transformation processes of pesticides (Meinhardt, 2003). Long-term studies indicated that persistence or accumulation of pesticide residues may occur in soils, resulting in very slow leaching to groundwater. For example, different studies showed that atrazine (and its metabolites) was detected in soils more than 4 months, 1 year and 8 years after application (http://www.speclab.com/compound/c1912249.htm), depending on application rates, environmental and biological factors that determine mobility and decay. Levy and Chesters (1995) used calibrated transport and degradation parameters to simulate long-term steady-state leaching of atrazine in a sandy-till aquifer overlying Cambrian sandstone. They concluded that steady-state concentrations are reached within 20 years and that it may take more than a decade to halve atrazine concentrations in groundwater after application cessation, due to slow degradation rates. Garmouma *et al.* (1997) found higher groundwater concentrations of atrazine and simazine in a study carried out from 1991 to 1993 compared to 1977, despite a reduction in maize cultivated areas during the period in the Melarchez agricultural basin (France). Dieldrin was found to persist in soils for >7 years, mainly due to its strong adsorption properties (Table 24.1) (http://www.speclab.com/compound/c60571.htm). Slow leaching of parathion was also documented at high levels of application, or after spills, where residues in the top soil were found after 6 and 16 years (http://www.speclab.com/compound/c56382.htm).

4 CONCLUSIONS

An evaluation of pesticide leaching and groundwater contamination was carried out using a semi-steady-state model that assumes a constant soil-water content. Sensitivity analyses to model inputs were carried out for ten priority pesticides in South Africa and for two soils typical of the Western Cape (South Africa).

Based on the simulations with VLEACH, the major factors affecting pesticide leaching and groundwater contamination are:

- depth to groundwater, recharge
- application rates
- pesticide properties, particularly the organic carbon partitioning coefficient
- soil organic matter content.

Soil bulk density and water content affect leaching to a lesser extent. The depth of soil incorporation of pesticides and soil porosity do not affect leaching rate or pesticide breakthrough. The VLEACH simulations also indicated that there is a good chance that, under specific conditions, the pesticides will be degraded in their path to groundwater. However, half-life values found in the literature were obtained under different conditions from those that were simulated, and these results should be interpreted with caution. Although VLEACH does not simulate the degradation of chemicals, the model gives a good indication of the role of sorption and it is a suitable tool for use in preliminary and theoretical studies.

It should be noted that the simulation results presented in this study depend greatly on the specific hydrogeological conditions and the specific pesticide properties. Different results would be obtained with different input data sets for different environmental conditions and pesticide species. There is a need for long-term monitoring data in order to test steady-state models like VLEACH.

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REFERENCES

Aharonson N 1987. Potential contamination of groundwater by pesticides. International Union of Pure and Applied chemistry. *Pure & Appl. Chem.* 59 1419–1446.

Boesten JJTI 1999. From laboratory to field: uses and limitations of pesticide behaviour models for the soil/plant system. *Weed Res.* 40 123–138.

Burrows HD, Canle ML, Santaballa JA and Steenken S 2002. Reaction pathways and mechanisms of photodegradation of pesticides. *Journal of Photochemistry and Photobiology B: Biology* 67(2) 71–108.

Gallagher DL, Johnston KM and Dietrich AM 2001. Fate and Transport of Copper-based Crop Protectants in Plasticulture Runoff and the Impact of Sedimentation as a Best Management Practice. *Water Research* 35(12) 2984–2994.

Garmouma M, Blanchard M, Chesterikoff A, Ansart P and Chevreuil M 1997. Seasonal transport of herbicides (triazines and phenylureas) in a small stream draining an agricultural basin: Melarchez (France). *Water Research* 31(6) 1489–1503.

Hantush MM, Marino MA and Islam M.R 2000. Models for leaching of pesticides in soils and groundwater. *Journal of Hydrology* 227(1–4) 66–83.

Hesterberg D 1998. Biogeochemical cycles and processes leading to changes in mobility of chemicals in soils. *Agriculture, Ecosystems & Environment* 67(2–3) 121–133.

Levy J and Chesters G 1995. Simulation of atrazine and metabolite transport and fate in a sandy-till aquifer. *Journal of Contaminant Hydrology* 20(1–2) 67–88.

London L and Myers J 1995a. General patterns of agrichemical usage in the southern region of South Africa. *South African J. Sci.* 91 509–514.

London L and Myers J 1995b. Agrichemical usage patterns and workplace exposure in the major farming sectors in the southern region of South Africa. *South African J. Sci.* 91 515–522.

Maharaj S 2005. Modelling the fate and behaviour of priority pesticides in South Africa. MSc thesis, Department of Earth Sciences, University of the Western Cape, South Africa.

Martins JM and Mermoud A 1998. Sorption and degradation of four nitroaromatic herbicides in mono and multi-solute saturated/unsaturated soil batch systems. *Journal of Contaminant Hydrology* 33(1–2) 187–210.

Meinhardt HR 2003. Evaluation of predictive models for pesticide behaviour in South-African soils. Water Research Commission Rep. No. 999/1/03, Pretoria, South Africa.

Muller K, Deurer M, Hartmann H, Bach M, Spiteller M and Frede HG 2003. Hydrological characterisation of pesticide loads using hydrograph separation at different scales in a German catchment. *J. Hydrol.* 273(1–4) 1–17.

Ravi V and Johnson JA 1997. Vlech: A one-dimensional finite difference vadose zone leaching model, version 2.2. User manual. Developed for the United States Environmental Protection Agency Office of Research and Development.

Renaud FG, Brown CD, Fryer CJ and Walker A 2004. A lysimeter experiment to investigate temporal changes in the availability of pesticide residues for leaching. *Environmental Pollution* 131(1) 81–91.

Ridel H 1997. Volatilisation of pesticides from soil and plant surfaces. *Chemosphere* 35(1–2) 143–152.

Vanclooster M, Boesten JJTI, Trevisan M, Brown CD, Capri E, Eklo Om, Gottesburen B, Gouy V and Van Der Linder A.M.A 2000. A European test of pesticide-leaching models: methodology and major recommendations. *Agric. Water Manage*. 44 1–19.

Weaver JMC 1993. A preliminary survey of pesticide levels in groundwater from a selected area of intensive agriculture in the Western Cape. Water Research Commission Rep. No. 268/1/93, Pretoria, South Africa.

Special topics

Editorial Comment: Special topics

The special issues in this section include groundwater quality problems associated with both anthropogenic and natural sources. There are a multitude of well-documented potential sources of groundwater pollution and this section illustrates that these also exist in the African continent.

The anthropogenic sources include discussions on pollution caused by sewerage and petrol station leakage and acid mine drainage. While these are largely point sources, the prevalence of these problems across Africa merits discussion. The biofouling paper from Bishop, highlights a common problem of natural impacts on groundwater quality and supply. These papers describe the processes in more detail and give insight into a small subset of potential groundwater issues of importance.

Several other natural causes of groundwater quality degradation have not been reported on in this section, although they are still very much problematic. These include elevated fluoride concentrations in groundwater, e.g., in region of Great Rift Valley, in the Northwest Province in South Africa and in southern Lesotho, naturally occurring radioactivity and seawater intrusion, which was alluded to in the preceding section of the book.

Not all the problems can be immediately revolved but understanding of their mechanisms should be promoted. The selected cases here are based on first hand information from the continent. Results presented give the reader a sense of the complex factors influencing groundwater quality problems in general.

It is important to note that this book is not comprehensive in terms of all the types of groundwater pollution. In urban areas, Africa should take cognisance of the priority pollutants and sources from North America and Europe. Potential sources should be identified, and proactive measure implemented to prevent contamination of the groundwater resources by toxic, carcinogenic and recalcitrant chemicals. It is with a note of caution that the absence of case studies on non-aqueous phase liquids (NAPLs) and groundwater remediation in this section is highlighted, with a view to pointing out the challenges ahead and the catastrophes Africa should avoid.

The value of groundwater in Africa cannot be overemphasized, and raising awareness to communities and decision makers regarding pollution sources, particularly those that may lead to more persistent groundwater quality degradation, should form an integral part of the landuse planning and regulation in this continent.

25

Contamination of the Abidjan Aquifer by sewage: An assessment of extent and strategies for protection

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ABSTRACT: In response to the problems of groundwater management in urban environments, UNESCO with the United Nations for the Environment Program (UNEP) jointly initiated the project entitled 'Assessment of pollution status and vulnerability of water supply aquifers of African cities'. This project consists of an assessment of the pollution status of the urban groundwater and proposing solutions for better aquifer protection. The methodology includes monitoring groundwater in terms of chemical and bacteriological quality through the preliminary determination of sites. The results show that the groundwater of Abidjan is affected by the progression of pollution in the urban groundwater from the south towards north, with a new trend of progression towards the west. This led aquifer management authorities to close certain water supply schemes, thus reducing the exploitation capacity of the aquifer. As strategies for protection, public awareness is raised by the periodic publication of bulletins on aquifer water-supply quality and workshops in collaboration with certain NGO's. The aim is to sensitise the authorities and the population on the dangers threatening groundwater in the medium and long term and to lead them to adopt strategies for better aquifer protection. In the same way a vulnerability map of the aquifer is established with the goal to better manage the land. As a solution, the creation of an integrated institutional framework for water quality management is proposed in order to establish lawful measurements in the various fields of water resource management.

Key words: integrated management, urban, groundwater, assessment, pollution

1 INTRODUCTION

The water crisis is primarily one of governance. Water should not be managed in a sectarian way with structures of decision that causes redundant actions and/or internal opposition. Water resources are consequently badly managed. This is the case in the Côte d'Ivoire, where the resources in general and the Abidjan Aquifer in particular are managed in a sectarian way and without joint actions and a single policy of resource management. The decision-making centres are multiple and most of the time antagonistic. These factors pose management problems for the Abidjan Aquifer, with the consequence of deteriorating the quantity and the quality of the resource. It is to face one of these problems that UNESCO jointly with UNEP initiated the project, specifically focussing on urban pollution of superficial aquifers in Africa. The goal is to better apprehend the state of groundwater pollution in large African cities and propose options for a better safeguard of surface and underground in the urban environments of Africa. We present here the results obtained from weekly monitoring of the water supply aquifer quality of Abidjan. This document is articulated in three parts. In the first part, we present the management problems of the Abidjan Aquifer. In the second part, consequences of bad governance and management of the aquifer are addressed, and we conclude by presenting recommendations for the protection of the Abidjan Aquifer.

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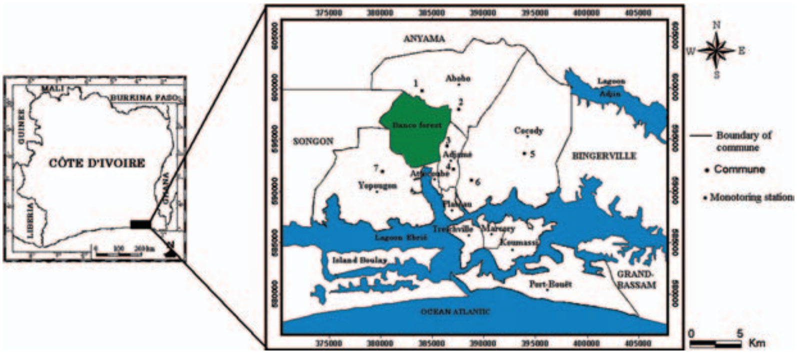

Figure 25.1. Location of the study zone.

2 PRESENTATION OF THE STUDY FRAMEWORK

Abidjan is located at the south of the Côte d'Ivoire and lies between the latitudes 5°00′ and 5°30′ north and longitudes 3°50′ and 4°10′ west (Fig. 25.1). It is the economic capital of the Côte d'Ivoire and extends 57.735 ha on surface. It is made up of ten communes with a population estimated at 2,877,948 (INS, 2001). On the geological level, the town of Abidjan belongs to the coastal sedimentary basin of Cretaceous to Quaternary age and it presents important groundwater potential. These groundwater resources consist of three aquifer levels of unequal importance (Quaternary, Continental Terminal and Maastrichtian). Only the aquifer of the Continental Terminal, commonly called 'Aquifer of Abidjan' is exploited for the drinking water supply (Aghui and Biémi, 1984). The Aquifer of Abidjan is unconfined and essentially constituted of sand (fine, medium and coarse).

The thickness ranges from 20 m in the north and 160 m to the south of the aquifer (SOGREAH, 1996). On the hydroclimatic plan, the zone of study belongs to an equatorial climate of transition, with two rainy and two dry seasons. Abidjan has a high annual rainfall (1600 mm of rain per year), which supports the recharge of the Aquifer of Abidjan.

3 METHODOLOGY

In response to the serious deterioration of the aquifer of Abidjan, the STEE laboratory, in collaboration with NGO H_2O, carried out a survey to identify the threats to the aquifer in the communes of Cocody, Abobo, Adjamé and Yopougon. A digital camera was used as a visual aid in the investigation and a documentary film was produced to raise public awareness. Within the framework of water quality monitoring, a series of *in-situ* measurements of physicochemical parameters such as conductivity, pH and temperature were carried out on selected sites (boreholes) every two weeks in collaboration with the SODECI (Society of Distribution of water in Côte d'Ivoire). Chemical analyses (nitrates, NO_3-N) and the bacteriology of borehole water were investigated by a certified laboratory twice per year (wet season and dry season). The test sample selection was made for various sites in collaboration with SODECI.

4 RESULTS

4.1 *Major threats to the urban aquifer*

4.1.1 *Absence of an institutional framework*

In the Côte d'Ivoire there is no institutional framework governing water resources. Resources are managed by a multitude of institutions. The organisation is partitioned in several ministries, companies of state,

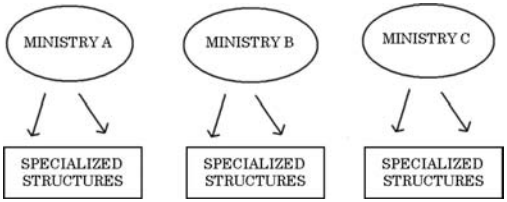

Figure 25.2. Partitioned organisational model of the water resources management in the Côte d'Ivoire.

Figure 25.3. Exploitation borehole encircled by dwellings in a hollow (Picture 1) and construction of dwellings beside a borehole of exploitation of the Abidjan Aquifer (Picture 2).

private organisations, universities and institutes of research, which are responsible for various aspects of the resources. All these structures work without any systematic coordination (Fig. 25.2). There is no joint action in the management of the water supply aquifer of Abidjan. It is under the supervision of the direction of the water resources that depend on the ministry for water and forestry and the direction of human hydraulics, depending on the ministry for economic infrastructures. This fact poses the problem of coordination in the actions of the various entities, as each structure is unaware of the others or their functions. There is no synergy between the structures. Information is disseminated and unknown to users of the resource. However, within the framework of integrated management of water resources, there is a willingness on the part of authorities to change this state of affairs.

According to Figure 25.2, Ministry A is for example charged with managing water in the Côte d'Ivoire, as are Ministries B and C. Thus, water is managed by several ministries, but there is no collaboration between them. Reforms have been undertaken since 1999 to reinforce the existing institutional framework. These reforms have resulted in the installation of an authority charged with water, which is responsible for the implementation of a national policy for water, supervised by the Ministry for Water and Forestry.

4.1.2 *Lack of respect for public property*

The town of Abidjan, as a harbour and coastal city, receives a great number of immigrants from other areas, searching for a better life. Since September 2002, as a result of the war, the city has received a flood of refugees from the zones of conflict. The consequence of this is the proliferation of precarious habitats at locations not recommended for living, such as hollows and the zones such as areas of public utility. This lack of respect for public property is manifested in anarchistic constructions that weaken the groundwater and infrastructures of drinking water supply to the town of Abidjan. The areas of groundwater collection (borehole protection zones) are invaded by urbanisation (Fig. 25.3) (Kouadio *et al.*, 1998).

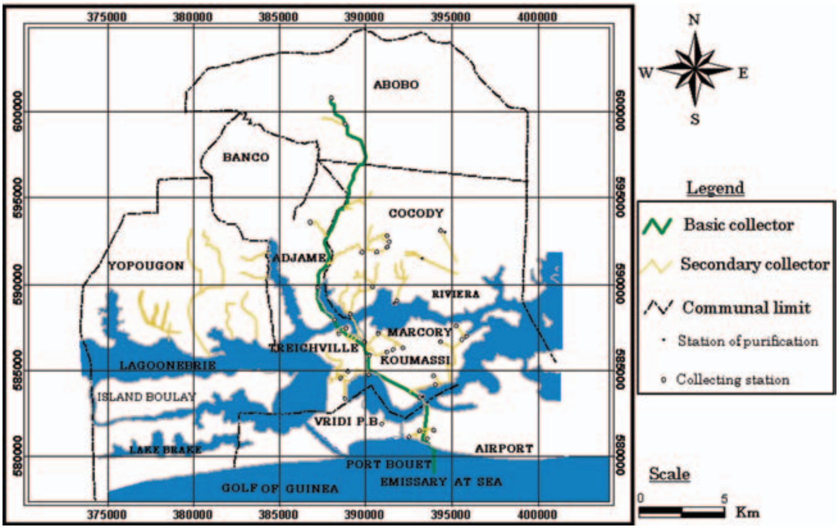

Figure 25.4. Collecting system of wastewater of Abidjan.

4.1.3 *Embryonic network of sanitation*

The problem of Abidjan is the virtual non-existence of a developed network of systems for collecting wastewater. A basic collection system (Fig. 25.4) crosses the city from north to south. Only a few secondary collectors are connected with the main system.

The consequence of this is that even districts equipped with adequate systems of sanitation (Plateaux, Cocody, Deux-Plateaux), the sewage emerges in valleys and are evacuated in the lagoon with the rains.

Some containments of sewage are open septic tanks, constituting a serious threat to the water supply aquifer. In some communes (Abobo, Koumassi, Marcory-Anoumabo, Biafrais Treichville, Attécoubé) the urban effluent sanitation system is practically non-existent (Fig. 25.5). The consequence is the adoption of unhealthy practices constituting serious threats to the quality of groundwater. In the municipality of Abobo, cesspools are drained and solid residue buried in holes that are closed after filling. In other communes, the storm basins and hollows are used as effluent disposal areas. There is not yet a clear policy of management for solid waste. A large city such as Abidjan produces an enormous amount of waste that requires an adequate framework for treatment. However, today the town of Abidjan does not yet have a controlled system of discharge. The refuse is poured into an aging and inadequate waste disposal system located at the groundwater surface. This discharge is not a risk for the Abidjan Aquifer. The above-mentioned refuse problems lead to the proliferation of water-borne diseases in Abidjan communes. Thus, the data on the water-borne diseases listed in the medical districts of Abidjan reveal that cholera is present in almost all the communes (Table 25.1). This was observed specifically in the communes of western Abidjan.

This part of Abidjan contains the communes of Yopougon and Songon. The rate of cholera cases suspected in 2001 is 838 for the only Yopougon Commune, with 15 deaths.

4.1.4 *Growth of the water demand and decrease of recharge in the climatic context of variability*

The growth demographic (3.7 per cent between 1988 and 1998 for Abidjan) and an increase in agricultural and industrial practices of this area have resulted in an increasingly demand for water from the Abidjan Aquifer (Fig. 25.6). Measurements available between 1977 and 1992 show a

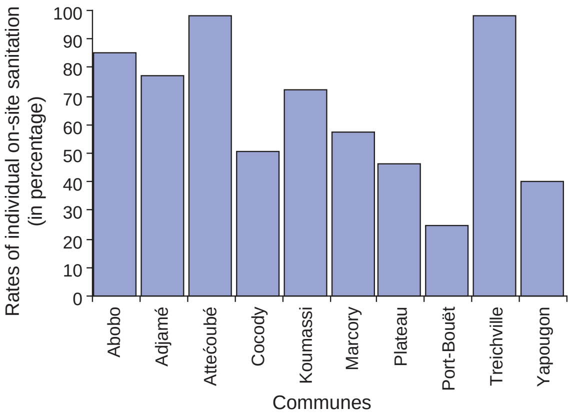

Figure 25.5. Rate of individual on-site sanitation in the various municipalities of Abidjan.

Table 25.1. Distribution of the suspect cases of cholera from 2001 to 2003 in the communes of Abidjan.

	Years					
	2001		2002		2003	
Communes	Cases	Death	Cases	Death	Cases	Death
Centre Abidjan (Plateau, Adjamé, Attécoubé)	421	9	230	0	29	0
East Abidjan (Cocody, Bingerville)	116	2	53	4	7	0
North Abidjan (Abobo, Anyama)	762	9	690	14	18	0
Western Abidjan (Yopougon, Songon)	838	15	1858	53	258	8
Southern Abidjan (Treichville, Koumassi, Marcory, Port-Bouët)	802	11	640	10	43	1
Abidjan Total	2938	46	3471	81	355	9

Source: Epidemiologic security service of INHP.

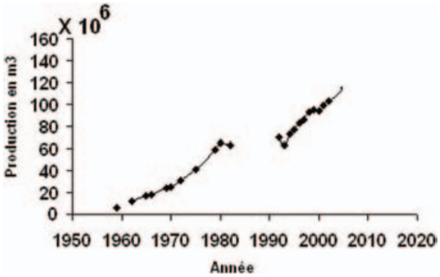

Figure 25.6. Evolution of annual volumes of water taken in the urban aquifer of Abidjan from 1959 to 2003.

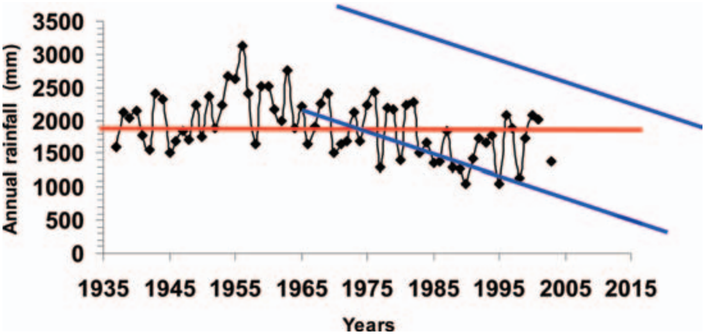

Figure 25.7. Variation of the annual rainfall from1937 to 2003 at the Abidjan airport station.

generalised fall in the level of the aquifer from 1 to 7 m with exceptional values exceeding 20 m (Jourda, 1987).

This has resulted in a decrease of exploitation observed for a few years in Abidjan and its surroundings. Work on the hydroclimatology of West Africa indicates a lower precipitation for the last few decades, with a rupture marked at the beginning of the 1970s, as well for precipitations as for the flows of the rivers (Paturel *et al.*, 1995; Servat *et al.*, 1997; Kafando and Achy, 1998; Aké; 2001 in Oga, 2004). In Côte d'Ivoire, climatic measurements from the Abidjan airport weather station situated along the coast from 1937 to 2003 confirm this regional tendency (Fig. 25.7). We observe on this figure a decrease in rainfall, marked by blue lines, while the water demand of the population increases each year. Annual precipitation dropped from 3129 mm in 1956 to 1374 mm in 2003, which is a fall of 44 per cent. With the rapid expansion of urban areas the construction of new residences is causing an increase in run-off and subsequently a decrease in recharge to the Abidjan aquifer. The reduction of the surface recharge to the aquifer poses a risk for the sustainability of the groundwater resource.

The governance system is inadequate, as informal settlements are created with little regard for regulation. Consequently, concrete appears to be found everywhere. The aquifer is weakened by both climatic variability and significant anthropogenic activity, which lead to a spectacular degradation of the groundwater resources in quantity and quality as will be seen in the following paragraph. It is thus urgent to set up a policy of monitoring for the groundwater through a network of suitable piezometers.

The piezometers measured constantly during 2001 reveal that the amount of piezometers decreased. The static levels are shallower in the south and significant in the north of the Abidjan Aquifer (Table 25.2).

5 ASSESSMENT OF POLLUTION STATUS

Since 2001, within the framework of the project financed by the UNESCO/UNEP, a study has been undertaken aiming to assess the state of the pollution of the Abidjan urban aquifer and at setting up an observatory of this groundwater by monitoring the physico-chemical and chemical level. The results can be seen in Figure 25.8. The analysis of the curves of variation of conductivity indicates three groups (Jourda *et al.*, 2004b):

- the first group consisted of drillings in the northern Zone 12 (ZN12), Anonkoua Kouté 6 (AK 6), eastern Zone 1 (ZE1) and Riviera Centre 6 (RC 6);
- the second group is formed by drilling the western Zone 6 (ZO 6);
- the third group is formed by drilling Adjamé 7 (AD 7) and Adjamé 2 (AD2).

Table 25.2. Measurements of the water table of the Abidjan Aquifer in 2001.

Piezometers	Water table (m)
East Zone P2 (Cocody)	51.29
North Riviera 4 (Cocody)	40.52
Centre Riviera 5 (Cocody)	71.73
Adjamé F1	58
Adjamé PZ9	41.4
IFAN (Plateau)	32.9
North Riviera F6	38
Plateau C4	33.3
Anonkoua Kouté PZ (Abobo)	48.38
Filtisac PZ (Abobo)	86.16
North Zone F4 (Adjamé)	72.47
North Riviera 3 (Cocody)	42.7

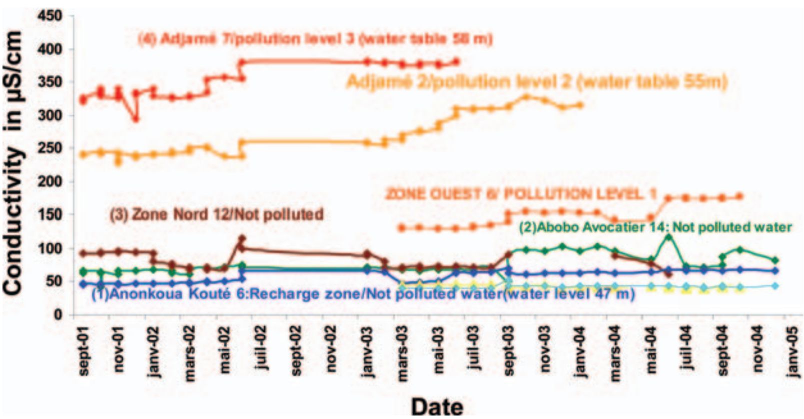

Figure 25.8. Variation of the conductivity of the Abidjan Aquifer from September 2001 to December 2004.

The first group characterises unpolluted water, and the second group corresponds to level 1 of pollution, which is water of intermediate pollution. The third group corresponds to level 3 of pollution, which is polluted water.

Measurements of the nitrate concentration (NO_3-N) in the exploitation boreholes of the SODECI from 1994 to 2004 are presented in Table 25.3. The representation of the nitrate variation rate is illustrated on Figure 25.9. The nitrates (NO_3-N) since October 1995 have exceeded the standard of potability in certain boreholes (the drinking water standard is 50 mg/L as N). In Côte d'Ivoire, European standards of drinking water are used. This pollution progresses towards the north in the direction of the 'clean' zone of recharge. These salt contents are expressed in the progressive pollution of the deep aquifer from the surface, which is severely polluted by the absence of proper sanitation and constantly increasing waste. Since July 1997, another area of progressive pollution (ZO6) of the deep aquifer appeared.

- Borehole closed means that the boreholes were closed or given up are no longer used for the water supply of the population. Among these boreholes are AD2, AD7, AD8 and PTC4.
- Borehole stopped means that the boreholes were stopped either as a result of a technical breakdown and could be repaired and re-used. In this case ZO6.

Table 25.3. Measurements of nitrate (NO_3-N) in mg/l on the boreholes.

Date	AK6	ZO6	ZN12	ZE1	AD2	AD7	AD8	PTC4
10/06/94	–	–	–	–	37.2	40.9	75.2	43.1
26/06/95	4.35	15.08	21.5	20.94	40.7	50	71.86	86.63
22/04/96	17.96	20.77	14.25	5.3	50	70	69	90
23/03/97	15	20	10.43	4	45	60	80	95
29/09/98	15	5.31	9	3	51	68	75	100
30/09/99	10	120	10	1.3	38	65	56	110
04/04/00	14.7	40	6	7	88	90	92	–
14/02/01	26	60	28	8	90	100	90	–
03/07/02	6	61	13	8	–	130	–	–
29/06/03	16	54	19	8	–	–	–	–
11/09/03	14	60	21	12.1	–	–	–	–
29/06/04	16	60	15	12.1	–	–	–	–
01/12/04	13	–	10	–	–	–	–	–

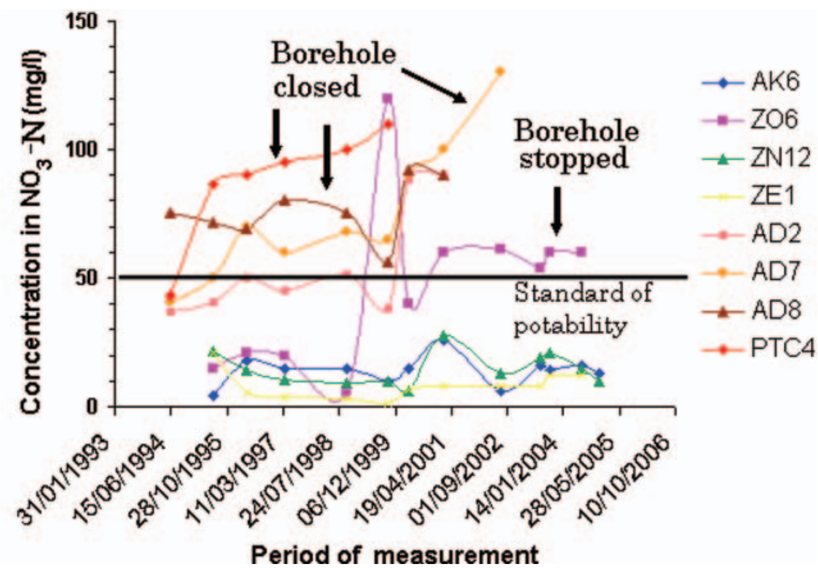

Figure 25.9. Variation of the nitrate rate in deep boreholes of the Abidjan Aquifer.

In terms of bacteriological level, monitoring shows good quality water (negative culture) for all the sites except for the western Zone 6 (ZO 6) which contained, in June 2004, a high rate (909 cfu/100 ml) of *Escheriechia coli*. The high percentage of nitrates and the presence of Escherichia coli in the water of site ZO 6 prove that pollution is the result of human activities and of recent origin, related probably to the infiltration of domestic wastewater amplified by the strong rains of May and June 2004. The consequence of the pollution of deeper boreholes is their closing, reducing the exploitation capacity of the aquifer and creating a further need for underground water treatment and more boreholes to satisfy the water supply demand.

6 DEVELOPMENT OF STRATEGIES FOR WATER SUPPLY AQUIFER PROTECTION

With the objective of a better water supply protection for the Abidjan Aquifer, the following recommendations are made:

- Identification of pollution hot spots and major threats to the water supply aquifer with the aim to draw the attention of stakeholders and the population to mitigate such threats.
- Establish urban water supply quality monitoring in order to prevent further pollution. In the long term the aim is to create an observatory for the water supply aquifer of Abidjan.
- Produce and disseminate periodic bulletins on the quality of the water supply aquifer for the authorities and population to raise consciousness on the vulnerability of groundwater resources and the need to protect them.
- In collaboration with NGO, we promote public awareness by documentary film projections and seminars.
- A vulnerability map of water supply pollution for the Abidjan Aquifer was created in order to help the authorities better manage the resource.
- Develop capacity building through the production of two M.Sc. theses and one Ph.D thesis in preparation.

7 CONCLUSION AND RECOMMENDATIONS

In conclusion, it is recommended that better governance and management is established for the water supply aquifer of Abidjan, which would include the following elements:

- The creation of an institutional framework for integrated water resource management in order to establish lawful measurements in the various applications of water protection.
- The creation of a network of reliable piezometers for the monitoring of the water table.
- The creation of synergy among the scientific community, the NGO, the various users of water and the authorities in charge of aquifer management for the collection and diffusion of information.
- The development of a policy of sanitation, with the creation of sewage systems and wastewater treatment with the goal of re-use.
- The installation of a waste disposal management strategy with the use of suitable sites observing all the conditions of a healthy environment.

ACKNOWLEDGEMENTS

We would like to thank the UNEP/DEWA and the UNESCO/PHI for their financial support for the realization of this study. Our thanks go to structures such as SODECI (Distributive Society of Water of Côte d'Ivoire), the Direction of Human Hydraulic (DHH) and the Direction of the Water Resources (DRE) for their collaboration.

REFERENCES

Aghui N. et Biemi J. 1984. Géologie et hydrogéologie des nappes de la région d'Abidjan et risques de contamination. Annales de l'Université Nationale de Côte d'Ivoire, série c, tome 20, pp.331–347.

INS 2001. Recensement Général de la Population et de l'Habitation (RGPH) 1998. Données socio-démographiques et économiques des localités, résultats définifs par localités, région des lagunes, vol.III, tome 1, 43 p.

Jourda J.P. 1987. Contribution à l'étude géologique et hydrogéologique de la région du Grand Abidjan (Côte d'Ivoire). Thèse Doctorat. Univ. scient. techn. et Méd. de Grenoble, 319 p.

Jourda J.P., Kouame K.F., Soro N., Kouame K.J., Kouadio B.H., Saley M.B., Kouame K. et Ahoussi E. 2004a. Aquifères superficiels et pollution urbaine en Afrique. Projet UNESCO/UNEP/UNCHS/ECA. Rapport décembre 2004, phase II, les activités du projet en RCI

Jourda J.P., Kouame K.F., Soro N., Kouame K.J., Kouadio B.H., Saley M.B., Kouame K. et Ahoussi E. 2004b. Bulletin sur la qualité des eaux souterraines de nappe d'Abidjan. N^{o} 4 et 5, juin et décembre 2004.

Kouadio L.P., Abdoulaye S., Jourda P., Loba M., Rambaud A. 1998. Conséquences de la pollution urbaine sur la distribution d'eau d'alimentation publique à Abidjan. Cahiers de l'Association Scientifique Européenne pour l'Eau et la Santé, vol. 3 n^{o} 1–1998.

Oga Y.M.S. 2004. Echange scientifique de courte durée. Rapport final, 38p.

SOGREAH 1996. Etude de la gestion et de la protection de la nappe assurant l'alimentation en eau potable d'Abidjan. Novembre 1996 (*Rapport 2*).

26

The impacts of coal and gold mining on the associated water resources in South Africa

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ABSTRACT: South Africa is the largest gold producer in the world and the fourth largest producer of coal in the world. The 224 million metric tons of coal produced per year directly supports employment for approximately 50,000 employees. Several water-related problems associated with mining, largely water quality deterioration due to pyrite oxidation, occur. The estimated post-closure water from the Mpumalanga coalfields is estimated to be in the order of 360 Ml/day (Grobbelaar *et al.*, 2001). The voids created by mining will all eventually fill up with water and surface structures can be long term sources of pollution of groundwater and associated surface water resources. In the gold industry numerous potential pollution sources exist including AMD (Acid Mine Drainage) from sand heaps, rock dumps and tailings, with other potential problems with radioactivity and cyanide. Several tools exist to predict the water quality degradation and examples of the application are shown in this paper. Due to the extent of mining in South Africa and the volumes of water associated with it, these aspects are very important considerations for preventing groundwater quality degradation.

1 INTRODUCTION

South Africa is one of the major mining countries of the world and activities are widespread (Figs. 26.1 and 26.2). South African mines produce a variety of ores e.g. chromite, coal, copper, diamonds, fluorspar, gold, iron ore, lead, limestone, manganese, nickel and platinum group metals. Mines also produce a variety of potential contaminants, depending on the ore deposit type, mining processes and mineral processing activities at specific sites.

Historically South Africa has been the largest producer of gold in the world. Only in 1996 a total volume of 377 million tons of mine waste was produced, accounting for 81% of the total waste stream in South Africa. These mine wastes contain large amounts of sulphide minerals (10–30 kg per ton) (Labuschagne *et al.*, 2005). Mining, as the name implies, is a non-renewable activity and the inevitable fact of starting a mine is that it will stop some time in the future (Hodgson *et al.*, 2001).

South Africa is the fourth largest producer of coal in the world and the 224 million metric tons of coal produced per year directly supports employment for approximately 50,000 employees. Unfortunately associated with mining several water-related problems, largely associated with water quality deterioration due to pyrite oxidation, occur.

Johannesburg is the centre of the gold mining industry and has been for many decades. Active gold mines are located in the West and East Rand areas in Gauteng, the Klerksdorp area in the North West Province and in the Free State Goldfields area. Gold mines are the largest employer in the mining sector with 39 operational gold mines in 2001 (Chamber of Mines, 2002). By-products from gold mines, include uranium and acid/pyrite, which are potential sources of groundwater pollution (acid mine drainage). One hundred and fifty nine mines (some still active) are believed to exist in the City of Johannesburg alone.

Coal mines are located in the northern and western parts of South Africa in the major drainage regions of the Vaal River, Olifants River and the northeastern escarpment within which four major

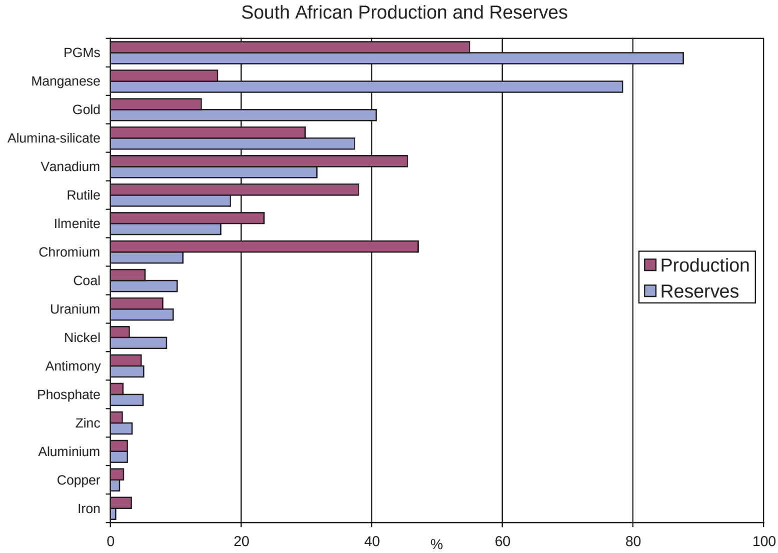

Figure 26.1. SA's role in world mineral reserves, production and exports, 2005 (Facts and Figures, South African DME as produced by SA Chamber of Mines, 2005).

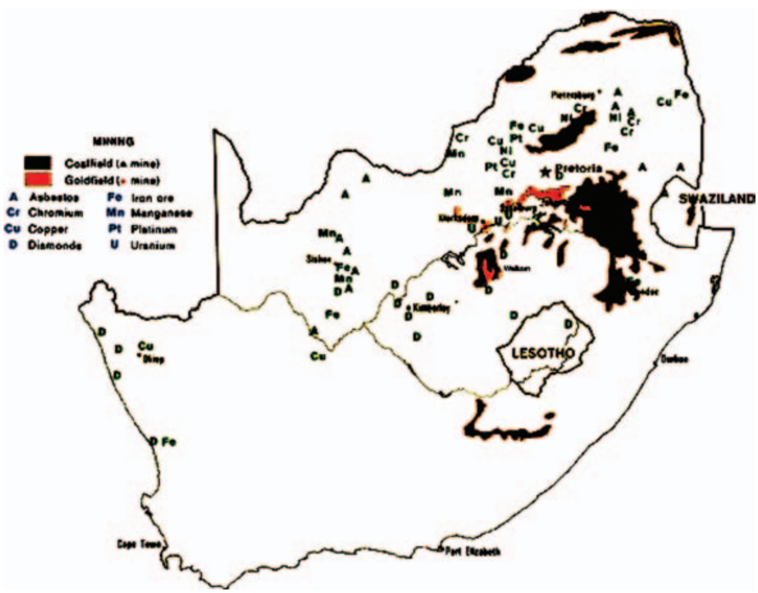

Figure 26.2. Mining in South Africa.

rivers flow. The South African Chamber of Mines recorded sixty-two operational coal mines in 2001, the third highest employer in the mining sector. Coal mining's impact on the water resources varies according to the life cycle of the mine. Coal mines may produce large quantities of acid or saline mine drainage.

Diamond mining occurs away from the major urban centres and is practiced mainly in Kimberley and surrounding areas as well as along the West Coast, into Namibia. Diamond mining operations are the most common, with 69 mines operating in 2001, but mining occurs on a smaller scale than gold and coal mining. The associated environmental impacts are generally less severe and the potential for groundwater contamination lower than for gold or coal mining, because the gangue minerals in diamond ores are generally less-reactive than those in gold and coal tailings (Usher *et al.*, 2004).

Other ores, including iron ore and base metals, are mined in the Northern Cape (e.g. Hotazel, Sishen, Okiep), North West (Rustenburg) and Mpumalanga (Barberton). Chrome and platinum group metals (PGM) are important mining enterprises in the Bushveld with 12 PGM mines and 13 chrome mines in operation in 2001. PGM's are the most significant growth area in terms of future projects in the mining sector and the second largest employer in the mining industry.

This paper will concentrate on the impacts of gold and coal mining particularly on the water quality of the adjacent areas.

2 WATER QUALITY IMPACTS FROM MINING

The impacts of the mining industry on groundwater are principally contamination (often from Acid Mine Drainage related processes) and impacts on quantity (initially dewatering and after closure decant back onto surface).

Associated with coal and gold mining in South Africa, the phenomenon of Acid Mine Drainage (AMD) occurs. Acid mine drainage occurs when sulphide minerals in rock are oxidised, usually as a result of exposure to moisture and oxygen. This results in the generation of sulphates, metals and acidity that can have manifold environmental consequences.

Pyrite (FeS_2) an iron disulphide (commonly known as fool's gold), is the most important sulphide found South African mines. Pyrite is a ubiquitous but trace mineral in many rock types. It is concentrated in many types of ore deposit by natural processes so that it may be the pervasive (but usually useless) mineral in the ore seam.

When exposed to water and oxygen, it can react to form sulphuric acid (H_2SO_4). The following oxidation and reduction reactions the pyrite oxidation that leads to acid mine drainage.

(1) $FeS_2 + 7/2\,O_2 + H_2O \Rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+$
(2) $Fe^{2+} + 1/4O_2 + H^+ \Rightarrow Fe^{3+} + 1/2\,H_2O$ (rate limiting step)
(3) $Fe^{3+} + 3H_2O \Rightarrow Fe(OH)_3$ (yellow boy) $+ 3H^+$
(4) $FeS_2 + 14Fe^{3+} + 8H_2O \Rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+$ (Stumm and Morgan, 1996).

In the many areas of South Africa there are co-existing carbonates such as calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$), which can neutralise the acidity generated. The reaction with calcite is given by:

$$FeS_2 + 2CaCO_3 + 3,75O_2 + 1,5H_2O \Leftrightarrow Fe(OH)_3 + 2SO_4^{2-} + 2Ca^{2+} + 2CO_2$$

The results of this are that many of the mine waters are not necessarily acidic, but often high in dissolved salts. Additionally, the sediments overlying the mines, can be fairly saline, and for example in the southern portion of the Mpumalanga coalfields, are high in sodium.

To put the importance of these issues into a global context, the estimates for AMD in other countries can be highlighted. In the USA it has been estimated that there are currently over 1.1 million acres of abandoned coal mine lands, over 9709 miles of streams polluted by acid mine drainage (AMD), 18,000 miles of abandoned highwalls, 16,326 acres of dangerous spoil piles and embankments, and 874 dangerous impoundments. Harries (1998) estimates that in Australia the operational costs of AMD-related actions run to approximately $60 million per year. He states that several Australian sites could require over $100 million each if the release of pollutants was to be reduced to a level with minimal ecological impact. Published costs of rehabilitating historic acid drainage sites have been estimated to be between $US 2 and 35 billion for the USA, and between $C 3 and 5 billion for Canada.

Acid generation can last for decades, centuries, or longer, and its impacts can travel many miles downstream. Roman mine sites in Great Britain continue to generate acid drainage 2,000 years after mining ceased (BC Mining Watch, 1996). Acid drainage from mining operations has a long history, dating back thousands of years to Phoenician times when the Iberian Pyritic Belt in Spain, from where the Rio Tinto (Red River) flows, was first exploited (Miller, 1998).

3 GOLD MINING IN SOUTH AFRICA

The gold-bearing Witwatersrand basin is massive and stretches through an arc of approximately 400 km (see Fig. 26.2). It has been mined for more than 100 years producing more than 41,000 t of gold and remains the greatest un-mined source of gold in the world. The gold is extracted from a 1–2 m thick tabular conglomerate layer. These conglomerates are pebbles of quartz in a sand matrix and contain about 3% of pyrite (FeS_2), as well as several other sulphide containing minerals, such as pyrrhotite (Fe_xS) and galena (PbS). Robb and Robb (1997) state that pyrite was the most abundant heavy metal constituent in the Witwatersrand reefs over the entire paragenetic sequence.

There are two areas of concern related to acid mine drainage in the Witwatersrand basin; the potential for valuable water resources to be polluted and danger related to sinkhole development.

3.1 *Sources of AMD*

Three principle sources of AMD can be identified in the gold mining industry of South Africa (Scott, 1995):

1. Sand Heaps – these are only associated with the older mines and are therefore predominantly found associated with mines in the northern part of the West Rand group of mines.
2. Rock dumps. Off-reef development such as shafts and underground access routes generate coarse rock waste. Such waste is accumulated in a dump at each shaft. In spite of there being no ore in these wastes, the rocks do contain a certain proportion (usually well below 1%) of pyrite. It is exposed to the atmosphere in the rock dump and will oxidise and generate small amounts of acid. Some suitably sited (close to demand and road transport) rock dumps have been used as a source of aggregate for road metal and other building applications.
3. Slimes dams are associated with all the metallurgical plants that have recovered metals in this area. Since the fine metallurgical residues are disposed of wet (hence the term slime), the dams have potential to generate significant seepage to underlying formations.

The slime is confined in a dam where water evaporates, is siphoned off, or seeps away.

The potential for AMD generation from sand dumps is high as they contain significant pyrite, their coarse grain size allows deep (+10 m) penetration of oxidising conditions and rapid circulation of water. Thus surface and subsurface seepage from sand dumps will be an aggressive, polluting solution.

The potential for AMD from rock dumps is relatively high since they are coarse grained and allow free circulation of oxygen and water.

The potential for AMD from slimes dams depends on the age of the dam. Those dams emplaced before 1940 will contain significant fine pyrite which will have potential to oxidise and so generate acid. Dams emplaced after 1940 and up to approximately 1980 may contain very little pyrite and thus will have little potential to generate acid. Recent slime disposal contains pyrite. Newer dams are designed and constructed keeping environmental protection in mind. This includes underdrains for leachate collection and toe drainage dams to collect lateral seepage. Rosner *et al.*, 2001, did a detailed assessment of pollution contained in the unsaturated and saturated zones below reclaimed tailings and rock dumps. The following hazardous trace elements were detected at significant concentrations in gold mine tailings: As; Cr; Cu; Ni; Pb; U; Zn. Soil underneath reclaimed tailings dams shows typical contamination due to AMD with acidic conditions (pH 3–4) in near surface soil samples become less acidic to a nearly neutral pH with depth. Groundwater quality beneath and in close vicinity to the investigated tailings dams is dominated by the Ca-Mg-SO_4 type, indicating acidic seepage. High TDS (up to 8000 mg/l) values occur mainly as a result of high salt loads (SO_4^{2-} and Cl^-) in the groundwater system. In most of the samples, groundwater pH values are fairly neutral due to the acid neutralising capacity of the dolomitic rock aquifer (Rosner *et al.*, 2001).

From this perspective the presence of dolomite, which is present in the northern arc of the Wits Basin, acts as a natural buffer to prevent acidity and thereby attenuates metals. This has been commonly observed on the West Rand although on the East Rand a great number of acidic discharges with high metal contents have been observed (Scott, 1995). This neutralisations however increases the risks of sinkhole formation, which has been a prominent groundwater problem on the West Rand (Usher *et al.*, 1999).

In the Free State goldfields (see Figure 26.2) significant water quality issues are also present. Due to the large quantities of water present in the mined Witwatersrand rocks, a large quantity of water (120–150 Ml/d) is pumped to the surface for accessibility of the mining area each day (Cogho *et al.*, 1992). This groundwater has an average electrical conductivity of 500 mS/m and cannot be used for drinking or irrigation purposes. A small portion is diluted with potable water and used as process water and the remainder is pumped to evaporation areas and pans. This has a negative impact on the localized shallow aquifers in the area. The groundwater quality impacts are associated with the large evaporation pans and areas, the water quality (Na-Cl – type) of the deep aquifers and the shallow low yielding Karoo aquifers underlying the mining activities, which keep the pollution localized surrounding the sources. The water is typical of a SO_4, Na, Ca–Cl type with associated low pH which cause metals to be mobilized in the groundwater (Cogho *et al.*, 1992). The groundwater pollution together with contaminated run-off contributes to the salination of the Sand River.

As an example of the potential impacts, in the KOSH mining area it is estimated that the almost 2600 Ha of tailings and 360 Ha of waste rock dumps can generate in the order of 150–200 tons of sulphate per day (Hearne, 1996, Labuschagne *et al.*, 2005), which significantly impacts on the aquifers and the adjacent streams and Vaal River.

3.2 *Other water quality issues in gold mining*

An additional problem related to gold mining is radioactivity. In certain instances the uranium was mined together with the gold while in others radioactive material has been disposed of with the waste. Uranium mobility is restricted due to the general Eh/pH conditions that exist, but radioactive pollution is a concern in a band stretching from Carltonville, through the Klipspruit catchments to the East Rand gold mines (Usher *et al.*, 1999).

Winde (2001) has reported on the effects of such occurrences in the Koekemoer Spruit of South Africa. Extensive research has been done on the potential toxic influence of uranium trial mining in the Karoo uranium province and highlights the fact that the small – scale mining occurs in close proximity to rural communities.

The danger associated with uranium is due to its radioactivity in the form of α, β and γ emissions that may be derived from any of the isotopes of uranium or daughter products in the uranium decay series. Thus pathways connecting uranium or its daughter products to unprotected animal or human tissue are of great concern. For example if dissolved in groundwater, after ingestion the radioactive material will be in intimate contact with sensitive tissue. Radon, a gas and one of the uranium decay series products, may be inhaled and via this pathway has intimate contact with animal or human tissue.

One of the most controversial aspects of gold mining involves the use of cyanide to extract over 90% of annual production of this precious metal (Mudder, 2005). The use of cyanide has a long history in the mining industry. For decades, it has been used as a pyrite depressant in base metal flotation, a type of beneficiation process. It also has been used for more than a century in gold recovery. It has been reported that 27% of mines in Africa use cyanide (Mining Magazine, 2000). In the 1950's, technology advanced that allowed large-scale beneficiation of gold ores using cyanide set the stage for the enormous increase in cyanide usage when gold prices skyrocketed in the late 1970's and 1980's. Continued improvements in cyanidation technology have allowed increasingly lower grade gold ores to be mined economically using leach operations. The use of cyanide in the leaching of gold ores has an increased potential to impact the environment because of the greater quantity that is used in leaching (EPA, 2000). The acute toxicity of cyanide coupled with impacts from a number of major incidents has focused attention on the use of cyanide in the mining industry.

Mudder (2005) states the case as follows "As the search for ore bodies moves into less developed countries and more remote and sometimes environmentally sensitive regions, a balance must be sought between the struggle to define sustainable development, manage limited resources, and alleviate poverty versus mining every newly discovered metal and mineral deposit. Public concern regarding the safety and environmental aspects of cyanide is valid and understandable, considering its historical uses and several recent mining incidents, which have involved cyanide." It must be pointed out that the risks of cyanide pollution compared to other potential environmental or water quality issues are considered low.

4 COAL MINING IN SOUTH AFRICA

4.1 *Overview of the Mpumalanga coalfields*

Coal mining in the Mpumanlanga region has been extensive and has been ongoing in the Mpumalanga Coalfields for more than 100 years. (Vermeulen, 2003). The depth of mining ranges from less than 10 m below surface to more than 100 m. The coal seams generally increase in depth to the south. Mining methods are bord-and-pillar, stooping and opencast. Opencast mining has been introduced during the late seventies. Underground mining on the 2 seam comprises in excess of 10,0000 Ha while opencast mining is expected to eventually exceed 40,000 Ha (Grobbelaar *et al.*, 2002). Coal is generally mined by opencast- or underground methods in South Africa. At depths down to 50 metres coal is normally extracted by surface mining, the extraction rate associate with type of mining being currently 85–90% (Hodgson and Krantz, 1998). At depths below 50 metres more conventional mining methods such as bord-and-pillar extraction have been used sine mining began.

4.2 *Geology*

The Karoo Supergroup in the Witbank region comprises the Ecca Group and Dwyka Formation. The total thickness of these sediments ranges from 0–100 m. The Ecca sediments consist predominantly of sandstone, siltstone, shale and coal. Combinations of these rock types are often found in the form of interbedded siltstone, mudstone and coarse-grained sandstone. Typically, coarse-grained sandstones are a characteristic of the sediments in the Witbank Area (Hodgson and Krantz, 1998).

Five coal seams, numbered from bottom to top as No. 1–5, are present. Only two of the seams are mineable over most of the area. These are the No. 2 and 4 Seams, which are usually separated by sediments of a total thickness in the order of 20–30 m. Seams 1 and 5 are, however, mined locally. Dolerite intrusions in the form of dykes and sills are present within the Ecca Group. Faults are rare. However, fractures are common in competent rocks such as sandstone and coal.

4.3 *Water in South African coal mines*

4.3.1 *Sources of water*

Several sources of water influx are expected in South African Collieries. In opencast areas, much of the influx is dependent on the state of the post-mining rehabilitation, while in underground mining factors such as the mining type, depth and degree of collapse and interconnectivity are important.

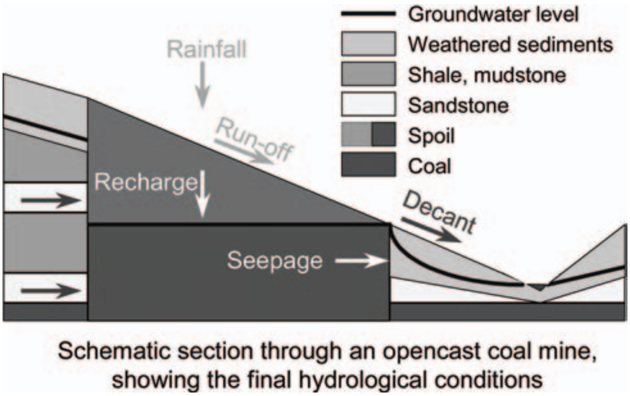

Figure 26.3. General geohydrology of opencast pits (Grobbelaar, 2001).

Table 26.1. Water recharge characteristics for opencast mining (Hodgson and Krantz, 1995).

Sources which contribute water	Water sources into opencast pits	Suggested average values
Rain onto ramps and voids	20–100% of rainfall	70% of rainfall
Rain onto unrehabilitated spoils (run-off/seepage)	30– 80% of rainfall	60% of rainfall
Rain onto levelled spoils (run-off)	3–7% of rainfall	5% of rainfall
Rain onto levelled spoils (seepage)	15–30% of rainfall	20% of rainfall
Rain onto rehabilitated spoils (run-off)	5–15% of rainfall	10% of rainfall
Rain onto rehabilitated spoils (seepage)	5–10% of rainfall	8% of rainfall
Surface run-off from surroundings	5–15% of total pit water	6% of total pit water
Groundwater seepage	2–15% of total pit water	10% of total pit water

Figure 26.3 illustrates the generalised hydrological conditions associated with an opencast environment. Normal groundwater movement still takes place in aquifers. Groundwater flow directions will necessarily be directed toward the pits, due to an artificial change in gradients on a local scale and a higher K-value in spoils (Grobbelaar, 2001). This flow, together with direct recharge into the spoils will create an artificial groundwater level in the heaped spoil until a decant level is reached. Water that decants out of the spoils as well as run-off from the surface of the spoils will follow the natural gradient and will flow to the nearest river or stream.

In terms of expected sources of water, Table 26.1 summarizes the most important information:

The high recharge percentage of around 20% is due to a multitude of factors such as ponding, areas of spoils exposure, restabilisation cracks and influx into ramp areas. Several researchers have confirmed these high recharges through decant measurements (Hodgson, 1999, van Tonder *et al.*, 2003). The high influx is naturally an important driver on the observed water quality in opencast pits.

In underground mines the following sources of water could be encountered:

– Water encountered in the seam as mining commences. This is fairly low except where fractures or fissures as they are known within the mining industry are encountered.
– Recharge through the roof lithologies. The magnitude of this varies depending on mining induced fracturing of the overlying sediments.
– Direct recharge where cracks from the collapse of mining areas, usually due to high extraction mining, run through to the higher-yielding transmissive aquifers nearer surface.
– Regional groundwater flow, which will usually flow along the coal horizon, due to its higher hydraulic conductivity compared to the surrounding sediments.

Table 26.2. Key mining and storage statistics for Mpumalanga coalfields.

Mining type	Current mined area (ha)	Future area to be mined (ha)	Total area to be mined	Extraction height (m)	Extraction rate	Volume coal mined (Mm^3)	Future volume coal to be mined (Mm^3)	Total volume to be removed	Storage volume (Mm^3)
U/G 5 seam	7842	7050	14,892	2.5	0.65	127	115	242	242
U/G 4 seam	13,485	2,833	16,318	3	0.65	263	55	318	255
U/G 2 seam	98,550	39,695	1,38,245	3	0.65	1922	774	2696	1887
U/G 1 seam	2525	0	2525	2.5	0.6	38	0	38	30
Opencast	13,557	14,480	28,037	3.5	0.9	427	456	883	221
Totals	1,35,959	64,058	2,00,017			2777	1400	4177	2635

– Influx through the floor lithologies. This can play an important role in areas where the floor is transmissive, but where the mining floor is close to the Dwyka such as where the No.1 or No. 2 seam are mined, such influxes are negligible.

The water encountered, as mining continues, will cause subsidence over time. In bord-and-pillar areas where mining has been completed, features such as roof bolts, drilled to stabilise the roof during mining, act as local sinks for water to drain to. Where such a roof bolt intersects horisontal or vertical fractures, increased influx is experienced. This is particularly problematic in areas adjacent to water storage compartments, where seals are installed to accommodate a head in excess of the mining height. As the water level rises, intersection of the naturally occurring bedding plane fractures occurs. These act as more transmissive conduits, which allow the water to flow more freely toward the locally created sink.

4.3.2 *Potential storage volumes*

Based on the mining areas, expected volumes of coal removed can be determined. Using the coal extraction ratios, the determined storage factors such as 25% for opencast mines and the storage factors determined for collapse in high extraction mining, an indication of storage in the Mpumalanga coalfields can be obtained (See Table 26.2).The expected storage volume is a resource that cannot be ignored in a water-stressed country.

4.3.3 *Summary of influx and effect on water quality*

The different mining types each have conditions which promote poor quality water, and others that should ameliorate the effects of pyrite oxidation. In all of the mining types, however, the local mineralogical conditions provide the most important driver on water quality. Trenches dug in the spoils clearly indicate the validity of this observation. In underground mining, this is no less important. A case study at an underground compartment shows that the roof sediments have a higher probability of acidification than the coal seam itself, implying that the high extraction areas pose the greatest risk. The balance between the increased recharge, with addition of alkalinity and faster inundation, against the higher surface area and lower net neutralising potential will determine what the final water quality will be. Table 26.3 summarizes selected issues from a hydrogeochemical perspective (Usher, 2003).

4.3.4 *Intermine flow and water volumes*

After the closure of mines, water in the mined-out areas will flow along the coal seam floor and accumulate in the lower-lying areas. These man-made voids will fill up with water and hydraulic gradients will be exerted onto peripheral areas (barriers) or compartments within mines. This results in water flow between mines, or onto the surface. This flow is referred to as intermine flow (Grobbelaar, 2001). Projections for future volumes of water to decant from the mines have been

Table 26.3. Key processes in different mining types that affect water quality.

	Opencast mining	Bord-and-pillar mining	High extraction mining
Recharge	15%–25% *Dilution can occur*	3%-–8% *Build up of oxidation products*	5%–15% *Rapid inundation with some dilution; also often an additional source of alkalinity*
Relative surface area	High	Low	Medium
S-value	25% *Flooding can occur to prevent excessive salt loads*	In line with extraction – 50–65% *Large amount of water – high water:rock ratio*	20% over the collapsed height *Smaller volume of water distributed over greater height*
Coal removal	90% *If coal seams are the most likely to acidify it reduces the risk*	50–65% *If coal seams are the most likely to acidify it has the greatest probability*	90% *If coal seams are the most likely to acidify it reduces the risk*

made by Grobbelaar *et al*., (2002). In total, about 360 ML/d will decant from all the mines in combination. On a catchment basis, it relates to the following (ML/d):

Wilge/Klip	Olifants	Klein Olifants	Vaal	Komati
23	170	45	120	2

4.4 *Salt generation in South African coal mines*

Salt generation rates were determined by Vermeulen (2003) by measuring outflows and chemistries from which tonnages of salt were calculated. Static and kinetic modelling has also been used in this investigation to improve the understanding of processes. Salt generation rates have been calculated for a number of collieries. The following are the main conclusions:

- The daily rate of sulphate generation in opencast mining is 5–10 kg/ha/d. An average daily value of 7 kg/ha/d has been used in salt balance calculations for individual collieries. The daily sulphate generation rate for underground mining is in the range of 0,4–2,7 kg/ha. These rates, which are lower than those for opencast mining, are mainly due to less available reactive surfaces in underground mines. Heavy metals in the coal are present throughout the Mpumalanga Coalfields. Even though they temporarily mobilized during oxidation, most precipitate under alkaline conditions. A number of the mines investigated have alkaline water. It is a common phenomenon that collieries enter into an alkaline phase while being flooded. The base potential in the coal left behind in the mine is usually sufficient to neutralise the acid mine water generated during the mining phase.

  As an illustration of the impact of AMD, in the Witbank Dam Catchment, a total sulphate production of 45–90 t/d (average 70 t/d) is produced by opencast mines, largely due to AMD (Hodgson and Krantz, 1998). Extrapolation to include future opencasting at existing mines can result in an escalation of the sulphate contribution to an anticipated value of 120 t/d. The latter translates into a sulphate concentration of 450 mg/L in the Witbank Dam. The cumulative consequence of AMD processes on a catchment scale is therefore significant enough to warrant detailed investigation into predictive and management tools. Grobbelaar (2001) and Grobbelaar *et al.* (2002) indicate that expected post-closure sulphate loads for the Witbank Catchment are in the order of 170–280 t/d, based on discharges of 100–160 Ml/d.

5 MINE WATER QUALITY PREDICTION TECHNIQUES

5.1 *Introduction*

Accurate prediction potentially offers the most cost-effective means of reducing the impact of AMD on the environment and the associated costs by allowing advanced planning for prevention and control. The objective of a prediction program is to reduce uncertainty to a level at which potential risk and liability can be identified and effective extraction, waste handling and, where necessary, mitigation and monitoring strategies can be selected (Price, 1998). Predictive tests vary in complexity of procedure and data interpretation, the time required to achieve a predictive result and cost. It is highly unlikely that any one test can produce all the information necessary to evaluate all mine wastes. Combinations of tests are required to provide a reliable assessment. The scope of a prediction program will depend on site-specific conditions and factors.

The approach required might include some or all of the following (Price, 1998):

1. Initial assessment and site reconnaissance.
2. Sampling.
3. Chemical, mineralogical and physical analyses.
4. Short-term leaching tests.
5. Geochemical static tests.
6. Geochemical kinetic tests.
7. Mathematical models.

Analytical tests used to determine the acid generating potential of rock samples are either static or kinetic in nature. A static test determines both the total acid generating and total acid neutralizing potential of a sample. The capacity of the sample to generate acidic drainage is calculated as either the difference of the values or as a ratio of the values as will be discussed in the section to follow. These tests are intended to predict the potential to produce acid and not the generation rate of acid. Static tests can be conducted quickly and are inexpensive compared to kinetic tests.

Acid-Base Accounting (ABA) is a screening procedure whereby the acid-neutralising potential and acid-generating potential of rock samples are determined and the difference (net neutralising potential) is calculated. Static tests are primarily intended to examine the balance between the acid-producing potential (AP) and acid neutralising potential (NP) of a sample. The net neutralising potential and/or the ratio of neutralising potential to acid-generation potential, are compared with a predetermined value, or set of values, to divide samples into categories that either require, or do not require, further determinative acid potential test work.

For material where the potential for acid generation is uncertain, kinetic test work is performed in an attempt to define acid generation characteristics. Kinetic tests are intended to reproduce the natural field processes of reaction, usually at enhanced rates so that an indication of water quality evolution and the important factors can be determined.

5.2 *The ABA characteristics of different lithologies in the Witbank Coalfield*

In order to obtain an indication of the contribution of different lithological layers to the overall acid-base account, ABA done as part of various projects was consolidated into a single database. Only approximately a quarter of all ABA results that could be obtained by researchers such as Hodgson (1997, 1998), Hodgson and Krantz (1998), Hodgson and Grobbelaar (1998) and Usher *et al.* (2002) have been sampled and analysed according to clearly defined lithological layers.

A total of 515 ABA samples were selected, which could be clearly recognised as belonging to a specific lithological unit. From this assessment (Fig. 26.4) it was found that the coal seams all have large variations in likely NNP, driven by the presence of pyrite nodules and the irregular occurrence of calcite, often as veinlets. The interbedded sediments have a far narrower range of probable values, with all averaging very near to zero. It can also be seen that the coal seams can

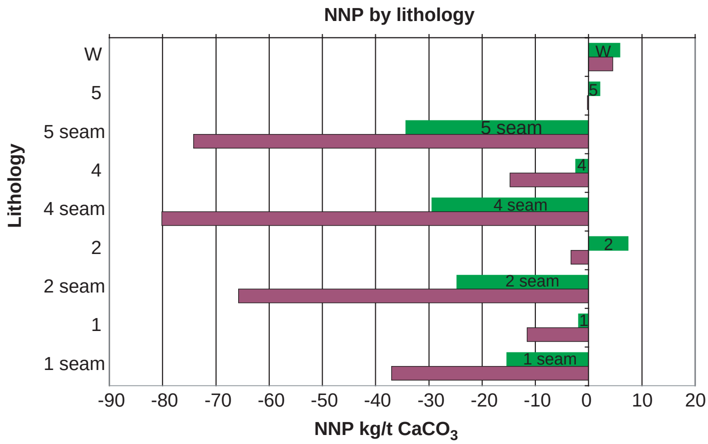

Figure 26.4. NNP by lithology.

be considered the most problematic in the Witbank Coalfield as far as acid potential is concerned. The layer between the 2 and 4 Seams has the most beneficial contribution in terms of neutralising acidity. The reasons for this are probably as much due to hydrogeological response of the system as the sediments mineralogy. The layers between the 2 and 4 Seams have lower pyrite content than the seams, but unlike the layers above the 4 Seam the carbonates have been leached out to a lesser degree by circulating groundwater.

5.2.1 *Validity of application*

Usher *et al*., 2004 compared different prediction techniques for water quality prediction, The conclusion from these assessments was that ABA, in most cases, provides information that can be correlated to the field, mineralogy and to kinetic. The use of multiple techniques leads to a higher degree of confidence in the use of the data. In separate investigation on underground collieries, Usher, Yibas and Zhao showed that analytical tests play an integral role in water quality predictions at mines. The results also showed that taking site conditions into account improves the predictions made and that these tests compare well to field observations.

5.2.2 *Suggested application*

Based on the research done on this problem, a suggested flowpath (Fig. 26.5) for the application of these tests for South African collieries was suggested by Usher *et al*., 2002.

6 CONCLUSIONS

The magnitude of the threat from mining activities is dependant on whether precautionary measures are taken to prevent contamination, but in many cases the scale of mining operations is such that groundwater pollution cannot be completely avoided. Abandoned mines pose a potentially even larger threat to groundwater resources, since most were operated for decades without any environmental controls. Recorded polluting incidents from mines have occurred primarily in Gauteng and Mpumalanga Provinces, where gold and coalmines produce acidic leachates, causing contamination of groundwater systems following infiltration (Usher *et al.*, 2004).

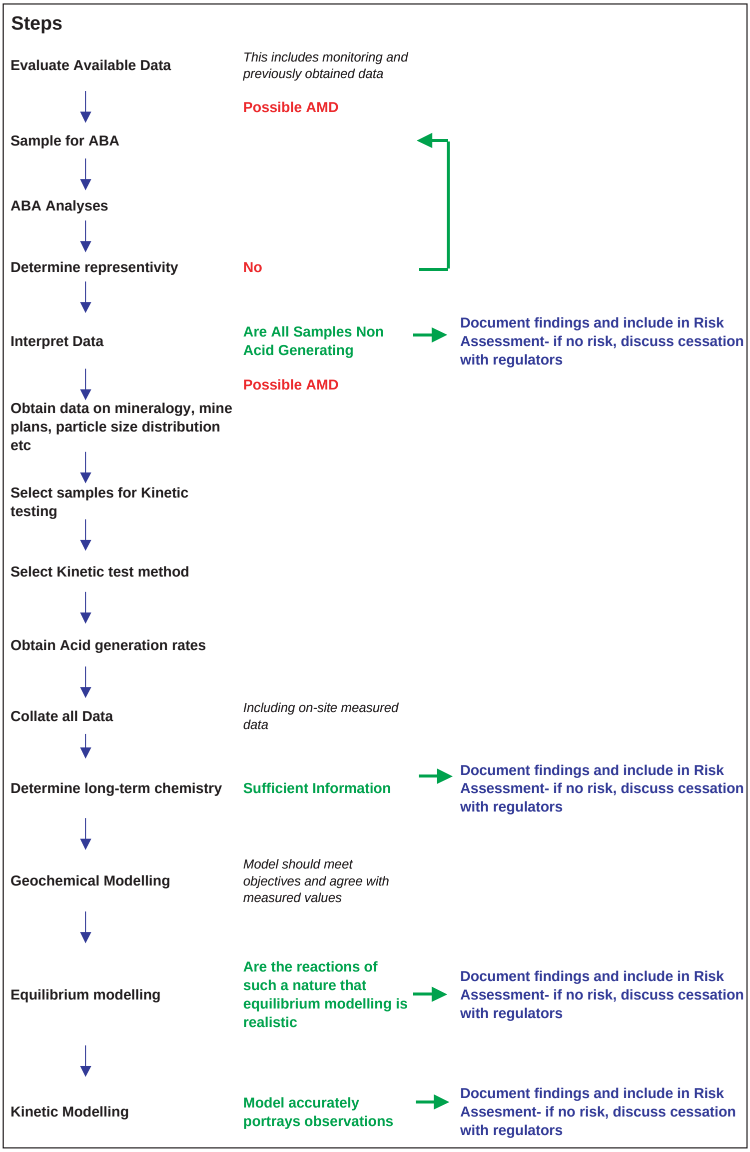

Figure 26.5. Suggested flow path for opencast coal spoils chemistry prediction.

Coal and gold mines, especially closed and abandoned mining operations, appear to be the most significant threats in terms of potential groundwater contamination from the mining sector in South Africa.

In a water-stressed country like South Africa all water must be regarded as a potential resource. Water can only be used if the quality thereof is fit for its intended use. The water management focus at the mines needs to change from a purely volume-driven focus to an associated water quality driven focus. If water qualities can be dealt with in a way that an array of qualities for different purposes can be provided, the cost-savings or rebates from the waste discharge costs will be such that the current waste becomes a prime asset for the mine and the country.

REFERENCES

Chamber of Mines of South Africa, Statistics of the Chamber of Mines of South Africa (2002) Johannesburg.

Cogho V.E., Van Niekerk L.J., Pretorius H.P.J. and Hodgson F.D.I. (1992) The Development of Techniques for the Evaluation and Effective Management of Surface and Groundwater Contamination in the Orange Free State Goldfields, WRC Report No. K5/224.

Department of Water Affairs and Forestry (2003) Water Quality Management Series. Sub-Series No. MS11. Towards a Strategy for a Waste Discharge Charge System. First Edition. Pretoria.

Douglas, J.T.K. and Moir, A.T. (1961) A review of South African Gold Recovery Practice. Transactions of the seventh Commonwealth Mining and Metallurgical Congress Vol. 3, SAIMM, Johannesburg. pp 971–1000.

EPA 910-B-00-001 (August 2000) Abandoned mine site characterization and cleanup handbook.

Grobbelaar, R. (2001). The long-term impact of intermine flow from collieries in the Mpumalanga Coalfields. Unpublished M. Sc Thesis. University of the Free State.

Grobbelaar, R., Usher, B.H., Cruywagen, L.,-M., de Necker, E. and Hodgson F.D.I. (2002) The Long-term Iimpact of Intermine Flow from Collieries in the Mpumalanga Coalfields, Water Research Commission Report.

Hearne, C. (1996) Klerksdorp Mine Managers Association – Phase 2 Investigation into the impact of diffuse seepage from gold mines in the Klerksdorp region on water quality in the Vaal River.

Hodgson, F. D. I. (1997) Current and long-term water and salt balances for Middelburg North Colliery. Unpublished report.

Hodgson, F.D.I. (1998) Water Qualities and salt balances for Kriel Pits 5 and 6. Unpublished report.

Hodgson, F.D.I. (1999) Intermine flow between Tavistock and Rietspruit Collieries. Unpublished report.

Hodgson F.D.I. and Grobbelaar R. (1998) Acid-base accounting and long-term mine water chemistry at Boschmanskrans Opencast, Unpublished report.

Hodgson, F.D.I. and Krantz, R.M. (1998) Groundwater quality deterioration in the Olifants River Catchment above the Loskop Dam with Specialised investigations in the Witbank Dam Sub-Catchment. Report to the Water Research Commission by the Institute for Groundwater Studies, University of the Orange Free State.

Hodgson, F.D.I., Usher B.H., Scott, R., Zeelie S., Cruywagen L.M. and de Necker E. (2001) Predictiontechniques and preventative measures relating to the post-operational impact of underground-mines on the quality and quantity of groundwater resources, 2001. Water Research Commission, report no. 699/1/01. Water Research Commission, Pretoria, South Africa.

Jackson, O.A.E. (1961) A review of modern milling practice at the South African gold and uranium mines. Transactions of the seventh Commonwealth Mining and Metallurgical Congress Vol. 3, SAIMM, Johannesburg. pp 936–970.

Labuschagne, P.F., Usher B.H. and Matfield F. (2005) Geohydrological management approaches for SITE closure in South African gold mines. Conference on Processing and Disposal of Industrial and Mining Waste. Minerals Engineering International, Falmouth, United Kingdom. 13–15 June 2005.

Mining Magazine, Mining Activity Survey (January 2000) Mining Journal Limited, London, pp 25–32.

Miller, S. (1998) Prediction Predicting Acid Drainage Groundwork, No.1 Vol. 2, Australian Minerals & Energy Environment Foundation.

Mudder, T.I. and Botz, M. (2005) Cyanide risks and realities Conference on Processing and Disposal of Industrial and Mining Waste. Minerals Engineering International, Falmouth, United Kingdom. 13–15 June 2005.

Price, W.A. (1998) Guidelines and Recommended Methods for the Prediction of Metal Leaching and Acid Rock Drainage at Minesites in British Columbia, British Columbia Ministry of Employment and Investment, Energy and Minerals Division, Smithers, BC, (April), p. 143

Robb, V.M. and Robb, L.J. (1997) The environmental impact of Witwatersrand Gold Mining. In: South African Mineral Resources.

Scott, R. (1995) Flooding of Central and East Rand gold mines: An investigation into controls over the inflow water, water quality and the predicted impacts of flooded mines. WRC Report no.486/1/95, Water Research Commission, Pretoria, South Africa.

Stumm, W. and Morgan, J.J. (1996) Aquatic Chemistry, 2nd Ed. John Wiley & Sons, Inc., New York, NY.

Usher, B.H. (2003) Development and Evaluation of Hydrogeochemical Prediction Techniques for South African Coalmines. Unpublished Ph.D thesis, University of the Free State.

Usher, B.H., Grobbelaar, R. and Hodgson, F.D.I. (1999) Acid-Base Accounting and the determination of long-term salt loads at Optimum colliery. Unpublished Confidential report to Optimum Colliery.

Usher, B.H., Cruywagen, L-M., De Necker, E. and Hodgson, F.D.I. (2002) On-site and Laboratory Investigations of Spoil in Opencast Collieries and the development of Acid-Base Accounting Procedures. Report to the Water Research Commission by the Insistute for Groundwater Studies.

Usher, B.H., Pretorius J.A., Dennis I., Jovanovic N., Clarke S., Titus, R. and Xu, Y. (2004) Identification and prioritisation of groundwater contaminants and sources in South Africa's urban catchments. WRC report no. 1326/1/04.

Usher, B.H., Yibas, B. and Zhao, B. (2004) The Application of Laboratory Geochemical Tests for Water Quality Prediction at Underground Collieries, Geoscience Africa 2004 Conference, Johannesburg, South Africa, 12–15 July 2004.

Van Tonder, G., Vermeulen D., Cogho V. and Kleynhans J. (2003) Prediction of the decant rate and sulphate concentration from rehabilitated open cast coalmines in South Africa. 6TH ICARD, Cairns, Queensland, 12–18 July 2003.

Vermeulen, P.D. (2003) Investigation of decant water from the underground collieries in Mpumalanga. Unpublished M. Sc thesis, University of the Free State.

Winde, F. (2001) Slimes dams as source of uranium – contamination of streams – the Koekemoer Spruit (Klerksdorp gold-field) as a case study, Proceedings Conference on 'Environmentally Responsible mining in Southern Africa.'

27

Monitored natural attenuation of petroleum hydrocarbons in a fractured environment – A case study

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ABSTRACT: A case study is presented where monitored natural attenuation coupled with active plume containment through pumping at the source, is the selected remedial strategy for petroleum hydrocarbon contamination. Of the multiple natural attenuation processes, biodegradation is considered the most important since it results in the destruction of the contaminant. The site under investigation is underlain by fractured siltstone and fine-grained sandstone, creating a complex aquifer. Monitoring wells and private boreholes surrounding the product release site are used for sampling purposes. In order to assess the Monitored Natural Attenuation (MNA) as a viable remediation option, bi-annual monitoring takes place for petroleum hydrocarbon concentrations and annual monitoring to assess the concentrations of electron acceptors (for oxygen, nitrate and sulphate) and the products of reduction (for manganese and iron). The results of the monitoring indicate that the dissolved phase hydrocarbon plume is shrinking. It also shows that secondary lines of evidence exist that biodegradation of petroleum hydrocarbons is taking place at the site under investigation and that the groundwater environment has the capacity to continue facilitating this process.

1 INTRODUCTION

Petroleum product releases to the subsurface environment is a common cause of groundwater contamination (Pretorius *et al.*, 2003). One of the aims of using the ASTM International method of Risk Based Corrective Action (RBCA) (ASTM, 2002) to deal with contamination is to use cost-effective methods to lower the potential (or actual) risk to receptors. For private companies responsible for cleaning up contamination, cost saving is an economic driver, while governments' efficient use of resources directly benefits society. Depending on the situation, i.e. where the pollution took place, which compounds are present and at what concentrations, it is sometimes possible to use Monitored Natural Attenuation (MNA) as a cost-effective remedial solution. On the other hand, there are some situations where MNA is not a choice, but the only viable alternative. This paper presents a case study in a fractured rock environment where most active remedial methods are likely to be ineffective. The depth to groundwater and bedrock make excavation unfeasible due to cost. Air sparging and vapour extraction techniques are limited in efficiency due to the small openings of the fractures as well as the depth to groundwater.

Biodegradation in addition to being one of the few alternatives, is considered an important natural attenuation process, since it results in the actual reduction in the mass of the contaminant (USEPA, 2004). This paper focuses on secondary lines of evidence to determine whether microbial action is contributing to the degradation of petroleum hydrocarbons at the study site.

2 SETTING

The site in question is a filling station in a small town along a major route. Several commercial properties are also situated in the immediate vicinity of the station. The rest of the properties, away

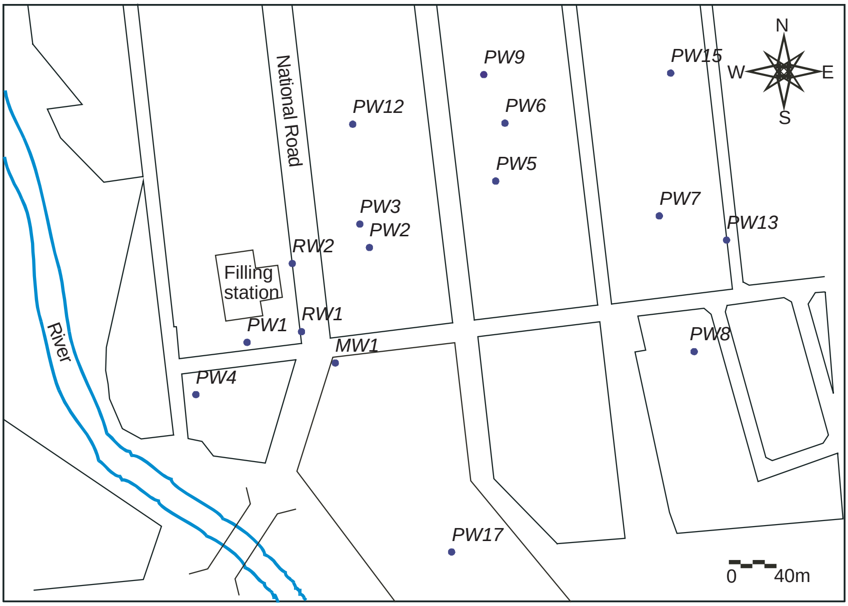

Figure 27.1. Layout showing the borehole positions in relation to the site.

from the main road, mostly comprise residential houses. Many residents in the town have their own boreholes that are mostly used for garden irrigation, while drinking water is supplied by the town. There is a non-perennial river close to the site, as shown in Figure 27.1.

Early in 2002, a resident close to the filling station found free phase product in his borehole (PW2). This resident pumped his well for almost a month before he was alerted to stop the practice. He collected sufficient petrol from his well to distribute it for use. The neighbour next door was away at the time and had asked somebody to start her borehole (PW3) pump every day to water the garden. The person looking after the property saw that the plants were wilting and started pumping more frequently, not knowing that free phase product was being pumped onto the plants. The result was that several plants died in the two gardens.

As a result of this leak, all the pipelines and tanks at the filling station were replaced, shortly after this primary source of pollution became apparent. According to the residents in the town, a similar incident had occurred on the same site during the 1980s. There is therefore a possibility that different pollution plumes are superimposed.

2.1 *Geology and hydrogeology – conceptual model*

Understanding the physical, chemical and biological processes that take place at a site forms part of the conceptual model. The construction of a conceptual model for a site is an iterative process, by which the model is constantly refined as more information is collected. Without a proper conceptual model it is impossible to identify key assumptions and areas of further data collection (Le Grand and Rosen, 2000). The choice of remedial action is also determined by the site conceptual model (USEPA, 1999).

The region of the case study is underlain by fractured mudstone and fine-grained sandstone (Botha *et al.*, 1998). With the water level at about 12 m below surface, a petroleum leak will cause the pollutants to migrate vertically downwards, spreading in both the larger fractures and matrix, which probably consists of microfissures and small pores (Botha *et al.*, 1998). When a borehole that intersects a fracture network is pumped, petroleum is mobilised along the fractures to accumulate in the pumping well. Although free phase is present in the well, the extent of product is limited to some degree by the preferential flow paths created by the fracture zones. However, some matrix flow does take place as found in low-yielding boreholes such as RW2 and PW3.

This conceptual model shows that off-site pumping to remove free phase product would probably lead to contaminants spreading along the fractures, which would create a large free phase plume. The high concentration of petroleum hydrocarbons in the fractures and fissures will create a positive diffusive gradient, and soluble constituents could dissolve into the matrix where storage takes place (Botha *et al.*, 1998). Over time, after free phase product has been removed, adsorbed contaminants will be released along with water from storage and contribute to a long-term dissolved phase plume, because of the multitude of loci where the hydrocarbons could reside in the subsurface (Lyman *et al.*, 1992).

2.2 *Active remediation*

Two boreholes were drilled on the forecourt of the site (RW1 & 2) and both contained free phase product. Pumping tests revealed that RW2 and PW2 were in direct hydraulic contact. Although the general groundwater flow direction is taken to emulate the surface topography towards the south, there is evidence that a NE-SW striking fracture zone exists. A Department of Water Affairs and Forestry (DWAF) official pointed out that the fracture zone could be associated with a similar striking dolerite dyke to the north of the site. Pump-and-treat was selected as an appropriate tool to contain the contamination, because the fractured rock environment and the depth of the water table did not leave much choice. It is important to point out that pump-and-treat is not a very effective remediation tool on its own, since the water flow in the aquifer does not influence the release rate of adsorbed contaminants. The diffusion rates are site specific and related to the physical setting and type of contaminants (Parker & Mohr, 1996). Pump and treat can be an efficient containment strategy, however.

A pump-and-treat system was implemented at the site in mid-2002. Three boreholes were pumped at different rates as determined by a basic 2D flow model. In essence this involved pumping RW1 at the highest rate followed by RW2 and then PW2, where the drawdown was never supposed to be more pronounced than at the site. The pumping of PW2 at the onset of the free phase release had already created a much larger secondary source area than would have been otherwise.

The water extracted was pumped through a separator system, which retained the free phase and discharged the waste water to the sewer system. Using a sewerage treatment plant for remediation of petroleum hydrocarbons proved to be effective (Riser-Roberts, 1998).

Although the pumping system was installed to contain the contamination, it was also decided to determine to what extent natural attenuation contributed to addressing the contamination. A specific aim was to establish whether microbial degradation was taking place, as this would speed up the remediation of dissolved phase contaminants.

3 WHAT IS MONITORED NATURAL ATTENUATION?

Natural attenuation takes place at all sites where contaminants are present. No contaminant plume is infinite in length and at some stage plume movement will be contained by the depletion of the source zone, limits in solubility, adsorption, dilution, biodegradation, hydrogeological boundaries or volatilisation, to name a few attenuation processes. The potential risk to receptors is limited or reduced through natural processes that take place and are dependent on the type of contaminant (USDOE, 1999). Depending on the contaminant type(s) of concern, daughter products formed through natural attenuation should be taken into account, as some compounds degrade to more toxic forms. Natural attenuation takes place to some degree without intervention and may take the form of physical, chemical or biological processes. Reducing the mass, toxicity, mobility, volume or concentration of contaminants in soil and groundwater translates to remediation. The question is whether these processes are taking place at a sufficient rate to be protective of human health and the natural environment. Also, monitored natural attenuation (MNA) is usually implemented along with some form of active source control (USEPA, 1999).

3.1 *Biodegradation of petroleum hydrocarbons*

An important natural attenuation process for petroleum hydrocarbons is biodegradation, where hydrocarbons are destroyed by aerobic and anaerobic bacteria in soil and groundwater (Kremesic *et al.*, 2000). Petroleum hydrocarbons present a source of electron donors, and the microbes need to utilise electron acceptors during the biodegradation process. Different electron acceptors offer different energy yields and microorganisms utilising electron acceptors with higher energy yields can often out-perform other microorganisms. Consequently, the highest energy-yielding electron acceptors are preferred.

Oxygen offers the highest energy yield for microbial metabolism. When the oxygen supply is limited, as can be expected in a fractured aquifer at depth, or if it is depleted due to bacteriological growth (Fetter, 1999), the anaerobic process becomes significant. Although aerobic degradation processes occur at a faster rate, the large supply of anaerobic electron receptors generally found in the subsurface can make this process more important in the long run, as well as being site specific (USEPA, 2004).

Nitrate, manganese oxides and iron oxides (forming Mn(II) and Fe(II) respectively when reduced) and sulfate follow in terms of a decreasing energy yield (Stumm & Morgan, 1996 and Schwarzenbach *et al.*, 2003). It is thus possible to assess by means of secondary lines of evidence whether biodegradation is taking place at a site. This is achieved by monitoring the concentrations of electron acceptors such as oxygen, nitrate and sulphate, as well as products of reduction such as dissolved manganese and iron. Monitoring the groundwater composition inside and outside a contaminant plume could provide evidence that biodegradation is taking place without having to complete microbiological studies.

3.2 *Monitoring requirements for monitored natural attenuation*

The objectives of the monitoring programme are:

- to keep track of the plume boundaries;
- to detect changes in geochemical parameters indicative of a change in natural attenuation rates;
- and to measure contaminant concentrations that could progress towards remedial objectives (USEPA, 2004).

Monitoring data are also used to ensure that no unacceptable risks to receptors develop, or that no new releases take place to change the remedial approach (USEPA, 1999). The monitoring programme is site specific and tailored to the situation, allowing changes in the remedial approach, if necessary. Without sufficient monitoring, it cannot be assumed that natural attenuation is taking place.

In order to prove that natural attenuation is taking place at a sufficient rate to remediate contamination, it is necessary that the following be achieved (USEPA, 1999):

1) Demonstrate that natural attenuation is occurring according to expectations,
2) Detect changes in environmental conditions (e.g. hydrogeologic, geochemical, microbiological, or other changes) that may reduce the efficacy of any of the natural attenuation processes,
3) Identify any potentially toxic and/or mobile transformation products,
4) Verify that the plume(s) is not expanding down-gradient, laterally or vertically,
5) Verify that there is no unacceptable impact to down-gradient receptors,
6) Detect new releases of contaminants to the environment that could impact the effectiveness of the natural attenuation remedy,
7) Demonstrate the efficacy of institutional controls put in place to protect potential receptors, and
8) Verify the attainment of remediation objectives.

4 SAMPLING METHODOLOGY

Groundwater samples were collected from the boreholes shown in Figure 27.5. Sampling for the presence of petroleum hydrocarbons at the site takes place every six months, while monitoring for degradation indicators occurs once per year. While it is usually not necessary to purge wells for petroleum hydrocarbon sampling as described by the American Petroleum Institute (API, 2000), boreholes were purged before sampling during inorganic sampling runs. During purging the dissolved oxygen, pH, Eh, EC and temperature were measured in a flow cell, with the samples taken only after the measurements stabilised within 10% of three consecutive readings taken at 2-minute intervals. Separate samples were taken for the petroleum hydrocarbon analysis, major ion analysis and dissolved metals (iron and manganese) analysis. Each sample for metal analysis was filtered (through a 0.45 micrometer filter) and acidified to a pH of below 2 to prevent precipitation of dissolved metals. Samples were kept on ice and analysed within their respective holding times.

5 MONITORING RESULTS

5.1 *Petroleum hydrocarbon results*

In May 2002 the dissolved phase plume extended over a large area, affecting PW3, 4, 5, 6 and 7. Several residents have, on request, stopped using their boreholes to prevent the significant spreading of the dissolved phase plume. In April 2003 dissolved phase was present in MW1, PW3, 5, 12 and 17. By September 2004 the dissolved phase plume was restricted to one borehole (PW5) outside the pumping zone and forecourt. The erratic pattern is partly due to the fact that uncontrolled pumping takes place in the town and also because the concentrations in some of the wells are close to detection limits.

The time series dissolved phase concentrations for PW5, where persistent dissolved phase concentrations have been found, and are shown in Figure 27.2. Apart from the total petroleum hydrocarbons (TPH), the data show asymptotic behaviour. It is expected that some seasonal fluctuation in dissolved phase concentrations will occur, especially since release from storage in the matrix will be unpredictable. PW3 previously had free phase product, while during the last sampling run even dissolved phase concentrations were below detection limits.

Free phase has been fairly persistent in the three wells used for recovery (RW1, RW2 and PW2), as well as in PW1, also situated on the forecourt of the filling station. Available evidence indicates

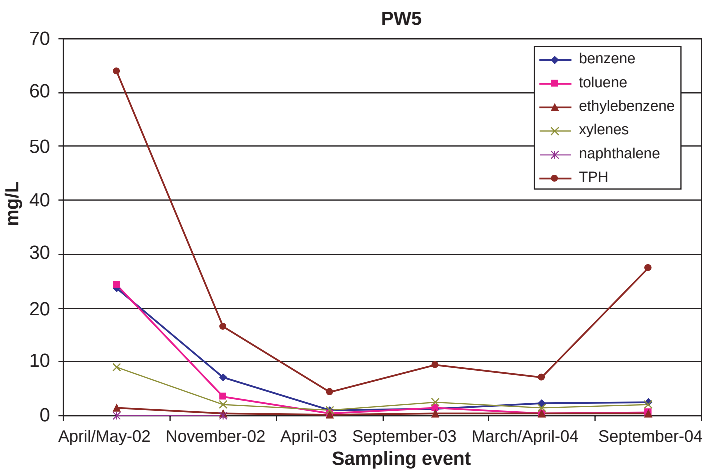

Figure 27.2. Dissolved phase concentrations over time in PW5.

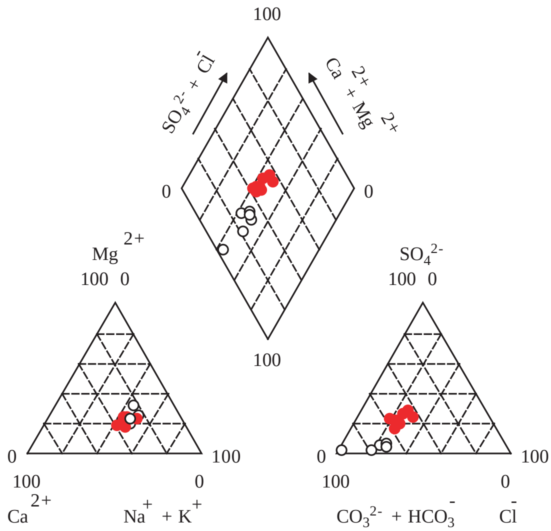

Figure 27.3. Piper diagram illustrating the relative major ion composition of the groundwater samples taken. Samples RW1, MW1, PW1, 2, 3 and 5 are indicated by the clear dots.

that the dissolved phase petroleum hydrocarbon contamination plume is contracting in the high permeability zones while pumping takes place close to the high concentration source area.

5.2 *Inorganic sampling results*

The relative major ion compositions (in terms of percentages of the total ion make-up) of the samples collected in September 2004 are illustrated in Figure 27.3 The Piper diagram indicates that the sampled boreholes fall into two groups, with the relative contributions of the non-conservative anions SO_4^{2-} and HCO_3^- distinguishing them. The observed differences in SO_4^{2-} levels between Boreholes RW1, MW1, PW1, PW2, PW3 and PW5 and the rest of the boreholes occurred not only in terms of relative concentrations as expressed by the Piper diagram, but also in absolute terms. SO_4^{2-} can be utilised as an electron acceptor, and with the information at hand, it is most likely that the observed pattern is due to the consumption of SO_4^{2-} by microbial processes in Boreholes RW1, MW1, PW1, PW2, PW3 and PW5.

The results of the afore-mentioned parameters of biodegradation, namely NO_3^-, SO_4^{2-}, Fe^{2+} and Mn^{2+}, are presented as radial diagrams. Figure 27.4 serves as an example of such a radial diagram and illustrates the chemical parameters chosen for each axis (note that the scale is not constant between the different axes). The dissolved oxygen concentrations measured after purging did not show any significant pattern. This is probably due to a sampling error. Since the chemical results indicate an anaerobic environment, it would have been expected that oxygen levels inside the dissolved phase plume would be lower than outside the plume. The variability could also be a result of aeration during pumping, where the borehole pumps of private residents were used. If aeration occurs during pumping, the possible effect of oxidation on the redox sensitive species must be noted. Eh values, which are indicative of the relative redox conditions, were plotted in place of the dissolved oxygen concentrations. A background graph (chosen to be represented by MW15 and representative of the inorganic chemistry of the samples outside the dissolved petroleum hydrocarbon plume) is indicated on the figure as the background values.

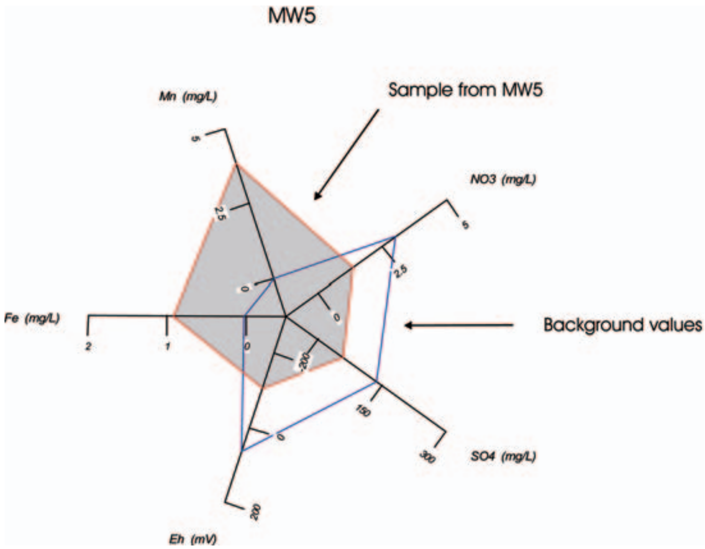

Figure 27.4. Comparison between background inorganic chemistry and the sample from MW5. Note that the scale on the different axes are not the same and that the Fe and Mn concentrations represent the Fe^{2+} and Mn^{2+} concentrations respectively.

The inorganic chemistry results represented in Figure 27.4 indicate that a reducing environment is present in PW5, and this is inferred to be a result of microbial degradation of petroleum hydrocarbons. The results for all the boreholes sampled during September 2004 are presented in Figure 27.5a as radial diagrams at the position of each sampling point. The grey area represents the measured parameters for each hole and the open polygon indicates background values as described in Figure 27.4. It also illustrates that the regional groundwater outside the dissolved petroleum hydrocarbon plume has a ready supply of electron acceptors, and that biodegradation is taking place within the plume and in the outer boundaries of the plume. The boreholes inside the plume have low levels of nitrate and sulfate and high levels of reduced manganese and iron. The regional groundwater should be able to supply electron acceptors to the affected aquifer, considering the groundwater flow speeds of approximately 35 metres per year, as calculated with the numerical groundwater model. The assumption of a steady supply of electron acceptors seems to be supported by the observation that the radial diagram profiles for MW1, PW6 and PW17 have changed towards a more oxidised environment since the September 2003 sampling run (Fig. 27.5).

6 CONCLUSIONS

The dissolved phase plume seems to be shrinking since monitoring started at this site. In addition to the apparent containment effect of pumping at the source area, sufficient evidence exists to indicate that biodegradation of petroleum hydrocarbons is taking place at the site under investigation and that the groundwater environment has the capacity to continue facilitating biodegradation. The pump and treat system is deemed to be invaluable in containing the spread of the free phase, whilst biodegradation is taking place in the plume, and the hydrodynamic control exercised by the pumps would continue to form part of the remedial strategy.

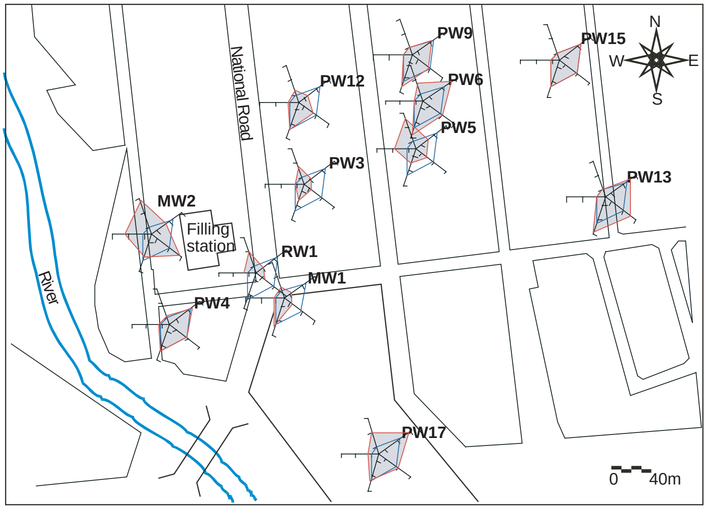

Figure 27.5. (a) Inorganic composition of water samples collected from wells in the vicinity of the site during the September 2004 sampling run.

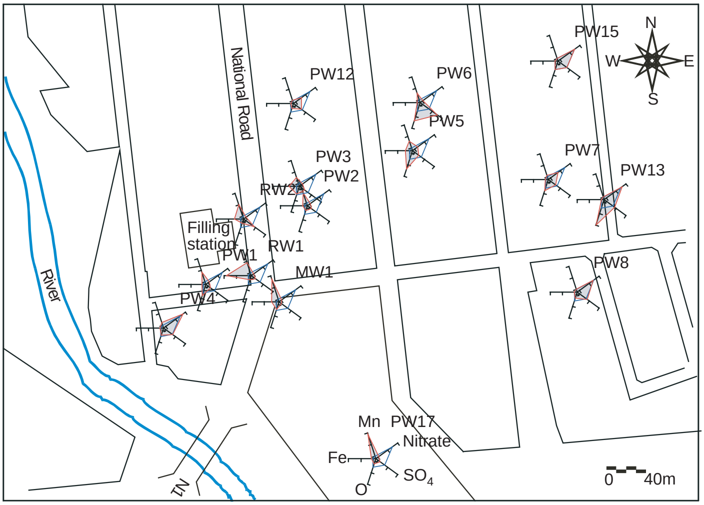

Figure 27.5. (b) Inorganic composition of water samples collected from wells in the vicinity of the site during the September 2003 sampling run.

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REFERENCES

API, 2000. American Petroleum Institute: *No-Purge Groundwater Sampling*, API Bulletin No 12, October 2000.

ASTM, 2002. Standard guide for risk-based corrective action applied at petroleum release sites, E 1739, ASTM International, USA.

Botha, J.F., Verwey, J.P., Van der Voort, I., Vivier, J.J.P., Buys, J., Colliston, W.P. and Loock, J.C., 1998. Karoo Aquifers – Their geology, geometry and physical properties. Water Research Commission Report No: 487/1/98. Water Research Commission of South Africa, Pretoria.

Fetter, C.W., 1999. Contaminant Hydrogeology. Second edition. Prentice Hall.

Kremesic, V., Yang X., Jeng C., Fisher B. and Curran L., 2000. Natural attenuation as a remedial alternative, Technical Guidance. BP Corporation (unpublished).

Le Grand, H.E. and Rosen, L., 2000. Systematic makings of early stage hydrogeologic conceptual models. GROUND WATER; V38 N6; P887–893.

Lyman, W.J., Reidy, P.J. and Levy, B., 1992. Mobility and Degradation of Organic Contaminants in Subsurface Environments. C. K. Smoley, Michigan.

Parker, T. and Mohr, T., 1996. Symposium on natural attenuation of chlorinated solvents in groundwater – A summary. HydroVisions On-line, Volume 5, No. 4, Winter 1996.

Pretorius, J., Usher, B. and Phophi, S., 2003. The Establishment of Regulatory Processes for dealing with Leaking Underground Storage Tanks in South Africa: Phase 1- Discussion Document . Draft report prepared by the Institute of Groundwater Studies for Department of Water Affairs and Forestry.

Riser-Roberts, E., 1998. Remediation of Petroleum Contaminated Soils: Biological, Physical and Chemical Processes. CRC Press.

Schwarzenbach, R.P., Gschwend, P.M. and Imboden, D.M., 2003. Environmental Organic Chemistry. Second edition. Wiley-Interscience.

Stumm & Morgan, 1996. Aquatic chemistry: Chemical Equilibria and Rates in Natural Waters. 3rd Edition. Environmental Science and Technology. A Willey-Interscience publication, John Wiley & Sons, USA.

USDOE, 1999. Decision-Making Framework Guide for the Evaluation and Selection of Monitored Natural Attenuation Remedies at Department of Energy Sites. Office of Environmental Restoration, US Department of Energy.

USEPA, 1999. Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites. OSWER directive number 9200.4-17P. http://www.epa.gov/swerust1/directiv/d9200417.htm

USEPA, 2004. How to Evaluate Alternative Cleanup Technologies for Underground Storage Tank Sites: A Guide for Corrective Action Plan Reviewers. Chapter 9: Monitored Natural Attenuation. (EPA 510-B-94-003 and EPA 510-B-95-007). http://www.epa.gov/swerust1/pubs/tums.htm

USEPA, 2004. Performance Monitoring of MNA Remedies for VOCs in Ground Water. EPA/600/R-04/027.

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Management strategies to mitigate clogging and biofouling in production boreholes

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ABSTRACT: An effective operational strategy is required to deal with the possibility of borehole clogging and biofouling in a production wellfield for potable water supply. Production boreholes become clogged to varying degrees of severity by naturally occurring bacteria capable of inducing biofouling in the aquifer, borehole screens, pumps and pipework. The clogging reduces the yield of the boreholes and ruins equipment. This is a worldwide problem and research to combat it or to rehabilitate affected boreholes has been conducted by others mentioned in the references. The most important issue to be highlighted is that once the potential problems have been identified, an effective management regime must be established on site to keep the boreholes in operational order. Success hinges on staff training and awareness to the problem of clogging and its detrimental effects. Three case studies are discussed to illustrate problems and operational strategy.

1 INTRODUCTION

This article examines the case for developing an effective operational strategy to deal with the possibility of borehole clogging and biofouling in a production wellfield for potable water supply. Management strategies are proposed which should limit and control the detrimental effects of clogging and deterioration of the boreholes.

The aim is to give some background to the problem of biofouling, its significance and effect on operational output and to outline the wellfield management initiatives needed under the circumstances to cope with the problem and maintain the best operational conditions.

Three case studies are used to illustrate biofouling problems in production wellfields supplying municipal undertakings, as opposed to citing research alone. The value of the case studies lies in being able to relate them directly to the concluding remarks, which contain a checklist of current best practice. Best practice is a high goal to attain, consequently each of the cases has some shortcomings due to particular local circumstances and where these are highlighted, it is not the intention of the author to criticize but to point out the consequent lessons for the future. The checklists therefore should be seen as a benchmarking tool.

One of the aims of this article is to give the reader sufficient material to understand the ramifications of the problem in the absence of being able to obtain all the references, other than from good library facilities.

2 THE BASICS OF CLOGGING AND BIOFOULING

Biofouling, in the context of water production boreholes, is defined as the proliferation of a biologically derived residue or product, which hampers the use of the resource. Generally, this is caused by thin biofilms containing consortia of bacteria in a protective slimy matrix which fixes and precipitates ions into minerals (combinations of amorphous, solid and complex deposits). In different parts of a borehole and depending on environmental conditions, the bacterial colonies range from aerobic to anaerobic and sulphate reducing to iron reducing, most commonly. These biofilms are very resistant to elimination by chemicals and grow in highly turbulent conditions, such as inside pumps, screens and gravel packs.

No natural water on earth is sterile, including groundwater from whatever depth and temperature (see Chapelle, 2001). Groundwater is generally in chemical and ecological equilibrium with its surroundings in the aquifer, which obviously includes the bacteria arising from the soil, sediments, rock substrates and recharge water. Heterotrophs utilize organic compounds and are the most widespread microorganisms in subsurface environments. Chemolithotrophic (meaning chemical rock eater) bacteria exist in extreme circumstances up to great depths in the pores and fissures in the earth's soil and rock utilising reduced inorganic or organic chemicals such as H_2, H_2S, Fe^{2+}, NH_3 or CO_2 from the available rock minerals for their metabolism. The controlling influence of aquifer microbiology on water chemistry has yet to be fully understood. This equilibrium can vary spatially and over time. The changes occur when groundwater comes to the surface either naturally or in this case, when pumped up the borehole, changes that are both chemical and ecological. This can cause certain bacteria to proliferate, which are the root of so-called biofouling (this term is probably a misnomer and could rather be thought of as an unstable ecological transition). One should therefore think of this ecological change as being an inevitable result of the action of changing the natural flow of the groundwater and by diverting it to the surface.

The presence of low levels of metal ions in solution, mainly iron and manganese, provides the nutrient source for biofouling to occur, when the right ecological conditions prevail. Cullimore & McCann (1978) report that concentration levels for Fe of 0.2 to 1.0 mg/litre and for Mn of 0.05 mg/litre upwards (no upper limit mentioned) in running water conditions (boreholes) are considered sufficient to provide the nutrients for clogging.

Biofouling is a condition, which develops in stages over time and involves colonisation of the well area and the aquifer by the various bacteria. The bacteria have various growth phases, which have been observed (Droycon Bioconcepts Inc.). These are briefly:

- Initial colonization by natural and introduced bacteria in the well.
- Primary void volume occupancy, occurs when consortial (community) biofilms develop in the porous media of the well and aquifer. This involves growth competition and fighting for space between these biofilms. Transmissivity is adversely affected by expanding colonies filling the porous matrix.
- Primary stabilisation occurs when the biofilms find their various niches in the matrix and reduce their size in pore space. Transmissivity stabilises and the well production settles down (becomes developed).
- Secondary void volume occupancy, occurs when a tri-phasic process of cyclic (a) growth, (b) sloughing off of excess biomass, followed by (c) restabilization, is set up. In this situation fluctuations of the transmissivity occur and well production is similarly affected. Chemical concentrations also show fluctuations.
- Clogging, occurs when the void volume is taken up by the colonies. This severely reduces well production.
- Total clogging, occurs when the clog formations mature and hard deposits of residue bind the porous formation or screen. Well production may even cease at this point. Generally, the biomass and its associated mineral precipitates plug up the interstices of the aquifer, gravel pack, screen and pump.

It is generally known that excessive pumping stimulates biofouling to take hold and exponentially increase. The yield of an individual borehole, which is sustainable or safe and does not lead to biofouling problems, is a conundrum facing many wellfield managers. Too often pumping tests have set optimistic yields in excess of sustainability. It is generally thought that laminar flow through the screen rather than turbulent flow is best practice to avoid biofouling, and similarly for flow

Figure 28.1. Photographs showing the evidence of iron related bacterial clogging at Atlantis in 1991; left, 'red water' is caused by the sloughing of the clogging products inside the borehole; centre, the clogging is encrusted around the submersible pump motor inhibiting effective cooling; right, the pump entry screen is clogged and ultimately the inside of the pump.

through the gravel pack or fissures. This is not an easy calculation because the flow paths are essentially random and non uniform, but the careful interpretation of step pumping tests does reveal the behaviour of the well screen and near regions of the aquifer and the transition from 'laminar to turbulent behaviour', which is difficult to relate to the hydrodynamic concept of Reynold's Number. Seasonal and regional level fluctuations in semi-arid climates also complicate matters.

Clogging caused by biofouling is a well-known phenomenon world-wide and it has often been ignored and ineffectually managed because of its intractability. The management initiatives needed for wells to combat biofouling associated problems have not generally been well understood in the wider community of Water Engineers and Geohydrologists. Microbiologists have provided the main thrust in the research of biofouling and its control. References as early as the 1930's mention the problems of iron bacteria and the fact that wells need treatment occasionally to remove and control the problem (Cullimore & McCann, 1978). Research reports dating back to the 1890's identify iron bacteria contaminating water supplies.

Research into the problem of biofouling has been done for many years but only gathered momentum by the early 1990's resulting in the current technology and understanding of the problem (Cullimore, 1992; US Army Corps of Engineers, 2000; Howsam, Mistear & Jones,1995).

Droycon Bioconcepts Inc (Cullimore) lists various techniques used in the diagnosis of failing wells:

- Early Techniques were based on the user observing deteriorations which could sometimes be dramatic. Such events typically being:
  - Losses in flow.
  - Serious increases in drawdown to achieve flow.
  - Generation of turbidity or colour in the water.
  - Appearance of taste or odour.
  - Increasing hygiene problems.
- Recognition of biofouling as an intrinsic concern covers three key aspect; water production, chemical & physical characterisation and biological aggressivity. The factors and parameter are typically:
  - Routine pump testing to determine the projected production capacity loss.
  - Iron content of the water changing with time, growth of deposits in pipes and tanks etc. Iron tends to bio-accumulate close to the well and screen. The critical range of concern being 0.1 mg/l (no obvious problem) and 1.5 mg/l (obvious problem).
  - Manganese content and the Fe:Mn ratio. Manganese tends to bio-accumulate further into the aquifer formation than iron. Hence when Mn dominates the ratio the clogging is likely to be

developed deeper into the formation and more resistant to treatment. The critical range of concern being 0.01 mg/l (not normally a major problem) and 0.5 mg/l (obvious problem).
- Total suspended solids may be seen as a reduction in clarity (turbidity increase) and/or colour change. This can be in the form of visible particles down to colloids. Analysis to determine the particle distribution will give clues as to bacterial growth and die-off phases.
- pH (acid/alkali) is an indicator of the suitability of the water for bacterial activity and its diversity. Optimal conditions for many microorganisms are in the range 7.2 to 8.8. However, many bacteria survive in extremely acidic or alkaline waters.
- Redox (reduction – oxidation potential) appears to be a reliable indicator to assess the potential for biofouling being in progress. The 'redox front', where conditions change from reductive (-50 mV) to oxidative ($+150$ mV), appears to be where the biofouling activities concentrate. The redox values in the borehole give some clues as to the location and state of biofouling activities.
- Temperature of the water in the formation has a direct influence on the types of bacterial able to colonise the water and the diversity.
- Total dissolved solids influence the range and diversity of microorganisms able to tolerate osmotic stress caused by salts. Generally the greatest diversity is found in the range 10 mg/l to 35,000 mg/l (sea water).
- Nutrients are obviously required to support bacterial cell growth. Three principal elements are associated in cell synthesis and metabolic strategies; carbon, nitrogen and phosphorous are required to be present in the aquifer. The C: N: P ratio has a bearing on the bacterial population present. Sampled water is often depleted of the key nutrients because the bacterial community present in the aquifer has already depleted it.
- Bacterial activity reaction (BART) tests. BART's are test kits developed by Cullimore (1992) and marketed commercially by Hach Chemicals, the purpose of which is to incubate water samples to test for the presence of various bactericidal species implicated in biofouling. Kits are available for iron related bacteria (IRB), slime formers (SLYM), sulphate reducing (SRB), nitrifying (NRB) and facultative (FACT). These test help to establish which forms of bacteria are dominant when biofouling is suspected.

Clearly, the design of the monitoring programme and methodology must be given careful attention to pick up changes and early warning signs related to clogging.

3 MONITORING IS A CORNERSTONE OF MANAGEMENT

Groundwater is a hidden resource and our knowledge of it comes from specific scientific observations, research and exploration at discrete points. Natural sources of groundwater, springs, seeps, wetlands and rivers provide us with easily accessible opportunities for observation at ground level, whereas manmade structures, wells, tunnels and boreholes give us opportunities for observing the hidden behaviour of the groundwater flow system.

In order to understand the groundwater resource and its ecology in the broadest sense, well selected and designed scientific observations and analyses are required. Clearly, monitoring in the sense of a structured continuous programme of scientific data collection is required.

The duty of care of our environment has been incorporated in law in most countries to formalises societal requirements, in particular for the sustainable use of groundwater. Sustainability can only be measured by analysis and diagnosis of monitoring results.

The success, or indeed the continued understanding of a management strategy designed for the sustainability of a potable groundwater supply, hinges directly on the ability of the monitoring programme to pick up relevant data suited to the management model.

Monitoring is an overhead that must be carefully managed to ensure that this expense is seen to be optimally beneficial to the sustainable utilisation of the resource. There is evidence of a tendency for managers in developing economies to expand the production capacity of a groundwater undertaking to keep up with demand without a commensurate increase in the monitoring. It is important for managers to grasp the strategic importance of monitoring and to provide the funding and infrastructure for it as an integral part of a groundwater abstraction scheme. If this is not done there is overwhelming evidence that borehole yields and water quality as well as viability will decline with time.

The case studies show up the fact that where monitoring is done systematically and continuously analysed clogging can be diagnosed and remedial action can then be implemented early to limit its detrimental effects on production and equipment.

4 DETERIORATION OF BOREHOLE YIELDS AND RELIABILITY

In general, when boreholes are established for water supply they are first subjected to various pumping tests to ultimately establish the yield of the particular borehole, along with the behaviour of the aquifer and hydraulic parameters. Once the safe yield is set and the pumping plant is designed to deliver this, the expectation of the borehole owner is that this yield will remain constant and be reliable, except possibly in extreme droughts. More often than not over time the yield and reliability deteriorates much faster than can be ascribed to failure of the borehole components and pumping equipment. This loss of performance can have severe cost and disruptive consequences if it is not anticipated or managed.

In the CIRIA Report 137 (Howsam, Mistear and Jones, 1995) the results of an international survey into this problem are summarized:

- Monitoring is given a low priority world-wide because of a lack of finance, organisational or logistical resources, or insufficient expertise or coordination between departments involved.
- The commonest causes of operational difficulties are sand ingress, fines migration and iron fouling, followed by encrustation, corrosion, anaerobic clogging and over-abstraction. Over-abstraction refers to inappropriate operation of the pumps such as one or more of the following; pumping individual boreholes or groups of boreholes at too high a rate, leading to interference between boreholes or adverse drawdown or oxygenation of the aquifer or regionally lowered water tables.
- Borehole operational problems often reflect poor design or construction, and operational and borehole pump problems outweigh those associated with the borehole structure itself.
- Although estimates of the percentage of boreholes which suffer from impaired performance ranged from 0% to 100% (*sic, presumably per respondent*), the available information suggests about 40% as a realistic figure for the number of boreholes world-wide with operational problems.
- Experiences of rehabilitation attempts are commonplace and there is a general awareness (as opposed to a good understanding) of available rehabilitation technologies.
- Approximately 66% of rehabilitation attempts are unsuccessful, owing to a lack of monitoring data resources or expertise, or to the degree of dilapidation of the boreholes.
- There are many reported instances where rehabilitation could have been attempted but this was not done because of a lack of finance, equipment and expertise; and also because it was felt that drilling a new borehole would be simpler and more likely to succeed.
- The survey also showed that attitudes are changing and that the need for a greater emphasis on monitoring, maintenance and rehabilitation is receiving recognition by international funding agencies. To summarise the current situation, it is increasingly recognised that groundwater is a vital resource and that boreholes are important assets whose efficiencies need to be monitored and maintained.

McLaughlin (2001) has provided a succinct and practical reference manual which explains the factors which cause the deterioration of due to fouling and corrosion. The purpose of this publication is to 'embed these (such as described in this article) understandings of the scientific basis of well deterioration processes within management frameworks to provide a comprehensive approach to dealing with water well deterioration'. This is highly recommended reading and a useful reference manual for operational staff.

Indeed, the author believes that these observations are still relevant and that the issue of borehole reliability is becoming more important.

5 THE ELEMENTS OF OPERATIONAL STRATEGY

Biological activity is present in all aquifers, which may become a problem over time when boreholes are drilled to abstract water, as outlined above. It is an inescapable conclusion that the geochemistry of groundwater is controlled by the aquifer lithology and microbial ecology and that abstraction via a borehole will upset this delicate balance (refer to Chappelle, 2001). To ignore this possibility carries the risk of an expensive lesson. However, blithe indifference seems to have prevailed as long as the borehole owner was able to establish that the water was potable or could be treated for some use. This is probably due to the problem being mainly hidden and inaccessible.

In the past, most effort was put into the development of reliable drilling methods and pumping technology. Little emphasis was placed on systematically addressing issues of borehole deterioration, maintenance and rehabilitation, or the improvements in borehole performance, reliability, cost-effectiveness and levels of service that can be achieved by the adoption of proper procedures (Howsam, Mistear and Jones, 1995).

In the briefest terms the operational stategy for groundwater abstraction must encompass the following elements for every borehole as well as the rest of the water supply system (adapted from Howsam, Mistear and Jones, 1995):

- Monitoring and Diagnosis. This includes, processes affecting the borehole performance, parameters to be monitored, monitoring methods, data handling and analysis, methods of diagnosis. Step drawdown pumping tests provide an important diagnostic tool with the methodology discussed below for clarity.
- Maintenance and Rehabilitation. This includes aims, objectives and methods.
- Design, Construction and Operation. This includes their influence on optimum performance and the understanding of best practice.
- Aquifer Characteristics. This includes the understanding of the aquifer and its propensity for borehole problems and the groundwater quality associated with it.
- Aquifer Recharge and Protection. This includes the regional context of the aquifer, groundwater flow paths, location and sources of recharge, groundwater pollution problems, protection requirements and land-use management.
- Policy and Legislation. This includes the prevailing regulatory control as well as the operating authority's management policy.
- Economic and Practical Benefits. This includes the evaluation of cost-effectiveness, capital investment analysis, replacement scheduling, labour and overheads.
- Staff Training and Efficiency. This includes qualifications, job-site training schemes, auditing, environmental awareness and reporting lines.
- Service Delivery. This includes service level agreements with customers and response plans.
- Developing an Integrated Strategy. This is probably the most important consideration because it must encourage a well-managed system involving preventative maintenance and reduce the dependence on reactive management responding to an event only when a system fails.
- Research and Development. This refers to the need to direct scientific work into developing tools such as early predictions of the potential for impaired performance of boreholes due to biofouling, physical and geochemical changes, and environmentally acceptable rehabilitation technology.

The three case studies below will demonstrate the varying effects of the problem and the strategies adopted while maintaining wellfield production.

Table 28.1 gives details of some techniques available for the removal and control of clogging problems. The most appropriate technique for a particular borehole requires a clear understanding of the borehole construction and the nature of the bacterial colonies present as well as the advice of skilled practitioners with local knowledge.

Table 28.1. Remediation techniques commonly in use to combat clogging.

Technique	Process	Where applied
AquaFreed®	Liquid carbon dioxide injection cleaning	*
Wessoclean®	Penetrating cleaning agents and bactericide	*
Calgon®	Polyphosphate sequestrating agent	*
Airburst®	Compressed air micro-burst cleaning	*
Vyredox®	In-situ iron fixation in the aquifer by redox adjustment	*
BCHT®	Blended chemical heat treatment (3 stage) using heated blends of surfactant, acid, chlorine, penetrating agent and neutralizing alkali	Atlantis and Klein Karoo
Acid wash	Dosing with an acid such as citric, hydrochloric, sulphamic and hydroacetic	Previously used at Atlantis and Klein Karoo and found to be of very short-term benefit.
Hyperchlorination	Very high dosage of hypochlorite solution to disinfect and oxidize bacteria	Perth
Hypochlorite generation	Electrolysis of high chloride natural water to periodically disinfect down hole during operation as an inhibitor	Perth
Reconstruction of bore	Sealing off unwanted water strikes to prevent cascading and oxygenation above drawdown level	Klein Karoo
Refurbishment of casing	Replacement of defective sections of casing and screen	Klein Karoo
Mechanical cleaning	Down-hole tools such as brushes, surge blocks, bailers, jetting pumps and packers in combination with the above methods to remove clogging waste	Atlantis, Klein Karoo and Perth

* Techniques used elsewhere in the world but not reported as part of this article, mentioned here because these have been considered by the author as potentially useful at Atlantis. These are patented chemical formulations, equipment or methods. Some of the license holders have websites for further information. There are more proprietary techniques than the ones mentioned above.

6 WELL PERFORMANCE TESTS

The purpose of well performance tests is to determine the behaviour of the well, its components (screen, casing, gravel pack) and the nearby region of the aquifer, all of which may be clogged to varying degrees. After the well has been developed, the initial step test is done to determine its performance and to provide a benchmark for comparison at later stages in the operational life of the borehole. When clogging or reduced yield is detected, a subsequent step drawdown test will indicate the extent of the problem and provide an analytical tool to decide on a course of action. The effects of the clean-up or rehabilitation of the borehole can be analysed by a follow-up step drawdown test. In other words, the original, before and after tests can be used to identify the clogging problem and determine the effectiveness of remediation, as can be seen from Figure 28.2.

Kruseman & de Ridder (1989) provides a comprehensive explanation of well performance tests and analysis such as the Hantush-Biershenk method, which is applicable to most borehole situations i.e., confined, unconfined or leaky aquifers.

The test is done by pumping the aquifer step-wise at increased discharge rates, without allowing the aquifer to recover between steps. The initial test is done with not less than 3 steps (preferably 5) up to the maximum yield of the borehole. In the subsequent tests the steps are repeated as closely as possible in order to be able to carefully compare results.

The specific drawdown curve can be used to obtain the constants B and C in the Jacob equation $(s_{w(n)}/Q_n) = B + C.Q_n$. The constants are, B, the linear aquifer-loss coefficient and C, the non-linear

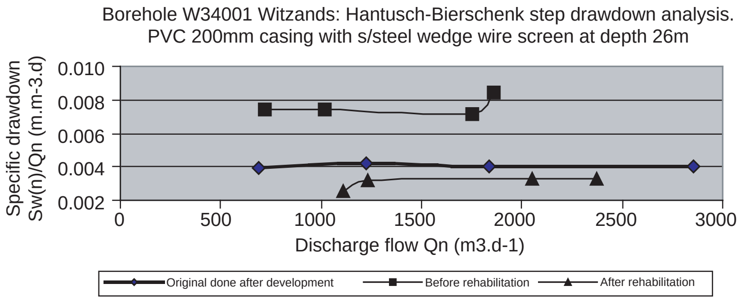

Figure 28.2. Typical result of a step drawdown test, showing the effects of borehole clogging and rehabilitation compared against its performance when first commissioned after development.

well-loss coefficient. The curve is a straight line, ideally, and departures from this indicate non-linear responses as the flux increases through the well screen and gravel pack or fissures. Abrupt changes of slope indicate possible defects in well construction or rearrangement of the gravel pack or aquifer formation. Clogging of the well will indicate values of the specific drawdown that are higher than the original test results.

Clogging is not normally expected to go far beyond the borehole into the aquifer and therefore each step (Δt) will not need to be done for longer than 100 minutes (60 minutes is suitable, in many cases). In the case of fractured rock this may be different. If interconnections exist with other boreholes or if fractures dewater leading to changes in water chemistry, case more extensive testing and analysis would be advisable in the initial test.

After the completion of the step drawdown test a recovery test may be done by recording the return of the water level (residual drawdown) in the borehole against time. This can be used to determine the skin effect (skin factor) and the transmissivity, which can be used to judge the extent of clogging when compared to the original test results.

7 CASE STUDY: ATLANTIS, WESTERN CAPE, SOUTH AFRICA

The town of Atlantis, part of the greater City of Cape Town, South Africa, has been supplied with potable groundwater from the primary coastal sand aquifer, which is a managed artificial recharge system, for the past 25 years. Consumption in 2004 was 6.2 million cubic metres of which 2.8 million was groundwater, the remainder being treated surface water from the Cape Town bulk supply.

Until 1999 the full supply was met from groundwater but declining yields placed the supply scheme in a highly stressed situation until the treated surface water could be made available to augment the supply. Generally, the declining borehole yields can be ascribed to the presence of iron bacteria and the need to maximise production. Lowering of the water table and consequent aeration of the aquifer formation are thought to have accelerated bacterial clogging. There is no clear evidence either that the artificial recharge using stormwater or that treated effluent influences clogging, mainly because remote parts of the aquifer that are still being naturally recharged are equally clogged (Tredoux, Cave and Bishop 2002).

There are 45 production boreholes, which yield from 3 to 15 litres per second (l/s), the majority in the 10 l/s range, with screens about 10 metres long at 25 metre depth in 40 metre thick unconsolidated sediments overlying bedrock (More Water c.c. 2002).

By 1997 the loss of yield had become a very serious problem, to the extent that boreholes which could potentially yield 15 l/s had declined to as little as 2 l/s. Based on work done at the Klein Karoo (described below), rehabilitation work was done on 39 boreholes using the patented Blended Chemical Heat Treatment (BCHT) process (Droycon Bioconcepts Inc. and Allford & Cullimore, 1999). In the majority of cases the yields were returned to the original determined when the borehole was first put into production.

BCHT is a relatively expensive process costing on average ZAR 50,000 per borehole, which is roughly half of the cost of developing and equipping a new borehole (ZAR 6.5 = US$ 1). It was necessary because the clogging had been allowed to go too far, requiring aggressive remediation. Regular cleaning as soon as deterioration is beyond an early threshold has been recommended as a more cost effective strategy for long-term sustainability.

The clear lesson learnt from this is that a strategy of monitoring to detect early signs of biofouling and yield drop-off must be adopted and coupled with appropriate equipment and staff training for rapid response. Regular pump removal together with simple borehole equipment cleaning, when clogging is detected, has to be adopted as a routine action to keep individual borehole yields at optimum values based on pump test analysis. When this was done only after the yield had been severely affected, expensive rehabilitation treatments had to be implemented, with consequent service.

8 CASE STUDY: KLEIN KAROO, WESTERN CAPE, SOUTH AFRICA

The Klein Karoo Rural Water Supply Scheme (KKRWSS) supplies household and stock water to the farming community in the Oudshoorn valley area of the Western Cape, South Africa. Groundwater is abstracted from the Table Mountain Sandstone (TMS) aquifer, which is an extensive fractured rock aquifer within the Cape Fold Belt mountain chain. The valley area is semi-arid, whereas the surrounding aquifer recharge mountain areas have some of the highest rainfalls in South Africa. Production comes from two wellfield areas on the east and west sides of the valley roughly 80 km apart.

The scheme was in full production by 1987 with a designed yield of 4.7 million m^3 per year. The yield has had to be revised to 1.1 million m^3 per year to account for pumping management requirements, borehole interconnectivity, clogging problems, spring flow impacts, recharge limitations, poor understanding of flow paths and inadequate borehole construction (Less, 2003 etc.). There are 18 production boreholes in fractured rock aquifer, with yields ranging from 0.5 to 19.4 litres per second with depths from 137 m to 249 m (Kotze, 2001).

The TMS aquifer consists of thick sedimentary deposits separated by aquitards, the upper Nardouw and lower Peninsular formations being the main water sources (Kotze, 2001). Dissolved iron concentrations are highest in the Nardouw formation and lowest in the Peninsular formation and fluctuate in all of the wellfield boreholes. Iron related bacterial clogging problems were experienced within a short time after production commenced, to varying degrees in the individual wellfields. This clogging severely reduced yields and compromised operational efficiency and service delivery.

Starting in 1991, attempts were mainly made by the Department of Water Affairs and Forestry (DWAF) to investigate the problem of clogging, followed by borehole reconstruction and the BCHT remediation process to improve the reliability of the scheme. Recommendations were made by consultants to implement management procedures aimed at reducing the impact of clogging and maintaining yield. Jolly (2000) reported that most boreholes could not be sustainably pumped at their maximum achievable pumping rates, only at 20%.

The initiator of the scheme, DWAF, handed over the routine operational responsibility to a local water authority, while still allocating annual funding for maintenance issues such as remediation of boreholes. The operational authority has not developed higher levels skills for managing biofouling and these are still provided by DWAF on an assistance request basis. These management inputs are often outsourced to consulting Hydrogeologists. The consequence is an arm's length management strategy for mitigating clogging, resulting in expensive rehabilitation measures such as BCHT when clogging becomes intolerable.

The DWAF project cost, including the development and building of custom designed equipment, came to ZAR 15,00,000 over the period from 1994 to 1997 (ZAR 6.5 = US$ 1). This covered the rehabilitation and reconstruction of 15 holes. The work included removing the PVC casings and slotted PVC screens, which were replaced with a smaller diameter PVC and stainless steel wedge wire screens over the production zone. Productive aquifer zones were determined from geophysical logging and packer pump testing.

The lesson to be learnt is that there must be effective training of the operational staff and local resources must be readily available in order to routinely tackle the problems of clogging. Furthermore, inappropriate construction of the boreholes could lead to reconstruction and rehabilitation which is very costly.

9 CASE STUDY: PERTH, WESTERN AUSTRALIA

The City of Perth annually consumes about 250 million cubic metres of potable water and 60% if this comes from groundwater. Since the mid 1970's there has been a significantly declining trend in rainfall and more recently severe drought. There is currently intense debate on water supply alternatives, effects on groundwater and conservation issues (The Australian, 4 Feb. 2005).

Perth is situated on the Swan Coastal Plain, comprising multi-layered aquifers of extensive sedimentary deposits to depths of 2500 m (Davidson, 1995). Potable groundwater is abstracted from the superficial unconfined aquifer and the deeper semi-confined and confined aquifers. Boreholes in the superficial aquifer are between 32 m and 66 m deep and yield up to 7000 m^3 per day. Boreholes in the confined aquifers are up to 1080 m deep and yield up to 10,000 m^3 per day. There are about 390 production boreholes operated by the Water Corporation supplying the Perth region (www.wrc.wa.gov.au). The groundwater is generally treated to remove dissolved and colloidal iron to potable standard at various water treatment plants.

A well-developed and effective management strategy has become an absolute necessity for Perth, which has been achieved by the joint efforts of the Rivers and Waters Commission (regulatory role), the Water Corporation (supply authority role) and the Western Australia Planning Commission (land-use regulatory role). Given the major contribution of groundwater to the wellbeing of the region, its effective and sustainable management is of paramount importance. The management focus is summed up as quantity and quality of groundwater (Water & Rivers Commission).

The Water Corporation has a dedicated staff compliment to do borehole tests and routine maintenance rehabilitation work. There are currently 80 boreholes with severe iron bacteria related problems. A purpose built rig is available for removing pumps and doing the cleaning operation. Other more complex operations are outsourced on long-term service contracts. The strategy is in a proactive state.

The routine cleaning of the boreholes, done on average every 6 to 12 weeks, costs around A$ 600 to A$ 1000 (A$ 1.25 = US$ 1) per borehole (Hodzic and Chudzik, 2005), for labour. The initiative to replace the conventional submersible motors with higher temperature rated ones has resulted in potential savings of A$ 1,50,000 to A$ 2,50,000 per year for 32 of the worst affected boreholes.

The lesson to be learnt here is that the investment in dedicated systems and resources in a routine strategy to cope with clogging has resulted in a high level of supply reliability from groundwater.

10 CONCLUDING REMARKS

10.1 *What the case studies reveal*

The three case studies highlight biofouling as being a key factor that has to be taken into account in the management of the undertakings in order to optimise production yield. There is often reticence amongst the operational staff to admit to this and inertia to change, implying that the strategy should rather be well thought out at the start of full-scale operation of the undertaking. Each case study is different from the point of view of the aquifer characteristics and scales of production, but the common factor is that each one is a vital economic resource, which could be drastically disrupted by the effects of clogging.

The Atlantis case shows that almost nothing was known about the potential for clogging that existed at the outset of the development of the resource and hence no systematic and stepwise procedure was available either to pick up the symptoms or to deal with the management of the problem. The resulting crisis management to maintain production has been costly and reduced the reliability of the system. The design of the borehole casing and screen also plays an important role in the ability of the clogging to become intractable or the borehole being a good candidate for rehabilitation.

The Klein Karoo case again shows that the potential for clogging was not clearly understood at the outset, that borehole design and construction rendered rehabilitation difficult and costly. Split management responsibility also makes for a lack of systematic and stepwise management, costly procedures and reduced reliability.

The Perth case shows that there is a strongly developed systematic procedure to identify problems and deal with them routinely. Perth's strategy proceeds step-wise, first with monitoring of production, providing evidence of clogging and loss of yield, followed by making use of different interventions to maintain a sustainable level of production.

In contrast, the Atlantis and Klein Karoo cases show that reactive interventions (only occurring once the problem is strongly evident) are not systematic or progressive and result in disruption of service and higher costs than the more successful strategy adopted in Perth. In each case there is a different operational strategy in place and of these only Perth seems to be successfully sustainable and relatively cost effective. Unfortunately, it has not been possible to clearly define the additional cost component associated with the prevention of clogging to be added to the basic treated water cost for each case study.

The case studies also show that borehole rehabilitation alone is not successful in maintaining expected production, unless it is coupled to a well-developed strategy actively promoted by management. Of the three examples, Perth is clearly ahead because the strategy is progressive, it is sufficiently funded and the operational staff is well trained and conversant with the objectives.

The aquifer types and characteristics in the case studies are distinctly different viz. primary shallow unconfined unconsolidated sands; fractured quartzitic sandstones; superficial unconfined overlying deeper semi-confined and confined sedimentary deposits. Recharge conditions are different in each case as well as water quality parameters such as dissolved iron. Extensive iron bacterial clogging is common to all three cases.

10.2 *General points to understand*

In its natural state an aquifer is in a quasi-steady state, ecologically and geochemically. The mineralogy of the aquifer constituents and the biomass within the aquifer have the majority influence on the geochemistry of the groundwater. This can be altered by geological changes, gradually over time or by sudden events such as volcanic activity or earthquakes. Man-made changes such as mine dewatering or abstraction boreholes will alter the ecology and the geochemistry. Any event or action that alters the water table levels in the aquifer and its rate of change will influence the ecology and geochemistry.

At the outset of developing a single abstraction borehole or wellfield there must be recognition and understanding of the influence that this will have on the aquifer from the point of view of hydrogeology (drawdown, flow paths, cone of depression and influence, yield etc.), likely geochemical changes and the possibility of biofouling (ecological changes).

Traditionally, the aspect of abstraction induced ecological and geochemical changes has not always received as much attention as the hydrogeological influences on the aquifer. The reason for this is that short-term tests can be successfully carried out which adequately determine the hydrogeological influences of abstraction, whereas the ecological situation is far more obscure and requires much longer term monitoring and tests.

Aquifer parameters must be determined by careful testing programmes, with regular repeating to pick up any possible changes. Very cautious assessments should be made of the maximum achievable abstraction rate and a lower sustainable yield for each borehole. The abstraction rate should be kept as low as possible to minimise turbulent flow condition within the aquifer and especially in the gravel pack and screen: this will reduce the probability of rapid clogging occurring.

When a borehole becomes severely clogged, it is necessary to resort to the extreme forms of rehabilitation using a powerful chemical and physical process such as BCHT, which is costly and does not guarantee complete success. It is preferable to avoid this by adopting a more frequent cleaning regime as soon as warning signs become apparent rather than to delay until clogging hinders production. A more severe process may the only be required at long intervals. This requires a routine structured and progressive approach. Provided the borehole construction is designed to accept regular cleaning activities and is robust, a good service life can be expected even with occasional heavy cleaning.

10.3 *Key points and actions required for management*

In order to responsibly and sustainably use, manage and effectively maintain the abstraction of groundwater from an aquifer, including the potential effects of clogging, the following key actions are required:

- *AT EXPLORATION STAGE*
  - Environmental scoping and impact assessment. Setting up basic monitoring protocols.
  - Exploratory drilling with thorough sampling and recording of lithology, mineralogy, water strikes and water quality as the borehole progresses down.
  - Hydrogeological investigation to establish aquifer parameters using pumping tests.
  - Borehole down-hole logging and geophysics at appropriate times during the drilling and on completion.
  - Hydro-census of the presumed zone of influence.
  - Clogging and biofouling census in the presumed zone of influence to uncover likely problems.
  - Careful design of the borehole to cater for the geology to be encountered, likely nature and positions of water strikes, keeping out unconsolidated material, space for pumps and monitoring equipment down hole, expected corrosion and erosion of the casing and screen and, lastly but most importantly, a design which makes for easy and effective rehabilitation.
  - Careful design of the well screen and pipework to minimise high flow velocities and turbulence which could promote clogging.

- *BEFORE ABSTRACTION BEGINS*
  - Analysing exploratory data collection and diagnosis of likely problems with water quality, borehole drilling, potential biofouling risk etc. A series of risk scenarios should be investigated as to the severity of clogging and the consequences together with possible interventions required to mitigate service disruptions.
  - Planning and initiating the monitoring protocols and infrastructure.
  - Advance training of the staff to be allocated to the operational management of the proposed wellfield or borehole, particularly in the basic understanding of the need for monitoring and recognition of the signs of well deterioration and biofouling.
  - Setting up of an aquifer management and development multi-disciplinary team, consisting of the following disciplines: hydrogeology, engineering, microbiology, geochemistry and ecology.

- *WHEN OPERATIONAL*
  - Analysis and diagnosis of the monitoring results by properly skilled engineers and scientists.
  - Feedback and on-going training to refine and optimise the understanding of the operational staff doing the routine work in the field.
  - Adequate budgeting to be able to run the operation organisation on a pro-active basis focussed on preventative maintenance.
  - Regular reviewing of operational practice and overall management of the aquifer and its recharge zone.
  - The monitoring protocol must be reviewed regularly and audited. It must be designed to be adaptive and cost efficient in order to be optimal.

- Routine testing of the boreholes to determine production yield and comparison with previous results using specific capacity as the measuring parameter in step drawdown pumping tests.
- Routine CCTV camera inspections to examine and record the condition of the borehole screens and comparison with previous results

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REFERENCES

Alford & Cullimore. 1999. *The Application of Heat and Chemicals in the Control of Biofouling Events in Wells*. Boca Raton: CRC Press Lewis Publishers.

Chapelle, F.H. 2001. *Groundwater Microbiology and Geochemistry*. John Wiley and Sons.

Cullimore, D.R. & McCann, A.E. 1978. The Identification, Cultivation and Control of Iron Bacteria in Groundwater. In Skinner and Shewan (eds), *Aquatic Microbiology*. Academic Press.

Cullimore, D.R. 1992. *Practical Manual of Groundwater Microbiology*. London: Lewis Publishers.

Droycon Bioconcepts Inc, website: www.bbi.ca/droycon.

Hodzic, M. & Chudzik, J. 2005. Personal communications regarding clogging in Perth. Water Corporation of Western Australia.

Howsam, P., Misstear, B. & Jones, C. 1995. *Monitoring, Maintenance and Rehabilitation of Water Supply Boreholes*, *Report 137*. London: Construction Industry Research and Information Association.

Jolly, J.L. 2000. Biofouling in boreholes of the Klein Karoo Rural Water Supply Scheme, Oudtshoorn, South Africa, and the effect on well scheme management. In O. Sililo (ed), *Past Achievements and Future Challenges; Proceedings of the XXX IAH Congress on Groundwater*. Rotterdam: Balkema.

Kotze, J.C. 2001. *Hydrogeology of the Table Mountain Sandstone Aquifer – Klein Karoo.* Unpublished PhD Thesis, Department of Geohydrology. Bloemfontein: University of the Free State.

Kruseman, G.P. & de Ridder, N.A. 1998. *Analysis and Evaluation of Pumping Test Data*, International Institute for Land Reclamation and Improvement, Netherlands.

Less, C. 2003 & 2005. Personal communication. More Water CC.

McLaughlin, R. 2001. Managing Water Well Deterioration. *IAH publication: International Contribution to Hydrogeology(ICH) Volume 22*. Rotterdam: Balkema.

More Water CC. 2002. *Atlantis Wellfield Rehabilitation: Consolidation Report of Phases 1 to 5, June 2002*. Prepared for the City of Cape Town, South Africa.

The Australian, Asa Wahlquist. 4 February 2005. *The West's water war*.

Tredoux, G., Cave, L. & Bishop, R. 2002. Long-term stormwater and wastewater infiltration into a sandy aquifer, South Africa, In P.J. Dillon (ed), *Management of Aquifer Recharge for Sustainability. Proc. 4th Intl. Symp. On Artificial Recharge of Groundwater, Adelaide, Australia*.

US Army Corps of Engineers. 2000. *EP 1110–1–27, Engineering and Design – Operation and Maintenance of Extraction and Injection Wells at HTRW Sites*. Washington DC: Department of the Army USA.

Water and Rivers Commission, Western Australia website http/www.wrc.wa.gov.au.

Western Australia Planning Commission and Water & Rivers Commission. 2001. *Gnangara Land Use and Water management Strategy Final Report*.

FURTHER READING

Davidson, W.A. 1995. *Hydrogeology and Groundwater Resources of the Perth Region Western Australia, Bulletin 142*. Geological Survey of Western Australia.

Engelbrecht, Pannie and Jolly, Jeff. 1999. *Iron precipitation problems in boreholes in the Klein Karoo*. Groundwater Consulting Services and CSIR, Report to the Department of Water Affairs and Forestry, South Africa.

Maherry, A.C., Tredoux, G. and Clark, S. 2005. An examination into the iron biofouling of the Atlantis aquifer and the factors influencing borehole clogging, *In Biennial Groundwater Conference 2005, Geological Society of South Africa*.

Smith, M. 2006. *Prediction, Control and Rehabilitation of Iron Encrustation in Water Supply Boreholes, Western Cape, South Africa*. Unpublished PhD Thesis, Department of Geological Sciences, Cape Town: University of Cape Town.

Smith, S.A. 1995. *Monitoring and Remediation Wells: Problem Prevention Maintenance and Rehabilitation*, Boca Raton: CRC Press Lewis Publishers.

Tredoux, G. and Cave, L. 2002. *Atlantis Aquifer, A status report on 20 years of groundwater management at Atlantis, ENV-S-C-069.* Council for Scientific Research, South Africa.

Saayman, I.C. and Adams, S. 2002. The use of garden bores in Cape Town, South Africa: lessons learnt from Perth, Western Australia. *Physics and Chemistry of the Earth*. Elsevier.

Van der Merwe, A.J. 1983. *Explorasie, ongunning en evalueering van grondwater van die sandafsetting in die Atlantisgebied vir watervoorsiening aan die Atlantisgroeipunt*. Unpublished Msc Thesis, Department of Geohydrology. Bloemfontein: University of the Free State.

Western Australia Planning Commission and Water & Rivers Commission. 2001. *Gnangara Land Use and Water management Strategy Final Report*.

Water Research Commission Report No.TT 158/01.2002. *A synthesis of the hydrogeology of the table mountain group – Formation of a research strategy*. Pietersen, K. and Parsons, R, (eds). South Africa.

Way forward

29

A South African perspective on the protection of groundwater sources

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ABSTRACT: This paper reviews present groundwater protection measures, points out major challenges and finally tries to indicate a way forward. Special emphasis is placed on groundwater protection zoning, a pro-active protection approach practised for drinking water supplies in most First-World countries. Because rapid learning in this strategic matter is important, readers from different countries are encouraged to share their own groundwater protection experience.

1 INTRODUCTION

The supply of clean water and the provision of sanitation were identified at the 2002 Johannesburg Summit as a central issue of the Millennium Development Goals. Because of the wide-spread nature of the problem and the massive backlogs, groundwater is playing an increasing role in community water supply. However, it has already become clear that groundwater supply must not be seen as the quick and low-cost option. Even more than the well-known problem of groundwater over-abstraction, groundwater pollution is leading increasingly to scheme failure, also in rural areas. In the case of groundwater, the old adage, 'prevention is better than cure' is probably truer than in many other situations, because problems often remain undetected for a long time and once they are noticed, remediation can be very costly and even impossible. While legislation is crucial to give a clear government signal of the importance of groundwater and the need for its protection, the ultimate protection of the widespread, localised resource lies in the education of communities.

South Africa has recently introduced new water legislation, which is very resource-protection focused. Still, after seven years, much still needs to be done to make groundwater supplies sustainable. The objective of the paper is to review present groundwater protection measures, point out major challenges and finally try to indicate a way forward. Special emphasis is placed on groundwater protection zoning, a pro-active protection approach practised for drinking water supplies in most First-World countries. Because rapid learning in this strategic matter is important, readers from different countries are encouraged to share their own groundwater protection experience.

2 CONTAMINANTS IN GROUNDWATER

South Africa has a semi-arid to arid climate and water resources are unequally distributed. At the present population level of around 42 million, there are just over 1200 kiloliters per person per year of available fresh water. This places South Africa on the threshold of the internationally accepted definition of 'water stress'. Groundwater is seen as a strategic water resource, as almost two thirds of South Africa's population depends on this resource for their domestic water needs. Because surface water resources are unevenly distributed and cannot cope with the growing water demand, the dependence on groundwater is considerably higher.

However, this growing importance of groundwater is not yet reflected in the improved management of the resource. This, together with the vulnerable nature of groundwater in many parts of the country, has led to widespread pollution.

Many cases of groundwater contamination in South Africa were reported by previous investigators (De Villiers, 1987; Xu *et al.*, 1991; Tredoux, 1993; Palmer Development Group, 1993; Weaver, 1993; Van Ryneveld, 1994; Xu and Braune, 1995; Cameron-Clarke, 1997; Engelbrecht and Cave, 1997; Conrad *et al.*, 1999; Rosnor *et al.*, 2000; Parsons and Taljard, 2000; Morris *et al.*, 2000; Ntsele *et al.*, 2000; Woodford and Chevalier, 2002; Verweij, 2003 and Usher *et al.*, 2004).

The case studies suggest that major contamination risks come from:

(1) on-site sanitation;
(2) solid waste dumpsites, including household waste pits;
(3) agricultural chemicals (fertilisers, herbicides and pesticides);
(4) cemeteries and graveyards;
(5) mining industry;
(6) transport;
(7) petrol service stations (underground storage tanks);
(8) wood processing and preserving;
(9) feedlot/poultry farms;
(10) manufacturing chemicals.

In rural and peri-urban areas, groundwater often becomes the sole source for addressing the historic backlog of community water supplies in terms of the Water Services Act of 1997 under the country's Reconstruction and Development Programme. In considering the potential of groundwater contamination in a rural situation, a comprehensive list of sources should include the following (Tredoux, 1993; Van Ryneveld, 1994; Xu and Braune, 1995; Fourie and Van Ryneveld, 1995; and DWAF, 2003):

- All types of on-site latrines, such as existing unimproved toilets and improved pit latrines, and any off-site sanitation systems including water-borne sanitation.
- Solid waste dumpsites, including household waste pits.
- Grey water disposal practices (often disposed of in the garden or in a pit in the yard), especially in the informal settlements.
- Cattle kraals or feedlots where cattle and other livestock are kept within confined spaces, and cattle dip tanks.
- Graveyards and community cemeteries.
- Certain small industries, especially motor vehicle repairs, food stalls and shops, and small manufacturing enterprises.

Contaminants associated with the afore-mentioned sources are dependent on a number of factors, including the level and age of facility use, the size of the waste build-up, local topography, and precautions to prevent contamination. In general, the following two groups are often reported, according to A Protocol to Manage the Potential of Groundwater Contamination from on-Site Sanitation (Edition 2) (DWAF, 2003):

(a) microbiological contaminants, typically viruses, bacteria, protozoa and helminths, and
(b) chemical contaminants, consisting of both inorganic and organic components. The inorganic components of primary concern are nitrogen and phosphorus. The organic components of primary concern are toxins and those that decay rapidly, depleting the oxygen in the carrier water or forming odorous byproducts.

Other groups of chemical contaminants that are less frequently observed, but may still be found in domestic wastewater (Fourie and Van Ryneveld, 1995), include:

(c) refractory organics, which in turn include:
- surfactants (e.g. detergents), particularly the non-ionic variety that has seen a rapid increase in usage in recent years in place of the previously popular anionic varieties. Non-ionic surfactants are potentially problematic, because they are considered less biodegradable than their anionic counterparts.
- pesticides and agricultural chemicals.
- cleaning solvents, e.g. benzene, toluene and carbon tetrachloride, which originate from sources such as toilet bowl cleaners, paintbrush cleaners and stove and oven cleaners.
- organics produced by processing natural organics (e.g. trihalomethanes).
- mineral oils (e.g. engine oil, PCBs).

(d) toxic inorganic ions
- heavy metals from small industrial activities such as metal plating, developing and printing photographs, or engineering and paint shops.

The microbiological contaminants are capable of being removed naturally by chemical, physical and microbiological processes occurring selectively within the aquifer system, while the chemical contaminants, particularly the inorganic components, are non-degradable in nature, more persistent and would usually enter the groundwater with some reduction due to adsorption, but without any change in form.

3 THE PRESENT GROUNDWATER PROTECTION SITUATION

Recognition in legislation is crucial as a basis for appropriate resource protection. For groundwater, this was only realised with the passing of the National Water Act, 1998. Under the previous Water Act, 1956, groundwater was classified as 'private water', allowing unlimited exploitation on one's own property. Under the new legislation, groundwater is classified as a significant resource and, together with surface water, is a common resource, with the state acting as public trustee. The Act includes excellent general resource protection measures and was followed by a special groundwater quality management strategy in 2000 (DWAF, 2000).

The challenge now lies in the implementation of the available approaches and instruments. The water-use licensing process, in terms of the National Water Act, 1998, which includes the licensing of waste discharge and disposal, offers the opportunity to manage potentially serious impacts on groundwater. What is still largely lacking is proper control of the conditions set for each licence that is given. Because of the need to prioritise, many uses with limited impact are presently dealt with through general authorisations, which are also, because of capacity reasons, largely uncontrolled.

It can therefore be said that, besides the limited licensed activities, groundwater protection is still largely in a first phase (tier 1) of awareness-building and guideline stage (Xu and Braune, 1995). This must be seen together with important changes in terms of water and sanitation service provision, which has been completely devolved to local government. While this is strictly a water-supply function, the protection of local resources cannot be neglected and also fits the local government role of integrated environmental management at this level.

The tier 1 approach has not yet been fully programmatic, but has shown a growing government concern for the protection of groundwater resources. Initiatives in the last five years include:

- Implementation of the Groundwater Protocol for the major National Sanitation Programme that the Department of Water Affairs and Forestry is putting into place. The protocol has moved from a guideline (Xu and Braune, 1995) to a protocol, which is included in the implementation of a sanitation scheme, initially mainly in rural areas and in the second version also in high-density settlements (DWAF, 2003).

- According to the differentiated protection strategy proposed in the guideline, contamination risks imposed by the provision of the community water supply and sanitation projects, should be minimised through the implementation of (1) minimum borehole construction standards, (2) minimum sanitation siting requirements or a consideration of other options to reduce or prevent the contamination of groundwater if socio-economic or geographic conditions would result in the choice of a sanitation system causing significant contamination of groundwater, and (3) minimum distances between boreholes and sanitation facilities.
- 'Minimum Guidelines and Standards for Groundwater Development' are included in the conditions of any government drilling contract.
- A feasibility study regarding the registration of all water-drilling professionals as a way of raising industry standards and improving the quality and accessibility of borehole information.
- A major Government of Norway-assisted programme with regard to 'Sustainable Development of Groundwater Sources under the Community Water Supply and Sanitation Programme in South Africa'. A large number of awareness and guidance materials has already been developed and piloted in three District Municipalities for programmatic roll-out to all Water Services Authorities country-wide.

The differences between urban areas, small towns and rural areas are graded rather than distinct, and are characterised by settlement density and available financial resources. Based on version 2 (DWAF, 2003), densities in our rural areas are generally less than ten houses/ha while densities of (low-income) urban areas tend to be around 30–50 houses/ha and can be even higher. In terms of financial resources, rural areas tend to have fewer financial resources than urban areas. A further note is that many rural areas are dependent on local groundwater resources, whereas in urban – and certainly metropolitan – areas, water is often obtained from further afield, and is therefore not subject to contamination by on-site sanitation systems in the same way.

It is expected that the planned classification of all water resources in South Africa in terms of the National Water Act, 1998 could greatly help with a systematic implementation of a groundwater protection programme. The goals for specific management classes are expressed as Resource Quality Objectives (RQO). These can take various forms, depending on local issues, e.g. groundwater levels or grades of a certain water quality that need to be maintained. A RQO can also prescribe the control of land-based activities, which could lead directly to groundwater protection zoning (Xu *et al.*, 2003).

4 GROUNDWATER SOURCE PROTECTION ZONING

Many countries worldwide have implemented groundwater protection zones, also called groundwater supply protection areas or wellhead protection zones, with the special focus of protecting domestic water supplies against pollution (Foster *et al.*, 2002). These source protection zones have to protect against:

- contaminants that decay with time, where subsurface residence time is the best measure of protection; and
- non-degradable contaminants, where flowpath-dependent dilution must be provided.

For absolute protection, it would be necessary to completely eliminate all potential pollution sources from a defined area of influence around a borehole or wellfield. This will often be untenable, because of the specific development needs in an area. The most common approach has therefore been to divide the capture zone in a number of subzones, with decreasing protection requirements with distance from the source. The vulnerability of a wellfield can be jointly measured by the vulnerability of the unsaturated zone and the ease with which the contaminant in the saturated zone can move towards an abstraction point under pumping conditions. The former is related to a vertical distance that the contaminant actually travels from its loading source through the unsaturated zone to reach the water table, and the time it takes to cover the distance. The latter is described by borehole capture zones, which are often related to specific travel times or die-off times for pathogens.

Because of the importance of proper protection for the domestic water supply and also the often conflicting situation between water resource protection and local development imperatives, it is important to have a reliable delineation of protection zones. The ideal situation of a steady abstraction from a homogeneous aquifer is complicated by non-homogeneities underground on land and with the abstraction pattern. Experience has shown that such problems can be overcome through appropriate scientific approaches for delineation. The choice of delineation technique will also be a function of (Foster *et al.*, 2000):

- the degree of understanding of the groundwater setting involved;
- the operational importance of the groundwater supply concerned;
- the human and financial resources available.

A 'programme for community water source protection' is part of the Departmental Policy Strategy for Groundwater Quality Management (DWAF, 2000), but has not been implemented to date. The implementation of protection zoning in South Africa and elsewhere in Africa and other developing countries is lagging behind what has become standard practice in Europe and the United States. There are many factors that have a strong bearing on a successful implementation, including a holistic approach, socio-economic conditions, the legal environment, institutional arrangements, appropriate public participation, and the human and financial capacity. It is among these that impediments to implementation in developing countries should also be considered.

4.1 *Holistic approach*

Groundwater protection zoning requires a planned approach in which resources are classified and characteristics like vulnerability to pollution and importance of resources are mapped as a basis for systematic and prioritised implementation.

Furthermore, the protection of water resources should not be viewed in isolation. At the highest level, the National Water Act, 1998 requires the determination and setting of the Ecological Reserve. Many aquatic ecosystems that need to be protected in this way are partially or wholly dependent on groundwater to sustain the required reserve flow, as well as ecological structure and function. In local pollution management situations one has to provide for surface water contributions to groundwater and vice versa.

In general, surface waters are commonly contaminated from a wide variety of sources as a result of human activities, often in the same way as those identified in the case of groundwater pollution. In particular, surface waters can become contaminated from sanitation systems, as in the following situations: blocked or broken sewer pipes; poor drainage properties of soils into which wastes are disposed, e.g. from septic tanks and digesters; rainwater intrusion into pit latrines that fill and overflow; and springs contaminated from nearby latrines. In many of these situations the resulting contamination, particularly in terms of bacteria and viruses, can result in significant health risks to downstream users of the surface waters.

In the reverse situation, polluted groundwater can impact springs and streams. During wet periods and rising water tables, pollutants including, for example, heavy metals, can also be incorporated in plants and create a serious health hazard.

4.2 *Addressing key uncertainties*

There will always be technical uncertainties in the protection of groundwater sources. These will have to be managed in a flexible approach (Foster, 2002).

- Scientific uncertainties should be addressed formally in the delineation process, for example through showing zones of uncertainty and by being completely transparent about the assumptions and uncertainties.
- In case of a considerable risk of pollution, which cannot be managed, e.g. abstraction within a high-density settlement, the relocation of the groundwater source to a low pollution hazard area, with the concomitant introduction of the appropriate land-use development controls, is advisable.

- While the definition of formal recharge capture areas will be appropriate for higher-yielding groundwater sources for the reticulated supply to larger villages and small towns, this is not practicable for a small community and individual private domestic wells, because their capture zones are very small. As a minimum, a systematic well head sanitary survey should be carried out and appropriate protection measures relating to the construction hazard and the surrounding environmental hazard implemented.
- An essential component of groundwater protection programmes is aquifer water-level and quality monitoring. Initial uncertainties should be addressed, and the effectiveness of the protection measures confirmed. This is distinct from the water quality monitoring that must be in place to safeguard the public drinking water supply.

4.3 *Socio-economic issues*

Demarcation of protection zones around a groundwater source area places restrictions on land use. This may be restrictive to local economic development. There may also be some resistance, because of alleged reduction in land value resulting from loss of opportunity or increased cost for industrial development or agricultural production (Foster, 2002).

In situations with poorly defined and readily enforceable property rights, the feasibility of protection zoning is questionable. This is still the situation in many developing countries, particularly in rural areas. Tenure rules make land transactions difficult and there is often a tendency to maintain the status quo. Cultural issues like meeting at the watering point and sharing water points for stock watering and domestic use may also present challenges to proper resource protection.

4.4 *Institutional arrangements*

It is crucial that various spheres of government, as well as government organisations cooperate to achieve successful implementation of groundwater resource protection. It normally requires the national government department responsible for water resources to draft the appropriate legislation and mount groundwater protection programmes. However, the required zoning of certain land for certain specified activities and exclusion of other activities can usually only be achieved through the by-laws of local government. The actual assessment of zones and ongoing management and control to achieve the desired resource protection should be at the level of the resource, with the full involvement of the local water services provider and the participation of stakeholders.

The progressive development of groundwater quality management in Botswana illustrates the different issues and strategies very well (McLaren, 1995):

- 1970s: Serious pollution of groundwater noticed in major villages (bacteriological and nitrate pollution);
- 1976: Policy of closing boreholes in the villages (establishing outside sources and piping water into villages);
- 1989: Rural water supply design manual, which includes groundwater protection measures;
- 1993: Start of a program of zoning for existing and potential surface and groundwater sources for protection;
- 1995: Revision of the Water Act underway to make provision for control measures within zoned areas.

Different views on the interplay between national and local-level regulation can be illustrated with the general approaches followed in England and Spain (Braune, 1994). Spain has a very detailed water law and more detailed regulations, which appear to slow down implementation. England feels that too strong statutory approaches do not allow the flexibility necessary in a changing environment. Because various institutions have to implement non-statutory measures, more central resources must focus on facilitation, including the provision of appropriate and dynamic information packages, which provide insight, suggest options and enhance decision making. Part of this pragmatic approach was implementation by means of a few key programmes, in particular vulnerability mapping for the whole country and the technical delineation of about two hundred protection zones countrywide on a priority basis. Implementation of protection measures within these zones is up to the responsible parties, which can be local planning authorities, landowners and others. Whether developing countries have the capacity and the required level of cooperative governance for this non-statutory approach, is an open question.

According to Foster (2002), it is important to note that, independent of the institutional approach, the technical measures of land surface zoning through maps combining aquifer pollution vulnerability classes and groundwater supply capture areas can be extremely valuable (with appropriate stakeholder participation) for:

- raising stakeholder awareness of groundwater pollution hazards;
- offering a credible and defensible groundwater input to land-use planning procedures;
- promoting public understanding of groundwater protection needs.

4.5 *Appropriate participation*

Appropriate participation of the key stakeholders and the general public is a requisite for sustainable development of a scarce resource. This is particularly the case for groundwater, because it is invisible and often poorly understood.

The participation of communities in the process of capture zone implementation is strongly encouraged. This participation should be incorporated into the health and hygiene education component of a water-supply and/or sanitation project in close consultation with the Department of Health (Environmental Health Offices). Particular areas where the community can contribute include:

- undertaking of a hydrocensus within and around the settlement;
- compilation of landscape map;
- assessment of existing threats to groundwater quality;
- evaluation of sanitation options;
- ongoing monitoring of the performance (with Department of Health).

This level of participation is considered essential for creating an awareness of the importance of protecting the groundwater resources in the longer term, and for establishing a programme of ongoing sanitary surveillance of all potential health risks within the settlements.

Water managers, especially farmers, need to be convinced that they are beneficiaries of these protection measures. The relevant government departments also need to be assured that clean water supplies effectively contribute to the reduction or even eradication of incidents of water-borne diseases such as cholera. Research of this nature will have little impact unless it is coordinated for the different stakeholders, and research findings can demonstrate the positive economic trade-off.

5 CONCLUSIONS

Protection of groundwater resources that serve as drinking water supplies should be an equally important target as creating new water supply infrastructures in a country.

Appropriate legislation, like the National Water Act, 1998 in South Africa, is important to ensure the protection of groundwater resources, but is not enough in itself. Plans, programmes and measures are required to put legislation into practice, which require adequate capacity and resources.

In the South African situation of widespread and highly localised groundwater occurrence and use, it will be physically and economically impossible to protect all groundwater to the same degree. For effective intervention, a differentiated protection approach is necessary, based on the vulnerability and importance of aquifers. Groundwater sources providing drinking water to communities have been identified as having the highest need for protection.

Groundwater protection zoning is well established in developed countries to protect valuable groundwater resources, but this practice appears to be slow to take off in developing countries.

To be able to make progress in developing countries like South Africa, special attention should be paid to the key factors, which could have a bearing on the successful implementation of protection zoning. These include a holistic approach, socio-economic conditions, the legal environment, institutional arrangements, appropriate public participation and the human and financial capacity.

Because protection zoning inevitably involves land-use decisions, which are outside the jurisdiction of water resources agencies, it is crucial to take the appropriate institutions at the national, provincial and local level on board.

Active participation of local stakeholders in the planning and implementation of protection zones in their areas is a prerequisite for success.

REFERENCES

Braune, E. 1994. Report on Overseas Visit to England, Israel, Spain and the United States of America to Investigate Policies and Strategies for Groundwater Quality Management in South Africa. Directorate Geohydrology Internal Report. Pretoria: Department of Water Affairs.

Cameron-Clarke, I.S. 1997. Current legislation to manage the impacts of mining on groundwater in South Africa. In: Chilton *et al.* (eds). Groundwater in the Urban Environment: Problems, Processes and Management. pp. 567–570. A.A. Balkema, Rotterdam.

Conrad, J.E., Colvin, C., Sililo, O., Gorgens, A., Weaver, J. and Reinhardt, C. 1999. Assessment of the impact of agricultural practices on the quality of groundwater resources in South Africa. WRC Report 641/1/99. Water Research Commission, Pretoria.

De Villiers, D.C. 1987. Evaluation of sites for on-site sanitation systems, The Civil Engineer in South Africa, August Issue, 1987.

DWAF 1997. A protocol to manage the potential of groundwater contamination from on-site sanitation (Edition 1), The National Sanitation Coordination Office & Directorate of Geohydrology at the Department of Water Affairs and Forestry, Pretoria.

DWAF 2000. Policy and Strategy for Groundwater Quality Management in South Africa. Water Quality Management Series. Department of Water Affairs and Forestry.

DWAF 2002. National Water Quality, Framework Policy, Water Quality Management Series Sub-Series No, MS 7 (Draft 2). Department of Water Affairs and Forestry, Pretoria.

DWAF 2003. A protocol to manage the potential of groundwater contamination from on-site sanitation (Edition 2), The National Sanitation Programme, the Department of Water Affairs and Forestry, Pretoria, March, 2003.

Engelbrecht, J.F.P. and Cave, L.C. 1997. Groundwater pollution from cemeteries. CSIR Internal Report ENV/S-I 97003. CSIR, Stellenbosch.

Foster, S., Chilton, J., Moench, M., Cardy, F. and Schiffler, M. 2000. Groundwater in Rural Development – Facing the Challenges of Supply and Resource Sustainability. World Bank Technical Paper no. 463. Washington: The World Bank.

Foster, S., Hirata, R., Gomes, D., D'Elia, M. and Paris, M. 2002. Groundwater Quality Protection – A Guide for Water Utilities, Municipal Authorities, and Environment Agencies. Groundwater Management Advisory Team (G.W. Mate). Washington: The World Bank.

Fourie, A.B. and Van Ryneveld, M.B. 1995. The fate in the subsurface of contaminants associated with on-site sanitation: A review, Water SA 21(2).

McLaren, D. 1995. Personal Communication.

Morris, R.D., Allerton, H.G. and van der Westhuizen, J.P.W. 2000. Practical and responsible approaches to soil and groundwater contamination management at a South Africa metal plating factory. In: Sililo *et al.* (eds). Groundwater: Past Achievement and Future Challenges. A.A. Balkema, Rotterdam.

National Water Act, 1998. Act No. 107 of 1998, Republic of South Africa.

Ntsele, Q., Trois, C., Schreiner, H.D. and Motsa, N. 2000. A review of the composition of leachates from landfill sites throught the Durban Metropolitan area. Biennial Conference, Sun City, 28 May–1 June, 2000. Water Institute of South Africa.

Palmer Development Group 1993. Urban sanitation evaluation, WRC Report No. 385/1/93. Water Research Commission, Pretoria.

Parsons, R. and Taljard, M. 2000. Assessment of the impact of the Zandvliet Wastewater Treatment Works on groundwater. Biennial Conference, Sun City, 28 May–1 June, 2000. Water Institute of South Africa.

Rosner, T., Vermaak, J.J.G., van Schalkwyk, A. and Viljoen, C.J. 2000. Assessment and rehabilitation of reclaimed gold tailings dams in South Africa. In: Sililo *et al.* (eds). Groundwater: Past Achievement and Future Challenges. A.A. Balkema, Rotterdam.

Sililo, O.T.N., Saayman, I.C. and Fey, M.V. 2001. Groundwater vulnerability to pollution in urban catchments. WRC Report No. 1008/1/01. Water Research Commission, Pretoria.

Tredoux, G. 1993. A preliminary investigation of the nitrate content of groundwater and limitation of the nitrate input. Report to the Water Research Commission, WRC Report No. 368/1/93, Water Research Commission, Pretoria.

Usher, B.H., Pretorius, J.A., Dennis, I., Jovanovic, N., Clarke, S., Titus, R. and Xu, Y. 2004. Identification and prioritisation of groundwater contaminants and sources in South Africa's urban catchments. WRC Report No. 1326/1/04, Water Research Commission, Pretoria.

Van Ryneveld, M.B. 1994. The current extent of coverage and the cost of water supply and sanitation provision in the urban areas of South Africa, ISSN 0378-4738, Water SA 20(2), April 1994.

Verweij, M. 2003. Leaking pipelines – Shell in South Africa, Part 3 of The Filthy Five. The Globalisation and Environment Project, Amsterdam, the Netherlands.

Weaver, J.M.C. 1993. A preliminary survey of pesticides levels in groundwater from a selected area of intensive agriculture in the Western Cape. WRC Report 268/1/93. Water Research Commission, Pretoria.

Wright, K. and Xu, Y. 2000. A water balance approach to the sustainable management of groundwater in South Africa, Water SA 26(2).

Woodford, A.C. and Chevalier, L. (eds) 2002. Hydrogeology of the Main Karoo Basin: Current knowledge and future research needs. WRC Report TT179/02. Water Research Commission, Pretoria.

Xu, Y., and Braune, E. 1995. A guideline for groundwater protection for community water supply and sanitation programme, Published by the Department of Water Affairs and Forestry, Pretoria, 1995.

Xu, Y. Braune, E., Colvin, C., Le Maitre, D., Pietersen, K. and Hatton, T. 2000. Comprehensive determination of resource directed measures for groundwater in South Africa, In: Sililo *et al.* (eds). Groundwater: Past Achievements and Future Challenges. Balkema, Rotterdam.

Xu, Y., Colvin, C., van Tonder, G.J., Hughes, S., le Maitre, D., Zhang, J., Mafanya, T. and Braune, E. 2003. Towards the Resource Directed Measures: Groundwater component. WRC Report No. 1090. Water Research Commission, Pretoria.

Xu, Y., Marais, S. and McCaffrey, L. 1991. A preliminary discussion of aquifer protection strategies for some areas in Bophuthatswana in Groundwater Quality and Pollution, Biennial Ground Water Convention, South Africa, 12–13 August 1991.

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