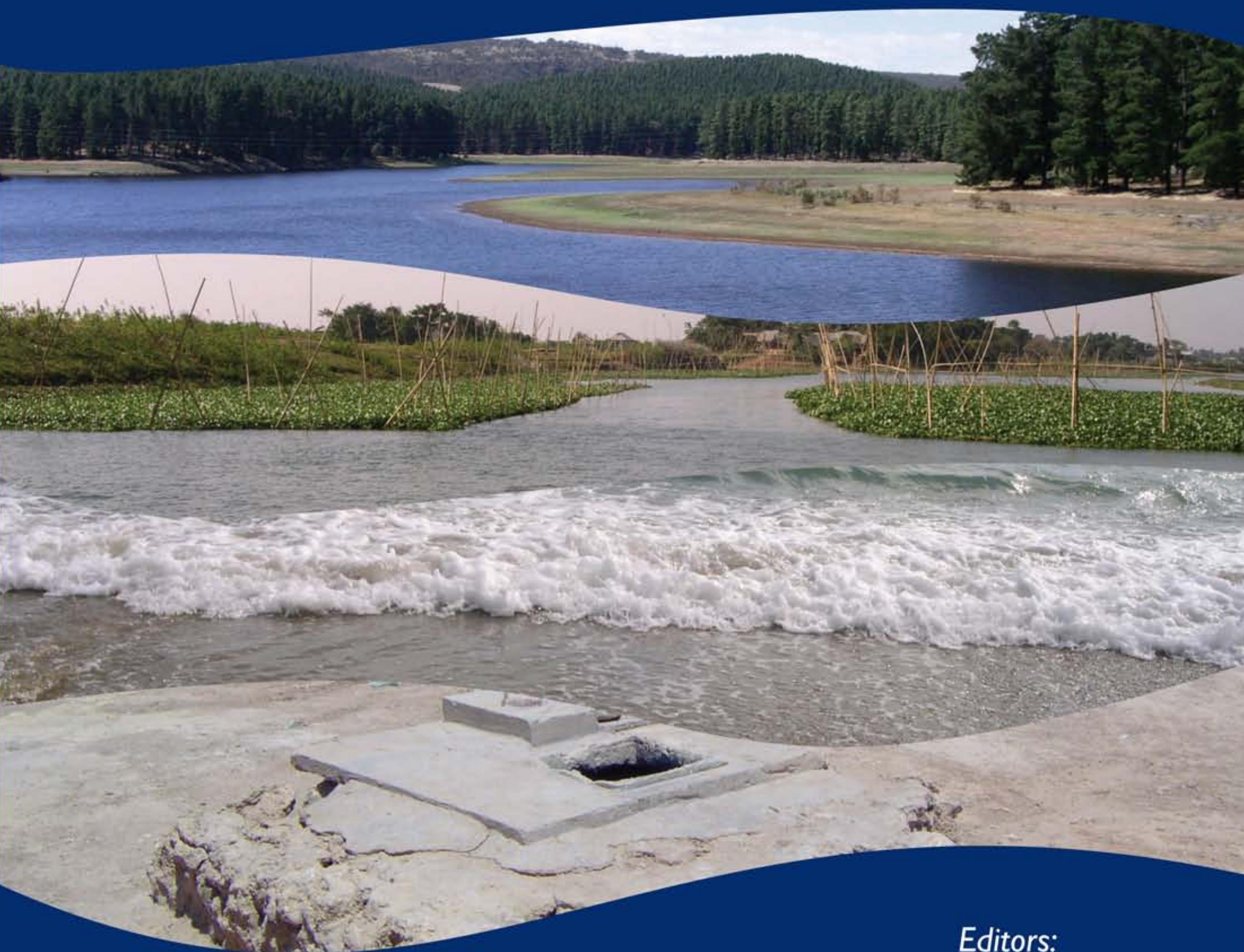
Groundwater for Sustainable Development

Problems, Perspectives and Challenges



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GROUNDWATER FOR SUSTAINABLE DEVELOPMENT: PROBLEMS, PERSPECTIVES AND CHALLENGES

BALKEMA – Proceedings and Monographs
in Engineering, Water and Earth Sciences

Groundwater for Sustainable Development: Problems, Perspectives and Challenges

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Taylor & Francis
Taylor & Francis Group
LONDON / LEIDEN / NEW YORK / PHILADELPHIA / SINGAPORE

ISGSD Special Publication

Taylor & Francis is an imprint of the Taylor & Francis Group, an informa business

Typeset by Vikatan Publishing Solutions (P) Ltd., Chennai, India
Printed and bound in Great Britain by Anthony Rowe (A CPI-group Company), Chippenham, Wiltshire

Published by: Taylor & Francis/Balkema
P.O. Box 447, 2300 AK Leiden, The Netherlands
e-mail: Pub.NL@tandf.co.uk
www.taylorandfrancis.co.uk/engineering, www.crcpress.com

Library of Congress Cataloging-in-Publication Data
Applied for

ISBN: 978-0-415-40776-2 (hbk)
ISBN: 978-0-203-89456-9 (ebook)

Dedication

Professor Emeritus Gunnar Jacks
71 years and over 40 years as a Hydrogeochemist

Gunnar Jacks was born on 13th April, 1937 in Grangärde, Sweden and married in 1963 with Birgitta, a pharmacist and licentiate in medical sciences. He graduated as a mining engineer from the Royal Institute of Technology (KTH) Stockholm in 1963. After working with rock stability issues for a couple of years he and his wife spent a year with a humanitarian Non-Governmental Organization (NGO) in Turkey where he realized the importance of water for societies and human life. He took a B. Sc. degree in Medicine from the Karolinska Institutet, Stockholm in 1971. After returning to Sweden and KTH, he studied hydrology and received the PhD in hydrochemistry in 1973. A turning point was when he assisted his professor in a short term groundwater investigation in Tamil Nadu, India. That led to a three year project work within Central Ground Water Board in Coimbatore in India and later on to an engagement in a project in Kerala, India. The chemistry of fluoride in groundwater was a major subject of his research interest and together with his Indian colleagues, he has published a number of research articles. While he has throughout years been a faculty member at KTH he has been on missions to Botswana, Vietnam, Thailand, and Laos.

Gunnar Jacks became Associate Professor in 1976 and got a professorship in 1985 in Groundwater Chemistry at the former Division of Land and Water Resources, Department of Civil and Environmental Engineering of the KTH. He took a very active part in teaching and research on acidification of soil and groundwater during the 1980s and 1990s. The quantitative study of weathering rates became an important issue and he was part of a group that used strontium isotopes for assessments. During this period he was also doing research on artificial recharge of groundwater as well as wetland hydrology and hydrochemistry. He has been involved with a number of international research projects since 1990s which include investigation on high fluoride groundwater in Rajasthan, India (1994–1997), research co-operation with the Department of Environment at the University of Botswana on pit latrines (1994–1998), local water supply and sanitation in suburban Dhaka, Bangladesh (1999–2003), hydrogeology in Eritrea, in collaboration with Dr. Risto Kumpulainen, Stockholm University (2002–2004), research co-operation with the University of Ghana in Logon on metal contamination in the water resources of Tarkwa region (2000–2006), and investigations on the possible deficiencies of zinc in soils, crops and food in northern Mali, sponsored by the Trasher Research Fund, Salt Lake City, USA (2003–2004). He has supervised a significant number of PhD students during his professional career at the KTH and serve as a peer-reviewer in a number of scientific committees and peer-reviewed journals.

Professor Jacks been actively involved in research on arsenic in soil and groundwater systems since 1991. During 1996, his research group at KTH was contacted by Dr. Debashis Chatterjee from the Kalyani University situated near Kolkata, India regarding the arsenic problems in West Bengal. That was the start of a still ongoing work in India, Bangladesh and several other parts of the world with a principal focus on research on the mechanisms of arsenic mobilization in groundwater environments, a work now headed by Prof. Dr. Prosun Bhattacharya. Gunnar Jacks has also in recent years been working on zinc deficiency in soils in the Niger inland delta in Mali. He is a guest professor at Åbo Akademi University, the Swedish speaking university in Finland. In 2006, he was elected as the President of the International Society of Groundwater for Sustainable Development (ISGSD).

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Foreword

Groundwater is an extremely important natural resource, more important than most people realize. Globally, the groundwater resources are subject to increasing stress both in terms of quantity as well as quality. Most rural areas in many developing countries depend solely on groundwater, however it also attaining significant importance in urban areas. The increased dependence on groundwater for human consumption calls for an understanding of the critical issues concerning the quality from public health point of view. This is anticipated to increase further during the next decades due to severe limitations on the availability of reliable quantities of surface water and the continuous degradation of surface water quality. The presence of geogenic contaminants in groundwater such as arsenic and fluoride at toxic levels is a major environmental health risk for the present century. In several regions of the world especially in different countries of Asia such as Bangladesh, Cambodia, China, India, Nepal, Pakistan, Taiwan, Thailand and Vietnam, as well as in many Latin American countries, the situation of arsenic toxicity is alarming and severe health problems are reported amongst the inhabitants relying on groundwater as sources of drinking water. Thus we need to improve our understanding on the genesis of high arsenic groundwater from the various aquifers in order to develop sustainable mitigation strategies to deliver safe drinking water to save millions from this disaster. An increased demand for groundwater also warrants application of improved tools for groundwater monitoring and vulnerability assessment to develop an integrated strategy to optimize groundwater resource management. This warrants a common platform for sharing knowledge and experience on groundwater matters on a worldwide scale for identifying and promotes optimal approaches for the assessment and management of groundwater resources for long-term sustainability of the aquifers as a resource for drinking water for global sustainable development.

The present volume '*Groundwater for Sustainable Development: Problems, Perspectives and Challenges*' being published as the '*Special Publication 1*' of the *International Society of Groundwater for Sustainable Development (ISGSD)*, is an important contribution that will assist to anticipate and recognize the problems and challenges for the use of groundwater for sustainable development. I am especially happy to dedicate this volume to mark the 70th birth anniversary of Professor emeritus Gunnar Jacks at the Department of Land and Water Resources Engineering, Royal Institute of Technology at Stockholm, Sweden for his services on the various aspects of research related to groundwater resources in developing countries.

I deeply appreciate the efforts of the editorial team of this volume for their untiring work and hope that the book will serve the aims for improving knowledge required for the protection and management of groundwater resources to fulfill the basic human needs.

Prof. Dr. Peter Gudmundsson
President
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Stockholm, Sweden
January, 2008

Preface

Water is an integral part of the environment and its availability is indispensable to the efficient functioning of the ecosphere. Since the dawn of civilization, water is an important lifeline for humans, animals and plants. Water is also of vital importance to all socio-economic sectors—humans, and economic development simply is not possible without a safe, stable water supply. The water resources are subject to increasing stress both in terms of quantity as well as quality. In recent decades, surface water resources have become highly contaminated with domestic and industrial wastes, as well as due to the fluxes of urban and agricultural runoff. Thus, water resources must be seen in the overall context of global sustainable development.

Groundwater has emerged as the most important source of water for drinking purposes and other domestic needs, industrial, and agricultural water supply in the world and its use has increased manifolds. Substantial amount of groundwater is used indiscriminately to fulfill the demands in the agricultural sector, especially in regions with rather dry and/or semi-arid climate to enhance crop production for sustain food supply. Groundwater also plays a key role in keeping wet ecosystems sustainable and sometimes as well in maintaining a suitable environment for human settlement. In order to achieve maximum benefit from the groundwater resources, substantial efforts are needed to explore the aquifer systems and to optimize its rational exploitation. However, attention is not only required for its exploitation, but as well for controlling a wide spectrum of problems related to groundwater.

Most rural areas in both developed and developing world depend on groundwater sources for drinking purposes. However, an exponential growth of population in urban areas has led to an increased exploitation of groundwater for domestic purposes and is thus increasingly important in urban areas as well. Its high dependence will increase even further during the next decades due to severe limitations on the availability of reliable volume of surface water and its continuous degradation in terms of quality. Thus, groundwater is in most parts of the world an extremely important natural resource, more important than most people realize. In many parts of the developing world, the population growth has created an unprecedented demand for water for drinking water purposes, competing for the same finite resource. The increased demand of groundwater during the last three decades has led to water scarcity in many parts of the world that demands proper assessment for suitable management of groundwater resources for sustainable development.

With the rising demand for groundwater, there is a need to look for improved strategies for aquifer management and augment water supplies. Keeping in view the increased abstraction of groundwater for sustaining the various needs of the society, it is important to develop an integrated strategy to optimize groundwater resource management. There are several means to augment the groundwater availability that includes artificial recharge of the aquifers for long term sustainability of the aquifers as a source for drinking water. Groundwater quality parameters, recharge technologies, transformations during transport through the aquifers, public health issues, economic feasibility as well as legal and institutional considerations thus needs to be addressed in order to ensure both quality and quantity of the groundwater resources.

It has been observed world-wide, that the coastal aquifers are vulnerable to salinization due to sea water intrusion caused by excessive pumping of groundwater for water supplies. As a consequence, it threatens the suitability of groundwater for drinking and other intended uses. Many coastal areas across the globe use groundwater as the main source of drinking, domestic and agricultural activities. With the population in the coastal region are increasing at alarming rate, the fresh water supply is being continually depleted, and warrants intense groundwater monitoring.

Natural calamities such as the large tsunami event on the 26th December 2004, has resulted in devastating effects on coastal ecosystems over a significant part of the coastal areas in Indonesia, southern India and Sri Lanka. The areas with low and relatively flat topography facilitated the saline water to flow over several kilometers inland that infiltrated through the sandy sediments and soils and directly hit the groundwater in the vast majority of the wells and rendering it in most parts of the shallow aquifers along the coastline became unusable for domestic and irrigation purposes.

Increased dependence on groundwater has also raised critical issues on its quality from the human health point of view. The presence of geogenic contaminants in groundwater for example arsenic and fluoride in toxic levels has posed major environmental health risks of the world population in the present century. Trace elements and metal(loids) occur in natural ecosystems in different concentrations and undergo recycling between different environmental compartments of the earth's crust. They are emitted from natural and industrial sources into the terrestrial, aquatic and atmospheric environments. In many cases, natural release of a metal(loid) is more than anthropogenic. For the last few decades, humans and domestic animals are suffering from toxic effects of metal(loids) and metals such as As, Pb, Cd, Cu, Se, Cr and many others. These elements are toxic to humans, animals and plants at high concentration. Several million people depend on arsenic- and fluoride contaminated groundwater for drinking purposes that endangers public health.

In several regions all over the world, mostly in developing countries, groundwaters from sedimentary and hard rock aquifers used for drinking are naturally contaminated with arsenic. In different countries in Asia such as India, Bangladesh, Cambodia, China, Nepal, Pakistan, Taiwan, Thailand and Vietnam, the situation of arsenic toxicity is alarming and severe health problems are reported amongst the inhabitants relying on groundwater as sources of water for drinking purposes. Arsenic occurrences in groundwater in Bengal Delta Plain of West Bengal, India and Bangladesh is one of the largest environmental health disaster of the present century, where over 50 million people are at risk of cancer and other arsenic related diseases. In these same countries, land and agricultural sustainability is threatened by the use of arsenic contaminated irrigation water. In several countries in Central and South America, for example in Argentina, Brazil, Chile, Mexico, Nicaragua and many others, elevated arsenic concentrations are reported in groundwater. In Argentina, at least 1.2 million people are affected. Elevated levels of natural arsenic in groundwater due to geogenic sources is therefore an issue of primary environmental concern, which limits the use of these resources for drinking or other purposes, and hinders the socio-economic growth. Hence, there is a need to improve our understanding on the genesis of high arsenic groundwater from the various aquifers in order to develop sustainable mitigation strategies to save millions from this calamity. Therefore urgent measures based on quality research and sound scientific investigations and reliable data are required to safeguard the drinking water supplies from groundwater sources.

Elevated concentrations of naturally occurring fluoride in groundwater are causing widespread health crisis in many parts of the world. Fluorine enters the groundwater system through natural processes especially due to the weathering and leaching of crustal rocks with high content of fluorine bearing minerals. However, significant influx of fluoride also comes from atmospheric deposition in the form of soil dust and the emissions of fluoride from the industry. The F intake via drinking water of 20–70 mg/d by adults may cause severe effects in the form of fluorosis due to displacement of Ca in teeth and bones. The potential growth retardation can be observed among animals due to F-deficiency. In addition, groundwater may also be contaminated by a wide spectrum of other toxic elements from natural and anthropogenic sources that are reported in different regions of the world.

As a fresh water resource, groundwater has major advantages over surface water. The groundwater is not only an important component of water resources but also the major supply source in hard rock area. The increased world population growth and the lack of surface water have caused over-exploitation of groundwater due to indiscriminate withdrawal renders large volumes of groundwater reserves to be depleted and that landscapes may turn dry and desolate by the decline of shallow water tables that also result in land subsidence. However, most of these problems tend to develop rather slowly, but controlling them is difficult, and many of them are practically irreversible. Over 99% of the world's fresh, available water is groundwater; yet, the vast majority of economic resources are directed to surface water found in rivers and lakes. This serious imbalance requires urgent readdress. Significant financial support is required for basic groundwater research if sustainable development is to be a realistic goal. In order to prevent further deterioration of groundwater quality and improve the status of aquifer systems, methodologies for vulnerability assessment and reliable groundwater management tools developed in recent years should be implemented by the developing countries for an improved management of groundwater resources.

These goals will never be realized without the commitment of world governments to exploration programs that can delimit and characterize aquifers, perform water balances, map water quality, and provide for long-term monitoring. Many aquifers extend across political boundaries. There is a critical need to promote intergovernmental coordination for developing joint management strategies for large trans-boundary aquifers. Ultimately, groundwater can deliver major socio-economic and ecological benefits but the aquifer systems that sustain the resource need to be adequately understood and responsibly managed. We require new technologies, and management policies that include effective strategies for water quality protection. Meeting these challenges require a serious commitment of funds by governments and various international donor agencies.

The 2nd International Groundwater Conference (IGC-2006) was held in New Delhi, India, between 1st–4th February 2006 that focused on a theme '*Groundwater for sustainable development: Problems, Perspectives and Challenges*,' with the goal for providing a platform to bring together earth scientists, professionals from chemical and engineering science disciplines, public health professionals and social scientists involved with the sustainable development of groundwater resources. The major purpose was to find out information related to exploration, qualitative and quantitative evaluation, and protection of groundwater resources and management. A number of technical themes were identified to cover the major aspects of groundwater and its role in sustainable development and are covered in six sections:

- Section I: Sustainable groundwater resources assessment and recharge processes (7 chapters)
- Section II: Water and environment (5 chapters)
- Section III: Groundwater modeling and its application in aquifer systems (5 chapters)
- Section IV: Coastal groundwaters and impact of tsunami (6 chapters)
- Section V: Arsenic and fluoride in groundwater (12 chapters)
- Section VI: Groundwater management (5 chapters)

This volume presents a state of the art in the recent developments in the field of groundwater that covers the technical, environmental, legal, economic, social, and gender aspects of groundwater resources management, contamination and protection recognizing the importance of integrated water resources management. The book is given out as '*Special Publication 1*,' the first publication of the International Society of Groundwater for Sustainable Development (ISGSD), conceived during July 2006, that would form a base for discussion and exchange of scientific ideas to identify future targets for research needed to improve the knowledge of an groundwater resources development, management and protection. Because of the multidisciplinary flavor of the chapters and the multinational constellation of the authors, this book will create interest among groundwater professionals, students, academicians, administrators, policy makers and executives, on the diverse problems associated with groundwater resources. This book will also serve as a catalyst to stimulate international cooperation on: i) developing common approaches for the assessment of groundwater resources; ii) augment the aquifer recharge; iii) understanding the various environmental factors that may affect the groundwater quality as well as treatment; iv) modeling of groundwater flow and application in aquatic systems; v) improved understanding of the coastal groundwater systems and their vulnerability due to natural diasters; vi) dynamics of natural groundwater contaminants such as arsenic and fluoride from the aquifers through groundwater to food chain and their impacts on the society; and vii) groundwater management needed to circumvent the environmental health disasters.

The organizers of the IGC-2006 would like to thank the Swedish International Development Cooperation Agency (Sida) in New Delhi, India for the financial support to the organization of the conference. We would also like to thank the former President of the Royal Institute of Technology Prof. Dr. Anders Flodström, the present President Prof. Dr. Peter Gudmundson, and

Prof. Ramon Wyss, advisor to the President, KTH for their keen interest in this important conference and the financial support. We deeply appreciate the support we have received from the Vice Chancellor of the Jawaharlal Nehru University, New Delhi as a co-organizer of the conference. Further, we would also like to thank the UNICEF, New Delhi, India especially Dr. Paul Devrill and Ross Nickson, the Department of Science and Technology, Ministry of Environment and Forest, Government of India and Dr. Bhanu Neupune from the UNESCO, New Delhi, India for their financial support to organize the conference. We would like to thank our colleagues, M. Vithanage, N. Singh, G. Jacks, M. Jakariya, O. Sracek, C. Mahanta, G. Baurne and C. Rammelt for their efforts with the timely review of the manuscripts of the chapters in this book. We wish to express our sincere thanks to them who contributed significantly to maintain the high quality of the contributions in this volume.

We would also like to thank the Swedish International Development Cooperation Agency (Sida-SAREC), Swedish Research Council (VR), and the Swedish South-Asian Studies Network (SASNET) for support during the editorial work of the volume. We are especially grateful to the organisational support and cooperation provided by the Royal Institute of Technology (KTH), Stockholm, Sweden and the Jawaharlal Nehru University, New Delhi, India, the University of Helsinki, Finland, Indian Institute of Technology-Mumbai, India, and the Integrated Expert Programme of CIM (GTZ/BA), Frankfurt, Germany, the Instituto Costarricense de Electricidad (ICE), San José, Costa Rica. Lastly, the editors thank Janjaap Blom and Lukas Goosen of the Taylor and Francis (A.A. Balkema) Publishers, The Netherlands for their patience and skill for the final production of this volume. We do hope that the volume and the chapters included herein will serve the broader purpose of improving knowledge required for the protection and management of the groundwater resources to fulfill the basic human needs for sustainable development.

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January, 2008

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Section I
Sustainable groundwater resources assessment and recharge processes

CHAPTER 1

Assessment of groundwater resources by using a simple hydrogeochemical tool in coastal aquifers of Krishna delta, India

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ABSTRACT: Groundwater in coastal regions is usually deteriorating by seawater intrusion. The towns and villages of Krishna delta are facing fresh groundwater problems because of transforming fresh groundwater to brackish/saline. Hydrochemical study has been carried out for the identification of hydrochemical parameters, which can be used for demarcating the fresh groundwater zones and identify the seawater intrusion processes. Groundwater analyses indicate that (1) HCO_3/Cl molar ratio and TDS are fairly reasonable indicators for the identification of fresh groundwater in coastal aquifers, (2) Sr and B are found to be indicators for such delineation, and (3) Na vs. Cl, Na vs. EC, Cl vs. EC and Mg vs. TDS have been found reasonably good for the estimation of the intensity of seawater intrusion and their spatial and temporal variations. It is observed that the influence of brackish water is more in the South Krishna delta comparatively in the North Krishna delta where the percentage of fresh water wells is as high as 79%. A plot between HCO_3/Cl molar ratio and TDS indicates good agreement with the results obtained from the trace elements (Sr and B) in the groundwater of the Krishna delta.

1.1 INTRODUCTION

Most of the deltas of India such as Krishna, Godavari, Cauveri, Mahanadi and also Gangetic plains have been facing fresh groundwater resource problem because most of its fresh groundwater aquifers are transforming to brackish (Bikshem et al. 1991, Subba Rao 2002, Saxena et al. 2002, 2003, 2004a&b, 2005). The brackish water changes to saline, it also affects the soil of the area, which may degrade the fertility, and intern reduces the yield (Paliwal & Maliwal 1981, Mericado 1985, Gupta 1994). Because of degradation of soil and enrichment of various types of undesirable chemical constitutions in groundwater, this can not only spoil the quality of water for potable purpose but also affect the crop growth rate and also force to go for selective crops, which can resist such extreme water quality (Howard & Mullings 1996). Such type of failure of crops has been indicated in some parts of Krishna delta where the sea-water occurs.

Krishna delta is located in the southeast of India and known for high yields of agricultural production. Vijayawada, Guntur, Tenali, and Machilipatnam are the principal townships located in this region. More than 70% population depends upon groundwater. Hand pumps, dug wells and dug-cum-bore wells used for withdrawal of the water. During the past few decades, the rapid increase of seawater intrusion caused the transformation of fresh groundwater to brackish/saline. Because of seawater intrusion some of the fertile land become wasteland and cultivation rate also reduced (Gupta 1994). Habitants of this region are facing problems for their needs either for portable or for irrigation. It is a fact that the soil of Krishna delta is very fertile and the rates of crop yields are very high. A large number of cultivations have carried out such as paddy, wheat, maize, betal, coconut, grains and various types of vegetable leaves along with cultivation of sugarcane has been carried out.

The purpose of this paper is to: 1) present of hydrochemical parameters, 2) demarcate of fresh groundwater resources and study the seawater intrusion of Krishna delta, and 3) also identify the hydrochemical parameters for seawater contamination process in the coastal environments.

1.2 MATERIALS AND METHODS

1.2.1 *The study area*

The holy river Krishna, close to the Bay of Bengal in the East Coast of India, forms the Krishna delta. River Krishna originates from Western Ghats near Mahabaleswar, Maharashtra. After covering a distance of 480 km in Maharashtra, 291 km in Karnataka and 510 km in Andhra Pradesh, with a total distance of 1281 km, it emerges into the Bay of Bengal and forms a delta (Fig. 1.1). The delta is almost plain and mostly covered with alluvium. The red sandy soils, mixed red and black soil are the main soil types of the area. The main source of recharge is precipitation with an average annual rainfall of 1050 mm. Annual average temperature is 28°C. The monsoon (June to October) has the maximum rainfall (75%). The main sources of groundwater are dug wells, hand pumps and bore wells. The flat area between Tenali and Repalle on western side of Krishna River represents the Upper deltaic plain with elevation of 13 m above mean sea level (Rao 1982, Ramesh et al. 1990).

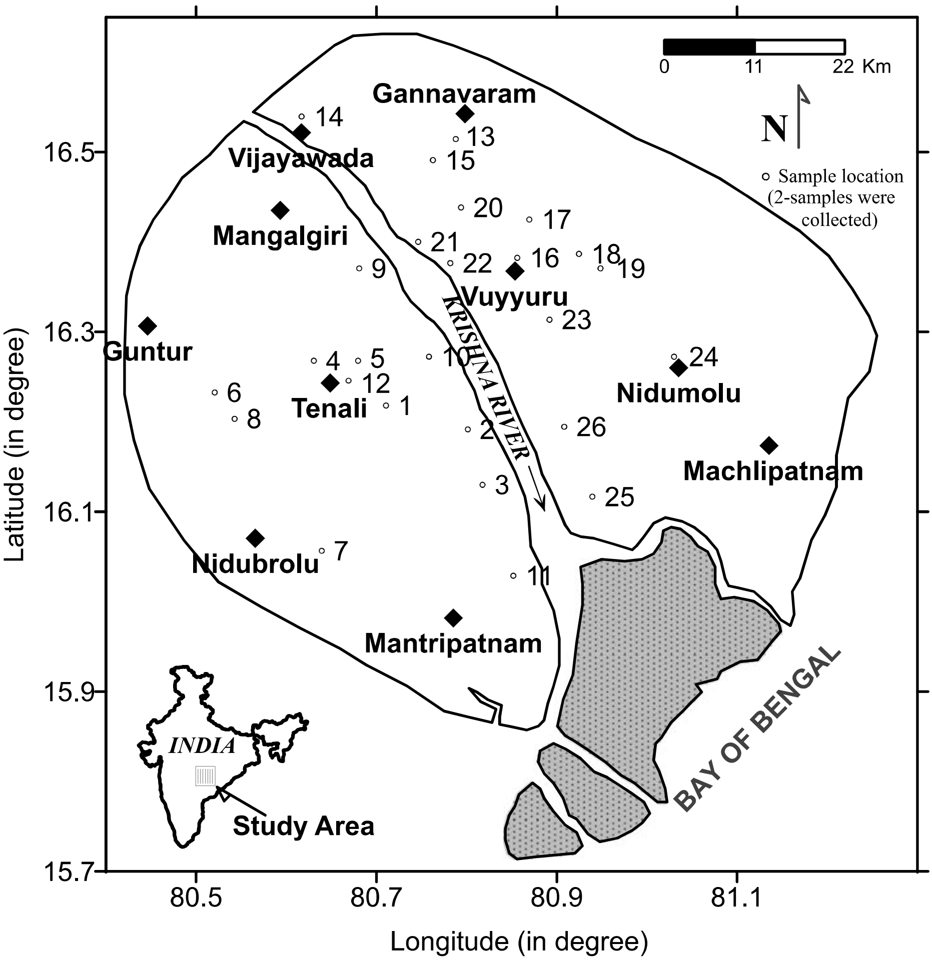

Figure 1.1. Location of the study area and the sites of groundwater sampling in Krishna delta.

1.2.2 *Sampling and analyses*

26 groundwater samples were collected from dug wells, hand pumps and bore wells in Krishna delta during pre-monsoon period (July 1998). The location of the sampling stations is shown in Figure 1.1. Out of these, 12 samples have been collected from south deltaic region, which is in the right side of the Krishna River and 14 samples from North Krishna delta, which is on the left side of the river. The water samples were collected in 1 L polythene bottles for major ion analyses. The groundwater from each dug wells was sampled at 0.5 m below the water table (depth to water level: 1.7–12.0 m, bgl) and, hand pumps were purged for 5 minutes prior to the collection of the samples. The water samples were used for the analysis of major cations (Na^+, K^+, Ca^{2+}, Mg^{2+}) and major anions (Cl^-, HCO_3^-, NO_3^- and SO_4^{2-}). Water samples were collected in 100 mL polythene bottles and acidified with 1 mL concentrated (14 M) HNO_3 for trace element analyses. The chemical analysis was carried out at the laboratories of the National Geophysical Research Institute (NGRI) following the standard procedures as outlined in APHA (1985). The major ions were determined within a week by mobile chemical laboratory and trace elements in the NGRI. During chemical analysis each sample was run two times and the instruments were calibrated time to time with standard solutions and care has been taken for accuracy and precision (Saxena 1987, Saxena & Mohan 1999). A reasonable precision for the chemical constituents was maintained by simultaneously analyzing the standard solutions and standard water (USGS-900) by repeating the chemical analysis. The concentrations of major ions and trace elements are shown in Tables 1.1 and 1.2 respectively.

Table 1.1. Chemical analysis[a] of groundwater, Krishna delta.

Sample No.	Location	EC	TDS	pH	Na	K	Ca	Mg	Cl	HCO_3	SO_4
1	Peddaruru	1250	775	7.5	112	10	81	30	188	281	46
2	Kolluru	2300	1665	7.8	150	20	170	80	465	280	185
3	Velluturu	2050	1230	7.8	142	22	170	75	260	275	140
4	Ansalkudu	1950	1215	7.5	285	20	140	45	235	410	175
5	Gurumula	2800	1915	7.5	485	25	100	75	685	425	140
6	Narakun	2100	1310	7.5	145	10	150	70	335	445	120
7	Induru	4500	2760	7.8	750	32	85	105	725	360	480
8	Chibrolu	2300	1650	7.6	140	10	265	70	440	250	455
9	Vallabha	2950	1785	7.9	400	25	100	80	625	325	70
10	Chadalwada	2900	3180	7.6	480	20	105	75	780	330	140
11	Reppale	2150	2340	7.8	165	10	180	45	460	455	70
12	Tenali	2100	1315	7.6	160	10	170	45	355	435	85
13	Kesarpalli	1250	830	7.6	100	10	105	50	250	255	55
14	Vijayawada	1900	1200	7.5	160	15	90	70	345	350	175
15	Uppuluru	3300	1980	7.9	345	20	125	140	750	385	275
16	Vyyuru	1850	1100	7.6	140	15	180	40	310	330	100
17	Katuru	2100	1405	7.8	165	10	165	55	405	490	60
18	Elarru	1450	915	7.5	95	20	105	40	205	270	160
19	Penamaru	2300	1435	7.8	160	10	160	82	305	550	75
20	Kankipadu	1050	665	7.8	85	15	80	20	150	245	30
21	Royyuru	950	630	7.4	85	10	65	20	145	250	30
22	Vall. Puram	1450	900	7.6	70	10	120	25	207	210	35
23	Kapileswaram	2300	1440	7.9	180	15	175	70	405	505	60
24	Nidumolu	3250	2110	7.9	400	20	120	165	685	365	280
25	Chellapalle	1150	835	7.8	130	10	60	50	255	265	50
26	Kudali	5050	3175	8.6	650	40	75	210	1160	475	545

[a] EC: in μS/cm, all concentrations are in mg/L.

Table 1.2. Chemical analysis of trace elements[b] in groundwater, Krishna delta.

Sample No.	Location	Sr	B	Cu	Zn	Fe	Ba
1	Peddaruru	775	104	8.6	<5.0	33.4	64.2
2	Kolluru	1665	369	7.1	<5.0	42.5	107.0
3	Velluturu	1230	166	8.3	<5.0	52.0	27.1
4	Ansalkudu	1215	147	7.9	<5.0	40.1	5.0
5	Gurumula	1915	211	6.5	5.4	48.0	10.5
6	Narakun	1310	199	9.1	<5.0	52.6	11.2
7	Induru	2760	422	8.3	<5.0	45.5	177.2
8	Chibrolu	1650	238	9.6	<5.0	604.0	25.9
9	Vallabha	1785	332	27.3	20.3	59.0	67.8
10	Chadalwada	3180	2929	749.8	43.5	499.5	21.2
11	Reppale	2340	366	32.3	40.0	38.9	132.0
12	Tenali	1179	188	28.6	14.9	29.2	13.5
13	Kesarpalli	306	102	31.0	16.1	298.0	23.2
14	Vijayawada	902	127	21.0	41.4	119.0	101.0
15	Uppuluru	1750	242	15.2	11.0	54.4	146.3
16	Vyyuru	947	151	46.5	22.3	91.2	101.3
17	Katuru	846	129	12.2	41.2	178.2	314.0
18	Elarru	840	156	56.0	38.8	181.7	26.3
19	Penamaru	983	159	25.5	1065	233.4	121.6
20	Kankipadu	253	179	21.5	89.0	79.4	66.8
21	Royyuru	602	71	25.0	399.8	526.5	139.7
22	Vall. Puram	23	94	6.5	<5.0	30.0	<5.0
23	Kapileswaram	1500	129	12.2	41.2	178.2	314.8
24	Nidumolu	1790	284	35.4	22.7	77.3	14.3
25	Chellapalle	203	143	16.8	14.7	504.3	40.1
26	Kudali	1679	695	7.3	<5.0	25.4	111.4

[b] All concentrations are in µg/L.

1.3 RESULTS AND DISCUSSIONS

1.3.1 *Hydrogeochemistry*

Groundwater samples were nearly neutral to alkaline with a pH ranging between 7.4–8.6 and revealed considerable variation in electrical conductivity (EC 950–5050 µS/cm). The variation in the EC values were reflected in considerable variation in the hydrogeochemical characteristics in different parts of the Krishna delta with different types of groundwater. Mostly the groundwaters are characterized by considerable variations in the concentrations of Na^+ (70–650 mg/L), K^+ (10–40 mg/L), Mg^{2+} (20–210 mg/L), Cl^- (145–1160 mg/L), HCO_3^- (210–550 mg/L) and SO_4^{2-} (30–545 mg/L). The concentration of HCO_3^- is inversely proportional to TDS. However, high TDS is directly proportional to the concentrations of Na, Mg and Cl. The groundwaters in Kudali, Nidumolu, Uppuluru, Induru, Vallabhapuram and Chadalwada have been found to have elevated EC values and may thus be affected by seawater intrusion. The Ca^{2+} is in moderate and it varies from 60–265 mg/L.

The concentrations of Ca^{2+} and HCO_3^- ions are high in the fresh groundwater compared to the samples of brackish/saline character. The groundwaters of Kudali, Induru, Nidumolu, and Uppuluru are being contaminated because they have very high contents of TDS, Na, Ca, Mg, Cl, HCO_3 and SO_4. All are more than permissible limits of potable waters (WHO 1984). These waters are suitable only for certain crops, which require more salt contents (Paliwal & Maliwal 1981, Gupta 1994). These groundwaters are classified in terms of TDS, into fresh (<1500 mg/L), brackish (1500–3000 mg/L) and saline (>3000 mg/L), Sr, B and HCO_3/Cl ratios (Tables 1.3, 1.4)

It is observed that the domination of brackish water is more in the South Krishna delta, but the situation is comparatively better in the North Krishna delta where the percentage of fresh water is

Table 1.3. Basis of groundwater classification in Krishna delta.

Parameters	Fresh (F)	Brackish (B)	Saline (S)
TDS*	<1500	1500–3000	>3000
Sr**	<1600	1600–5000	>5000
B**	<200	200–500	>500
HCO_3/Cl	>1.0	0.5–1.0	<0.5

* TDS in mg/L, ** Sr and B in μg/L

Table 1.4. Groundwater samples in Krishna delta and their compositional categories*.

Sample No.	TDS	Sr	B	HCO_3/Cl	Sample No.	TDS	Sr	B	HCO_3/Cl
1	F	F	F	F	14	F	F	F	F
2	B	B	B	B	15	B	B	B	B
3	F	F	F	F	16	F	F	F	F
4	F	F	F	F	17	F	F	F	F
5	B	B	B	B	18	F	F	F	F
6	F	F	F	F	19	F	F	F	F
7	B	B	B	B	20	F	F	F	F
8	B	B	B	B	21	F	F	F	F
9	B	B	B	B	22	F	F	F	F
10	S	B	S	S	23	F	F	F	F
11	B	B	B	B	24	B	B	B	B
12	F	F	F	F	25	F	F	F	F
13	F	F	F	F	26	S	B	S	S

* F: Fresh, B: Brackish and S: Saline

Table 1.5. Percentage of wells representative of the compositional categories in Krishna delta.

Water quality	South Krishna delta	North Krishna delta
Fresh	42	79
Brackish	50	14
Saline	8	7

Table 1.6. The percentage of wells classified according to the prevalent water-types.

Water quality	South Krishna delta	North Krishna delta
Na-Cl	33	14
Na-Ca-Cl-HCO_3	42	43
Na-Mg-Cl-SO_4	8	7
Mixed	17	36

as high as 79% (see Table 1.5). The seawater contamination is mainly caused by the intrusion of seawater along with Mg^{2+} and SO_4^{2-} etc, and these are more in the South Krishna delta. However, these regions have more demand towards urbanization which intern may have excessive withdrawal of groundwater. A complete deteriorated groundwater has been found in 7–8% of samples, which can also affect the soil fertility and may be responsible for crop failure (Saxena et al. 2003).

On the basis of percentage of chemical constituents these are classified into: i) Na-Cl, ii) Na-Ca-Cl-HCO_3, iii) Na-Mg-Cl-SO_4, and iv) mixed types. The percentage of water samples in each type is presented in Table 1.6.

1.3.2 *Hydrogeochemical variability on a regional scale*

To understand the hydrogeochemical behavior of seawater intrusion in a regional scale, the TDS contours have been drawn and shown in Figure 1.2. This shows the TDS value is observed more in the central parts of the north Krishna delta in particular southeast of Yuyyuru and north-east of Niduvolu, north-south-east of Tenali, and north of Machlipatnam and these are important towns of the Krishna delta. However, a high TDS profile is indicated in the south-east of Tenali and surrounding areas of Nidumolu-Mantripatnam, which is known to be an area with good agricultural productivity. This may indicate the possibility of high rate of seawater ingression/intrusion in these parts of the delta where a high rate of withdrawal of groundwater is also observed. It is noted that the central part of the delta both in right and left sides of the Krishna River is comparatively more habitated along with larger agricultural fields.

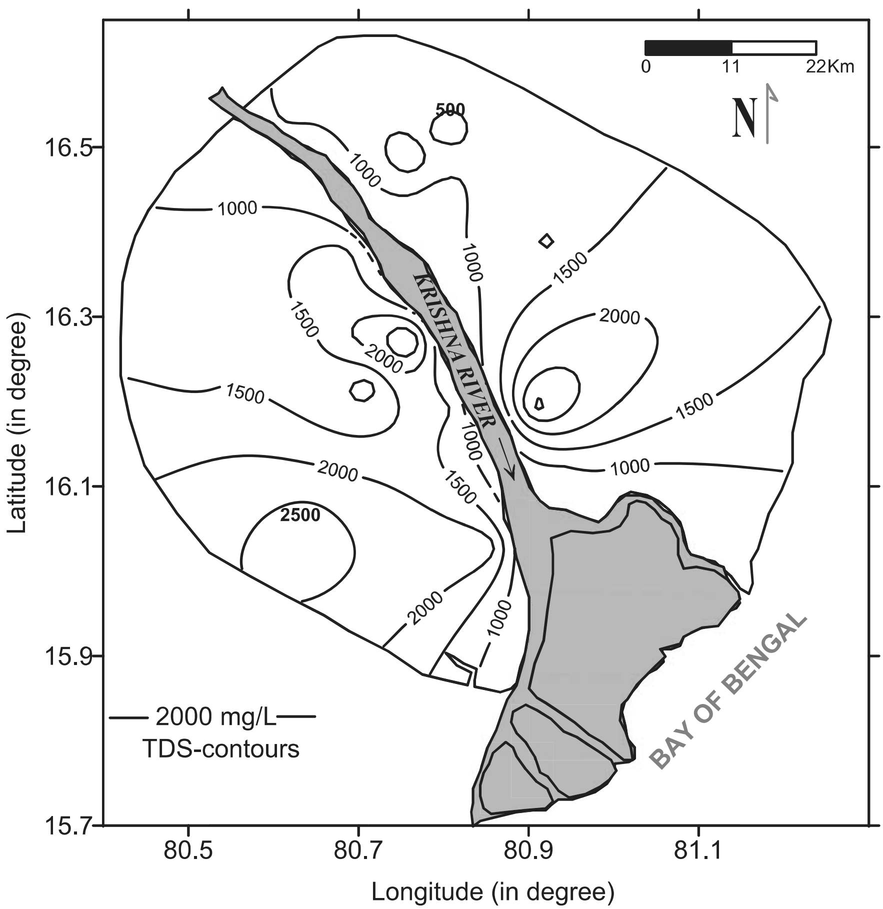

Figure 1.2. Spatial variability of the distribution of TDS in groundwater of Krishna delta.

1.3.3 *Interrelationship between the hydrogeochemical characteristics and delineation of fresh groundwater*

The trace element data on the groundwaters reveal good correlations between TDS and Sr and B (Fig. 1.3). In general, Sr and B concentrations were low in fresh groundwater, but high in brackish and saline waters (Table 1.3). The spatial variation of Sr and B distribution are shown in Figure 1.4. It is observed that the effects of seawater intrusion are less in the central part of the delta close

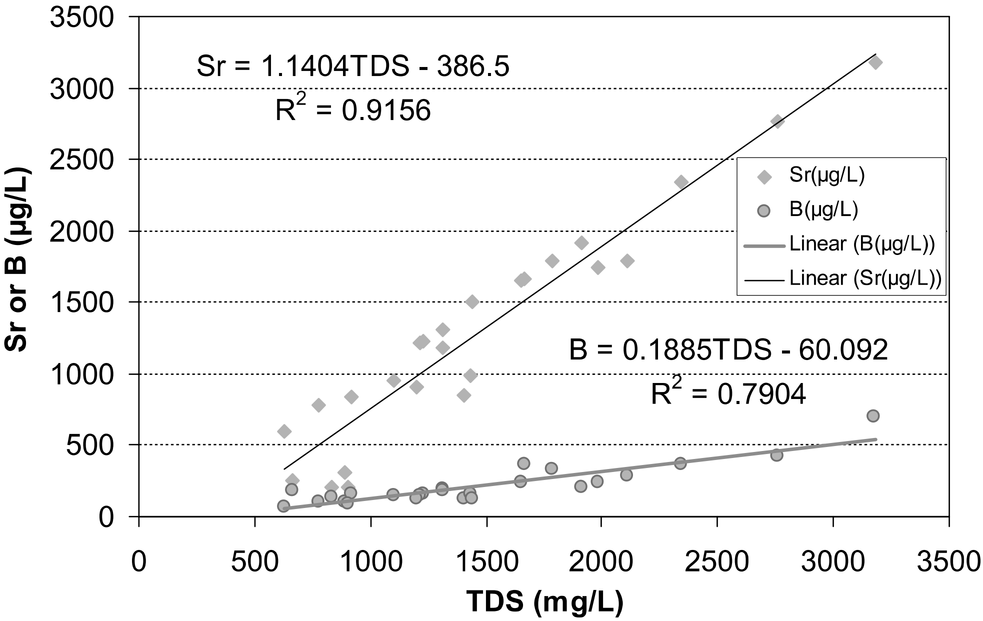

Figure 1.3. Relationship between TDS with the distribution of Sr and B in groundwaters of Krishna delta.

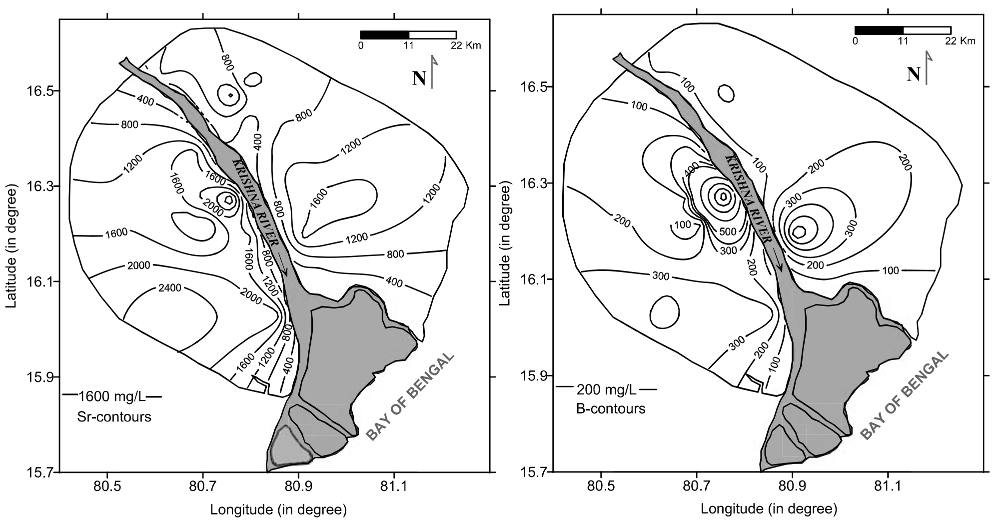

Figure 1.4. Spatial variation of the distribution of Sr and B in groundwater of Krishna delta.

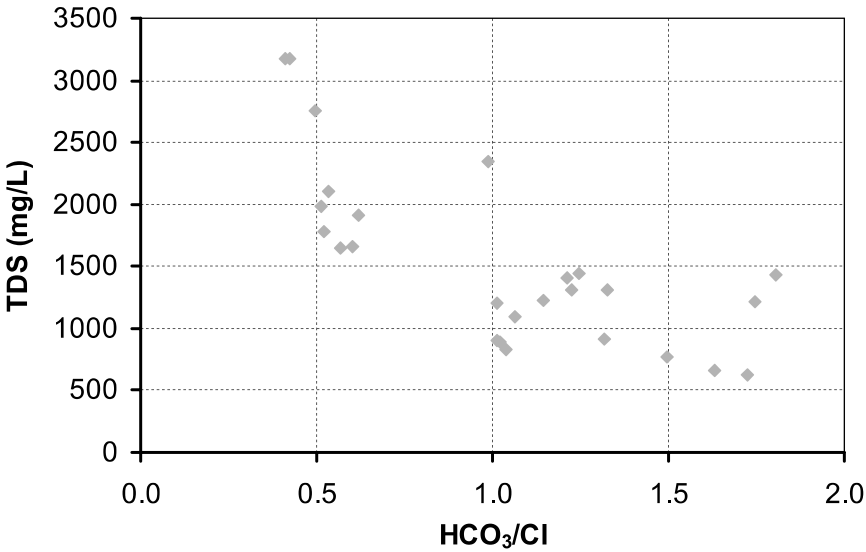

Figure 1.5. Relationship between TDS and HCO_3/Cl ratio in groundwaters of Krishna delta.

to Vyyurru-Royyuru-Ellurru regions. Groundwaters near Nidumolu and Kudali region indicate elevated concentrations of both Sr as well as B and reveal seawater intrusion. In South Krishna delta, the situation is little different because of high ingression rate observed in Chadalwada-Reppale-Induru tie lines. Spatial variability of B in groundwater shows that the concentration of B is more towards the eastern part of the Krishna river in the North Krishna delta. In the middle part of the delta the seawater intrusion is more, however, the withdrawal of water is also indicated more in this part.

The aqueous ionic species such as HCO_3^-, Cl^- and TDS have been considered as useful chemical parameters for the delineation of fresh groundwater resources in the coastal environments. A plot between HCO_3/Cl ratio and TDS (Fig. 1.5) shows a more or less systematic variation in the chemical behavior of the HCO_3/Cl ratio with TDS. In particular, it is more effective below 1500 mg/L (TDS) and 1.0–2.0 (HCO_3/Cl ratio). However the results obtained by HCO_3/Cl and TDS when compared to B and Sr (Table 1.4), the results are found to be in good agreement.

1.4 CONCLUSIONS

In general, Sr and B were observed less in fresh groundwater, but high in brackish and saline waters. The aqueous ionic species such as HCO_3^-, Cl^- and TDS have been considered as useful chemical parameters for the delineation of fresh groundwater resources in the coastal environments. Good correlations have been found between TDS-Sr and TDS-B. The seawater contamination process is mainly caused by the intrusion of NaCl, Mg, SO_4 and other trace elements; these are comparatively more in South Krishna delta. The groundwaters of Kudali, Nidumolu, Uppuluru, Induru, Vallabhapuram and Chadalvada have been found to be intensely affected by seawater intrusion.

ACKNOWLEDGEMENTS

The authors wish to thank to Dr. V.P. Dimri, Director, NGRI, Hyderabad for his kind support and permission to publish this paper. The authors are thankful to Dr. A.L. Ramanathan and Dr. Prosun Bhattacharya for their critical comments which helped us to improve the earlier draft of this manuscript.

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CHAPTER 2

Groundwater contribution to total runoff using baseflow separation: A case study in southwestern Iran

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ABSTRACT: Estimating the monthly and annual volume of groundwater contributions to runoff within a watershed is a critical aspect of many hydrogeologic investigations. In this study, groundwater contributions to total runoff as Baseflow Index (BFI) were estimated at fourteen stream gaging stations throughout the Basins in south west of Iran. Two baseflow separation techniques, local minimum and recursive digital filter with range of 0.9–0.975 filter parameters, were used to separate daily runoff data into direct runoff and baseflow. The relative performances of the techniques were evaluated using the results obtained from recession analysis and visual inspection. Comparison of baseflow separation results indicated that recursive digital filter method with the parameter value of 0.925 has more accurate baseflow values. Temporal variability of BFI through the region was evaluated using the results of recursive digital filter method. Some possible applications of the results of this research in the study area were discussed.

2.1 INTRODUCTION

The total streamflow can be viewed as consisting of two parts, direct runoff and baseflow. The direct runoff is the storm runoff that results from rainfall excess. The baseflow is the water discharged from groundwater aquifers. Estimating the monthly and annual volume of groundwater contributions to runoff within a watershed is a critical aspect of many hydrogeologic investigations including water supply, irrigation planning, wastewater dilution, navigation, hydropower generation and aquifer recharge and characterization (Brutsaert & Nieber 1977, Troch et al. 1993, Dingman 1994, Szilagyi 2004). Baseflow, which is a major component of streamflow, can be considered a reasonable approximation of groundwater discharge to a stream (White & Sloto 1990, Holtschlag & Nicholas 1998, Smakhtin 2001). This is particularly useful for numerical groundwater flow modeling researches. Some researchers (Hoos 1990, Rutledge & Daniel 1994, Mau & Winter 1997) have shown that baseflow can be used to derive basin average estimates of groundwater recharge. In addition, an understanding of the outflow process from groundwater is also essential in water budgets and catchment response analysis.

In order to model the temporal variability of groundwater discharge at the basin scale, baseflow data series can be applied. In this study two automated baseflow separation techniques, local minimum and recursive digital filter with the filter parameter of 0.9–0.975 using WHAT model (http://pasture.ecn.purdue.edu/∼what), were used to estimate daily baseflow data series in fourteen different basins in south west of Iran.

Correlation analysis between the two methods has shown the nearly close results. A comparison of baseflow separation results of these two different approaches with the results of recession analysis and visual inspection has carried out to find the most accurate procedure. In this study, groundwater contributions to total runoff as Baseflow Index (BFI) were estimated in fourteen stream gaging stations, with thirty complete water years of daily streamflow data, throughout the Basins in south west of Iran.

The overall objective of this research is to identify the most accurate automated baseflow separation technique, and to gain a better understanding of the contribution of groundwater component of streamflow at different spatial scales. This study is intended as an important step toward improving the knowledge of the temporal and spatial baseflow characteristics of southwest basins of Iran, which is crucial to prevent groundwater depletion and pollution and to define a rule for establishment of conjunctive use in the region. Some possible applications of the results of this study were discussed regarding to analysis of low flow data series and baseflow estimation at the ungaged basins in the study area.

2.2 METHODS AND MATERIALS

2.2.1 *Study area*

In this research fourteen basins that are located in south west of Iran were selected (Fig. 2.1). The basins covers an area of approximately ranges from 24,141 to 37 square kilometer, with elevations between 600 m at the outlet of the Tangepanj hydrometric station and 4400 m at the high mountains in Pole Shaloo basin. The mean annual precipitation of the area is about 500 to 650 mm. Low flow season generally begins in July and ends in November, which dominated by baseflow. Baseflow contribution during flooding season is very significant. The selected basins

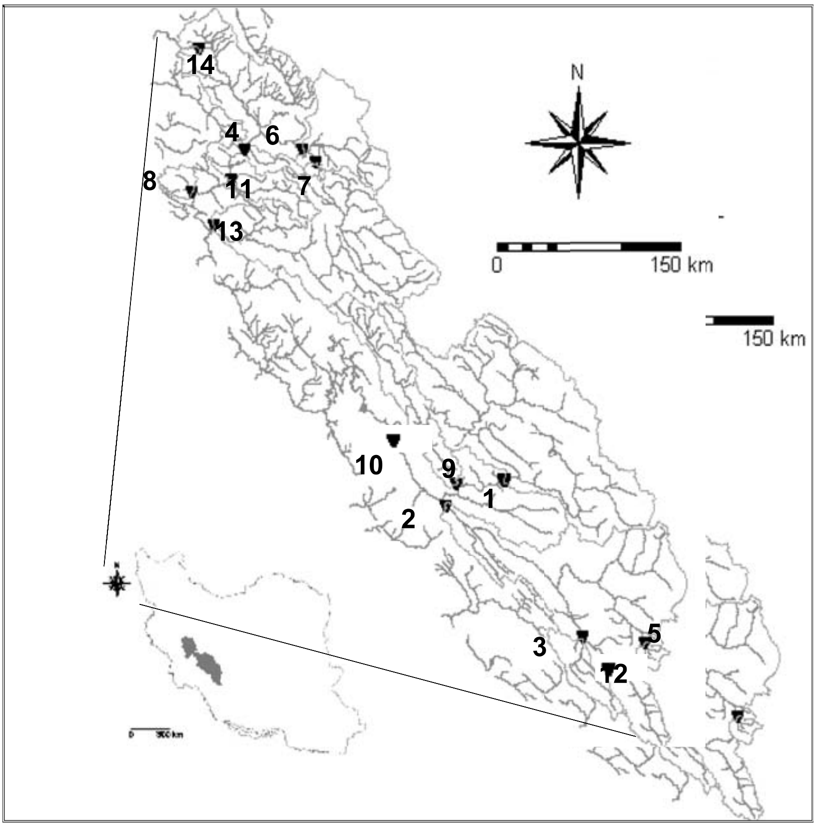

Figure 2.1. Location of the study area in Iran.

in this study have different hydrological and topographical characteristics. Details of the basins under study including average annual runoff, area, and elevation of hydrometric stations are listed in Table 2.1.

2.2.2 *Data*

The runoff data employed in this study consist of daily flows, in cubic meter per second at fourteen basins depicted in Figure 2.1, in a period of thirty years from 1969 to 1998. The data have gathered from Water Resources Research Organization (TAMAB Co.) in Iran. Temporal variation of daily runoff as an example for the outlet of the two largest and smallest basins, Pole Shaloo and Vanaii respectively, has shown in Figures 2.2 and 2.3 for the water year of 1979–80. Figure 2.2 shows a typical daily hydrograph for small basin with high elevation in study area that snowmelt consist the most part of annual flow. In Vanaii basin, for instance, much of precipitation during autumn and winter is snowfall. Therefore groundwater discharge as baseflow dominates total flow in this period. A typical daily hydrograph for large basin with different hydrological regimes is shown

Table 2.1. Details of basins under study.

No	Basin	Area (km^2)	Average annual runoff (m^3/s)	Elevation (m)
1	Armand	9900	100.2	1050
2	Barez	8999	126.8	815
3	Batari	885	16.6	1560
4	Chamchit	345	7.03	1290
5	Dehkadeh	200	5.3	2220
6	Daretakht	2185	4.4	1800
7	Kamandan	78	1.56	2080
8	Keshvar	335	5.73	770
9	Morghak	2146	74.3	860
10	Pole Shaloo	24141	341.9	800
11	Sepiddasht	7134	45.6	970
12	Shahmokhtar	1187	22.2	1730
13	Tangepanj	6390	146	600
14	Vanaii	37	2.35	2000

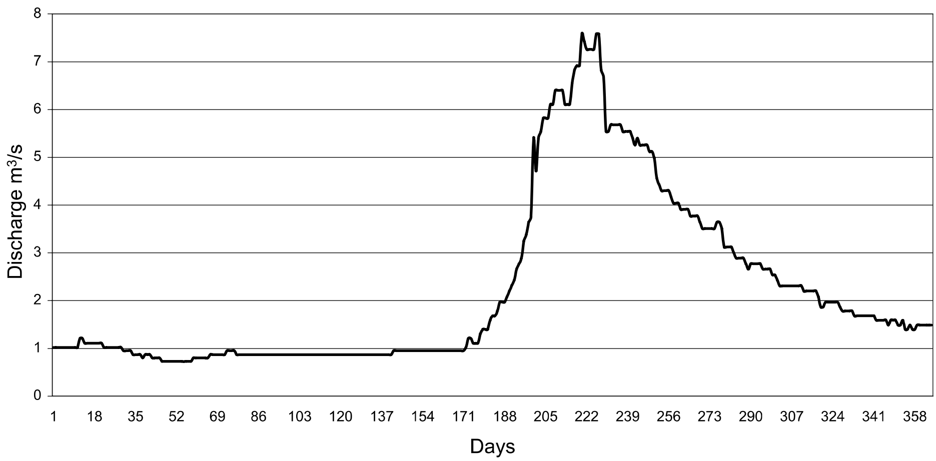

Figure 2.2. Temporal variation of daily streamflow Vanaii hydrometric station for the water year 1979–80.

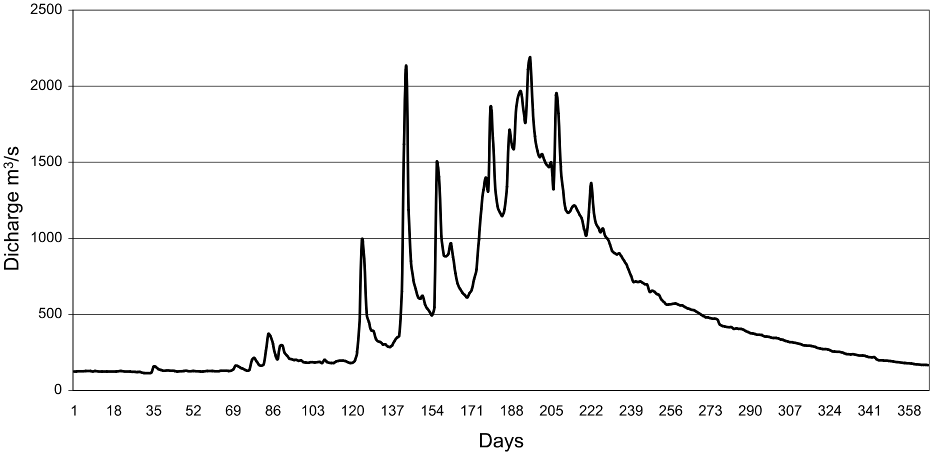

Figure 2.3. Temporal variation of daily streamflow in Pole Shaloo hydrometric station for the water year 1979–80.

in Figure 2.3, with daily hydrograph for Pole Shaloo hydrometric station during water year of 1979–80.

2.2.3 *Baseflow separation*

Baseflow separation techniques divide a stream hydrograph into two major components: baseflow and direct runoff. In this research two different methods, local minimum and recursive digital filter, for separating of baseflow from total runoff were used as an index of groundwater contribution to runoff. The first method is local minimum method. The local minimum method checks each day of a period of daily streamflow record to determine if it has the lowest discharge in one half the interval minus 1 day [$0.5(2N^* - 1)$ days] before and after each day being considered. If it is satisfied, the discharge value for that day is considered a "local minimum" value, and is connected to other local minimum values. The baseflow for all days between local minimums is estimated using linear interpolation.

The second method that was used to estimate baseflow data series is recursive digital filter method. Nathan & McMahon (1990) applied recursive digital filter as an automated approach for partitioning streamflow into baseflow and direct runoff. Recursive digital filter can be written as:

$$f_k = \alpha f_{k-1} + \frac{(1+\alpha)}{2}(y_k - y_{k-1}) \tag{2.1}$$

where f_k is the filtered quick response at the kth sampling instant, y_k is the original streamflow, and α is the filter parameter and $(y_k - y_{k-1})$ is the filtered baseflow. The filter is passed over the data three times. The parameter α affects the degree of attenuation, and the number of passes determines the degree of smoothing. Nathan & McMahon (1990) found that the value of the filter parameter that yielded the most acceptable baseflow separation was in the range of 0.9–0.95. The simple digital filter with a parameter set to 0.925 produced more accurate yields and repeatable results compared to the simple smoothing and separation rules. They found that the results were to those obtained by using traditional graphical techniques.

2.2.4 *Evaluation of baseflow separation techniques*

The relative performances of the techniques used in this study need to be compared and evaluated based on reliable criteria. Although it can be said that all baseflow separation techniques are arbitrary and subjective, however, recession analysis could be considered as the most reliable evaluation measure of baseflow separation techniques. Perhaps hydrograph separation based on the use of chemical or radioactive tracers is also an accurate tool for separating different parts of streamflow, which is unfortunately, not justifiable in application and in economical point of view. Therefore, other possible and yet reliable measures should be used.

In this research graphical inspection and recession analysis were used to compare the results of different automated methods and to find the most accurate baseflow separation technique in the study area. Relative performance of the methods were compared based on mean absolute error (MAE) and root mean square error (RMSE) criteria with respect to the results of recession analysis:

$$MAE = \frac{1}{n}\sum_{i=1}^{n}[z(x_i) - \hat{z}(x_i)] \tag{2.2}$$

$$RMSE = \sqrt{\frac{1}{n}\sum_{i=1}^{n}[z(x_i) - \hat{z}(x_i)]^2} \tag{2.3}$$

where,

n: number of data observations
$z(x_i)$: observed values by recession analysis
$\hat{z}(x_i)$: estimated values using automated techniques

2.2.5 *Baseflow index*

The baseflow index (BFI) is a dimensionless ratio, which developed by Lvovich (1979) and the Institute of Hydrology (1980). BFI can then be calculated as the volume of baseflow divided by the volume of total runoff for each year or for total period of record. This index can present some information about the proportion of the runoff that originates from stored sources. The index can be calculated from streamflow data or estimated from basin geology (Institute of Hydrology 1980). The first step in estimation of BFI is separating of baseflow from surface runoff. BFI has many practical applications, especially in rainfall-runoff modeling. Also it can be used as a basin characteristic to compare the flow characteristic of different basins. In this research the BFI values for fourteen basins using the selected baseflow separation method were calculated. A discussion of BFI determination results is given in the following section.

2.3 RESULTS AND DISCUSSION

In this research both the local minimum and recursive digital filter with the filter parameters of 0.9, 0.925, 0.95 and 0.975, were used to estimate baseflow data series in daily and monthly time scale in fourteen basins in south west of Iran. Figure 2.4 shows an example of the graphical output of baseflow separation for Pole Shaloo hydrometric station using the local minimum method for the water year of 1979–80. Figures 2.5 to 2.8 have shown the results of baseflow separation using recursive digital filter method using filter parameters of 0.9, 0.925, 0.95 and 0.975 respectively.

Figures 2.4 to 2.8 could be used for a graphical comparison between the above methods with respect to the performance of baseflow separation ability. As can be seen in Figure 2.4, local minimum method slightly overestimates the baseflow values. It gives a good estimate of the baseflow rise but it fails to predict the end of surface runoff. This method also estimated a very high baseflow in the troughs of multi peaked events and in the troughs between two events that affected each other. On the other hand, as can be seen in Figure 2.8, recursive digital filter method with

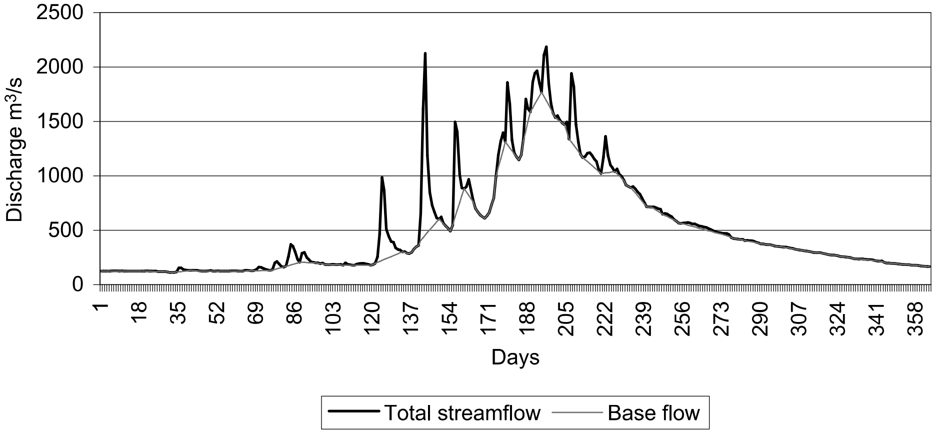

Figure 2.4. Total streamflow and separated baseflow, using local minimum method, in Pole Shaloo hydrometric station for the water year of 1979–80.

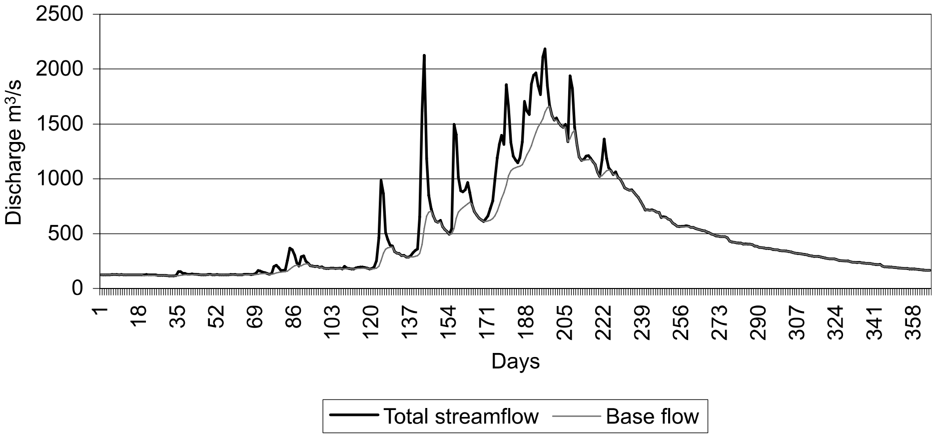

Figure 2.5. Total streamflow and separated baseflow in Pole Shaloo hydrometric station for the water year of 1979–80, based on recursive digital filter with the parameter value of 0.9.

the parameter value of 0.975 underestimates the baseflow values. Underestimation of baseflow is very clear and significant during high flow season, which occurred at the end of winter and during spring. Underestimation of the baseflow can be distinguished also when the parameter value of 0.95 was chosen (Fig. 2.7). With respect to the graphical comparison, recursive digital filter with the parameter values of 0.9 and 0.925 have more reliable results (Figs. 2.5 and 2.6, respectively).

As a comparative study, recession analysis was carried out to a portion of the data (three years of the data series from 1996 to 1998) and it was considered as a measure of comparisons between the automated methods. Baseflow recession curve itself contains valuable information about ground water flow and it is widely used in hydrological models and other water resource applications. Values of Qn and $Qn + 1$ are plotted against each other and the lower envelope line of all points is considered to indicate the k-values, which is the recession constant of the particular basin. A recession curve equation can be written as:

$$Q_b = Q_0 \cdot k^t \tag{2.4}$$

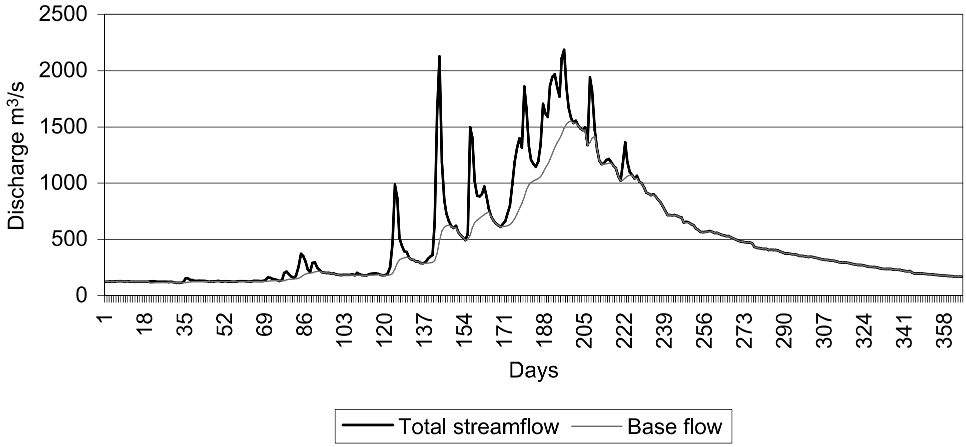

Figure 2.6. Total streamflow and separated baseflow in Pole Shaloo hydrometric station for the water year of 1979–80, based on recursive digital filter with the parameter value of 0.925.

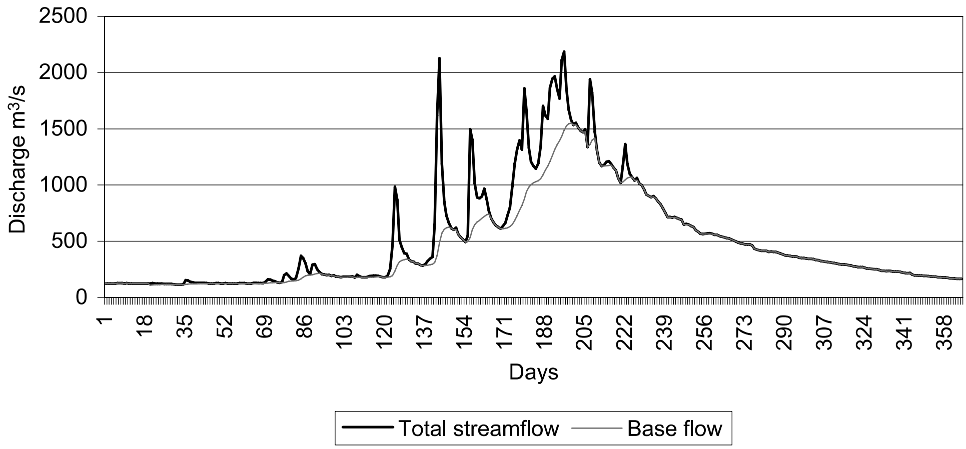

Figure 2.7. Total streamflow and separated baseflow in Pole Shaloo hydrometric station for the water year of 1979–80, based on recursive digital filter with the parameter value of 0.95.

where Q_b is baseflow, Q_0 is initial value of baseflow, k is recession constant and t is time. First of all, the runoff hydrographs were plotted on semi-log paper, which flow is on log scale and time on natural scale. The start and end of surface runoff were identified using change of recession constant (k) or slopes of the semi-logarithmic plots. Different procedures can be adopted for determining the shape of baseflow hydrograph. In this case the separation of the baseflow from the direct runoff achieved by drawing linear segments at the points of start and finish of direct runoff. This method is almost impractical and tedious when it should be applied to a large number of basins furthermore they cannot be formulated. Therefore, it is necessary to evaluate the performance of the automated baseflow separation techniques, which are simple, consistence and does not need subjective efforts. This procedure will assure applicability of the baseflow values as independent variables for future water resources researches.

Error estimation criteria including MAE and RMSE have been used to compare the results of automated baseflow separation techniques with the results of recession curve analysis technique. Table 2.2 gives the error estimates obtained for each methods, local minimum, recursive digital

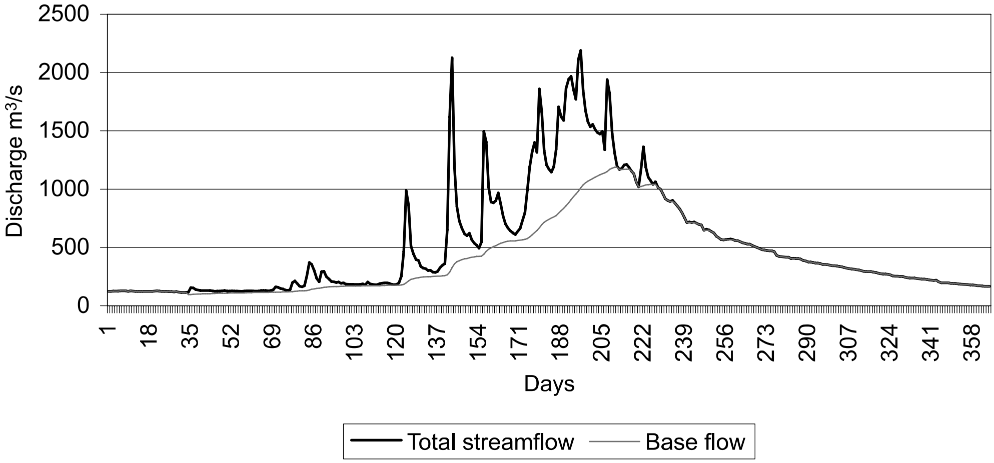

Figure 2.8. Total streamflow and separated baseflow in Pole Shaloo hydrometric station for the water year of 1979–80, based on recursive digital filter with the parameter value of 0.975.

Table 2.2. Performance analysis of the results of automated baseflow separation techniques in Pole Shaloo basin using error criteria.

		Error	
No.	Method	MAE	RMSE
1	Local minimum	8.34	13.92
2	Recursive digital filter (0.9)	8.77	19.07
3	Recursive digital filter (0.925)	8.19	15.10
4	Recursive digital filter (0.95)	10.37	17.83
5	Recursive digital filter (0.975)	27.02	47.18

filter with the parameter values of 0.9, 0.925, 0.95 and 0.975 in comparison of recession curve analysis results for Pole Shaloo hydrometric station during period of 1996 to 1998. As can be seen in Table 2.2, recursive digital filter method with the parameter value of 0.925 (method number 3) present more accurate results with respect to the recession curve analysis.

If the runoff process is conceptualized in a way that flow from ground water aquifers begins on the hydrograph recession, it will be necessary to select the point on the recession curve where direct runoff ends. As McCuen (1998) stated, the most common conceptualization uses the inflection point on the hydrograph recession. In this research as other graphical inspection method, the inflection point on the recession curve was used to compare the ability of the automated baseflow techniques. The inflection points were identified using change of slopes of the semi-logarithmic plots. Figure 2.9 shows the way that was used to extract the inflection point on the recession curve of hydrographs. Figure 2.10 shows the different performance of the automated baseflow separation methods used in this study to accurately define the inflection point. As can be seen in Figure 2.10, methods of local minimum (No. 1), and recursive digital filter with the parameter values of 0.95 and 0.975 (No. 4 and 5) have not found accurately the inflection point. Recursive digital filter method with the parameter values of 0.925, 0.95 (No. 2 and 3) have slightly similar results. However the third method, recursive digital filter method with the parameter value of 0.925 has slightly more accurate result than the second method (Fig. 2.10).

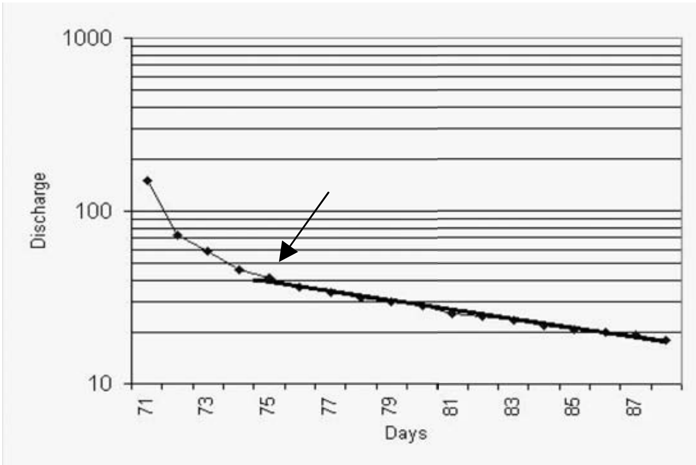

Figure 2.9. Recession curve analysis for finding inflection point on semi-logarithmic plot.

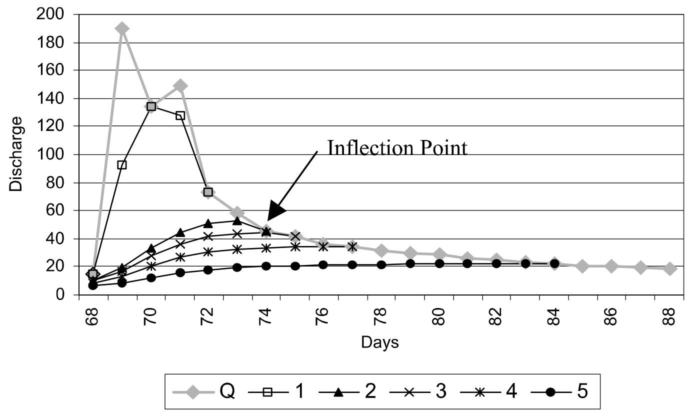

Figure 2.10. Comparison of baseflow separation techniques regarding to the place of inflection point.

Table 2.3. Summary of BFI parameters in Pole Shaloo, Batari and Vanaii basins.

	Basin		
Parameter	Pole Shaloo	Batari	Vanaii
Mean	0.88	0.82	0.89
Max	0.93	0.93	0.92
Min	0.82	0.74	0.79
St. Dev	0.02	0.05	0.03

In this research, the monthly and annual BFI data series for all fourteen basins under study were extracted using recursive digital filter method with parameter value of 0.925. Table 2.3 shows summary statistics of annual BFI values such as mean, maximum, minimum and standard deviation of BFI values for some of basins with different spatial scales. Temporal variability of the annual BFI values as an example can be seen in Figure 2.11 for Pole Shaloo hydrometric

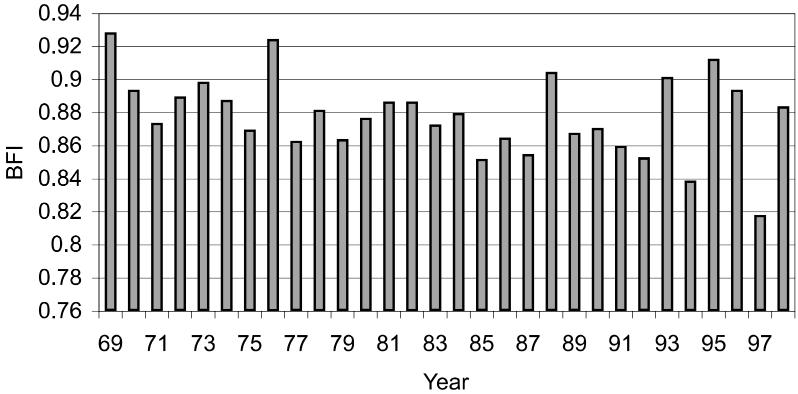

Figure 2.11. Temporal variability of the annual BFI values in Pole Shaloo basin for the period of 1969 to 1998.

station. Figure 2.11 show that considerable variations occurred in the calculated values of BFI from year to year in that particular basin. Pole Shaloo, Batari and Vanaii Basins are some example of typical large, medium and small-scale basins in the study area. As can be seen in Table 2.3, annual average of BFI varies from 0.74 to 0.93. It is evident that more than 80 percent of the annual runoff could be considered as baseflow or groundwater discharge in the study area and similar basins.

2.4 CONCLUSIONS

In this research, two automated baseflow separation techniques, local minimum and recursive digital filter with different parameters, were used to determine baseflow data series at fourteen stream gaging stations, in south west of Iran. Statistical comparison of the results of these different methods with recession curve analysis and some graphical inspections indicated that digital filter method with the parameter value of 0.925 has more reliable and accurate results. Application of automated recursive digital filter is easy, accurate, consistence and does not need subjective efforts, which will assure applicability of the baseflow values as hydrological variables for future water resources management during low flow season and other applications.

Annual and monthly variability of BFI through the region examined using the selected approach. Estimating the contribution of groundwater recharge is an important aspect of many hydrogeologic and water resources investigations, especially numerical modeling and water budget analysis, at the watershed scale. In the case of water quality availability, baseflow estimates can be integrated with this information to estimate groundwater loading. As an important application, groundwater discharge could be used in conjunction with land use change trends to investigate the effects changes in land management systems at the watershed scale, which is a critical aspect in water resources planning and management. The results of this study could be used to define contribution of groundwater and baseflow data series for ungauged basins in the region using regionalization procedures regarding to importance of low flow seasons with viewpoints of water quality and environmental consequences of drought periods. On average, groundwater discharge represented approximately more than 80% of total annual streamflow for the stations modeled. This highlights the importance of managing surface water and groundwater as a single interconnected resource.

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CHAPTER 3

Groundwater resources sustainability in Qatar: Problems and perspectives

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ABSTRACT: The State of Qatar is a peninsula located in the Arabian Gulf. It has no surface water and the only accessible and renewable conventional source of water is groundwater that is being recharged mainly from the sparse rainfall within the Qatari territories. The abridged groundwater balance explains that the abstraction by the Qatari farms is always greater than recharge from rainfall apart from few exceptional years and the accumulated deficit was estimated to be more than 2180 Million Cubic Meters (MCM) by the year 2003/2004. The aim of this paper is to identify the existing pressing problems that are currently being faced for the management of this important source and hence suggest some applicable and practical solutions in order to reach groundwater resource sustainability. Results showed that the gap between the available groundwater resources and the agricultural demand can be bridged and sustainable use of groundwater can be achieved by applying the proposed groundwater resources management plan.

3.1 INTRODUCTION

The State of Qatar is a peninsula located in the Arabian Gulf. The total area including the attached offshore islands is about 11493 km^2. The population was around 111,000 in 1970 and leaped to 744,000 in 2004, living mostly in Doha, the capital of the country (ASA 2004). The climate is typically arid type, characterized by scanty winter rainfall with an annual mean of 81 mm for the period from 1972 to 2004, high summer temperature, high evaporation, very strong wind and high relative humidity (Agro-hydro-meteorological Year Book 2004). It has no surface water and the only accessible and renewable conventional source is groundwater that is being recharged from the light rainfall within the territories of Qatar.

The abridged groundwater balance explains that the abstraction by the agricultural sector is always greater than recharge from rainfall apart from few exceptional years and the accumulated deficit is estimated to be more than 2180 Million Cubic Meters (MCM) by the year 2003/2004.

This apparent deficit could be assigned to certain particular reasons. These reasons are summarized as follows: High rate of population increase, Very limited water resources, severe climate conditions, inefficient use of water in most of the Qatari farms, existing technical and institutional obstacles, fast rate of urbanization, and ambitious agricultural and industrial national development plan. Over and above, one of the most important reasons is the absence of a clear groundwater resources management and development strategy.

The paper consists of four main parts; the first one is a review about the groundwater resources in the State of Qatar, the second part presents the groundwater use among all sectors, the third part introduces the groundwater balance and identifying the main problems, the fourth part presents is groundwater sustainability in Qatar, and the fifth part is suggesting the solutions in a form of proposed management plan. The results showed that the groundwater demand can be satisfied if the proposed solutions are considered.

3.2 GROUNDWATER RESOURCES: OCCURRENCE AND REVIEW

Qatar has no rivers or lakes and the only available and renewable conventional source of water is groundwater that is being recharged from the scarce rainfall within Qatar while very small part of it from outside the Qatari territories in the Kingdom of Saudi Arabia.

Two main and three secondary groundwater basins have been identified: the main ones are the northern and the southern ones and the three secondary are Abarug (Alat) basin near Abu Samra, Doha basin in Doha area and the Aruma basin within southwest Qatar. Figure 3.1 shows different groundwater basins in Qatar.

3.2.1 *The northern groundwater basin*

The partial absence of shale in north of Qatar has made hydraulic continuity between Rus and the upper Umm Er Rhadouma Formations more active which resulted in a water complex system. Generalized lithostratigraphic succession of the geological formations in Qatar and their descriptions are shown in Figure 3.2. The northern basin exists as a fresh lens in both formations and it is considered as the most important groundwater source of acceptable quality in Qatar. Water quality in this region is of calcium-bicarbonate type.

Fresh-salt water intrusion and up-coning conditions are controlled by the degree and extend of the overall exploitation and recharge mechanism. Under current conditions, the extent of this zone is increasingly being reduced by sea water intrusion along coastal margins, predominantly along the eastern coast. However, in the northern groundwater zone, the thickness of the Rus formation varies from 20–80 m of which saturated zone is about 30–60 m. The depth of water varies from 10–40 m from ground surface.

The northern groundwater zone is separated from the southern groundwater zone by a V-shaped structure that its apex is nearly at Rodat Rashid. North of Rodat Rashid, the Rus Formation is of predominately carbonate facies with only residual deposits of interbedded gypsum.

3.2.2 *The southern groundwater basin*

Within this zone, the Rus formation with the depositional sulphate facies contains a comparatively thick evaporate beds of high solubility. The existence of Midra Shale Member in the overlying Dammam Formation has retarded the dissolution of the existing beds. This resulted in a poor aquifer of irregular unconfined or perched water bodies of varying quality, and lack of aerial hydraulic continuity with the underlying Umm Er Rhadouma aquifer (Eccleston et al. 1981).

Nevertheless, the southern groundwater zone is of far less recharged quantity than that of the northern basin and covers slightly more than half of the land area of the State Qatar. Within most of the zone, water levels are in excess of 30 meter below ground surface.

The northern and the southern zones are recharged locally from the deep percolation of the limited rainfall. Figure 3.3 represents the annual total rainfall for the state of Qatar from the year 1972 to the year 2004. The average annual recharge amount for these two basins, from the year 1971/1972 to the year 2003/2004, is 56.80 MCM.

3.2.3 *Doha basin in greater Doha area*

Although, groundwater levels in the main basins has recently shown rapid decline since the 1970s, water levels in greater Doha area is in continuous rise. The Doha basin is of very recent origin and is being recharged mainly from excess landscape irrigation, leaking from water pipes, rainfall in great Doha area and leaking from septic tanks. The annual amount of water collected by the drainage network from the Doha basin is estimated to be 23.4 MCM at the year 2004. Although, this water is of moderate water salinity, it is contaminated with municipal sewage water in many locations. Now, it is being disposed to the Gulf at points with low topography as an economically best-acceptable solution.

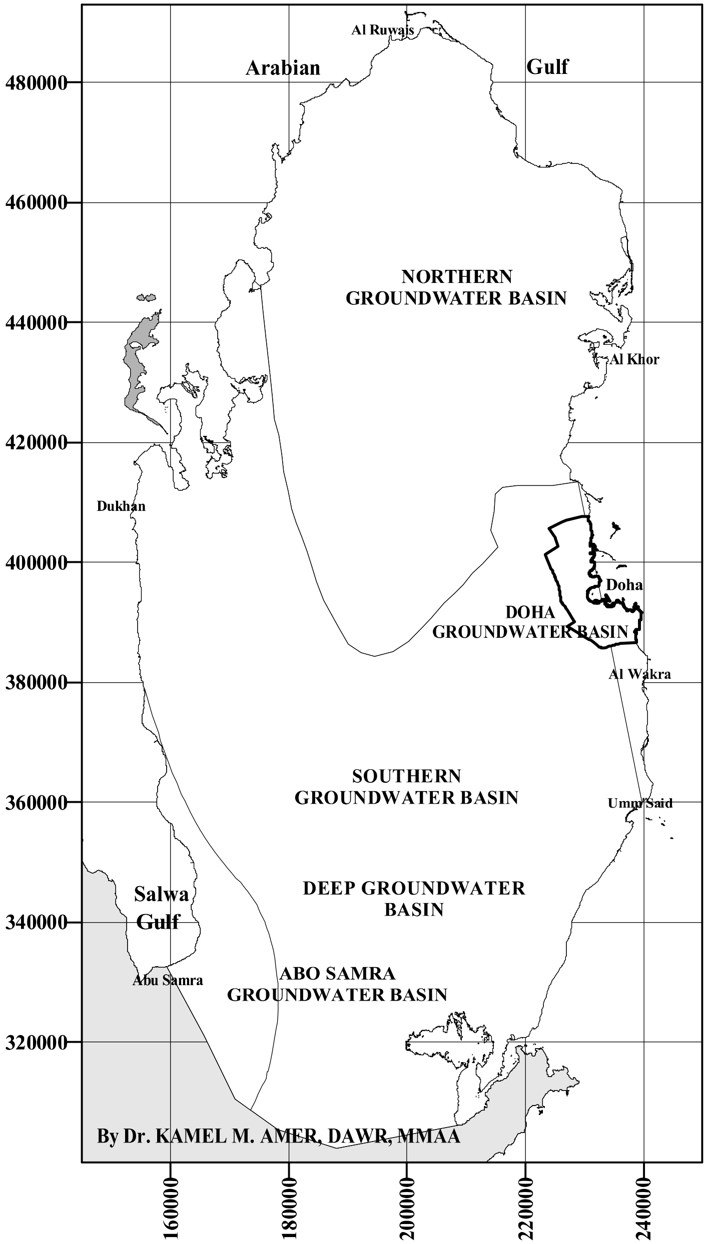

Figure 3.1. Groundwater basins in Qatar.

3.2.4 *The upper Dammam aquifer (Alat aquifer near Abu Samra)*

The Abarug Member (equivalent to the Alat in the Kingdom of Saudi Arabia) of the upper Dammam formation occurs mainly in the southwest of Qatar. The Alat aquifer, which is an artesian one, located at the extreme south western region of Qatar, in the vicinity of Abu Samrah, occurs only on the flanks of the Dukhan Anticline where it reaches about 2 m in thickness. It is thin (from 10 to 25 m) and rests upon the Abarug Marl which is also about 10 m thickness and acts as the

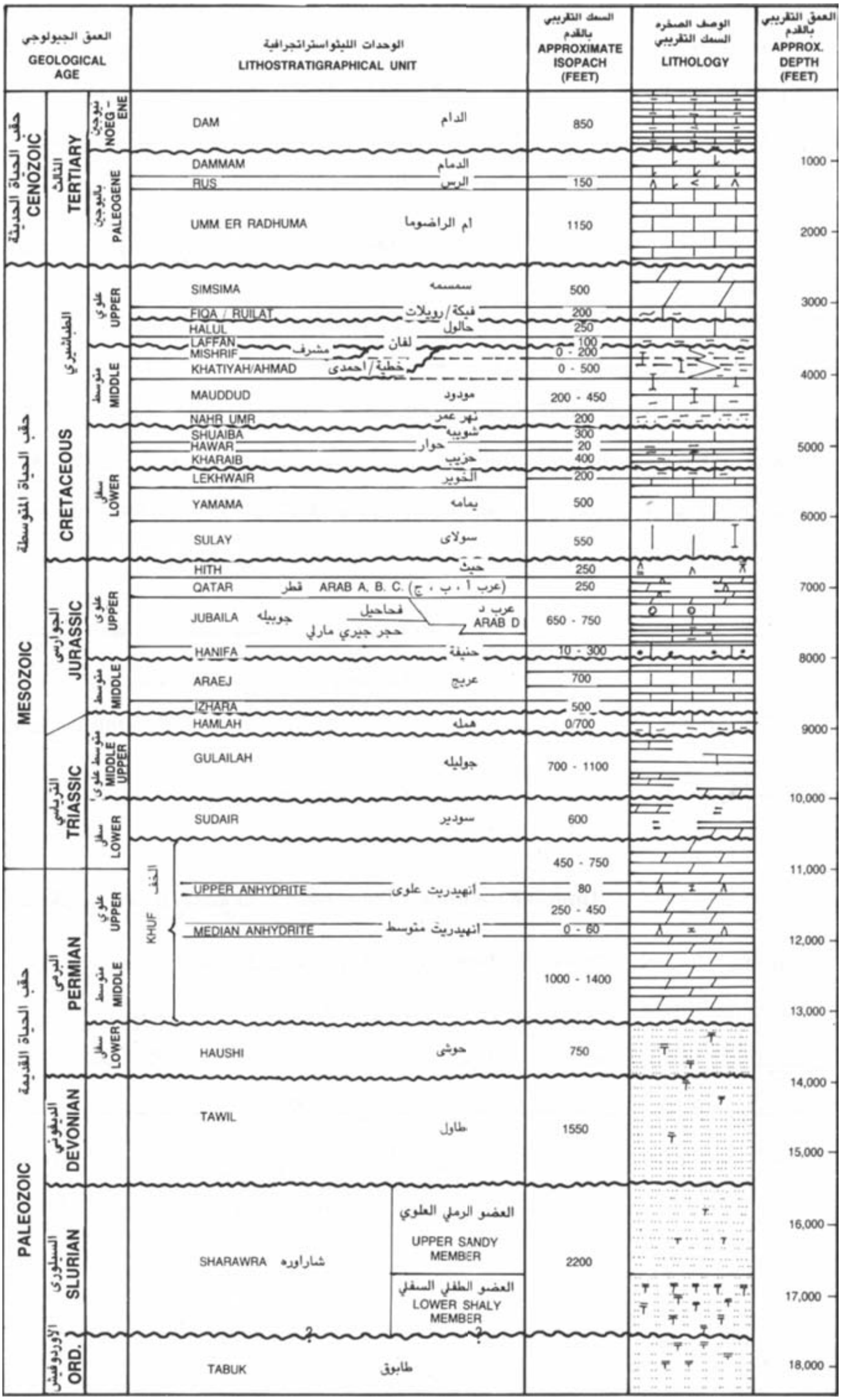

Figure 3.2. Generalized lithostratigraphic column of Qatar, after QGPC.

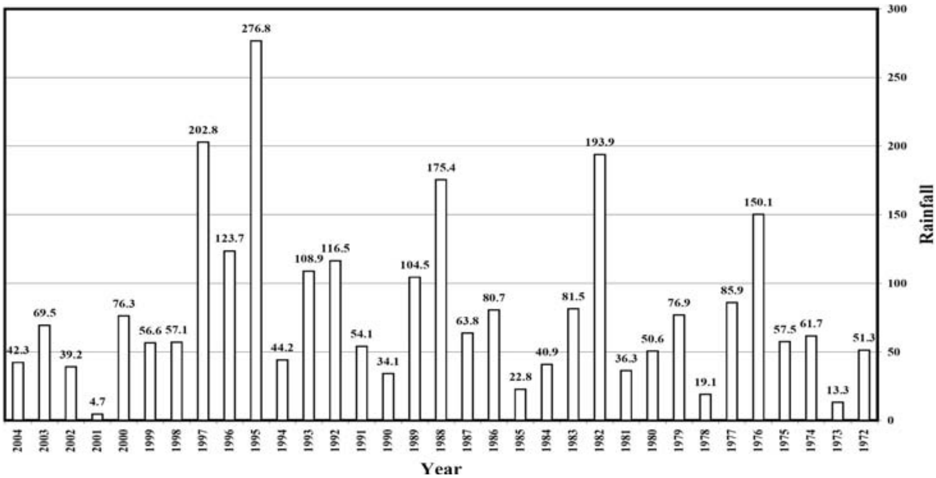

Figure 3.3. Annual total rainfall in the state of Qatar (1972–2004).

supporting aquiclude. Recharge occurs over the outcrop of the upper Dammam in the Kingdom of Saudi Arabia and the piezometric gradient appears constant at about 0.0015 as far as the border area west of Salwa where an increase to 2 to 3 times this value is apparent.

This aquifer is of limited extent and with an average thickness of about 15 meters, front flow in Qatar of 20 km and an average transmissivity of 200 m^2/day. Its average horizontal permeability is about 13 meter per day. It is believed to have an annual safe-yield of up to 2.2 MCM of variable quality brackish groundwater. The direction of groundwater flow in the Alat aquifer is assumed to be east—northeast wards, following the regional pattern, towards the centers of extraction.

3.2.5 *The Aruma deep aquifer within southwest Qatar*

The Aruma aquifer within southwest Qatar comprises approximately 130 meters of limestones belonging to the Aruma Formation. The top of the aquifer is deep and exceeds 450 meters within southwest Qatar. The aquifer is confined. It is overlain by thick and relatively impermeable deposits of the Lower UER and is underlain by a sequence of shales of up to 100 meters thickness belonging to the Lower Aruma Formation. There is evidence to show that the aquifer thickens towards the north and becomes less isolated hydraulically.

The aquifers have a transmissivity value within the range 40 to 60 m^2/day. The storativity value is approximately 5×10^{-5}. The geometry of the aquifer, its hydraulic character and the prevailing hydraulic gradient suggests that the annual volume of throughflow is small and is estimated to be about 0.5 MCM. On account of its depth and isolation from surrounding water-bearing lithologies, this recharge is unlikely to be within Qatar. Groundwater flows in a northerly direction although this needs further confirmation.

The quality of groundwater declines along the hydraulic gradient in a northerly direction. In addition, there is limited evidence that a further component of decline occurs towards the east.

3.3 GROUNDWATER USE

Till the middle of the last century, groundwater comprised the main source of Qatar's water supply for both domestic and agricultural purposes. The Government of Qatar decreased the municipal water's share of groundwater from 25% in 1960 to less than 1% by the year 2004. Nevertheless, the present accustomed strategy in Qatar is to use groundwater resources mainly for agriculture

Table 3.1. Groundwater withdrawal from northern and southern basins.

Year	No. of active farms	No. of working wells	Total abstraction for agricultural use (MCM)	Total abstraction for municipal use (MCM)	Total abstraction (MCM)
59/1960	N.A.	N.A.	3.0	1.0	4.0
64/1965	N.A.	N.A.	22.0	1.2	23.2
69/1970	N.A.	N.A.	36.0	2.0	38.0
75/1976	259	660	51.2	5.3	56.5
79/1980	337	806	67.0	1.5	68.5
90/1991	780	2496	142.5	2.5	145.0
95/1996	899	2769	234.4	2.6	237.0
00/2001	908	3340	234.6	2.3	236.9
03/2004	909	2981	218.4	2.4	220.8

N.A.: Not Available.

and construct desalination plants for supplying drinkable water while the treated sewage water is constrained to the irrigation of forage crops and landscape ornamentals.

During the pre-oil era, and because of the limited water resources in Qatar, people concentrated their activities towards the sea; thus agricultural development was extremely limited (Al-Nasr & Al-Sheeb 1999). But with the accumulation of vast oil revenues, the Government paid some attention on expanding the sector in order to achieve some extent of food security (Al-Kuwari 1996).

Table 3.1 shows the total groundwater withdrawal, from both the northern and southern groundwater basins. Groundwater withdrawal increased steadily. It was about 4.0 MCM by the year 1959/1960 and reached 220.8 MCM by the year 2003/2004, which means 55 times increase. Furthermore, the number of working wells increased from 660 in 1975/1976 to 2981 wells in 2003–2004. Accordingly, this resulted in huge stress on groundwater resources and caused severe groundwater levels decrease and groundwater quality deterioration.

Farms are concentrated geographically in the north and the middle of the state of Qatar where a groundwater reserve is with acceptable quality, on the contrary to the south, where groundwater is more saline. This is also indicated by the distribution of wells all over the country: 42% of wells are in the north, 51% in the centre and 7% in the south (Hashim 1995, MIA 1984).

3.4 GROUNDWATER BALANCE AND ASSOCIATED PROBLEMS

The performed groundwater balance is a quantitative summing up of all inputs and outputs to and from the groundwater system—for both the northern and southern zones as they are the most important ones. Input in the form of recharge by rainfall is weighed against outputs of withdrawal and other losses. Luckily, this water balance has been carried out since 1971/1972.

It showed that the utilization is often greater than recharge, with a factor that ranges, at the present, from four to five, except for few years. Consequently, the mounted up groundwater deficit is estimated to be more than 2180 MCM by the end of the year 2003/2004. It represents more than 87% of the early estimated total accessible fresh groundwater storage in Rus and Upper Umm Er Rhadouma formations, which is about 2500 MCM. A cut down groundwater balance is presented in hereafter Table 3.2.

Obviously, it is proved that the main problem facing groundwater in Qatar is the provoked imbalance between demand that is in rise and replenishment that is in short supply. In case this situation lasts, the over-utilization will cause entire depletion of accessible fresh groundwater resources within the next five years.

Table 3.2. Groundwater balance from 1971/1972 to 2003/2004 (MCM).

Year	Groundwater from rain	Gross consumption	Annual balance	Accumulated balance
1971/1972	30.360	42.580	−19.580	−19.580
1975/1976	77.410	56.500	17.820	−87.100
1979/1980	40.690	68.500	−30.150	−206.210
1990/1991	22.450	145.000	−104.300	−688.430
1995/1996	161.846	237.000	−34.546	−1002.268
2000/2001	53.059	236.900	−143.190	−1710.242
2003/2004	30.531	220.800	−153.699	−2181.705

This problem resulted mostly from the extreme withdrawal for agricultural activities as its share now is more than 99% of the total groundwater extraction. This fact along with the absence of metering and enforced regulations encourages the uncontrolled consumption. In addition to that, groundwater in Qatar is accessible to farmers free of charge while they pay only for pumping cost.

Over and above, groundwater quality deterioration is potentially one of the most damaging of all of the influences because it commonly takes significant time before the influences commence to become known. Eventually, if the resource is not managed suitably salinity levels will increase and the groundwater will rapidly turn out to be too saline to use for irrigating many crops.

Also, this problem is revealing itself in the abandonment of farms in coastal areas and consequently their rapid soil degradation. It is stated that out of 1265 registered farms in 2003/2004, 356 were abandoned. However, If the present practice continue, not only farms near the coast will face salty water problems, but also farms farther inland will face similar ones.

Management of this limited water resource must comprise a comprehensive knowledge of the cause—result relationship that the human behavior has, not only on groundwater but, on the broader surroundings as well.

The groundwater system in Qatar is exceptional and requires sensitive planning and management approaches, so as to deal with the consequences of groundwater over use. These consequences contain the following issues:

- decrease in groundwater levels,
- up conning of saline water and to seawater intrusion,
- deterioration in groundwater quality,
- decrease in yield,
- decrease in groundwater storage,
- salinization of soil and,
- damage to ecosystems.

3.5 GROUNDWATER SUSTAINABILITY IN QATAR

Sustainable groundwater resources management is defined as the group of measures, which avoids an irreversible or quasi-irreversible destruction of groundwater and any other natural resource depending on it such as soil and ecosystems. Such management permits that resource to continue its service, together with ecological service, over very long periods of time. The withdrawal from a groundwater aquifer should in the long term not be larger than the long-term average recharge.

A number of pioneering tactics may be undertaken in order to improve the sustainability of the available groundwater resources. They include some arrangements of aquifers use such as the re-use of reclaimed water, artificial recharge of rain water using recharge wells, and employing the integrated water resources management principals.

However, In order to diminish groundwater deterioration in Qatar, it is suggested to limit the annual groundwater withdrawal from both the northern and southern groundwater basins to about 60 MCM, which is considered the upper ceiling of the annual groundwater recharge. In the meantime, groundwater augmentation and maximizing the use of the other potential water sources is a main objective. This will allow the country to accomplish the maximum allowable level of food security without scarifying the groundwater resources in the northern and southern groundwater basins.

3.6 PROPOSED GROUNDWATER RESOURCES MANAGEMENT PLAN

In order to ensure groundwater sustainability, a tangible and challenging plan should be proposed. This plan must be away from any exaggeration and overestimate that will block or delay its implementation arrangements.

To apply this above mentioned plan, it is encouraged to put the solutions in a form of a wide-ranging scheme of two modules. The first is a short-term strategy that could be put into operation in the near future with some immediate actions. The second is a long-term one that should have a comprehensive preparation for some eternal measures through it.

In the route of these two strategies, sufficient financial resources and precise cash-flow are required to be due to allow carrying out this above mentioned plan and its conjugate measures.

This plan should be in agreement with Qatar's general national development plan in order to accelerate its progress (the general national development plan) and conform to its main concerns. Besides that, this plan is supposed to be in synchronization with the widespread surroundings, limitations and uniqueness of the State of Qatar whether the physical, the environmental or the social ones.

3.6.1 *The short-term strategy*

The proposed short-term strategy can be defined as all prompt procedures that are compatible with and will attain protection and conservation of the readily available groundwater resources. Actually, it is the first step of any sustainable groundwater resources management and development plan. Logically, the software measures always come before the hardware solutions. The following measures can be put into action as soon as possible:

- Consider groundwater as a national asset and suspend licensing of establishing new farms or adding new areas to the existing ones and Stop any additional withdrawal from the groundwater aquifer in a distance more than 15 km from the shoreline.
- Identify the wells located in the environmental protectorates and abandoned farms either in or outside towns to close them up. Conclusively, this will reduce the number of licensed wells.
- Close some of the existing non-productive farms that use groundwater and closing their wells. The owners can then be subjected to either financial or replacement compensation. The replacement compensation has to be with farms supplied with any of the non-conventional water resource such as treated municipal or industrial sewage effluent.
- Fix the permitted irrigated area in each farm and restrict the crop to certain types that recommended by Qatar's agricultural policy. The priority should be given to crops of lower water consumption and tolerant to high salinity.
- Improve irrigation efficiency by expanding modern irrigation techniques within a time frame of 5 years for the whole of the country. Besides, installing water meters in order to observe the total water abstracted from groundwater by each farm.

It is anticipated that the above-mentioned measures will lead to save about 40% of the total abstracted amount from the groundwater aquifer (Hashim & Abdulmalik 1997).

- Develop the administrative and institutional arrangement of the Department of Agricultural and Water Research, The Water Research Section which is responsible for groundwater resources management and development in Qatar.

- Support farm owners to intensify their agriculture so as to help them improving the efficiency of both the unit volume of water and the unit area of land.
- Securing agricultural loans to help farm owners in carrying out farm modernizations.
- Study the proposal of applying nominal fees on the consumed water by each farm in order to use it as a rationalization mechanism for water conservation.
- Strengthen the public awareness as the public can play a very important role to achieve the proposed groundwater strategy. To attain superior assistance and participation, the public needs to realize the water conditions and has to be conscious of its task. This should be done by using effective formal and informal methods of communication (United Nations 1993).
- Activate the Permanent Water Resources Committee in order to perform its designated role.

3.6.2 *The long-term strategy*

The long-term strategy is meant to be those measures and solutions that need a significantly lengthy time to be implemented and they are normally being performed in the course of national projects and/or long-term governmental course of actions.

These measures and solutions concentrate on two pivots. The first one is the readily available groundwater resources augmentation to decrease the gap between the available groundwater resources and the agricultural demand. The second is new non-conventional water resources exploration to be used in agriculture in order to diminish the stress on the groundwater resources. Under this framework, the following measures could be proposed:

3.6.2.1 *Re-use of Treated Sewage Effluent (TSE)*

3.6.2.1.1 Re-use of the municipal TSE

Re-use of the municipal TSE is a chief water resource. It has a double advantage as it protects the environment from severe problems in case of direct dumping and is a new, permanent water resource; it should not be discarded. This water resource is distinguished by endless growth with time. Furthermore, it is the only source of water that increases with population growth and development, contrary to the other water resources. Therefore, the country has to optimize the use of this valuable source of water to improve and protect the already deteriorated groundwater aquifer whenever possible.

It is estimated that by 2004, the annual volume of municipal TSE in Qatar attained more than 50 MCM and the used amount is constrained to the irrigation of fodder crops and landscape ornamentals. According to the Department of Drainage Affairs, it is expected that municipal TSE will reach 133.40 MCM by the year 2020 and 161.15 MCM by the year 2030. Although tertiary treated sewage water is being used now in landscaping in Doha, its use in irrigating different food crops or recharging the groundwater aquifer is neither socially nor technically acceptable. Consequently, this water resource can be included in any future national water resources plan. Yet, the key barriers that stands in front of the re-use of municipal TSE in Qatar are the long distance between treatment plants and areas with potential use in agriculture and the absence of an integrated water resources management approach that is policy driven. Over and above, there is no joint strategy or action plan that harmonizes different tasks of the involved institutions.

3.6.2.1.2 Re-use of the gas to liquid industrial TSE

Gas to liquid is a refinery process that converts natural gas or other gaseous hydrocarbons to longer-chain hydrocarbons. Water is also produced as a by-product during this process. The amount of the produced industrial TSE can attain 1 barrel of water from each barrel of liquefied gas. It is expected that the annual volume of this water will reach about 22 MCM by the year 2010 and its ceiling which is about 40 MCM will be reached in few years time according to the initial official estimates (Amer & Al-Mahmoud 2003).

Nevertheless, this type of water has good salinity level as its Total Dissolved Solids (TDS) does not exceed 200 part per million but its source as an industrial TSE plays an important role in contaminating it with some other serious pollutants. These pollutants compose an obstacle in making the best use of it. As a result, detailed technical and institutional discussions must start to manage and put an in-depth plan for the use of this water taking its quantity, its quality, the needed level of treatment, the cost of needed treatments, the responsible organization for treatments, the payer of treatment cost, the field of use, the place to use and the conveyance cost into consideration. This source of water is very promising and it is expected to cover a considerable part of water deficit in Qatar.

3.6.2.1.3 Re-use of the water collected by the drainage network under Doha city

Before the fifties of the last century, pre-oil era conditions, shallow groundwater under Doha city was several meters below ground level. The rapid development of Doha was coupled with a significant rise in groundwater table. ASCO (1983) proved that 40.6% of the source of this water is return from irrigation, 27.3% is leakage from potable water distribution network, 19% is leakage from sewerage system and septic tanks, 12.4% is from rainfall and 0.7% is from leakage from tank distribution system. In 1983, the total annual amount of water produced for municipal and industrial use was about 60 MCM (KM, 2003) while in 1999 it was about 133.8 MCM and the losses is about 54.2 MCM i.e. about 40.5% of the total amount produced (ASA, 2001). Accordingly, it is anticipated that the current greater part of the water collected by the drainage network under Doha city is initiated from the leakage of potable water distribution network and this is not a sustainable source of water.

The amount of water collected by this network reaches 22.63 MCM/year for the year 2003 with salinity ranging from 6000 to 14000 part per million. It is being sent to the Gulf as the most inexpensive accessible scenario up till now. Yet, the Department of Drainage Affairs is performing thorough investigations so as to accurately identify the quantity, the quality and the source of this water to be able to make use of it in the water resources management and development plan. In the time being, many institutions and individuals have shown interest to intermediately re-use this water in landscaping after treating it.

3.6.2.2 *Increasing recharge to the groundwater aquifer with rainfall using drilled wells*

Formerly, some wells were drilled to a depth reaching the water bearing formation in few suggested major watershed depressions with the aim of; mainly mounting the recharge to the groundwater aquifer with the standing water at the bottom of depressions (vertical drainage) and, secondary acting as flood safeguard structures for the nearby active farms. It was found that recharge increased with a very considerable amount. For that reason, a decision was made to drill 341 recharge wells in some of the recommended depressions of the country. It is supposed that, with employing some complete evaluations, proper routine and emergency maintenance plan and suitable rehabilitation strategy to the existing 341 recharge well and subsequently making the best use of the revealed results and recommendations to put a new design and choose new locations to cover the country, this will improve the annual recharge with about 50% of its original value (Amer & Al-Mahmoud 2003) to reach about 90 MCM. Nevertheless, the original value of groundwater recharge from rainfall is about 10% of the total amount of rainfall coming to the country. Yet, there is a wide range of enhancement to the groundwater recharge especially from rainfall. The time limit is suggested to be about 14 years from the beginning of the studies.

One of the potential methods to use in order to augment rainfall is cloud seeding. In the cloud seeding process, a chemical driving force is presented into the cloud system. This adjusts the microphysics and/or dynamics of the clouds, with the aim of endorsing precipitation. Qatar has particular uniqueness that should be examined first in order to plan for such a study. Some of these characteristics are; the prevailing wind direction, the dominant cloud types in the region, the accuracy of the process regarding the dimensions of the peninsula, the proportional cost with the other potential sources such as seawater desalination. Essentially, since the seventies, intensive experimentations in the use of the technology of cloud seeding were made in different locations in the GCC, Middle East and other countries. The estimated tentative cost of the increased run-off by one cubic meter for the Utah Cloud Seeding Project, Utah, USA, is US$ 0.00083 (Stauffer & Williams 2000).

In order to maximize recharge by rainfall to the groundwater aquifer, recharge wells may be coupled with cloud seeding process. The cloud seeding process will be performed where and when the conditions are favorable regarding recently maintained wells, type of formation, water quality in the bearing formation etc. Obviously, this will be applied only in case that the feasibility of the cloud seeding process being proved.

3.6.2.3 *Seawater desalination*

Qatar depends mainly on seawater desalination to meet 99% of its domestic and industrial water demand. The annual production capacity of that source of water has amplified from 0.25 MCM in 1953 to about 161.8 MCM in 2003 (ASA 2004) and expected to reach about 258 MCM by the year 2025 (KM 2003).

Formerly, it was recognized that investing in agriculture, whilst having desalinated seawater as a source of water, is not feasible activity. In the time being, because of the remarkable scientific progress done in the field of desalination during the last few decades, the cost of desalinating 1 cubic meter decreases from about 9 $\$/m^3$ in 1960 to about 0.9 $\$/m^3$ in 2000. This desalination cost is likely to reach about US$ 0.3 by the year 2025 (Zhou & Tol 2004). Thus, a detailed feasibility study has to be commenced to examine the opportunity of supplying one third to half of the registered active farms and block their wells. The produced desalinated water can be used to artificially recharge the groundwater aquifer and that will improve groundwater quality and ultimately supply active farms.

3.6.2.4 *Freshwater importation from outside the country*

Water importation is defined as the action or procedure whereby water is brought into the country from different one rather than building new water projects within the area. Typically, it refers to the artificial transport of water from one water basin to another.

Importation of water presents a potential choice for future water supply in Qatar in order to bridge the gap between supply and demand or to be recharged to the groundwater aquifer to enhance groundwater quantity and quality. In the past, the idea was investigated through a comprehensive study to transfer water from Iran by a pipeline but no decision was taken until now.

On the other hand, many means are available for importing water from anywhere, i.e. by pipelines, giant water bags, old and new oil tankers and transporting ice bergs from the north pole. An extensive significance can be given to such an option after performing a comparative feasibility study between the different means of water importation and between water importation and the other available water resources.

3.7 CONCLUSIONS

Assuming the application of the entire suggested short-term strategy procedures as preliminary and vital action, then, the application of the main group of long-term strategy measures that include: re-use of the municipal advanced treated sewage effluent, re-use of gas to liquid industrial treated sewage effluent after solving the problems that inhibit its usage, and increasing recharge to the groundwater aquifer with rainfall using drilled wells after achieving the necessary studies and executing the subsequent phases of drilling the maximum number of recharge wells all over Qatar, it is anticipated that Qatar will be able to bridge the gap in groundwater resources within about 8 years as exposed in Table 3.3.

Nevertheless, as a permanent plan, so as to reach groundwater sustainability in Qatar, it is advised that the short-term strategy and the main group of long-term measures be employed immediately. However, careful assessment needs to be done prior to the adoption seawater desalinisation and/or import of freshwater before endorsing any of them as an alternative of fresh water supply.

Table 3.3. Hypothetical groundwater balance from 2003/2004 to 2015/2016 MCM based on the short-term measures and (6.2.A.1 + 6.2. A.2 + 6.2.B) of the long term measures.

Year	Groundwater recharge from rain	Gross consumption	Annual balance	Accumulated balance
2003/2004	30.531	220.829	−153.699	−2181.705
2004/2005	56.807	220.829	−120.871	−2302.576
2005/2006	56.807	220.829	−109.319	−2411.895
2006/2007	56.807	220.829	−92.293	−2504.188
2007/2008	56.807	220.829	−75.266	−2579.454
2008/2009	56.807	220.829	−58.239	−2637.693
2009/2010	56.807	220.829	−47.764	−2685.456
2010/2011	56.807	220.829	−32.764	−2718.220
2011/2012	56.807	220.829	−25.764	−2743.984
2012/2013	56.807	220.829	−18.764	−2762.748
2013/2014	56.807	220.829	8.711	−2754.037
2014/2015	56.807	220.829	22.336	−2731.701
2015/2016	56.807	220.829	36.336	−2695.364

ACKNOWLEDGEMENTS

The author thanks the organizers of the 2nd International Groundwater Conference (2nd IGC-2006) for giving this opportunity to present this work. The author also expresses his deep sense of gratitude to Professor D. Chandrashekharam, Dr. Jochen Bundschuh and Dr. Prosun Bhattacharya for their critical comments on an earlier draft of the manuscript.

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CHAPTER 4

Application of resistivity imaging for delineation of aquifer configuration

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ABSTRACT: Electrical resistivity imaging survey has been conducted to define the geometry of aquifer configuration in Pathri-Rao watershed situated in the Piedmont zone of Himalayan foot hill region, Uttaranchal (presently Uttarakhand), India. Resistivity image profiling data at 9 sites were recorded using IRIS imaging system with 72 electrodes deployed at 10 meter spacing in each site. The profile length at each site was 710 m, oriented in different directions in field as per the survey design and accessibility of space. 2D inversion of each profile data was carried out with topography data using RES2DINV code. Interpreted 2D resistivity image were analyzed in terms of geological formation and geometry of aquifer system in the area. Interpreted resistivity data reveals the unsaturated surface layer resistivity range 100–800 Ω-m. Clay formation is characterized by low resistivity (10–25 Ω-m). Resistivity of aquifer zone in the study area varies from 40–150 Ω-m. Depth, thickness, surface elevation, possible groundwater flow direction and the interaction of shallow and deeper aquifers were also inferred from resistivity image. The results are in general agreement with the results obtained from isotope technique and other hydro-geological data obtained from the area.

4.1 INTRODUCTION

Acquisition of resistivity data, their presentation and interpretation are much simpler relative to other, geophysical methods. Collecting resistivity data requires putting an electrical current into the ground and measuring the potential difference, normally using four electrodes system. The electrodes system moved for each measurement. A set of measurements are possible, either by maintaining the constant spacing between electrodes and the locations are moved across the ground surface (profile) or by varying the spacing between electrodes around a central location (sounding). By spreading the electrodes further apart for each measurement, the resistivity method measures deeper into the subsurface. The measured resistivity data can be related to geology in the subsurface, by modeling various geological features.

The introduction of multi-electrode cable completely revolutionizes the resistivity measurement, analysis and interpretation methodology. The programmable microprocessor controlled switching system with data collection memory built into a resistivity meter has strengthened the use of multi-electrode system. The equipment accelerated data collection from about five minutes per measurement to approximately four measurements per minute. Loke & Barker (1996) has developed a rapid least squares data inversion method for the data collected by multi-electrode system. Thus the advances in field equipment design capability, and the development of computer assisted algorithms, the electrical imaging has now become cost effective and more commonly used technique for solving the problems related to delineation of aquifer system and efficient development of groundwater resources. Various workers have used electrical resistivity commonly used technique for solving the problems related to delineation of aquifer system and efficient development of

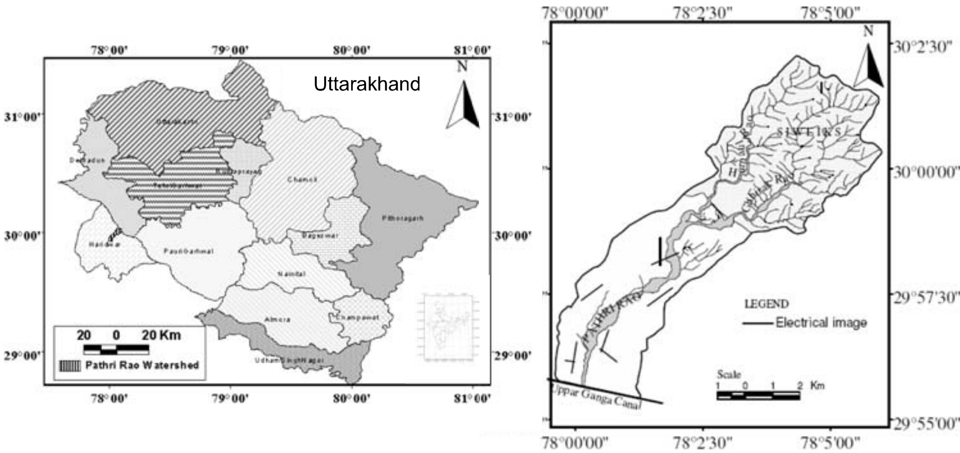

Figure 4.1. Map of the Pathri-Rao watershed showing locations of resistivity image profiles.

groundwater resources. Various workers have used electrical resistivity methods for solving hydrogeological problems. Venkateswara Rao & Briz-Kishore (1991) have used it for the estimation of groundwater potential index at various survey locations. Shahid & Nath (1999) have also used integration of remote sensing, and electrical sounding data for spatial hydrogeological modeling of a soft rock terrain.

Electrical resistivity imaging is a powerful tool and plays a vital role in ground water exploration, and delineation of aquifer configuration. It maps aquifer systems, clay layers, lateral and vertical extent of aquifer, qualitative estimation of ground water flow direction. The objective of present study is to use electrical resistivity imaging for the delineation of aquifer system and its geometrical configuration in the Pathri-Rao watershed of Himalayan foothill region in the state of Uttaranchal (presently Uttarakhand), India. The study area is shown in Figure 4.1, located between latitudes 29° 55′–30° 03′ N and longitudes 77° 59′–78° 06′ E covering an area of 52 km^2. The geological and geohydrological setting of the area have been discussed in the paper by Kachhwal et al. (in press), therefore to avoid duplication the same has not been discussed here.

4.2 METHODOLOGY

Resistivity image profiling at 9 selected locations in accessible area were recorded using fully automatic IRIS multi-electrode resistivity meter. Locations of these profiles are shown in Figure 4.1. The direction of each profile line is governed by the availability of site and generally ori-ented along and perpendicular to the Pathri-Rao river. The image profiling data at each site were recorded with 72 electrodes placed at 10 m spacing. The total of 895 data recorded at each site with Schlumberger-Wenner configuration in 710 m long profile. A typical data acquisition sequence is shown in Figure 4.2.

4.3 RESULTS AND INTERPRETATIONS

Final inverted 2 D model at each electrical image profile (EIP) indicates the resistivity variation along the profile line up to about 100 m depth. In order to discuss the geohydrological feature systematically, these profiles are numbered from north to south direction in increasing order (Fig. 4.2). EIP-1 is located in northern part of study area; in the restricted area where the Pathri Rao is bifurcated into two smaller channels (Harnaul Rao and Chirak Rao, Fig. 4.1). The profile line passes through

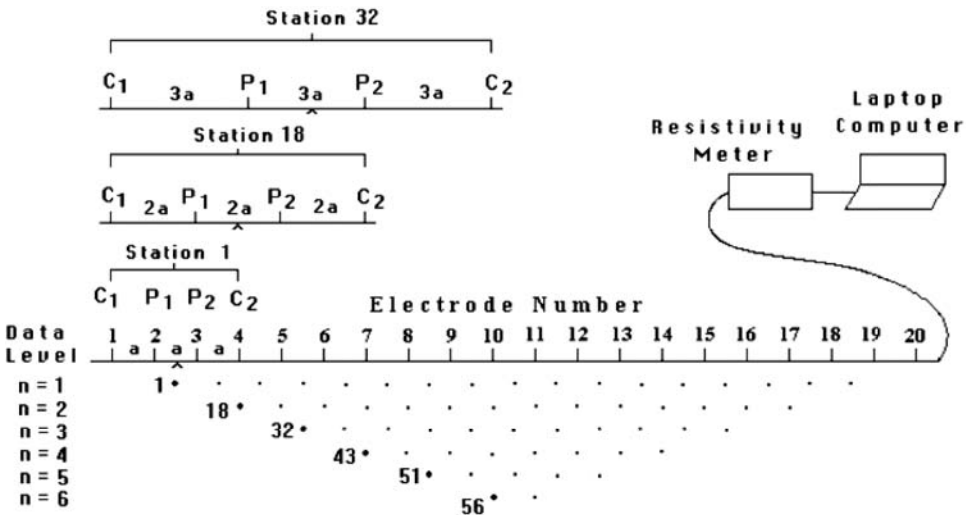

Figure 4.2. A typical acquisition sequence of measurements to build up a pseudosection.

Table 4.1. Resistivity values of unconsolidated sediments in Bhabhar formation.

Lithology	Resistivity value (Ω-m)
Top soil	100–900
Saturated clay	10–25
Saturated sand with pebbles	30–100

one channel (Harnaul Rao). Electrically the area is represented by a very high resistivity of near surface material (900 Ω-m) representing the unsaturated coarse material and boulder transported from Siwalik hill. Resistivity decreases to about 100 Ω-m below the depth of 20 m indicating the finer material at depth. Further decrease of resistivity to a value of about 50 Ω-m at a depth of about 30 m indicate aquifer zone. Due to the presence of small channel (Harnaula Rao) a lateral variation in the resistivity values is also observed. Higher resistivity values (80 Ω-m) observed at Harnaul Rao, indicate the presence of coarser material in comparison to the adjacent zones. EIP-2 is located southward to EIP-1 and passes through the Pathri Rao river bed along NE-SW direction.

The top layer near surface high resistivity (600 Ω-m) formation is continued. Underneath the top highly resistive layer a consistent aquifer zone dipping towards southwest is visible. The effect of Pathri Rao is seen in slightly uplifting the aquifer zone underneath the bed. Another profile (EIP-3) is located in further southward direction to the previous profiles the accumulation of the finer material in near surface zone resulting the formation of shallow saturated zone in perched conditions more consistent and decreased sloping deeper aquifer zone. In further southward direction , EIP-4, passes along N 100 E western bank of Pathri-Rao river indicates the formation of two aquifers zones, at shallow depth (10 m) and a deeper depth (60 m) separated by a low resistive zone.

The low resistive zone here represents less permeable clayey formation. Thus middle part of the study area is represented by relatively finer material and existence of two aquifer zones at shallow and deeper level respectively. This is a typical example of the two aquifers (shallow and deeper) separated by a thick impermeable layer (clayey formation). In southern part of the study area at EIP-5, the two aquifer system is continued however, the low resistivity zone is replaced by slightly higher resistivity (50 Ω-m) materials in almost middle of the profile indicating the two aquifers zones are connected by more permeable formation. Similarly other profiles oriented in different

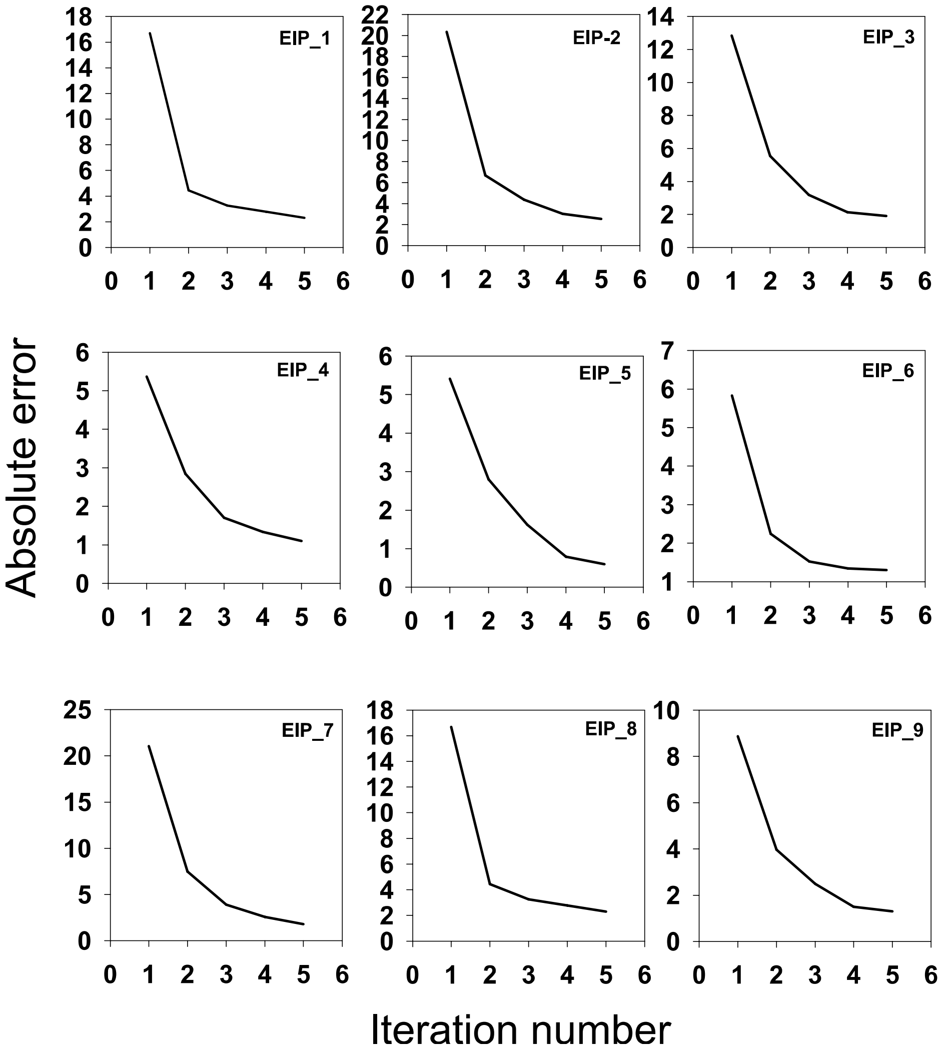

Figure 4.3. Convergence of inverted solution as a function iteration number for all 9 profile data.

direction show the configuration of aquifer zone and its surface elevation. Aquifer depth and its geometrical configuration are also revealed in eastern bank of Pathri Rao and in perpendicular direction. The depth of shallow aquifer has also been verified by water level monitoring data from the area. Table 4.1 shows the resistivity ranges for various lithological units in the area obtained by comparing the resistivity variation with the available borehole data.

4.4 CONCLUSIONS

The application of resistivity imaging in determining the aquifer configuration in the Pathri-Rao watershed of Himalayan foothills region in Uttarakhand, India has been demonstrated. The resistivity is found to be highly variable in the area. Northern part unsaturated formation is highly resistive where as the resistivity decreases as we move from northern zone to southern zone in the study area. The change is resistivity is correlated with the coarser and finer material in various saturation conditions. The identified electric boundaries separating the zone of different resistivity may or may not coincide with boundaries separating layers of different lithological composition. This may result if the electro-stratigraphy varying from the gross lithostratigraphy. This limitation is taken into account by calibrating the resistivity values with representative borehole data in the study area which will help defining resistivity values for each litho-unit which is subsequently used in direct determination of aquifer system in the area. Present study indicates the presence of an aquifer in deeper part in northern zone of study area two aquifers (shallow and deeper) were delineated in middle and southern zone of the area. The results broadly agree with the available groundwater monitored and hydrogeological data.

ACKNOWLEDGEMENTS

The authors deeply appreciate the efforts of the reviewers for their comments. We are thankful to the organizers of the 2nd International Groundwater Conference and the editors of the volume for their patient assistance.

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CHAPTER 5

Hydrogeologic analysis of Kathajodi river basin, Orissa, India

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ABSTRACT: The Bayalish Mouza located in the Kathajodi River basin of Orissa is a typical river island surrounded by the Kathajodi River and its branch Surua. Agriculture is the main occupation of the inhabitants and groundwater is a major source of irrigation. Ample water is available in the monsoon season and even some area remains waterlogged during this time. But farm ponds dry up towards the end of February and groundwater is not enough to meet the agricultural water requirements during the post-monsoon and summer season. The present study focuses on the hydrologic and hydrogeologic analyses of this basin to explore the possibility of enhanced and sustainable groundwater supply. The streamflow analysis showed that the river flow is highly seasonal and it reduces very much during summer season. The hydrogeologic analysis indicated that a confined aquifer exists comprising medium to coarse sand. The aquifer depth ranges from 20 to 40 m below ground level with its thickness varying from 12 to 56 m. Overall groundwater flow is from north-west to south-east direction. The groundwater level drops by 3 to 6 m during dry periods, with April-May being the most critical months. The weekly groundwater-level data show a good correlation with the rainfall, suggesting significant groundwater recharge from rainfall. Apart from rainfall, groundwater aquifer has good interaction with the Kathajodi River also. Construction of a series of check dams with spillways along the main natural drain, construction of a few more water harvesting structures in the upper portion of the basin and numerical analysis is recommended for exploitation and sustainable use of water resources in the basin.

5.1 INTRODUCTION

Groundwater is a very important and invaluable natural resource. It is renewable but finite resource, which is generally characterized by stable temperature and chemical composition. It's unique qualities that it is generally free from pathogens, easily accessible and free from suspended particles has made it the most important and preferred source of water for agricultural and domestic use. However, overexploitation of groundwater can result in adverse effects on the local and regional ecosystems. A growing number of regions are facing increasing water stresses owing to burgeoning water demands, profligate use, and escalating pollution worldwide (Rodda 1992, Biswas 1993, Falkenmark & Lundqvist 1997). The comprehensive assessment of freshwater resources of the world (Shiklomanov 1997) illustrates the magnitude of the global water problem and estimates that about one-third of the world's population is currently afflicted with moderate to severe water stress. By 2025, approximately two-third of the world population will be at risk of facing water stress, if present trends continue (Kuylenstierna et al. 1998). The experiences in the field of water management in India have shown that unbalanced use of water resources have either lowered groundwater level or caused waterlogging and salinity in different parts of the country (Jha et al. 2001). Particularly, in the canal-dominated regions of North India, there has been increase in groundwater levels due to seepage from the canals. Excessive pumping on the other hand has led to alarming decrease in groundwater levels in several parts of the country. This in turn has increased the cost of pumping, caused seawater intrusion in the coastal areas and has raised questions about the future availability of groundwater.

In the state of Orissa, total annual replenishable groundwater resource is estimated to be 21 km^3. Exploitation of groundwater resources is not appreciable in Orissa. Groundwater development varies from 5% in Malkanagiri district to about 42% in Balasore district (Pati 2004). Due to the confinement of rainfall to 4 to 5 months, there is excess water during rainy season and water scarcity during non-rainy season. Hence there is need for integrated management of surface water and groundwater in which the excess surface water in the rainy season can be stored either on the surface or by groundwater recharge and use them in non-rainy season. The inherently random nature of surface water supplies and the natural recharge to an aquifer give groundwater stocks an important role as a contingent supply for times when the flow components of supply are below average. Optimal inter-temporal allocation of groundwater used conjunctively with surface water will give a higher value to the surface water than it would have on an unmanaged basin. In a well managed basin, an uncertain surface water supply can be almost as valuable as a certain supply equal to the mean of the uncertain supply, the only difference being the extra pumping costs (Burt 1976).

A study area has been selected at Bayalish Mouza in Kathajodi River basin of Orissa. The purpose of the study is to do hydrologic and hydrogeologic investigations to explore the possibility of enhanced and sustainable groundwater supply. The idea about hydrology and hydrogeology will help in integrated water management planning of the study area. The paper presents the results of the field investigations and the pertinent data analysis.

5.2 STUDY AREA

Bayalish Mouza is located in Kathajodi River basin of Orissa (Fig. 5.1). It is a typical river island surrounded on both sides by the River Kathajodi and its branch Surua. The site is located between 85°54′21″ to 86°00′41″E longitude and 20°21′48″ to 20°26′00″N latitude. The total area of the river island is 35 km^2. The area experiences a tropical humid climate with an average annual rainfall of 1535 mm of which 80% occurs during the months of June to October. The normal mean monthly maximum and minimum temperatures of the region are 38.8°C and 15.5°C in the month of May and December respectively. Similarly, the normal mean monthly maximum and minimum evaporation of the region are 202.9 mm and 80.7 mm for the month of May and December respectively. The average annual normal evaporation is 1502 mm.

Agriculture is the major occupation of the inhabitants. Total cultivated area in the region is 2445 ha. Out of that 1365 ha is irrigated land. Total low lands in the region are 408 ha, medium lands are 1081 ha and high lands are 956 ha. All the low lands, medium lands and 618 ha of total high lands are used for paddy cultivation. Rest of the area is used for vegetable cultivation. All the irrigated lands are irrigated by lift irrigation points drawing groundwater from the underground aquifers. Groundwater resources development in the area is good. There are about 100 government tubewells in the area, which are the major source of groundwater withdrawal. These are constructed and managed by Orissa Lift Irrigation Corporation. Now, slowly they are being handed over to the local level water users associations. Out of the hundred tubewells, at present 65 tubewells are operating.

Even though Kathajodi River flows on both sides of the study area, there is water shortage during the dry periods. The flow of water in Kathajodi in the post-monsoon season is less. The farm ponds also dry up during this period. The groundwater is not sufficient to meet the water requirement of the command area. During the monsoon season, there is no water shortage. But a different problem in the form of waterlogging is encountered in the monsoon season. Embankments have been provided on the banks of the rivers to prevent the entry of river water into the inhabited area during flood events. So, the entire rainwater of the region is drained through the Malia drain and discharged at a single outlet into the river. But during the flood event, the river water enters into

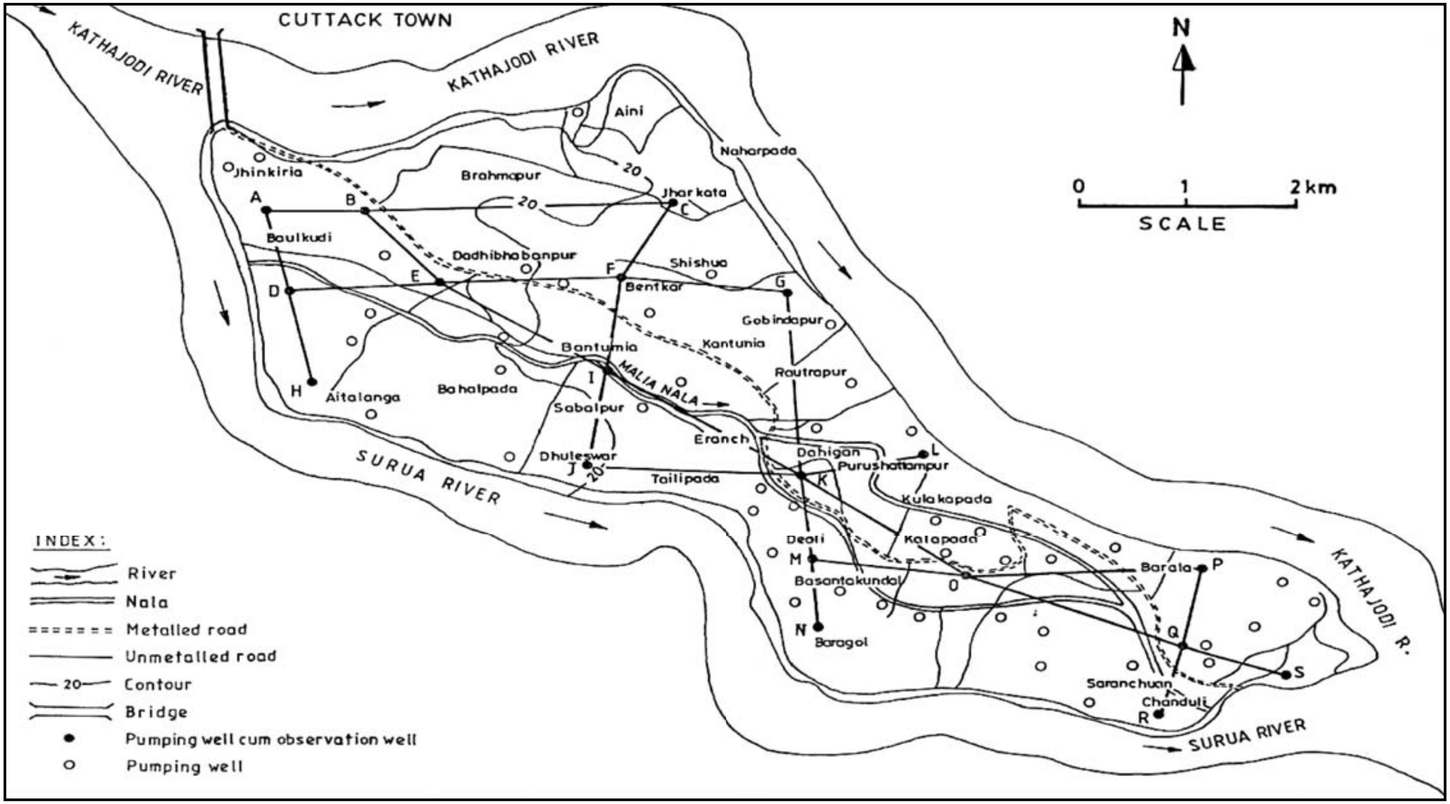

Figure 5.1. Location of pumping wells at Bayalish Mouza in Kathajodi River basin of Orissa.

the inhabited area through the outlet. So, a sluice gate is provided at the outlet so that it can be closed during the flood events. During this period the rainwater is unable to drain and it creates a waterlogging condition in the southern side of the study area. This damages the crops and creates innumerable problem to the farmers and residents.

5.3 MATERIALS AND METHODS

5.3.1 *Groundwater level monitoring*

For monitoring of the groundwater levels, nineteen tube wells were selected in the study area. The groundwater level was monitored on a weekly basis from February 2004 onwards. The location of different tube wells in the study area is shown in Figure 5.1. The tube wells were selected in such a way that they can represent approximately four north-south and four east-west cross-sections.

5.3.2 *Data collection*

The daily streamflow data of Kathajodi River at the highway bridge gauge station was collected for 4 years (2001–04). Using the daily streamflow data, annual maximum discharge, minimum discharge, 95-day discharge, ordinary discharge, low discharge, droughty discharge and mean discharge were calculated for the four-year period (2001–04). Here it should be noted that, after arranging the daily streamflow data of one year in a descending order, the 95/96th day (96th day for a leap year), the 185/186th day, the 275/276th day, and the 355/356th day discharges are, respectively, known as 95-day, ordinary, low, and droughty discharges.

The lithologic data at 50 sites of the study area was collected from Orissa Lift Irrigation Corporation office. The lithologic data offers unique opportunities to gather information about the geology and groundwater conditions of the site. The above information is necessary for developing plans for optimal utilization of groundwater resources. Using the lithologic data, detailed geologic cross-sections along the sections as shown in Figure 5.1 were drawn for further analysis. Availability of aquifers at different depths was plotted and aquifer thicknesses in the respective depth were found out.

5.3.3 *Data monitoring and analysis*

Daily rainfall data of the study area was monitored by installing a raingauge station at the site. Weekly groundwater level fluctuation of different tubewells were plotted and superimposed with the weekly rainfall data. The groundwater level data were correlated with the rainfall data. Samples were collected from the tubewells and water quality analysis was done. The samples were analyzed for parameters like pH, EC, Ca^{2+}, Mg^{2+}, Na^{+}, Cl^{-} and HCO_3^{-}.

5.3.4 *Stream-aquifer interaction*

Stream-aquifer interaction studies are useful for studying the conductivity between the aquifer and stream. Two flood events were encountered in the Kathajodi River during monsoon season of 2005 one during first week of August and the other during third week of September. During both the flood events, simultaneous monitoring of water level in the river and a well near the bank were monitored for 3 and 4 days respectively at 3 sites. The river level data and well level data were compared to study the conductivity between the river and the aquifer.

5.4 RESULTS AND DISCUSSION

5.4.1 *Streamflow analysis*

Figure 5.2 illustrates the daily variation of streamflow in the Kathajodi River for a period of four years (2001–2004) at highway bridge gauging station. It is observed that the steamflow reaches the peak during the period July to September (Fig. 5.2). It starts decreasing from the month of October and becomes very negligible from December onwards when rainfall events are very less. The streamflow varies appreciably over the four year period with least streamflow in year 2002. The streamflow values were appreciably more in the years 2001 and 2003.

Table 5.1 summarizes some important flow characteristics for the 2001–04 period. The maximum streamflow in the years 2002 and 2004 is significantly less than the four year mean. Although years 2001 and 2003 experienced relatively high maximum flows, minimum flows were lower. The 95-day, ordinary and low flows vary greatly over the years, irrespective of annual peak flow. The droughty flow and minimum flow were non-zero only in the year 2003. Quite low flows in the non-monsoon months show that surface water resource is significantly reduced in the dry periods.

5.4.2 *Basin geology*

The lithologic data analysis along the marked cross-sections (as shown in Fig. 5.1) showed that a confined aquifer exists in the groundwater basin. Figure 5.3 shows the geologic cross-section across

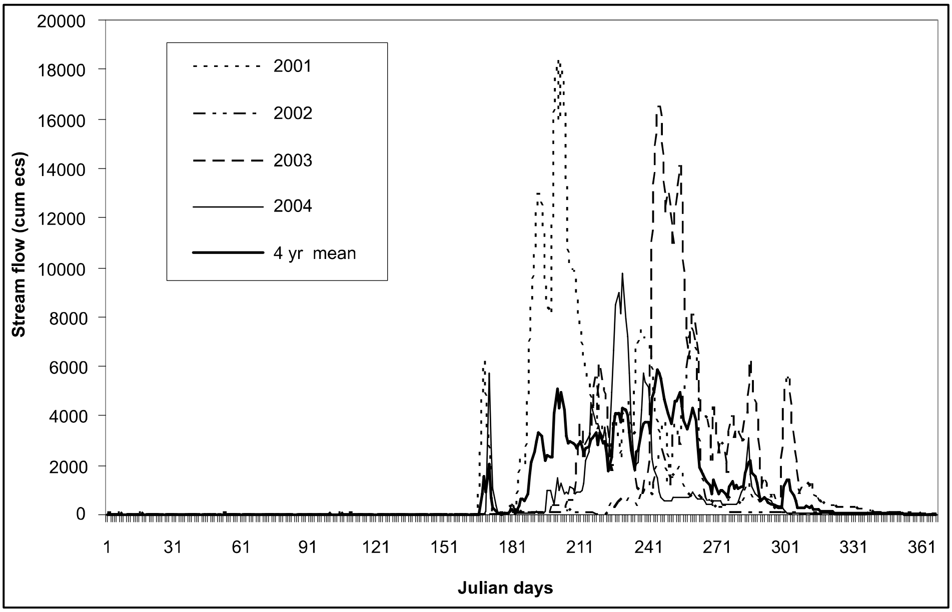

Figure 5.2. Annual discharge hydrograph of Kathajodi River at highway bridge gauging station.

Table 5.1. Streamflow characteristics of the Kathajodi River for 2001–04 period.

	Streamflow (m^3/sec)				
Type of flow	2001	2002	2003	2004	4-yr mean
Maximum flow	18380.0	7557.0	16530.0	9727.0	13048.5
95-day flow	643.7	59.0	959.1	431.9	523.4
Ordinary flow	33.3	20.7	24.9	20.9	24.9
Low flow	17.1	13.2	17.5	3.4	12.8
Droughty flow	0.0	0.0	17.0	0.0	4.2
Minimum flow	0.0	0.0	12.1	0.0	3.0
Mean flow	1345.2	283.0	1338.1	587.9	888.6

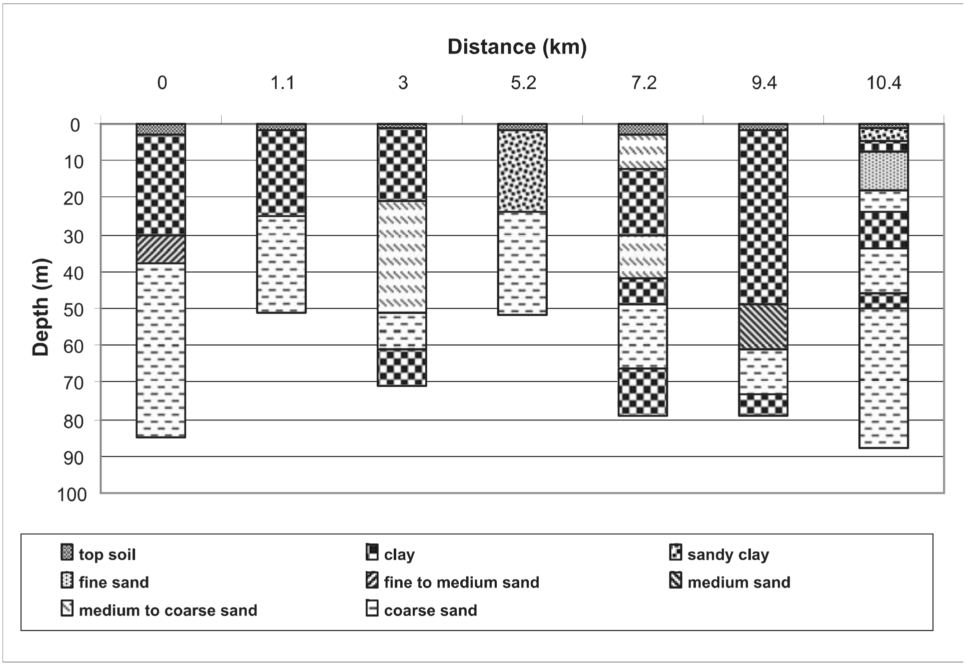

Figure 5.3. Variation of lithology along central section.

Table 5.2. Variation of aquifer thickness at various depths.

Section	Depths 40 m	50 m	60 m	70 m
North	22–56 m	24–56 m	24–56 m	35–49 m
West	21–56 m	21–56 m	19–56 m	19–56 m
South	12–51 m	14–51 m	14–51 m	24–51 m

the central section. The top layers are mostly composed of clay or sandy clay. The thickness of the top aquitard varied from 16 to 49 m. The exceptions being the site of W-c, where water bearing formation is available at a shallower depth and the site of W-h, where the clay layer is extended upto 66 m. Apart from clay and sandy clay, the aquitard consisted of patches of fine to medium sand. The water bearing formations consisted of mostly coarse sand, medium to coarse sand and pebbles with the coarse sand being the dominant formation. The depth of the impermeable layer below the aquifer varied between 47 m to 88 m. The availability of aquifers at different locations was plotted for different depths of 40 m, 50 m, 60 m and 70 m. The thickness varied in the range of 12–56 m. The variation of aquifer thickness at different depths at different sections of the basin is shown in Table 5.2.

5.4.3 *Groundwater level fluctuation*

Monthly variation of groundwater levels at 19 sites over the Bayalish Mouza basin (Fig. 5.1) for the 2004 water year (1 April 2004 to 31 March 2005) are shown in Figures 5.4a & b respectively. Generally the water level rises in the month of June with the onset of monsoon and reaches the peak in the months of July-August. In the month of September also, high water level is observed

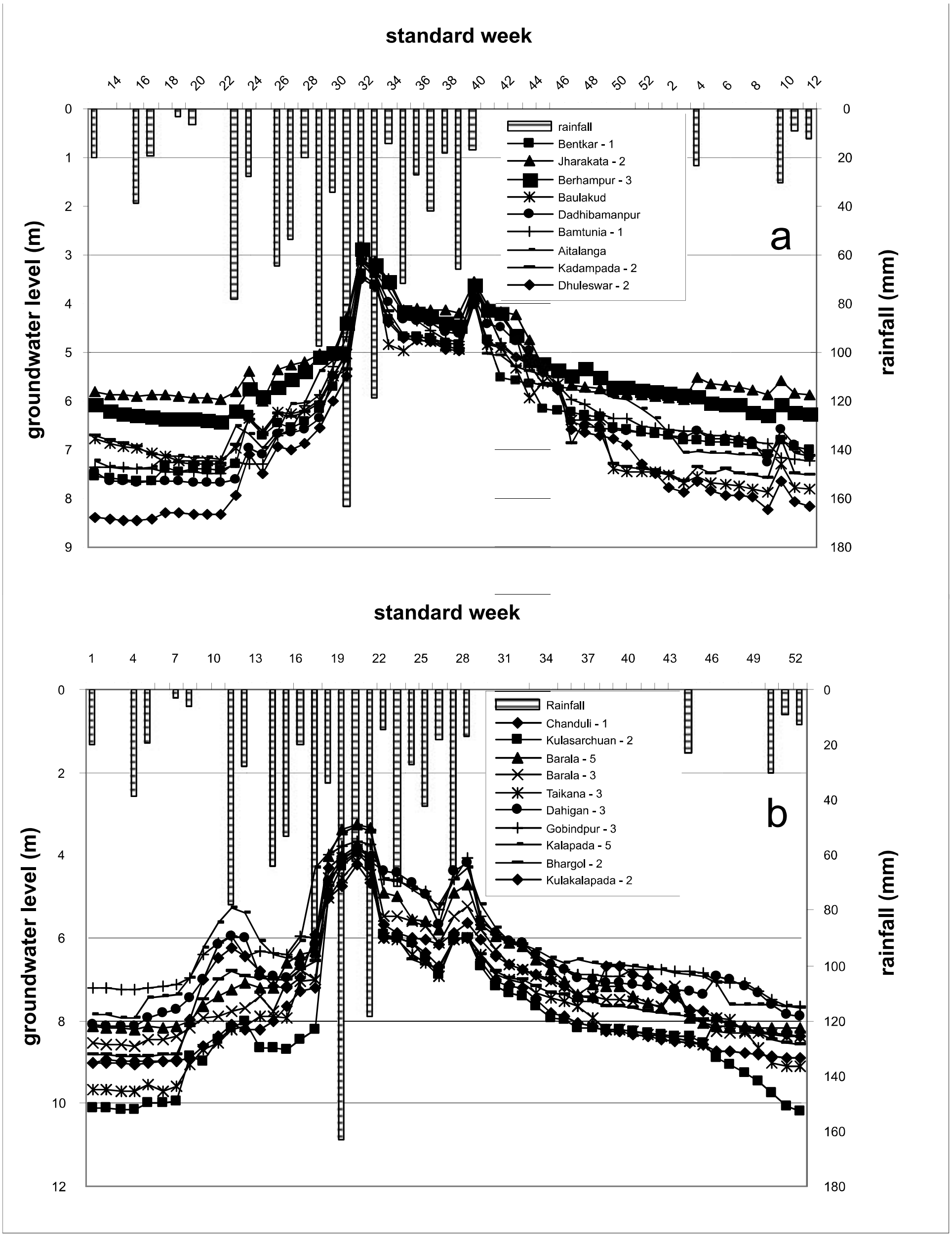

Figure 5.4. (a & b) Monthly groundwater level fluctuation at the 19 sites together with bar graph of rainfall distribution.

and from the month October onwards it starts declining to reach the minimum level in the months of April and May. There is slight increase in groundwater level in the month of March due to some significant rainfall. Overall groundwater flow is from north-west to south-east direction. The groundwater level drops by 3 to 6 m during dry periods. The weekly groundwater-level data show a good correlation with the rainfall, suggesting significant groundwater recharge from rainfall.

5.4.4 *Stream-aquifer interaction*

As the underlying aquifer in the Bayalish Mouza basin is alluvial, it is likely that there will be hydraulic connection between the Kathajodi River and the aquifer. However at present there is not adequate knowledge of stream-aquifer interaction which is important for efficient management of water resources. In fact, the interaction between groundwater and surface water bodies is extremely complex and dependant on a number of factors such as topography, subsurface hydraulic properties, groundwater flow patterns, temporal variations in rainfall, streambed geometry and stream stage (Hewlett & Hibbert 1963).

To study the interaction between the stream and aquifer, simultaneous monitoring of the river stage and a nearby well stage was monitored at three locations during two flood events. The data during the first flood event showed that there is decrease in well stage with decrease in river stage. During the second flood event, there was heavy rainfall in the basin which resulted in surface water storage near the wells. Hence proper trend of variation of well stage in relation to river stage was not observed. But the first flood event data show that there is likely to be some stream-aquifer interaction.

5.4.5 *Groundwater quality*

Groundwater quality was monitored at 8 pumping wells in the pre-monsoon season. Different parameters like pH, electrical conductivity, calcium, magnesium, sodium, chloride and bicarbonate were measured. The pH values varied from 6.8 to 7.3, whereas the electrical conductivity varied from 0.16 to 0.26 dS/m. The calcium concentration varied form 0.8 to 2.0 meq/L and the magnesium concentration varied from 0.2 to 0.9 meq/L. The sodium concentration varied from 0.1 to 0.3 meq/L whereas the chloride concentration varied from 0.7 to 1.8 meq/L. The bicarbonate concentration varied from 0.2 to 0.6 meq/L. The analysis shows that all the values are within the tolerance limit and the water is very good for irrigation purpose.

5.4.6 *Suggested action*

As there is waterlogging in the downstream side of the area in the monsoon season, construction of a series of check dams along the main drain (Malia drain) can reduce the waterlogging in the downstream side to some extent. It can create intermediate surface water reservoirs from which farmers can pump water for irrigation purpose. The intermediate surface water storage can cause more groundwater recharge in the upstream side of the area. For further reduction in waterlogging during flood events, water can be pumped out of the area into the river. Suitable land development in the walerlogged area (Sahoo et al. 2004) can increase the crop productivity of the land. The method suggests digging of a small portion of the land and filling up the rest portion. The dugout land can be used for storing excess water and fish culture whereas the elevated land can be used for crop cultivation. To study the effect of check dams on groundwater recharge and reduction of waterlogging, numerical analysis is recommended.

5.5 CONCLUSIONS

A hydrologic and hydrogeologic investigation was carried out in the Kathajodi River basin of Orissa for efficient utilization of water resources. The streamflow analysis of the Kathajodi River shows that maximum flows are most likely during the period July to September. The surface water resources are significantly reduced during dry periods. The geologic investigations show that a confined aquifer exists in the basin. The aquifer thickness ranges from 12 to 56 m. The groundwater monitoring in the region shows peak water level in the months July–August with April–May being the critical period. There is good correlation between groundwater level and rainfall, suggesting significant groundwater recharge from rainfall. The stream-aquifer interaction study shows that there is interaction between the Kathajodi River and the aquifer. The groundwater quality analysis in the pre-monsoon season shows that the water is very good for irrigation purpose.

It is suggested that construction of a series of check dams along the main drain can to some extent reduce the waterlogging in the downstream side and increase the groundwater recharge in the upstream side. Numerical analysis is suggested to study the effect. The results of the study would help develop integrated surface and ground water management plan in the study area.

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CHAPTER 6

Mechanisms and rates of recharge at Timbuktu, Republic of Mali

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ABSTRACT: The mechanisms and the rate of recharge have been assessed in the surrounding of Timbuktu, Mali, by the use of chloride budgets. The area has a Sahelian climate and receives a mean rainfall of 225 mm. Two clusters of wells and three soil profiles were sampled for the purpose. The well cluster in the vicinity of the river Niger showed effects of induced infiltration. For the well cluster further away from the river and near the town of Timbuktu a mean recharge of 3.7 mm was found. The recharge occurs preferably in depressions between dunes due to channeling by soil crusts on the slopes of the dunes. Occasional contents of high nitrate contents in soil and groundwater may be due to N-fixation by cyano-bacteria found on the soil surface.

6.1 INTRODUCTION

The mechanisms and amount of the groundwater recharge to the uppermost aquifer in recent sands have been studied in the vicinity of Timbuktu in northern Sahel in the Republic of Mali. Timbuktu lies 15 km north of the river Niger where the river forms a bend touching the southern limit of the Saharan desert. The mean yearly rainfall in Timbuktu is 225 mm, falling during the months of July, August and September while the potential evapotranspiration is 1800 mm (Fontes et al. 1991). The mean yearly temperature is 30°C. The area around Timbuktu is a sand dune landscape. The aquifer studied is one in the sand dunes with a groundwater level of 10–15 m below the ground level. This is a more or less perched aquifer above the Jurrassic-Cretacaeous Continental Intercalcaire aquifer which was studied by Fontes et al. (1991). The dunes are mobile within a distance of 4–5 km from the town due to excessive vehicle traffic while they are stabilized further ahead (Jacobberger-Jellison 1994). The dunes carry a sparse vegetation of *Balanites aegyptiaca, Acacia tortilis, Leptadenia pyrotechnica, Calotropis procera* and *Zizyphus spina christi.* The density is in the order of 15 trees/ha. The lowland in between the dunes does not carry any perennial vegetation but only ephemeral grasses like *Cenchrus biflorus* and *Panicum laetum* during and just after the rainy season, Occasionally small nebkha dunes (Tengberg 1995) with *Leptadenia hastata* are found in the depressions.

The aim of the study is to assess the groundwater recharge and compare the figures with the need of water by the local population and their herds.

6.2 METHODS

Chloride in groundwater and in soil water profiles in relation to the chloride in precipitation has been used for assessment and groundwater recharge in many investigations (Allison & Hughes 1983, and Gieske 1992). Rainwater was collected during the rainy season 1997 from July to October. Well water was collected from 32 wells situated outside habitations. Two clusters of wells were sampled, one close to Timbuktu, and the other about 10 km southwards closer to the river Niger. Soil samples were taken down to 2.2 m depth in three pits in depressions ("mare" in French) between sand dunes. The samples represented intervals of 0.1 m each. Soil moisture was determined and the chloride was extracted with de-ionized water. Chloride, nitrate and sulphate were determined with ion chromatography on a Dionex DX 120.

6.3 RESULTS AND DISCUSSION

6.3.1 *Rainwater chemistry*

It was impossible to use the chloride measured in rainwater due to the abundance of local terrestrial dust in the air containing attached salt particles. Dry deposition collectors in the form of textile filters in frames 0.2 by 0.2 m turned brown from dust even clear days with little wind. Thus the rainwater chemistry rather reflected the local soil environment than long range transport. In surface sand from dune areas needles of gypsum could be observed in the microscope explaining the sometimes high calcium and sulphate contents found in rainwater analyses. An assessment of the chloride content in rainwater from a similar climatic setting in the Republic of Niger gave 0.6 mg/L (Bromley et al. 1997a). From a wetter site, also in the Republic of Niger, Freydier et al. (1998) have arrived at a volume weighted content of 0.4 mg/L. A similar result from the same region is given by Modi et al. (1995) and by Galy-Lacaux & Modi (1998). A recent synthesis of the rainfall chemistry in Niger and northern Nigeria gives a range of chloride contents from 0.26 to 2.8 mg/L (Gon et al. 2001). This article presents an analysis showing that a cumulative rainfall of 225 mm in this region should contain a weighted chloride concentration of 0.4 to 0.8 mg/L. This level seems to be representative for continental sites in Africa. Jonnalagadda et al. (1994) recorded 0.4 mg/L in Zimbabwe and Gieske (1992) 0.5 mg/L in Botswana. Thus in the absence of an own estimate we have used 0.6 mg/L as a reasonable average for a continental site like Timbuktu.

6.3.2 *Recharge from well water chemistry and soil profiles*

The mean groundwater recharge derived by the chloride budget in 13 wells near Timbuktu was found to be 3.7 mm/year. 19 wells in a setting closer to the river Niger showed a recharge rate of 27 mm/year, most probably due to induced recharge from the river. The three soil profiles dug to 2.2 m depth in dune areas near Timbuktu (Figure 6.1) gave recharge rates of 2.1, 3.8 and 4.2 mm respectively. These figures are point values and should be viewed in the light of the runoff-runon regime which has been observed in the dune landscape (Gaze et al. 1997). The profile with the highest recharge, 4.2 mm/year, is the one where the most visible signs of runoff-runon were observed in terms of erosion channels on the slope. Leduc et al. (2000) have calculated a recharge rate of 2–3 mm/year for a large area in the Republic of Niger which receives from 0–350 mm of rainfall. Recharge rates of 11–17 mm/year have been recorded in continental parts of Africa with about 500–600 mm rainfall (Bromley et al. 1997a, Nkotagu 1996). Most of the water losses to the atmosphere are through evaporation. Nizinski et al. (1994) found that the transpiration of a thorn shrub consisting of *Acacia tortilis* and *Balanites aegyptiaca* with 150 trees/ha in a slightly wetter climatic setting in Senegal was 340 mm. Thus the 15 trees/ha should transpire around 50 mm. The exposure to wind will be higher in a thinner stand elevating the transpiration rate. Soil evaporation takes a major toll. Wallace & Holwill (1997) registered 80% soil evaporation out of 260 mm rainfall in patterned woodland in Niger.

6.3.3 *Crust formation in sand dune areas*

Crust formation on the lower slopes of sand dunes plays an important role, promoting the groundwater recharge creating a runoff-runon regime (Gaze et al. 1997). The crusts are formed by filling in of the voids between the sand grains with finer particles through water transport, thus clogging

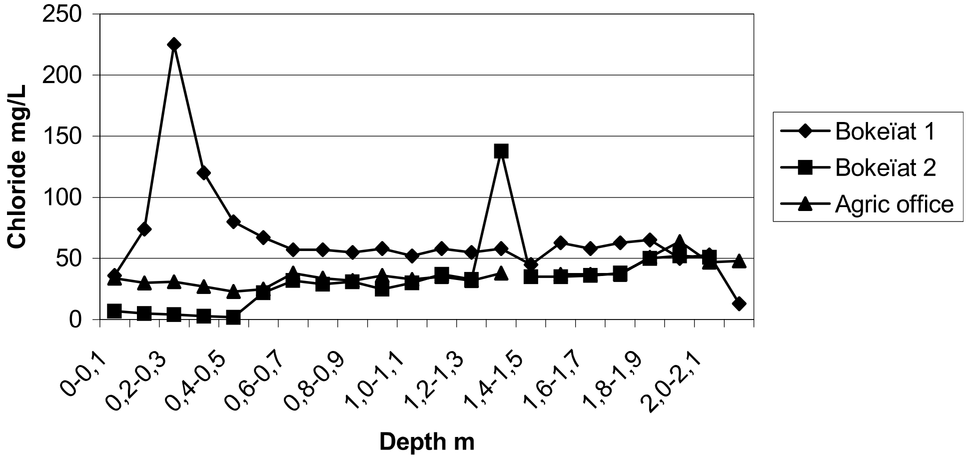

Figure 6.1. Chloride profiles in three depression areas between dunes.

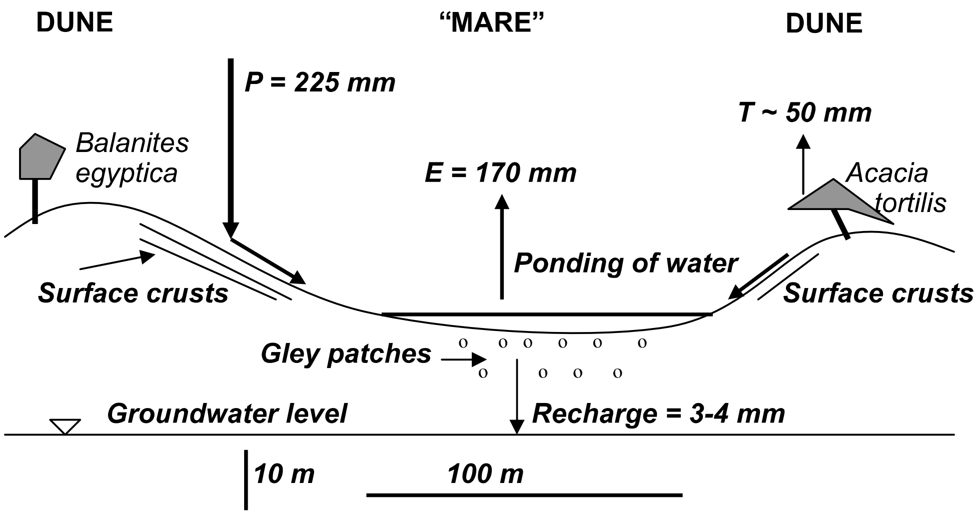

Figure 6.2. Water cycling in the dune landscape near Timbuktu. P = precipitation, E = evaporation, T = transpiration.

the surface (Heil et al. 1997, Valentin & Bresson 1992). Multiple crust are usually found, separated by a vertical distance of a few millimeters.

While the surface flow dominates the lateral transport, sub-surface flow may contribute substantially to the solute transport (Ribolzi et al. 2000). In the dune areas near Timbuktu the crusts on the lower slopes of the dunes have a funnelling effect causing pooling of water in the depressions (Figure 6.2).

The crusts are considered as negative features in dryland cultivation causing water losses from millet and sorghum fields (Rockström & Valentin 1996, Rockström et al. 1998, Daba 1999). Losses by surface runoff may be 25–50% for intense rainfall events (Rockström 1999). However, Bromley et al. (1997b), Issa et al. (1999) and Heimaux & Gerard (1999) have found that such crusts acts as water collectors for strips of shrub vegetation ("brousse tigré" in French) in areas with about

500 mm yearly rainfall. The depressions between the dunes are called "mares" in French actually meaning pools. The pools exist for a few weeks during and immediately after the wet season. Their existence is witnessed by gley phenomena in the subsoil of the mares. The gley patches indicate anoxic conditions in the mares during a period of the year and this is probably the reason for the absence of perennial vegetation like trees and bushes in the mares. In the mares there is only ephemeral grass vegetation, mainly *Cenchrus biflorus* or *Panicum laetum.* Few tree species in the semi-arid areas can stand anoxic conditions in the root zone. *Acacia nilotica* is one of them (Maydell 1986), but this specie is not present in this region. The trees and bush vegetation exists only on the dunes. This means that a chloride profile above the zone of capillary influence in the mare area can be used for assessment of recharge rate. The sandy character of the soil excludes any capillary raise of water to the surface.

The importance of the crusts is seen in Figure 6.1. The Bokeïat 2 site showed the pronounced crusting with visible patterns of surface runoff on the slopes from the previous rainy season. The Bokeïat 2 profile shows a low chloride content in the upper part indicating a considerable percolation during the rainy period from July to October immediately preceding the sampling in November. Two short profiles, each 1 m deep, one dug in the slope of the dune and one in the mare showed mean contents of chloride of 13 mg/L respectively 47 mg/L. This is a qualitative indication of the importance of the runoff-runon regime and stresses that the recharge the assessing that the assessment of recharge through the chloride content of the water in the vadose zone requires far more data than has been collected in this investigation.

6.3.4 *Nitrate in the vadose and groundwater zones*

Both groundwater and soil water contains occasionally high contents of nitrate, up to as much as 350 mg/L. Similar concentrations have been detected by Edmunds & Gaye (1997) and Deans et al. (2005) in northern Senegal in Sahel and by Tredoux & du Plessis (1992) in South Africa. Three sources can be figured out, symbiotic N-fixation by vegetation like acacias, animal excreta and cyano-bacteria growing in the mares (Garcia-Pichel & Belnap 1996, Défarge et al. 1999). A low C/N ratio, well below 10 in the surface soil in the mares indicate that the cyano-bacteria are an important source. Such crusts do not prevent infiltration of the water (Eldridge et al. 1997) while they may prevent erosion (Défrage et al. 1999, Issa et al. 1999). The erratic climate with rainfall initiating N-fixation followed by dry spells with mineralization, nitrification and subsequent heavy showers flushing the nitrate below root depth can be visualized as a reasonable mechanism. The abundance of nitrogen fixing organisms like the Acacia-Rhizobium symbiosis and the cyano-bacteria in this environment is probably explained by the fact that the nitrate formed during the the dry season is often flushed below the root zone of plants by the onset of the rainy season. In spite of the scarcity of rainfall it often comes in the form of heavy showers (Rockström 1999). Under such conditions nitrogen fixing capacity is a competitional advantage.

6.4 CONCLUSIONS

About 2% of the precipitation forms groundwater recharge. Surface crusts are important features that promote groundwater recharge as they funnel surface runoff to depressions where the water can percolate. The pools formed ("mare" in French) are anoxic at least for a period which is displayed by gley patches in the subsoil. This may explain the absence of perennial vegetation like shrubs in the mare areas. High contents of nitrate is common in the groundwater and soil water, one important source may be the cyano-bacteria which are observed in the mare areas. The present groundwater abstraction from the uppermost aquifer in recent sands for household use and watering animals is sustainable. In case of droughts the main threat to the rural population is the difficulty in finding grazing for their herds. An increase in the groundwater recharge might be achieved by tilling the mare areas to break up any crust and form furrows to increase the surface for recharge (Cattle 1999).

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CHAPTER 7

Evaluation of natural and artificial recharge using tracers in a semi-arid region of Kurnool district, Andhra Pradesh, India

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ABSTRACT: The Kallugotla watershed in Kurnool district of Andhra Pradesh, located in semi arid region, experiences scarcity of water for drinking and irrigation. The watershed (25 km^2) covered by limestone-quartzite terrain of Upper Precambrian age. The area receives an average annual rainfall of 550 mm. Integrated geohydrological investigations comprising resistivity surveys, hydrogeological surveys, bore hole pumping tests and natural recharge studies were carried out in the watershed for selection and design of artificial recharge strategies for sustaining the yield of irrigation wells. Based on the investigations, few watershed specific artificial recharge strategies were suggested and implemented in this watershed. Natural recharge rate due to combined effect of rainfall and applied irrigation was measured at 6 sites in the watershed area during 2000 monsoon using tritium tracer technique. The result indicates average recharge of 105 mm for the seasonal rainfall of 585 mm and irrigation input of 377 mm. The natural recharge and water level fluctuation data obtained in this watershed with the recharge measurements made in the adjacent watershed was used for estimating natural recharge due to rainfall and return flow due to applied irrigation independently. Dye and chemical tracers were used in tracer experiments for delineating flow pathways during the time of groundwater recharge through the artificial recharge structures. The tracer data obtained was used for determining the influencing zone and beneficiary wells due to artificial recharge structures. The tracer studies indicated that the recharge occurs along the gently dipping joint planes.

7.1 INTRODUCTION

Quantification of recharge to groundwater through various sources in a watershed or basin is necessary for systematic development and management of groundwater resources. The various sources of natural recharge are precipitation, return flow from applied water irrigation, seepage through streams, ponds etc. Infiltration of monsoon precipitation through unsaturated zone is the principal source of natural recharge. Due to failure of monsoon and overexploitation of limited groundwater potential, serious water table decline is observed in low rainfall semi arid region of southern peninsular India. The bore wells drilled in these areas for irrigation and domestic purposes are showing reduction in yield or going dry within short span of time. During the last two decades, artificial recharge programs were undertaken by central, state, NGO's and research institutions, for developing a suitable methodology/replenishing the aquifers, in order to sustain the water supply for drinking and irrigation.

The applications of tracer offer a wide scope for evaluation of natural and artificial recharge. Both radioactive and chemical have increasingly becoming popular in the hydrogeological investigations. In this paper we present the results of tracer studies carried out in Kallugotla watershed located in limestone terrain in semi arid region of Andhra Pradesh for evaluation of natural recharge and in studying the impact of artificial recharge structures implemented in the watershed.

7.2 STUDY AREA

Kallugotla watershed (25 km^2) is about 30 km south of the city of Kurnool in Andhra Pradesh in Southern India (Fig. 7.1). The watershed has hilly terrain in the south and northwestern parts and has a general slope from south to north. The watershed is covered by limestone and quartzite's belonging to Cuddapah and Kurnool Group of Precambrian age. The watershed is sloping towards north, while formations exposed are dipping opposite to the drainage direction (NGRI 2001a, b). The central part of the watershed area is a flat terrain with an average elevation of 400 m. A single major stream known as Saddara Vagu drains the catchment area. The soils are loamy and black soils in the central part and gravelly soils near the foothills of quartzes. The Watershed can be sub divided into structural hills, buried pediments (moderate) and buried pediments (shallow). Groundwater occurs under water table condition in the weathered or cavernous shaley limestone and attains a semi-confined condition in the fracture or cavernous zone at deeper depths. Groundwater in the watershed is exploited by open wells, private bore wells and also by Andhra Pradesh State

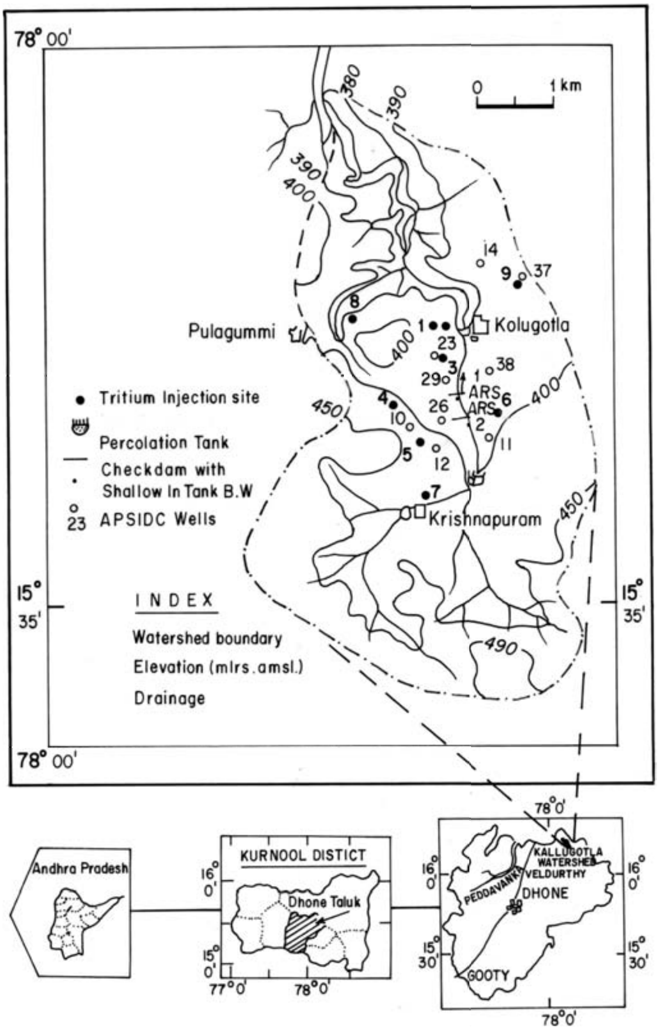

Figure 7.1. Location map of the Kallugotla watershed, Kurnool district, Andhra Pradesh, South India.

Irrigation Development Corporation (APSIDC) irrigation wells drilled up to the depth of 60–90 m. The average annual rainfall in the watershed is about 650 mm.

7.3 NATURAL RECHARGE EVALUATION IN THE WATERSHED USING TRITIUM TRACER

Tritium (tracer) injection method was applied in the watershed area for evaluating annual replenishment due to rainfall/rainfall together with the applied irrigation during 2000 monsoon season. The tritium injection method of estimating natural recharge is based on Piston-Flow model for movement of moisture in the unsaturated zone which has been developed by Munnich (1968). This technique was used for measurement of natural recharge in several watersheds and basins located in different hydrogeological provinces of India (Rangarajan & Athavale 2000). In this technique, the moisture at a certain depth in the soil profile, is tagged with tritiated water. The tracer moves downwards along with the infiltrating moisture, due to subsequent precipitation or irrigation. A soil core from the tracer-injected site is collected after a chosen interval of time and the moisture content and tracer concentration are measured for various depth intervals. The peak in its concentration distribution indicates the displaced position of the tracer. Moisture content of the soil column between the injection depth and displaced depth of the tracer in the soil is the measure of recharge to groundwater over the time interval between injection of tritium and collection of soil profile. The piston flow model has been verified in laboratory and field conditions for various soil types through multiple Tritium injection experiments (Rangarajan 1997).

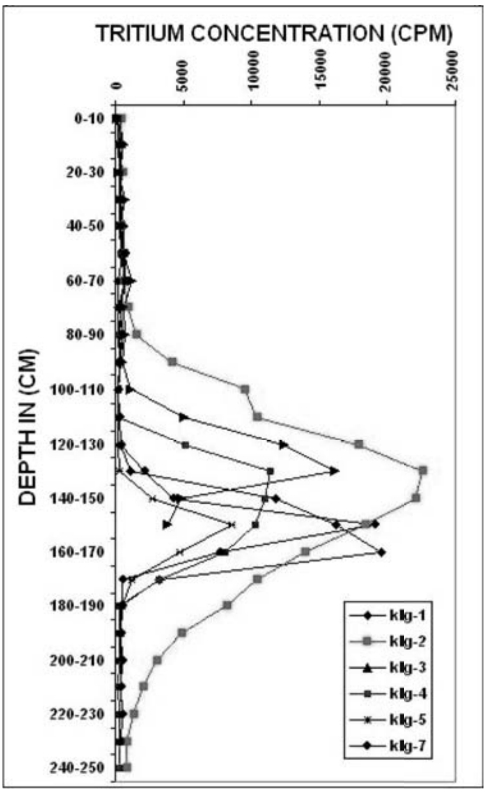

Figure 7.2. Tritium profiles at recharge sites.

Natural recharge measurements were made at 9 sites in the watershed area. The sites are located in agricultural plots where ID crops are grown and the slope of the surface is less than 1%. Tritiated water having activity of 150 microcurie (3.33 × 108 dpm) was injected at 60 cm bgl at each site in the month of June 2000 before the monsoon season. The sources of input at these sites are from rainfall or rainfall + groundwater applied irrigation. Vertical soil profiles collected from the tracer injected sites after the completion of monsoon were analysed in the laboratory and the data obtained were used for computing recharge at each site. The tritium profiles observed at 6 natural recharge sites is shown in Figure 7.2.

The natural recharge values obtained at 9 tritium injected sites in the watershed area vary from 69.1 mm to 128.9 mm, with a mean value of 105.4 mm and a standard deviation of 22 mm. The seasonal rainfall recorded during June 2000 to January 2001 (effective rainfall) in the Watershed area was 585.8 mm. Average input at tritium sites by applied irrigation during the same period was computed as 377 mm. The total input due to rainfall and irrigation is therefore 962.8 mm. The natural recharge computed is thus equivalent to 11.0% of the total input due to precipitation and irrigation.

7.4 IMPACT OF ARTIFICIAL RECHARGE STRUCTURES IN THE WATERSHED USING DYE AND CHEMICAL TRACERS

The irrigation bore wells drilled by APSIDC in the watershed area have shown drastic decline in yield particularly over a period of one year utilization for ID crops cultivation practices. Integrated geohydrological investigations were carried out in the watershed and several artificial recharge

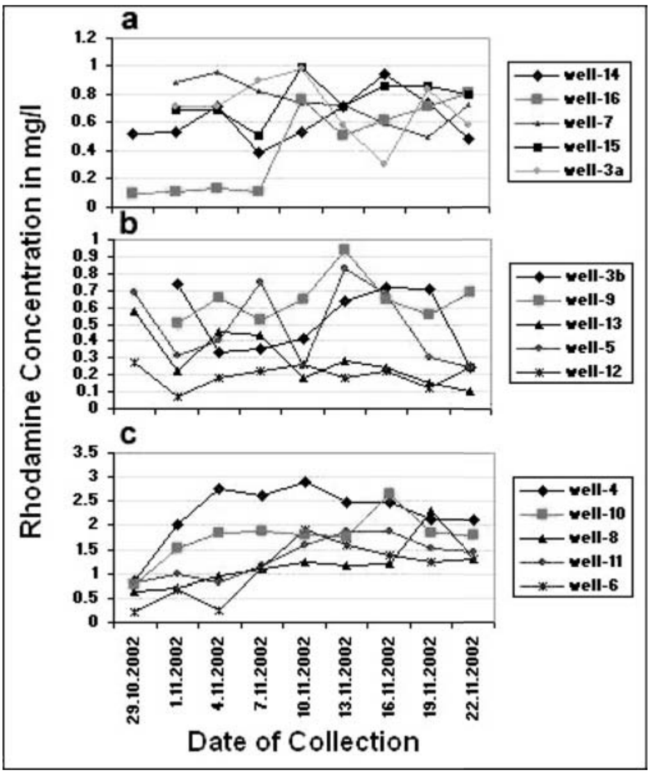

Figure 7.3. Rhodamine-B tracer concentration in observation wells in Kallugotla watershed.

structures and strategies were recommended mainly for sustaining the yield of APSIDC irrigation bore wells (NGRI 2002). APSIDC executed structures such as mini percolation tank and cascade of check dams with in tank shallow bore wells (ARS-1, ARS-2) as shown in Fig. 7.1. The mini percolation tank in the upstream part where two-second order streams converge and cascading check dams (2 numbers at favorable locations) are in the central part of the watershed along the stream course. The percolation tank was constructed for the purpose of storage runoff water caused by high intensity rainfall events and as a source of water for check dams (through gates) in the downstream side. Low resistivity zones having thickness of 6 m were encountered below 12 m depth at check dams locations. In order to recharge this zone, bore holes up to a depth of 25 m, filter bed assembly was constructed in the storage of these two check dams (ARS1, ARS2). It was expected that the aquifer from a depth of 14–20 m gets recharged by this process and will be able to supply water to the aquifer which are subjected to pumping by the APSIDC bore holes.

Rhodamine-B tracer was injected to the in tank shallow bore well ARS-1 and bromide in the form of Potassium Bromide was injected in to the in tank shallow bore well ARS-2 on 29.10.2002 (Fig. 7.1). Groundwater samples from the APSIDC bore wells and nearby private bore wells were collected periodically and tested for arrival of Rhodamine-B and Bromide tracers. Rhodamine-B concentration in well water samples was determined by Aquaflor Florimeter and bromide concentrations in samples by WTW ion selective bromide electrode.

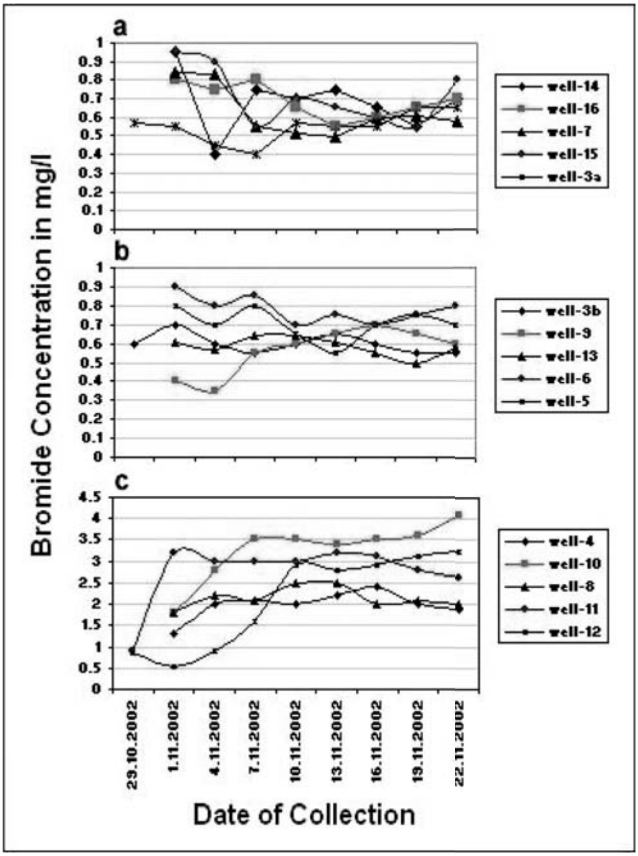

Figure 7.4. Bromide tracer concentration in observation wells in Kallugotla watershed.

The bore holes 6, 4, 10, 8 and 11 located in north, west and south of injection well ARS-1 have shown the arrival of Rhodamine tracer concentration (Fig. 7.3c). The observation bore holes 4,10,8,11,12 located in north, west and south of injection well ARS-2 have shown the arrival of Bromide concentration (Fig. 7.4c). Based on the distance of the nearest well and clear Rhodamine and Bromide tracer signals observed in wells, the velocity of groundwater is calculated as greater than or equal to 28 m/d. High velocity (28 m/d) implies that the subsurface limestone aquifer is highly fractured and transmissive in nature. This is supported by significant water level rise in bore wells during the monsoon season even though the natural recharge rate is only 11%. The tracer data also indicates that some of the bore wells around the recharge structures are benefited (Figs. 7.3c, 7.4c) and some are not benefited (Figs. 7.3a, 7.3b, 7.4a, 7.4b). Injected tracer concentration observed in the wells located in the upstream side of artificial recharge structures (southern side) also indicates that recharge is taking place through (3°–5°) dipping planes towards south and moving against the topographic slope and stream flow direction. The investigations also demonstrated that the artificial recharge strategies identified on scientific investigations basis are proper and improves the groundwater resources of these watersheds for sustainability of irrigation wells.

7.5 CONCLUSIONS

Integrated geohydrological investigations involving resistivity, hydrogeological surveys, bore hole pumping tests and natural recharge studies were carried out in the watershed for selection and design of artificial recharge strategies for sustainable yield of the irrigation wells. Based on the investigations, few watershed specific artificial recharge strategies were suggested and implemented in the Kallugotla watershed in Kurnool district of Andhra Pradesh. Tracer technique is found to be a potential tool for evaluation of natural recharge and tracing the flow path during artificial recharge process. The influencing zone and beneficiary wells due to artificial recharge structures can be delineated through tracer application.

ACKNOWLEDGEMENTS

The authors deeply appreciate the meticulous comments of Gunnar Jacks and D. Chandrashekharam on the manuscript. We are thankful to the organizers of the 2nd International Groundwater Conference and the editors of the volume especially Prosun Bhattacharya for the editorial assistance.

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Section II
Water and environment

CHAPTER 8

Hydrogeochemical studies around the Bhalswa landfill in Delhi, India

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ABSTRACT: The study of the hydrogeochemical processes in the vicinity of Bhalswa landfill was carried out with an objective to identify the geochemical processes and its relation with groundwater quality besides understanding the impact of landfill leachates on groundwater quality and to get an insight into the hydrochemical evaluation of the aquifer system. The various graphical plots and statistical analysis has been carried out using chemical data to deduce a hydrochemical evaluation of aquifer system based on the ionic constituents, water types, hydrochemical facies and factors controlling groundwater quality from May 2003 to January 2005. Total thirty three groundwater samples were collected from shallow and deep aquifers around the landfill covering an area of 15 km^2. Groundwater sampling was carried out twice during pre-monsoon and post-monsoon seasons each in these periods. The study also reveals the suitability of groundwater for irrigation and drinking purpose as well as water quality variation of groundwater. The contour line around landfill indicates that the landfill is a point source of pollutant. Geology of study area is mainly made up of alluvium; there is a high probability of leaching of pollutant into groundwaters. The concentrations of metal contaminants in the landfill leachates are generally high with maximum concentrations of Fe (∼22 mg/L), Mn (∼20 mg/L), Cu (∼10 mg/L), Pb (∼2 mg/L), Ni (∼0.25 mg/L), Zn (∼10 mg/L), Cd (∼0.2 mg/L). The concentration of the major ions such as Cl^- (∼4000 mg/L), SO_4^{2-} (∼3320 mg/L), PO_4^{3-} (∼4 mg/L), NO_3^- (30 mg/L) and F^- (∼50 mg/L) were also much higher than the regulatory standards. The study reveals that landfill is in depleted phase and it is affecting groundwater quality in the vicinity of landfill and surrounding area due to leaching of contaminants and posing a serious threat to the groundwater in the adjoining areas.

8.1 INTRODUCTION

The study of hydrogeochemical processes in groundwater was carried out in the vicinity of an unplanned Bhalswa landfill in Delhi, India, with an objective to understand the impact of the landfill leachates on groundwater quality, which is used for drinking, agricultural and industrial purposes. It helps to understand and distinguish the rock water interaction and anthropogenic influences i.e. the leaching of pollutant from landfill. The geochemical properties of groundwater also depend on the chemistry of water in the recharge area as well as the different geochemical processes occurring in the subsurface water.

The geochemical processes occurring within the groundwater and the reaction with aquifer minerals have a profound effect on water quality with addition of leaching of pollutant from landfill create further alterations in quality of groundwater (Srivastava & Ramanathan 2008). The quality of water along the course of it's under ground movement is dependent on chemical and physical properties of surrounding rocks, the quantitative and qualitative properties of through flowing water bodies and products of human activity (Mathess 1982). Groundwater chemically evolves by interacting with aquifer minerals or internal mixing among different groundwater along flow-paths in the subsurface. Therefore, spatial distribution of chemical species gives some idea about the direction of groundwater movement. Schultz & Kjeldsen (1986) indicated that increases in solute concentrations in the groundwater were caused by spatial recharge, governed by micro-topographic controls. Generally groundwater at the discharge zones tend to have higher mineral concentration compared to that at the recharge zones due to the longer residence time and pro-longed contact with the aquifer matrix (Freeze & Cherry 1979). Further, the weathering of primary and secondary minerals is brought by the release of cations and silica into it. Several conventional methods of data analysis are available for the simple interpretation and presentation of results (Matthess 1982, Hem 1989), e.g. histograms, trilinear, semi-logarithmic diagrams and many others. Multivariate analysis, such as factor analysis is used simply as a numerical method of discovering variables that are more important than other data for representing parameter variation or demonstrating hydrochemical processes. It relies on a set of assumptions about the nature of the present population from which samples are drawn. These assumptions provide the rational for the operation that are performed and the manner in which the results are interpreted. Viewed from an international perspective of '$<$1700 m^3/person/yr' as water stressed and '$<$1000 m^3/person/yr' as water scarce, India is water stressed today and is likely to face severe water scarcity by 2050 (Central Pollution Control Board 2001). Delhi being the rapid growing capital (population 10.3 million people in 2004–05) city of Asia, is facing problems concerning both the groundwater quality and quantity and is presently forced to meet their 50% of water supply requirement from groundwater (Central Pollution control Board 2001). Except some reports on flourosis and selected water quality parameters, no published information is available on the extent of population that is exposed to contaminated groundwater in different parts of Delhi. As per an estimate the landfills of National Capital Territory, Delhi cumulatively generate a significant quantum of 8,14,800 m^3 of leachates annually, which is alarming from groundwater point of view (Central Pollution Control Board 2001). The geology of the study area makes it more susceptible for leaching.

Central Pollution Control Board (2001) reported indication of the increased level of pollution due to leaching from the unplanned landfills in various parts of the country. Hence it needs more detailed study about this specific landfill. A vast amount of literature is available on presence of trace elements in solid wastes (Tisdell & Brestin 1995, Shivhare & Pandey 1996). Acidity of the soil solution and solubility of metals are closely related (Tyler et al. 1987). According to Tyler et al. (1987), the acidity of soil increases by 3 to 5 fold with increase of metal concentration. This causes high leaching of hazardous metals (Blais et al. 1993). The solid waste containing metals at low pH has a high pollution potential to contaminate groundwater (Olaniya et al. 1992). Groundwater contamination is degradation of the natural quality of groundwater as a result of human activity. The adverse impacts of landfills leachates and adjacent surface water and groundwater have prompted great number of studies. Since Bhalswa landfill region is one of the most urbanized area of the capital of India, and it influence a huge population. Therefore it needs much more attention to study groundwater quality in its vicinity. Hence in the present study, a detailed investigation was carried out with the objective of identifying the hydro-geochemical processes and its relation to present groundwater quality, to identify the hydrochemical evaluation of the aquifer system. The study also attempts to assign the weightage to the factors that controlling water chemistry and general suitability of groundwater for irrigation and drinking purpose.

8.2 STUDY AREA

Bhalswa landfill is located in one of the most urbanized area in northwest Delhi, India in between latitudes 28°42′30″ and 28°45′N and longitudes 77°7′30″ and −77°11′54″E (Fig. 8.1). Sampling site is shown in Figure 8.1.

8.2.1 *Climate and precipitation*

The climate is semi-arid nature due to marked diurnal differences of the temperature, high saturation deficit and low-moderate rainfall. The climate is markedly periodic and is characterized by a dry and gradually increasing hot season between March and June, a dry and cold winter from October to February and the warm, monsoon period from July to September. The average rainfall is about 721 mm/year (Indian Meteorological Department, IMD, 1990–2005). The minimum (average)

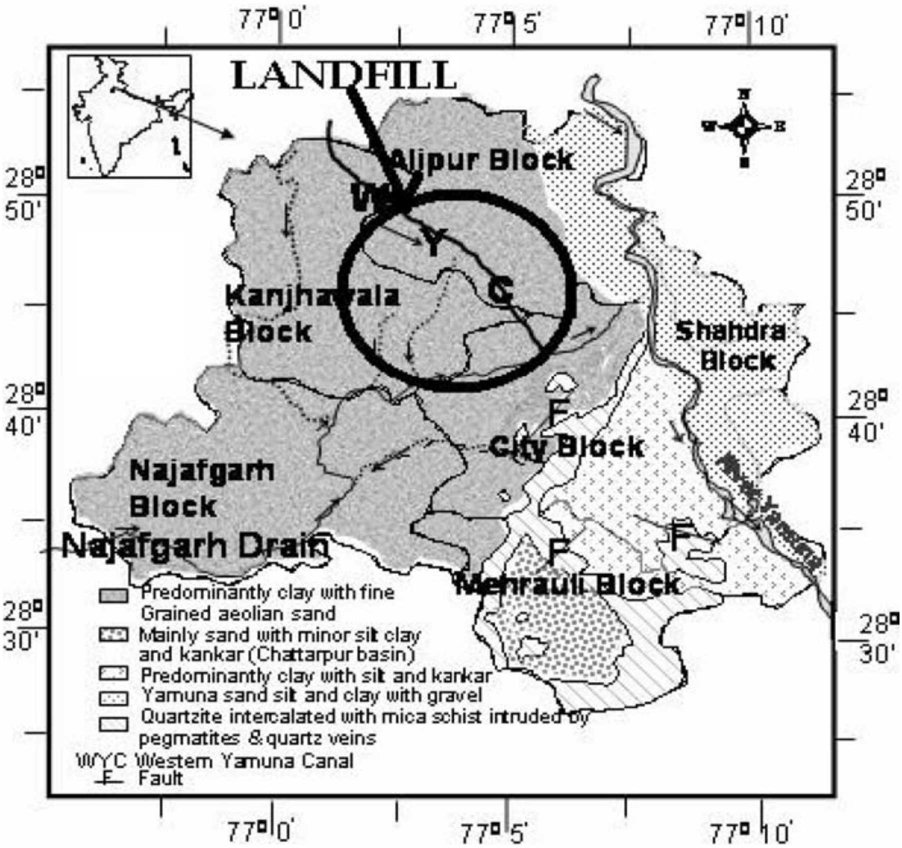

Figure 8.1. Location and geology of study area around the Bhalswa Landfill site in Delhi.

and maximum (average) temperatures are 19.2°C and 31.5°C respectively, with daily maximum temperatures during the hottest months commonly exceeding 41.2°C.

8.2.2 *Geology*

The Delhi region is a part of Indo-Gangetic Alluvial Plain, at an elevation ranging from 198–220 m above mean sea level. Lithologically, the area is transected by a quartzite ridge forming the northern extension of the Aravalli Hills that passes through the southern border of Delhi and ending to the north on the west bank of the Yamuna River (Fig. 8.1). Physiographically, the region shows four major variations, including the Delhi Ridge. A prolongation of the Aravalli hills consisting of quartzite rocks and extending from the southern part of the territory to the western bank of Yamuna for about 35 km. The alluvial formations overlying the Quartzitic bedrock have different natures on either side of the ridge. The nearly closed Chattarpur alluvial basin covering an area about 48 km^2 that is occupied by the alluvium derived from the adjacent quartzite ridge alluvial plains on the eastern and western sides of the ridge and Yamuna flood plain deposits. These are of recent origin, also termed as newer alluvium. The Newer alluvium is characterized by the absence of permanent vegetation due to periodic flooding and lack of kankar. Thickness of alluvium overlying the bedrock is increasing in the direction away from ridge and reaches 300 m in the western parts of Nazafgarh, Kanjhawla block and in the northern part of Alipur block. The thickness of alluvium in the region of the Yamuna River in the east of the ridge is about 165 m (Central Ground Water Board 2001).

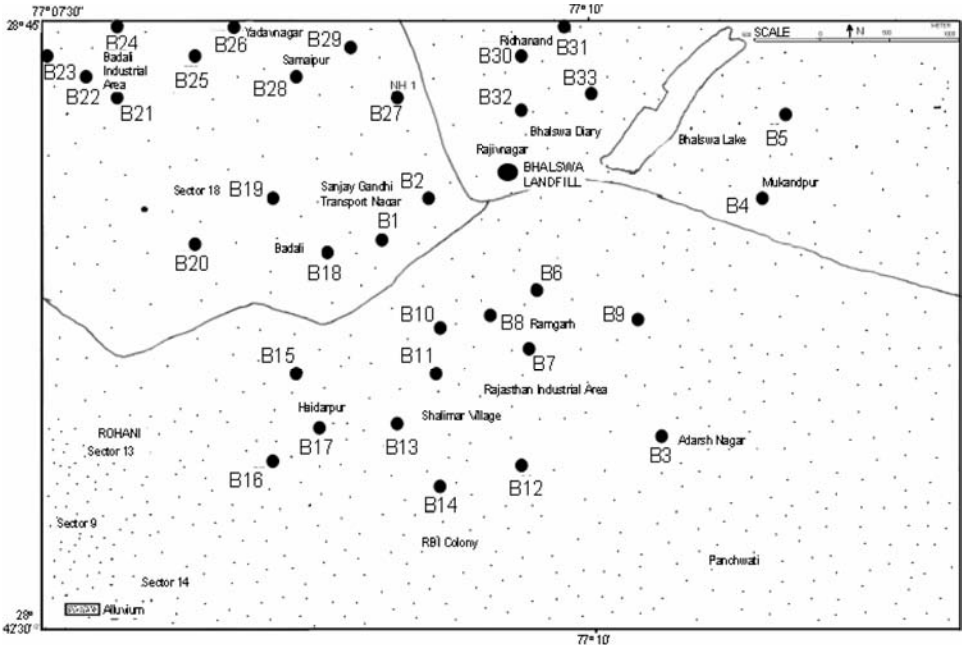

Figure 8.2. Location map of the sampling sites around the Bhalswa landfill.

Land utilization in Delhi has changed significantly over the years due to conversion of agricultural land for urban needs. A large part of the area has alkaline and saline soils with abundant calcrete depositions at places (Wadia 1981). The alkaline soils contain sodium bicarbonate and carbonate among the soluble salts, while the saline soils are impregnate with sodium chloride and sulphate as the main soluble salts. The distribution of these soils divides the study region into four geographical sub-regions. The northwest region is covered by calcareous, silt clay loam; the northeast soils are calcareous, silt, clay and sandy-loam type, the southern parts.

8.3 METHODOLOGY

To understand the general variation in groundwater chemistry over the study area, a well inventory survey was carried out during March 2003 and electrical conductivity (EC) and pH were measured. A global positioning system (GPS) was used for location and elevation reading and supported by topographic sheets made available from geological survey of India. These data were used to select the representative wells, hand pumps for groundwater sampling. Sampling wells were selected in such a way that they represent different geological formations as well as land use pattern at varying topography of this area (see Fig. 8.2). Sampling has been carried out in an area of about 15 km^2 around the landfill about two times in a year. The sampling was done in pre-monsoon and post-monsoon four times form May 2003 to January 2005. Sampling wells were selected in such a way that they represent different geological formations as well as land use pattern at varying topography of study area. Groundwater samples collected in 500 mL bottles. The parameters such as pH, ORP, DO, temperature, electrical conductivity (EC) and total dissolved solids (TDS) were analyzed in the field using Consort NV water-analyzer kit.

For the analyses of cations, 100 mL samples were filtered using 0.45 μm filters and preserved with ultra pure HNO_3, while HBO_3 was used as preservative for nitrate analysis and stored at 4°C temperature to avoid any major chemical alteration for various chemical analysis (APHA 1995).

Sodium and potassium were analyzed by AIMIC, PE I Flame Photometer following the standard methods. Trace elements (Mn, Fe, Cu, Ni and Zn) and alkaline earth metals (Mg and Ca) were analyzed by Atomic Absorption Spectrophotometer (AAS-900). Anions (SO_4^{2-}, NO_3^-, F^- and Si) were analyzed by using JENWAY 6505 UV/Vis Spectrophotometer following the standard method as mentioned in APHA (1995). Bicarbonate and chloride were analyzed by titration method using standard procedure (APHA 1995).

8.4 RESULT AND DISCUSSION

The average, maximum, minimum and standard deviation for each water quality parameters analyzed for both pre-monsoon and post-monsoon for year 2004 and 2005 is presented in Table 8.1.

8.4.1 *General water chemistry*

Groundwater pH around Bhalswa landfill is slightly alkaline due to influx of HCO_3^- in subsurface water with rainwater through various ion exchange processes in aquifer system (Table 8.1). Out of many samples, few samples in Bhalswa Dairy village reported exceptionally very high conductivity around 5000 μS/cm. However, a periodic sampling of the remaining wells for continuous two years shows conductivity in between 1000 μS/cm to 3000 μS/cm (Table 8.1). The groundwater samples collected from Bhalswa Dairy village, Shalimar Village, J.J. Colony, Badali and Ramgarh showed comparatively high electrical conductivity. According to classification scheme of Wilcox (1955) around 63.6% sample are in permissible limit, 6.06% in good condition while 30.30% samples shows much higher conductivity are in doubtful condition. Most of the TDS values obtained in the study area are beyond the permissible limits, rendering the water unsuitable for various domestic activities. The high value of total dissolved solid in post-monsoon in comparison to pre-monsoon is due to dissolution of various ions with rainwater.

8.4.2 *Major anions*

The average value of chloride and other anions (SO_4^{2-}, NO_3^-, F^-) are higher in post-monsoon than pre-monsoon perhaps due to the rise of water table in post-monsoon season which dissolved salts coated in soil with rainwater (Ramesam 1982, Ballukraya & Ravi 1999, Jalali 2005) (Table 8.1). The high concentration around the landfill indicates possible anthropogenic input of these contaminants i.e. leaching of these anions from landfill. The landfill leachates contain high concentration of Cl^- ($\sim$4000 mg/L), NO_3^- ($\sim$30 mg/L), F^- ($\sim$50 mg/L) and PO_4^{3-} ($\sim$4 mg/L), which is higher than the recommended value of Central Pollution Control Board (2001), (Table 8.2). The Bhalswa Dairy Village, Ramgarh and Yadva Nagar shows comparatively high concentration may be due to migration of contaminant plume with groundwater flow. Generally the major source for nitrate in groundwater is domestic sewage, run off from the agriculture field, and leachates from the landfill sites (Lee et al. 2003, Jalali 2005).

The groundwater samples collected around landfill show, nitrate concentration in groundwater is very close to concentration observed in leachate (Tables 8.1 and 8.2). This indicates that vertical soil profile of landfill is highly saturated with nitrate concentration which helped seeps out whole quantity of nitrate from landfill to groundwater. In general, Bawana and J.J. Colony, show very high concentration of nitrate indicate anthropogenic input in groundwater aquifer system. The spatial and temporal variations of nitrate concentration around landfill indicate possible anthropogenic source in this area.

8.4.3 *Major cations*

The major ion concentration in groundwater samples collected around Bhalswa landfill is higher than the standard value as given by USPH and WHO guidelines for drinking water quality and

Table 8.1. Summary of hydrochemical parameters analyzed in the groundwater samples around the Bhalswa landfill site.

	Year 2004									Year 2005						
	Pre-monsoon				Post-monsoon					Pre-monsoon			Post-monsoon			
Parameter	Min.	Max.	Mean	SD	Min.	Max.	Mean	SD	Min.	Max.	Mean	SD	Min.	Max.	Mean	SD
Temp. (°C)	22.5	35.1	26.8	nd	21.2	32.2	25.4	nd	22.7	36.2	27.8	nd A	19.2	28.4	23.4	nd
pH	6.34	7.87	7.06	0.26	6.29	7.64	7.17	0.29	6.61	7.91	7.56	0.26	7.09	8.49	7.82	0.35
Eh (mV)	140	178	160.3	9.6	146	177	161.6	19.6	144	188	167.3	9.80	165	186	167.2	9.76
EC (μS/cm)	748	3256	18406	8136	694	2974	1671	724.3	768	3600	2140	843	648	3171	1691	785
TDS (mg/L)	477	2042	1171	503	464	2014	1149	502	487	2059	1184	503	494	2064	1184	507
Cl^- (mg/L)	142.4	1019.7	434.0	225.6	133.5	1174.8	445.6	214.5	162.4	1129.7	467.0	215.6	150.1	1197.4	474.1	234.9
SO_4^{2-} (mg/L)	61.5	551.5	238.9	129.0	56.1	534.7	222.1	118.2	58.45	591.47	248.9	139.0	69.7	612.5	254.8	122.5
HCO_3^- (mg/L)	66.	284.3	150.9	54.2	72.05	295.4	159.9	51.3	66.02	304.27	154.9	59.21	73.2	313.2	168.1	50.5
NO_3^- (mg/L)	8.03	43.5	20.0	9.62	8.80	49.4	23.7	8.17	9.52	46.46	29.90	10.62	10.0	55.2	33.7	11.0
F^- (mg/L)	2.15	8.45	5.58	2.16	2.01	9.24	5.89	2.11	2.15	9.45	5.68	2.10	2.41	9.76	6.22	2.17
Na^+ (mg/L)	62.5	683.5	300.8	164.1	45.5	736.0	284.1	153.7	72.49	783.5	360. 8	154.07	67.8	758.9	275.06	161.1
K^+ (mg/L)	4.98	41.66	22.02	10.1	3.71	34.4	19.2	9.31	7.98	49.7	28.02	9.09	6.1	45.1	27.47	11.1
Mg^{2+} (mg/L)	14.5	48.1	24.98	9.3	11.3	42.4	20.7	8.49	15.45	58.07	34.9	9.54	13.4	51.5	33.3	8.95
Ca^{2+} (mg/L)	68.9	215.5	101.1	40.9	54.48	200.4	104.8	40.2	88.9	315.49	156.1	60.9	73.5	291.8	146.5	49.2
Mn (mg/L)	0.02	1.90	0.51	0.57	0.01	1.61	0.44	0.48	0.02	2.02	0.51	0.57	0.01	1.9	0.32	0.26
Fe (mg/L)	0.83	6.34	2.3	1.59	0.64	6.0	2.10	1.63	1.12	7.14	2.3	1.59	0.89	6.97	2.16	1.55
Zn (mg/L)	0.12	2.27	0.94	1.05	0.09	3.06	0.85	0.99	0.15	3.67	1.24	0.87	0.18	3.32	1.09	0.93
Cu (mg/L)	0.01	0.08	0.06	0.02	0.01	0.07	0.03	0.02	0.01	0.11	0.09	0.03	0.01	0.09	0.07	0.02
Ni (mg/L)	0.04	0.29	0.17	0.07	0.02	0.25	0.12	0.06	0.07	0.49	0.34	0.06	0.05	0.35	0.23	0.05

Table 8.2. Summary of the analyzed concentrations of the metal contaminants and major ions in Delhi landfill leachate based on the present study and their comparison with the regulatory standards.

Parameters	Concentration (2004)	Concentration (2005)	Recommended	References
pH	4–13	5–13	4–12	CPCB, 2001
Fe	201	22	–	Not Available
Pb (mg/L)	<2	<2.2	<2	CPCB, 2001
Cd (mg/L)	0.2	0.25	<0.2	CPCB, 2001
Cu (mg/L)	<10	<12	<10	CPCB, 2001
Ni (mg/L)	<3	<3.5	<3	CPCB, 2001
Zn (mg/L)	<10	<11	<10	CPCB, 2001
Mn (mg/L)	20	25	–	Not Available
Na^+ (mg/L)	2640	3150	–	Not Available
Ca^{2+} (mg/L)	687	7031	–	Not Available
Mg^{2+} (mg/L)	151	167	–	Not Available
K^+ (mg/L)	119	1321	–	Not Available
F^- (mg/L)	<50	<54	<50	CPCB, 2001
SO_4^{2-} (mg/L)	3320	33751	–	Not Available
NO_3^- (mg/L)	<30	<451	<30	CPCB, 2001
Cl (mg/L)	4000	4500	–	Not Available
PO_4^{3-}(mg/L)	4	4.4	–	Not Available
HCO_3^- (mg/L)	412	431	–	Not Available

thus indicate anthropogenic influence. Sodium concentration ranges from 62.5 mg/L to 684 mg/L (pre-monsoon) and 45.5 mg/L to 736 mg/L (post-monsoon). In pre-monsoon high concentration of sodium was obtained in comparison to post monsoon due to evaporation during summer season and dilution by rainwater in post-monsoon in groundwater. In pre-monsoon high concentration of potassium was obtained in comparison to winter and post monsoon because of increased concentration in groundwater due to evaporation during summer. Further, Delhi soils have significant amounts of illite, which fixes K from water. Hence ground water in Delhi has low K values.

The concentration of Ca in groundwater was higher than the standard USPH and WHO maximum permissible limit for drinking water (Table 8.1) because of input of calcium salt by anthropogenic source like leaching of pollutant from landfill. In general, high concentration of calcium was obtained during pre-monsoon in comparison to post-monsoon period because of increased concentration in groundwater by evaporation of water in summer and dilution of groundwater in post-monsoon by rainwater later (Rajamohan & Elango 2004). The Mg concentration in groundwater also shows similar trend to Ca because of similar natural origin, but spatial variation in their concentration shows anthropogenic input to groundwater. The high concentration of Mg is probably due to the input of salt by anthropogenic source like leaching of pollutant from landfill. High concentration of magnesium was obtained during pre-monsoon in comparison to post-monsoon period because of evaporation of water in summer season. The lateral variation of magnesium in groundwater shows, high concentration in Bhalswa Dairy region with continuous decrease from landfill along the groundwater flow direction outward. It indicates possible influence of landfill leachate on groundwater quality which is supported by counter lines of temporal and spatial variation of EC all around the landfill.

8.4.4 *Hydrochemical characteristics and patterns of variation*

The concentration of sulphate ion is much higher than the expected value as geology of the study area possess high content of gypsum. As most of the sample have Na/Cl ratio around or above 1 in pre-monsoon and less than 1 in post-monsoon (Kumar et al. 2006), indicating that ion-exchange process is prevalent in the study area (Fig. 8.3a). The plot between ($Ca^{2+} + Mg^{2+}$) and ($SO_4^{2-} + HCO_3^-$) is close to 1:1 equilibrium line if dissolution of calcite, dolomite and gypsum

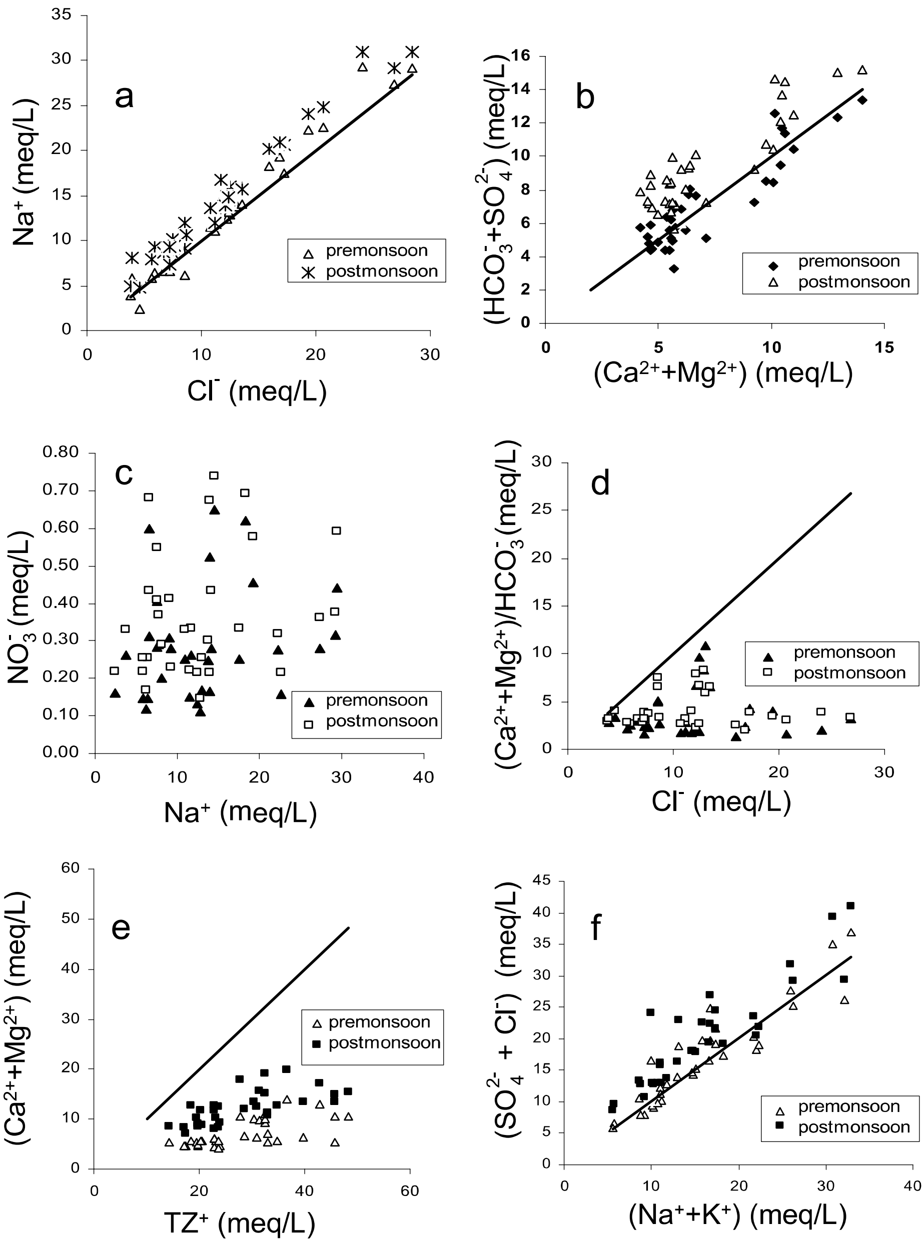

Figure 8.3. Plots of the salient hydrochemical characteristics of groundwater in vicinity of Bhalswa landfill.

are dominant reaction in the system. Ion exchange tends to shift the points towards right due to the excess of ($SO_4^{2-} + HCO_3^-$) ions, which may be derived from the anthropogenic to the groundwater system (Cerling et al. 1989, Fisher & Mulican 1997). The bivariate plot between ($Ca^{2+} + Mg^{2+}$) and ($SO_4^{2-} + HCO_3^-$) shows most of samples falls above the 1:1 ratio line (Kumar et al. 2006), except for few post-monsoon samples which indicate predominance of ion-exchange process in groundwater system as a result of possible leaching of these anions from landfill (Fig. 8.3b). The plot between Na^+ and NO_3^- shows most of the groundwater samples scattered and falls above the equiline 1:1 (Lee et al. 2003, Subba Rao 2002), demonstrating a clear anthropogenic input in groundwater system, because nitrate is released in groundwater mainly through anthropogenic source (Fig. 8.3c). The major source of pollutant in this area is Bhalswa landfill, where possible leaching of nitrate from landfill is justified from the nitrate contour lines around landfill.

The source of Ca and Mg in the groundwater can be deduced from $m(Ca^{2+} + Mg^{2+})/m(HCO_3^-)$ ratio. If Ca and Mg originate solely by dissolution of carbonate in aquifers as well as due to the weathering of accessory minerals like pyroxenes and amphibole where this ratio would be about 0.5 (Sami 1992). As this ratio increase with salinity Ca and Mg releases to the groundwater aquifer is faster than bicarbonate indicating the anthropogenic input to the aquifer system (Kumar et al. 2006) (Fig. 8.3d). The plot between TZ^+ (total cation) vs. $(Ca^{2+} + Mg^{2+})$ shows most of samples far below the theoretical line (1:1) (Fig. 8.3e), depicting an increasing contribution of alkali elements as the dominant major ions in groundwater, due to anthropogenic input from the landfill (Subba Rao & Devadas 2003). While the increase in alkali elements correspond to simultaneous increase in $(Cl^- + SO_4^{2-})$, suggesting a common source of these ions due to the presence of Na_2SO_4 and K_2SO_4 in the soils (Datta et al. 1996), indicate leaching of these ions from landfill leachates (Fig. 8.3f). The dominance of Na^+ an index of weathering suggests that the ions result from dissolution of soil salts or derived from landfill leachates, which also suggests that the higher concentration of alkalis is from sources other than precipitation (Singh & Hasnain 1999). All this indicates significant impact of landfill leachates on hydrogeochemical process occurring in groundwater aquifer systems in vicinity of Bhalswa landfill.

The concentration of fluoride in groundwater is principally governed by climate, the composition of the host rock and hydrogeology (Sujatha 2003). The major source of fluoride in study area is due to the availability of fluorite minerals in bedrock of Aravalli ridge, which is located very close to Bhalswa landfill and passing through midway of the Delhi. The areas with a semiarid climate, crystalline rocks and alkaline soils are mainly affected by fluoride (Handa 1975). This is probably due to higher TDS in groundwater resulting in an increased ionic strength and higher fluorite (CaF_2) solubility in groundwater. Fluorite is mainly the principle bearer of fluoride and is found in granite, gneiss, and pegmatite (Rama Rao 1982). Fluoride is released to the soil and groundwater by the process of weathering of primary rock or leaching of landfill contaminant (Sujatha 2003). When released into the soil and groundwater the fluoride concentration may increase until saturation with fluorite is reached (Handa 1975). The alkaline water can mobilize fluoride from fluorite (CaF_2) with the simultaneous precipitation of $CaCO_3$, because of the solubility of CaF_2 increases with an increase in $NaHCO_3$ rather than with salts NaCl, Na_2SO_4, $MgCl_2$ and $MgSO_4$ (Rammohan Rao et al. 1993, Saxena & Ahmad 2001, 2003).

$$CaF_2 + 2HCO_3^- \longleftrightarrow CaCO_3 + 2F^- + H_2O + CO_2 \tag{8.1}$$

$$CaF_2 + 2NaHCO_3 \longleftrightarrow CaCO_3 + 2Na^+ + 2F^- + H_2O + CO_2 \tag{8.2}$$

Minerals rich in $CaCO_3$ can also favor the dissociation of fluoride from F-bearing minerals, as given below:

$$CaCO_3 + H^+ + 2F^- \longleftrightarrow CaF_2 + HCO_3^- \tag{8.3}$$

$$CaF_2 \longleftrightarrow Ca^{2+} + 2F^- \tag{8.4}$$

$$K = \{a(HCO_3^-)\}/\{a(H^+) * a(F^-)\} \tag{8.5}$$

where, K is an equilibrium constant and a, is the activity. It is evident that if the pH is constant, the activity of fluoride is directly proportional to bicarbonate. The major source of fluoride in Delhi is fluorite and mica bed-rock, which releases by ion exchange process between OH^- ion from water and F^- ion form mica bed (Central Ground Water Board 1995). It is also possible that the evaporation/weathering process would cause an increase in concentrations of all species in groundwater water but the spatial distribution of these species shows that the major source of chloride, nitrate, fluoride and other heavy metals is landfill leachates.

8.4.5 *Spatial distributions*

The iso-concentration plots presented in Figure 8.4, give an idea about spatial distribution contaminant in groundwater and also help in prediction of movement of contaminant plume with the flow of groundwater. In these study contour lines indicate landfill as anthropogenic source of contamination of groundwater. The area around Bhalswa Lake (mainly Bhalswa Village, Burari) is highly polluted. The concentration of all parameter crosses 5 to 10 time of standard value provided by WHO and USPH for drinking water. The area around Badali, Yadav Pur and Shriddhanand Park is highly contaminated.

The spatial and temporal variation of Cl^- and EC around the Bhalswa landfill is shown in Figure 8.4. The variation occurs due to migration of contaminant from landfill leachates indicated by the clustering of contour lines in and around the landfill due to spatial and temporal factors.

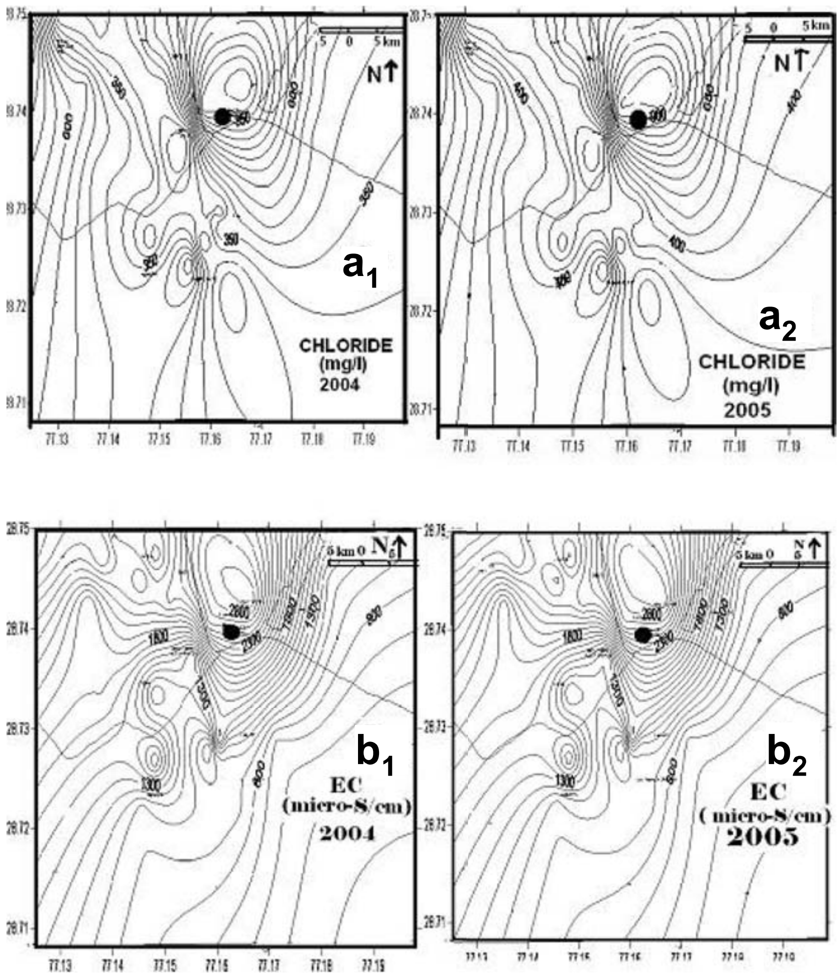

Figure 8.4. Spatial distribution of EC and chloride around Bhalswa landfill.

Few samples in Bhalswa Dairy village showed exceptionally high EC ($\sim$2800 μS/cm), located close to landfill where the leachates rapidly infiltrate down and mix with the groundwater. In general, EC is high in Bhalswa Dairy village, similarly in other areas like Shalimar Village, J.J. Colony, Badali and Ramgarh, groundwater samples also showed comparatively high EC value. Chloride and sulphate concentration is also highest in this region i.e. around 1000 mg/L and 550 mg/L respectively. The contour maps also indicate high concentration around Bhalswa Lake because it is acting as a recipient pool from the leachates from Bhalswa landfill.

The chloride ion is an indicator of anthropogenic inputs in groundwater because there are very less known natural source of chloride. The spatial variation plots (Fig. 8.4a, b) shows high concentration of chloride in Bhalswa Dairy Village, Badli Industrial Area, J.J. Colony and Yadav Nagar. The chloride ion concentration varies with the movement of groundwater. The contour line shows movement of Cl dominant plume occurs along with the flow of groundwater. It also indicates landfill as a point source of contamination (Fig. 8.4a).

The major source of fluoride in Delhi is attributed to the presence of the pegmatite veins comprising mica. In certain wells, especially around Swami Shriddhanand Park, Bhalswa Dairy Village and Burari area, high concentration of fluoride are encountered in the wells. Northwest area of landfill shows high concentration of fluoride in comparisons to southeast because fluoride concentration increases due to fertilizer used in agriculture (Jalali 2005).

The conductivity of groundwater indicate amount of dissolve constituents present in water and used in determining the suitability of water for irrigation, drinking, and industrial uses. Most of the samples show high value of EC due to leaching of pollutant from landfill to subsurface water. The contour line indicate high conductivity in Bhalswa Dairy village and near by area (Fig. 8.4b). These values of conductivity decrease with the distance as we move away from landfill in radius as well as variation with plume of groundwater movement. These variations of conductivity with the distance indicate leaching of pollutant from landfill because conductivity decreases as we move away from landfill.

8.4.6 *Trace elements (Fe, Mn, Ni, Cu, Zn and Pb)*

The major anthropogenic source of Fe in groundwater is steel industry waste (Central Pollution Control Board 2001). The steel industry generally dumped their effluents near by landfill, which contain high concentration of Fe. With time, the Fe seeps in groundwater from landfill with rainwater. The high Fe concentration in untreated ground waters may turn changes its taste and may be rejected by consumer. The permissible level for Fe concentration in drinking water as given by WHO is 0.3 mg/L and by the Bureau of Indian Standards (BIS) is 1.0 mg/L.

The spatial variation of Fe concentration in groundwater around Bhalswa landfill varies between 0.83–6.97 mg/L (Table 8.1). In some areas such as the Badali Industrial Area, Rajasthan Industrial Area and Bhalswa Dairy village, the concentrations of Fe are considerably high due to the seepage of the anoxic landfill leachates to groundwater (Srivastava & Ramanathan 2008). The geochemical environment at the landfill sites are generally complex, where reducing condition is often predominant. The increase indicates the leaching of iron from landfill because iron concentration decreases with the radial distance from landfill. High iron concentration is observed in Bhalswa Dairy village area and it's near by location.

The concentration of Mn in ground water sample varies from 0.02 mg/L to 1.9 mg/L (pre-monsoon) and 0.01 mg/L to 1.61 mg/L (post-monsoon) (see Table 8.1). The spatial variation of Mn, shows possible influence of landfill leachate on groundwater quality as indicated by Srivastava & Ramanathan (2008*) also in there study. The concentration of Mn in all seasons shows higher value in the eastern part of the study area. This may be attributed to anthropogenic activity such as waste discharge from industrial units. In addition, Mn occurs in different forms in air which reacts with SO_2 and NO_2 and dissolved in rainwater causing higher concentration in the study area (Kabata-Pendias & Mukherjee 2007). Riemann & Caritat (1998) reported bulk deposition of Mn in Germany as 30 to 720 g/ha/yr. The aerial pollution of Mn was also reported in mass analysis of particulates in Norway ranging 28 to 2100 mg/kg (Kabata-Pendias & Mukherjee 2007).

The concentration of Ni in ground water varies from 0.04 mg/L to 0.29 mg/L (pre-monsoon) and 0.02 mg/L to 0.25 mg/L (post-monsoon) (Table 8.1). The spatial variations of Ni concentration around the Bhalswa landfill indicate leaching of pollutant from landfill. Sanjay Enclave, Jahagirpuri, Bhalswa Dairy village shows higher concentration in comparison to other area, because they are located very close to landfill indicating the impact of landfill leachate on groundwater quality. Ni concentration is higher in the southern part of study area, mainly samples taken from Rajasthan Industrial area and Badali Industrial area because industrial effluent from these industries anyhow contaminates ground-water may be by leaching from landfill or by industry dumping itself in groundwater.

The concentration of Cu in ground water varies from 0.01 mg/L to 0.08 mg/L (pre-monsoon) and 0.01 mg/L to 0.07 mg/L (post-monsoon) in year 2003–04 (Table 8.1). While, in leachate Cu concentration is around 9.8 mg/L, which indicate leaching of pollutant from landfill to groundwater. The spatial distribution of Cu shows the leaching of Cu as point source of pollutant. It is justified by high concentration of Cu near by landfill area like Bhalswa Dairy village, Jahagir Puri, Badali Industrial Area etc. In general pre-monsoon shows higher concentration in comparison to post monsoon because dilution of groundwater takes place in post monsoon indicating landfill.

The concentration of Zn in ground water samples varies from 0.12 mg/L to 2.27 mg/L (pre-monsoon) and 0.09 mg/L to 3.06 mg/L (post-monsoon) (see Table 8.1). While in landfill leachate Zn concentration is 9.8 mg/L much higher, showing the possibility of leaching of heavy metals from landfill to sub-surface. On seasonal basis, pre-monsoon generally shows high concentration than post-monsoon because of dilution of groundwater during monsoon period.

8.4.7 *Graphical representation of water quality data*

In the present study, Piper (1944) trilinear diagram has been used to decipher the geochemical evolution of groundwaters. The diagram consists of two triangular fields and a central diamond shaped field. In the two triangular fields, the concentrations of major cations and anions are plotted separately and then projected on to the central field for the representation of overall characteristics of water. The cations, anion triangles and central diamond-shaped areas in these diagrams (Fig. 8.5) can be divided into several designated sub area, each of which is assigned on area number codes which represent water of certain geochemical character.

The major ion compositional plots of groundwater around Bhalswa landfill (Fig. 8.5) shows dominance of mainly sulphates, chloride, sodium and calcium that indicate pollution due to anthropogenic sources. Concentrations of all these parameter increased from May 2003 to January 2005,

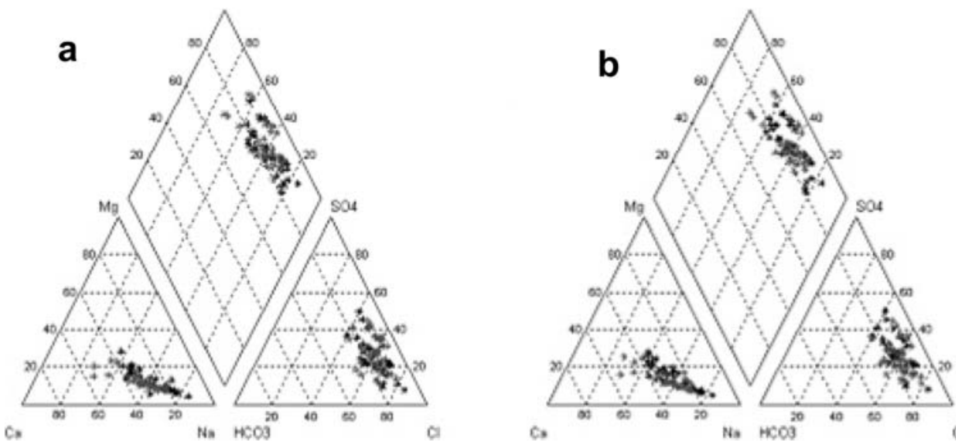

Figure 8.5. Major ion composition of the groundwater from Bhalswa landfill plotted on a Piper diagram (red = pre-monsoon, black = post-monsoon).

which indicates increase in pollution with time (Fig. 8.5). Dominance of sodium and chloride in piper diagram indicates saline nature of groundwater it may be due to leaching of pollutant from landfill. This prediction of contamination of groundwater in vicinity of Bhalswa landfill is justified by contour line drowns of different parameter around landfill (Fig. 8.4).

8.4.8 *Statistical analysis*

The numerical analysis of hydro-geochemical data has been attempted to determine the geochemical parameters of groundwater (Lawrence & Upchurch 1982). Correlation and factor analysis are widely used in statistical or numerical concepts for parametric classification of modeling studies (Balasubramanian et al. 1985). Statistical data generally have better representation than graphical representation because (a) there is a finite number of variable that can be considered (b) variable are generally limited by convention to major ions and (c) superior relation ship may be introduced by use of certain procedure.

8.4.8.1 *Correlation matrix*

The correlation matrix for groundwater samples collected around Bhalswa Landfill is shown in Table 8.3. Good correlation observed between Ca-Cl^-; Ca-Mg; Ca-Na; Ca-SO_4^{2-}; Ca-K; Na-Mg; Mg-Cl^-; Mg-SO_4^{2-}; Ca-F^-; Na-Cl^-; Na-F^-; Na-SO_4^{2-}; Na-K etc shows all of them have same origin source. In all most all seasons, groundwater samples show similar correlation. In general, highly polluted ground-water samples have low oxidation-reduction potential because of reducing atmosphere. Most of sample in vicinity of Bhalswa landfill shows low oxidation-reduction potential; it indicates reducing condition in groundwater in the vicinity of Bhalswa landfill.

8.4.8.2 *Factor analysis*

Factor analysis as applied to widely differing sets of groundwater chemical data, appears to be moderately successful as statistical tool for revealing hydrochemical and hydro geological feature. The aim of the factor analysis of the hydrogeochemical data is to explain the observed relationship in simple terms expressed as new set of data called factors. Factor analysis model is assumed to represent an overall variance of the data set and structure expressed in this pattern of variance and co-variance between the variables and similarities between the observations (Davis 1980). Contribution of a factor is said to be significant when the corresponding eigen value is greater than unity (Briz Kishore & Murali 1992). In general the factor will be related to the largest eigen value and will explain the greatest amount of variance in the data set. The factor analysis of groundwater around Bhalswa landfill was shown in Table 8.4.

Factor I of principle component factor matrix of groundwater around the Bhalswa Landfill is characterized by strong loading of Fe, Na^+, K^+, Cl^-, F^-, HCO_3^-, EC and TDS and account for

Table 8.3. Correlation matrix of Bhalswa landfill.

	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-	HCO_3^-	SO_4^{2-}	NO_3^-	F^-	$H_4SiO_4^-$
Ca^{2+}	1									
Mg^{2+}	0.97	1								
Na^+	0.71	0.65	1							
K^+	0.56	0.61	0.48	1						
Cl^-	0.81	0.74	0.97	0.43	1					
HCO_3^-	0.04	0.02	−0.16	0.52	−0.21	1				
SO_4^{2-}	0.95	0.92	0.60	0.62	0.67	0.23	1			
NO_3^-	−0.33	−0.25	−0.08	0.27	−0.20	0.50	−0.18	1		
F^-	0.74	0.67	0.67	0.10	0.74	−0.38	0.68	−0.39	1	
$H_4SiO_4^-$	0.02	0.03	−0.03	−0.35	0.01	−0.30	0.14	−0.57	0.11	1

Table 8.4. Principle component matrix of hydrochemical parameters.

	Component					
Parameters	1	2	3	4	Initial	Extraction
pH	−0.64	0.10	0.36	0.31	1	0.67
ORP	−0.82	0.32	0.01	0.11	1	0.75
EC	0.88	−0.33	−0.001	−0.11	1	0.84
TDS	0.88	−0.33	−0.003	−0.1	1	0.81
Na^+	0.74	0.35	0.25	0.28	1	0.66
K^+	0.79	−0.04	−0.06	0.11	1	0.82
Mg^{2+}	0.25	0.52	−0.65	0.23	1	0.87
Ca^{2+}	0.31	0.77	−0.34	0.26	1	0.82
Mn	0.31	0.47	0.57	−0.42	1	0.77
Fe	0.70	0.43	0.13	−0.26	1	0.83
Zn	0.87	0.18	0.19	−0.06	1	0.67
Cu	0.27	−0.69	−0.34	0.02	1	0.81
Ni	0.02	0.17	0.87	0.19	1	0.83
F^-	0.81	−0.29	−0.24	−0.17	1	0.82
Cl^-	0.78	0.38	0.18	0.18	1	0.90
SO_4^{2-}	0.17	0.84	−0.39	−0.03	1	0.83
NO_3^-	0.31	−0.48	0.12	0.70	1	0.67
HCO_3^-	0.77	−0.23	0.11	0.10	1	0.67
Total	6.60	3.35	2.24	1.18		
% of Variance	38.80	19.72	13.20	6.92		
Cumulative %	38.80	58.52	71.72	78.64		

38.8% of the total variance. Strong loading of Na^+ and K^+ indicate natural weathering of rock mineral and various ion-exchange processes in groundwater system in vicinity of landfill (Drever, 1997). While strong loading of chloride and iron with alkali metals indicates anthropogenic input in groundwater system, it may be due to leaching of industrial effluents from landfill. The strong loading of HCO_3^- ions, with alkali and alkaline earth metals are support the view of natural weathering sources. The high variance in Na^+, K^+ and F^- indicate anthropogenic input in aquifer system (Mahlknecht et al. 2004). Since there were no known natural source of fluoride in Bhalswa landfills. It was justified by spatial distribution of F^- ion around landfill (Table 8.1). Factor II of principal component factor matrix of groundwater around Bhalswa landfill is characterized with strong loading of Ca^{2+}, Mg^{2+} and SO_4^{2-} ions accounts for 19.72% of total variance. The strong loadings of Ca^{2+} and SO_4^{2-} indicate weathering of gypsum which is supposed to be available in geology of Delhi. Factor III is characterized by strong loading of Nickel with an account for 13.2%.

The strong loading of Ni indicate anthropogenic input in groundwater system because there are no known natural source of Ni in the study area. Factor IV is characterized by strong loading NO_3^- ions indicate anthropogenic input in groundwater system it may be due to leaching of fertilizer from agriculture land (Mahlknecht et al. 2004) or from leaching municipal landfill. While spatial distribution of NO_3^- around landfill indicates possible leaching of NO_3^- from landfill. Factor V indicates groundwater Chemistry is controlled by the pH variation in the aquifer system.

8.5 CONCLUSIONS

Around 15 km^2 area was selected for this study and it covers around 2 km radius with center Bhalswa landfill. All sampling site were chosen such a way that it gives maximum representation of study area. Since Bhalswa landfill is located in most urbanized area of Delhi, it influences a huge population of the Capital of India. Bhalswa landfill is unplanned, in flood plain of Yamuna and its geology makes it more susceptible for contamination by leaching of pollutant.

The studies indicate high contamination of groundwater due to leaching of pollutant from landfill with temporal and spatial variation of groundwater quality around landfill, that is justified by statistical interpretation. Most of samples reported have high conductivity, high trace metal concentration, chloride, nitrate and other cations/anions indicates anthropogenic input i.e. by leaching of pollutant from landfill. Since pollutant concentration decrease in groundwater flow direction, along the radius as we move away from landfill indicates Bhalswa landfill act as a point source of contamination. The various graphical plots and statistical analysis have been applied for the chemical data based on the ionic constituents, water types, and hydrochemical facies to deduct the impact of landfill on groundwater quality. The statistical analysis, spatial and temporal variations indicate leaching of contaminant from landfill to groundwater aquifer system. The concentrations of trace metals in leachate are much higher than the standards.

The study suggests that landfill influenced the groundwater chemistry of Delhi aquifer system, it is high time to check it and plan for a possible alternative site. The site developed for future dumping should have planned engineering structure to control/minimize the impact of Landfill leachate on ground water quality around the landfill site.

ACKNOWLEDGEMENTS

SKS would like to thank University Grant Commission of India, for providing the research grants in the form of a Junior Research Fellowship. We deeply appreciate the help we have received from the Central Pollution Control Board, Central Ground Water Board, Survey of India and Indian Meteorological Department for necessary data whenever required. The author ALR is thankful to MOWR for the support in the form of Project. We would thank to Dr. Prosun Bhattacharya, KTH, Sweden, for his critical reviews on the earlier drafts of this manuscript. At last, we would like to thanks all coworkers for providing necessary assistance during the study at the School of Environmental Sciences, Jawaharlal Nehru University, New Delhi, India.

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CHAPTER 9

Assessment of the potential hazards of nitrate contamination in surface and groundwater systems in Hooghly district of West Bengal, India

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ABSTRACT: We made an inventory of nitrate (NO_3-N) enrichment in surface and groundwater systems in the Hooghly district of West Bengal, India owing to intensive farming with high fertilizer doses as a function of quantity of fertilizers use, soil characteristics, types of crop grown, depth of groundwater sampling and also N-load in soil profiles. Water samples were collected from different sources at 70 odd sites spread over in 17 blocks of the district along with representative soil profiles. The NO_3-N content both in surface and groundwater varied from 0.10 mg/L to 3.24 mg/L, being well below the threshold limit of 10 mg/L fixed by WHO for drinking purpose. The content decreased with increasing depth of wells ($r = -0.29^*$) and clay content of soil profiles ($r = -0.51^*$). It, however, increased with increasing rate of fertilizer application ($r = 0.91^{**}$), NO_3-N load in soil profiles ($r = 0.509^*$) and was higher in areas where shallow—rather than deep-rooted crops are grown.

9.1 INTRODUCTION

Among the fertilizers applied to the soil for increasing crop production, nitrogenous ones are most important. In arable soils, these are rapidly converted to nitrate (NO_3) form, which are readily available to plants, but are highly soluble and hence easily leachable. When quantity of nitrogen added to the soil exceeds the amount that the plants can use, the excess NO_3 does not get much adsorbed by soil particles, leaches out from the root zone by water percolating through the soil profile and ultimately accumulates into the groundwater. The magnitude of such leaching, however, depends upon soil characteristics, types of crop grown, management practices followed etc. It also goes to the surface water by runoff and causes eutrophication. These result in an increase in the level of nitrate in the drinking water, which causes harmful biological effects. A high level of NO_3 (>45 mg/L or >10 mg/L NO_3-N) in the drinking water can cause methemoglobinemia or blue baby syndrome in infants and gastrointestinal cancer in adults. Although a lot of attention has already been paid world over particularly in the developed countries on this phenomenon, in developing countries like India its importance is being felt recently because of sporadic reports of NO_3 enrichment in groundwater in a few agriculturally intensive areas in Punjab (Singh et al. 1995, Bajwa et al. 1993), Haryana (Singh & Singh 2004), Delhi (Gupta & Chandrasekharan 1999), Maharashtra (Deshpande et al. 1999), Andhra Pradesh (Srinivas Rao 1998), West Bengal (Kar et al. 2003). They have attributed the high NO_3-N in groundwater to high nitrogenous fertilizer use. However, so far there has not been any systematic attempt to find out the extent of nitrate in different soil and water resources and to quantify the contribution of the fertilizer sources towards nitrate contamination of aquifers in India. Recently, the Indian Council of Agricultural Research has initiated a systematic study through a network project for the assessment of potential nitrate contamination hazard in surface and groundwater systems in the heavily fertilized and intensively cultivated districts of India. The present work is a part of the study restricted to Hooghly district of West Bengal, one of the most heavily fertilized and intensively cultivated districts of India. Attempts have been made here to relate NO_3-N content in water sources particularly groundwater with soil characteristics, N-load in soil profiles, types of crop grown in cropping system and quantity of fertilizer applied for raising the crops.

9.2 BACKGROUND OF THE DISTRICT

9.2.1 *Physiography and geology*

The district forms a part of Bengal basin developed by the riverine deposits of Ganga and its tributaries. The principal rivers Hugli, Damadar and Dwarakeswar with their tributaries (i.e., Behula, Kunti, Saraswati, Kana Damadar, Mundeswari, Jhumjhumi, Amodar and Tarajoli) have a gradual descent from north-west to south and south-east almost parallel to each other following the natural trend of the landscape. According to genesis and evolution of land forms, the district can broadly be divided into two division viz., i) old alluvial plain occurring in the west of river Dwarakeswar and ii) the monotonous level alluvial plain in the east which is further divided into a) natural leeve, b) meander flood plain and c) alluvial plain. The entire district is built up with mixed alluvium. The eastern parts are clayey, stiff and deep; while the western parts are loamy, permeable and easily friable.

9.2.2 *Climate and drainage*

The district has humid subtropical climate with an average rainfall of 1599 mm, being more in the eastern part than in the western. Majority portion of the rains ($\sim$80%) is received during monsoon season. The soil temperature regime is hyperthermic. The mean annual air temperature is 26°C. The drainage pattern is dendritic, in general. Owing to imperceptible slopes, the area thus is partly well drained and partly under-drained. Moderately well drained soils occupy about 45.4% area of the district followed by well drained (38.3%), imperfectly drained (13.2%) and poorly drained (3.1%) soils.

9.2.3 *Soil type*

The soils of the district belong to 3 orders, with Inceptisols covering 64.4%, Entisols 23.1% and Alfisols 12.4% of the total area. They are dominantly loam in texture (53.9%) followed by silty clay (23.3%), silty clay loam (22.1%) and sandy loam (0.71%) in nature. Surface soils of 72% area of the district are acidic, followed by slightly acidic (14.8%), neutral (8.9%) and mildly alkaline (4.9%). Organic carbon status is usually low to medium.

9.2.4 *Land use*

It is one of the most intensively cultivated with high agricultural productivity districts of India. The cropping intensity is as high as 215%. Rice is the principal crop in all the blocks of the district, followed by potato, jute, vegetables and orchard crops. Rice and potato are grown in almost 59.2% and 20.8% of the gross cropped area followed by jute (6.7%) and wheat (3.2%). Banana is the most important plantation crop grown in the district. There is, however, little forest (0.3%) and pasture land in the district. Land use with all these crops necessitated consumption of increased rate of fertilizers.

9.2.5 *Fertilizer use*

As mentioned earlier, it is one of the highest fertilizer consuming districts in India. The growth of fertilizer consumption per ha per year during the last ten years is presented in Table 9.1. It is observed that as much as 277.7 kg of fertilizer is applied per ha per year in the district vis-à-vis 128.0 and 92.0 kg per ha for the state of West Bengal and India respectively. Of the applied amount, nitrogenous fertilizer constitutes about 49.4%. Of late there is a surge in consumption of nitrogenous fertilizer in the district. Such huge application of N for a longer period may lead to contamination of NO_3 in groundwater.

Table 9.1. Average fertilizer (NPK) consumption in Hooghly district during the years 1995–2005.

Year	N (kg/ha)	P (kg/ha)	K (kg/ha)	Total fertilizer (kg/ha)
1995–96	110.2	56.9	48.9	216.0
1996–97	116.8	56.8	52.0	225.6
1997–98	117.2	51.4	37.0	205.6
1998–99	135.0	56.5	40.7	232.2
1999–00	135.7	65.8	47.5	249.1
2000–01	98.9	91.1	60.0	250.0
2001–02	103.6	104.0	72.3	279.8
2002–03	115.0	120.3	78.5	313.7
2003–04	118.6	119.2	77.4	315.2
2004–05	137.1	78.5	62.1	277.7
Range	98.9–137.1	51.4–120.3	40.7–78.5	205.6–315.2
Mean	118.8	80.1	57.6	256.5
SD	13.3	26.8	14.9	38.8

9.3 MATERIALS AND METHODS

Water samples were collected during the pre-monsoon season of 2005 from ponds, khals, canals etc. (for surface water) and from dugged wells, shallow, mini deep and deep tube wells at 70 odd sites spread over 17 blocks of the district following appropriate statistical design (sampling intensity increases with increasing rate of fertilizer use). While selecting the wells, priority was given to those, which are used as sources of water both for irrigation and drinking purposes. Samples were collected in acid washed polythene bottle after allowing the wells to flow for a few minutes. One mL of purified H_2SO_4 was added to each polythene bottle before collecting the samples to bring the sample pH down to 2.0 or less for preserving it for determination of NO_3-N in the laboratory. pH and EC values of the samples were measured *in situ* immediately after collection; while NO_3-N was measured in the laboratory following phenol disulphonic acid method with the help of a spectrophotometer. Horizon-wise (0–20, 20–40 and 40–60 cm) soil samples were also collected from representative sites (from where water samples collected) for their analyses for clay, sand and silt, organic carbon, pH, total N and NO_3-N, to ascribe the possible causes for variations in the observed concentration of NO_3 in the water samples. Data on cropping pattern, rainfall, quantity of fertilizer and irrigation water used and depth of aquifers sampled were also collected. Results thus obtained were analysed following appropriate statistical methods using the software SPSS 10.0.

9.4 RESULTS AND DISCUSSION

Analysis of twenty five representative soil profiles of the district showed an increasing clay but decreasing organic C and NO_3-N and total-N content along depth. However, silt content and EC values recorded little variations. In general, average clay and NO_3-N contents were higher in these soil profiles (Table 9.2). Analysis of both the surface and groundwater showed that pH and EC of the samples ranged from 6.2 to 8.6 and 0.011 to 0.824 dS/m respectively (Table 9.3). On an average, the values of pH and EC were higher for the surface (7.3 and 0.54 dS/m) than for the groundwater (7.1 and 0.23 dS/m).

The NO_3-N content both in surface and groundwater also varied significantly in different blocks of the district, the magnitude being 0.27 to 3.12 and 0.10 to 3.24 mg/L with mean values of 0.96 and 0.81 mg/L respectively. This indicated that the content was higher in the surface than in the groundwater. It was also observed (Figure 9.1) that the average NO_3-N concentration in water

Table 9.2. Physicochemical characteristics of a few soil profiles of Hooghly district.

Properties/Layer	0–20 cm Range	Mean	SD	20–40 cm Range	Mean	SD	40–60 cm Range	Mean	SD
Sand (%)	1.2–50.6	17.6	16.1	1.4–44.9	14.6	13.8	1.7–48.6	14.7	13.4
Silt (%)	18.0–77.8	47.8	18.2	17.0–81.7	46.3	17.4	17.1–72.1	43.6	13.2
Clay (%)	10.0–75.5	34.6	20.8	10.9–76.2	39.1	19.9	10.0–75.7	41.8	17.1
pH	4.5–7.9	6.4	0.9	5.8–7.7	7.1	0.6	6.1–7.8	7.1	0.6
EC (dS/m)	0.09–0.62	0.29	0.15	0.04–0.33	0.20	0.09	0.02–0.88	0.23	0.21
Organic C (%)	0.23–1.29	0.73	0.26	0.10–0.50	0.32	0.13	0.10–0.59	0.30	0.14
NO_3-N (mg/L)	3.22–44.15	22.64	11.8	1.48–44.8	9.23	9.85	2.73–17.7	7.46	3.62
Total N (%)	0.08–0.22	0.14	0.05	0.05–0.21	0.12	0.05	0.04–0.18	0.09	0.05

Table 9.3. Average pH, EC and NO_3-N contents in water from different sources.

Properties	Surface water Range	Mean	SD	Groundwater Range	Mean	SD
pH	6.3–8.6	7.32	0.48	6.2–7.2	7.14	0.30
EC (dS/m)	0.011–0.696	0.543	0.193	0.019–0.824	0.230	0.226
NO_3-N (mg/L)	0.27–3.12	0.96	0.65	0.10–3.24	0.81	0.72

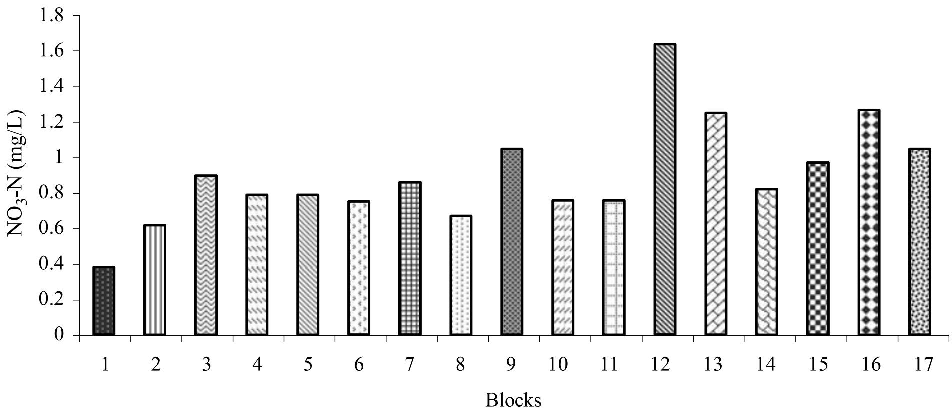

Figure 9.1. Average concentration of NO_3-N in water collected from different blocks. 1 = Chinsura-Mogra, 2 = Balagar, 3 = Pundua, 4 = Polbadadpur, 5 = Serampore, 6 = Chanditala-I, 7 = Chanditala-II, 8 = Singur, 9 = Haripal, 10 = Goghat-I, 11 = Goghat-II, 12 = Khanakul-I, 13 = Khanakul-II, 14 = Arambag, 15 = Tarakeshwar, 16 = Pursura, 17 = Dhaniakhali.

of different sources from only five blocks viz., Khanakul-I (1.64 mg/L), Pursura (1.27 mg/L), Khanakul-II (1.25 mg/L), Dhaniakhali (1.05 mg/L) and Haripal (1.05 mg/L) was greater than 1.0 mg/L. The samples from the Chinsura-Mogra, on the other hand, contained the lowest amount (0.38 mg/L) followed by Balagar (0.62 mg/L), Singur (0.67 mg/L) and Chanditala-I (0.75 mg/L). A relatively higher concentration of NO_3-N in groundwater of the above mentioned five blocks was possibly related to the intensive potato cultivation there with liberal and heavy application of fertilizers, particularly nitrogenous ones.

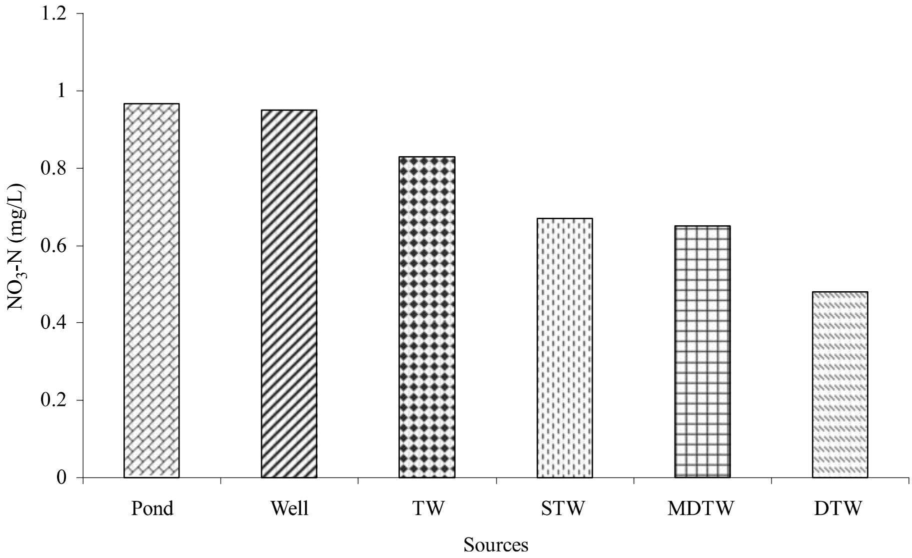

Figure 9.2. Average NO_3-N concentration in water from different sources. Abbreviations: TW = Tube well; STW = Shallow-tube well; MDTW = Mini-deep tube well; DTW = Deep-tube well.

Among the sources the NO_3-N concentration was highest in water from dugged-well followed by: tube-well > shallow-tube well > mini-deep tube well > deep-tube well in decreasing order (Figure 9.2), the different wells, however, having different depths. Average depth of the wells were as follows: deep-tube well (115 m) > mini-deep tube well (100 m) > shallow-tube well (65 m) > tube-well (60 m) > dugged well (25 m). Results thus indicated that the concentration of NO_3-N decreased with increasing depth of the wells from which sampling was done. On an average, wells having depths >90 and <30 m had NO_3-N content of 0.48 and 0.95 mg/L respectively. Statistical analysis of the data showed a significant negative correlation (−0.29*) between NO_3-N content in water and depth of sampling aquifers. This is a typical feature in nitrate contamination of groundwater observed worldwide (Power & Saikh 1995).

Out of the 108 samples analyzed, only 3.7% contained NO_3-N higher than 3.0 mg/L, 9.3% higher than 2.0 mg/L and 26.8% higher than 1.0 mg/L. Results thus indicated that all the water samples contained NO_3-N below the 10 mg/L, the threshold limit fixed by WHO for drinking purpose. As such presently there is no risk involved in drinking water from underground aquifers of the district. Similar low NO_3-N content in groundwater samples from an adjoining agriculturally developed district (Nadia) in West Bengal was also reported by Kar et al. (2003).

An attempt was made to find out relationship between the concentration of NO_3-N in water and the rate of application of fertilizers, especially nitrogenous ones. It was observed that the maximum (3.1 mg/L) and minimum (0.23 mg/L) mean values of the NO_3-N in water occurred in samples from those sites associated with fertilizer application rate >300 kg/ha and <100 kg/ha respectively. This was corroborated by the existence of a significant positive correlation ($r = 0.91^{**}$) between the concentration of NO_3-N in groundwater and the rate of fertilizer application (Figure 9.3). Such relation indicated an overriding influence of applied fertilizer for increasing NO_3-N concentration in groundwater.

The concentration of NO_3-N in groundwater was also found to be related with the type of the crops grown in different locality. It was observed that this was higher in areas where shallow rooted crops viz., rice, potato, onion etc. are prevalent than in the areas where deep-rooted crops viz., jute, sesame, brinjal, cabbage etc. are grown. For example, in Chinsura-Mogra, Balagar, Singur, Chanditala-I, Serampore etc. blocks, where jute, sesame, brinjal, cabbage are an integral part of

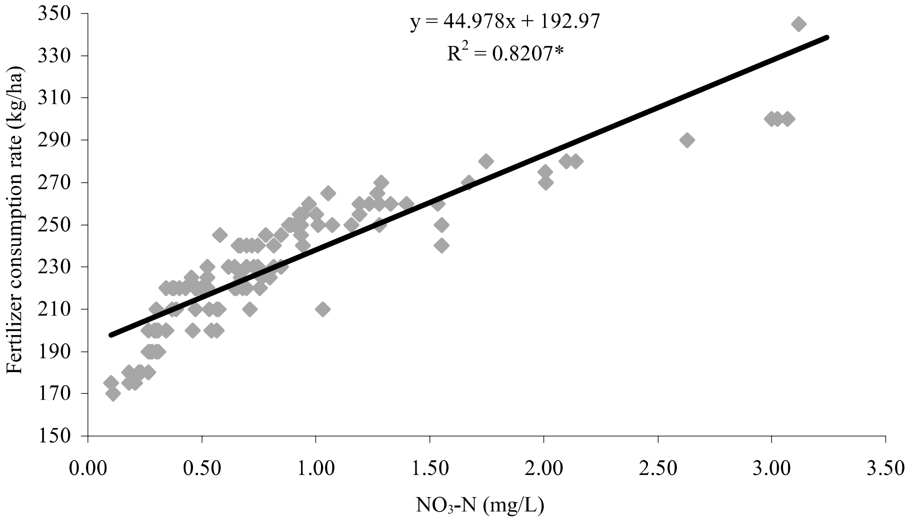

Figure 9.3. Relationship between fertilizer consumption rate and NO_3-N content in water.

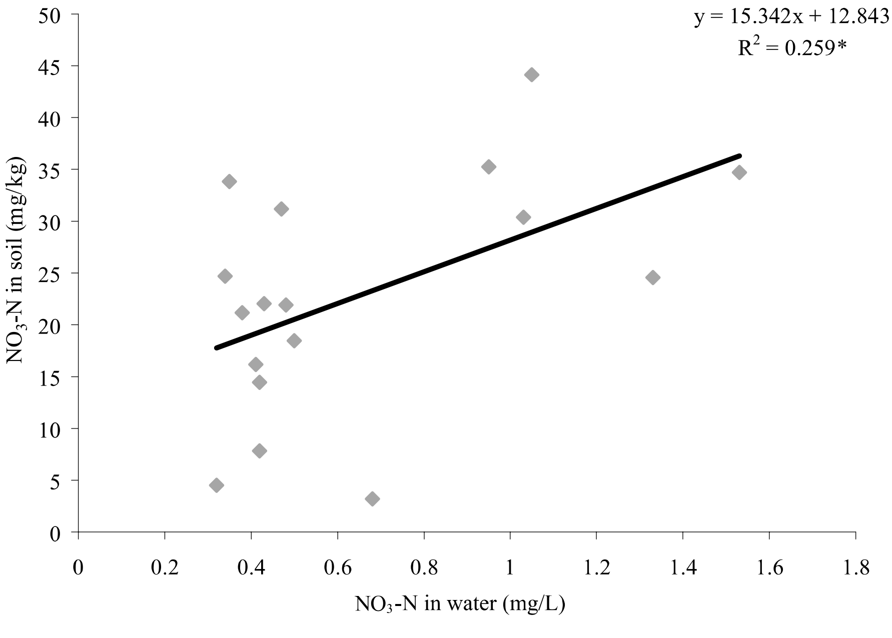

Figure 9.4. Relationship between NO_3-N load in soil profiles and NO_3-N content in water.

the existing cropping systems have a low NO_3-N concentration in their groundwater. Contrarily, in Khanakul-I, Pursura, Khanakul-II, Dhaniakhali, Haripal, Tarakeswar, Pundua, Arambag, Chanditala-II blocks, where shallow rooted crops are prevalent, have comparatively high concentration of NO_3 in their groundwater. Bajwa et al. (1993) also reported that under shallow rooted heavily fertilized vegetable crops, nitrate losses as a result of leaching could be very large.

The concentration of NO_3-N was also found to be lower in groundwater in areas where lowland rice is cultivated vis-à-vis the other crops. This was possibly due to increased denitrification under the anaerobic condition and decreased leaching of nitrate under rice cultivation with puddling.

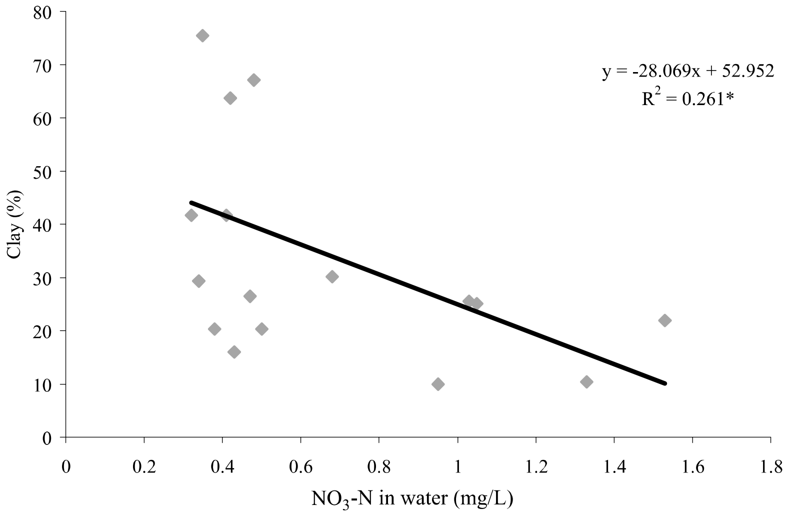

Figure 9.5. Relationship between clay content in soil profiles and NO_3-N content in water.

Ladha et al. (2005) also opined that in lowland rice fields with fine-textured soils, leaching loss of N are low because of restricted percolation, whereas losses from coarse-textured permeable soils can be substantial.

Nitrate loading in groundwater depends not only on the quantity of nitrogen fertilizer applied to soil but also heavily on soil characteristics. Nitrate mobility along a soil column to groundwater is, in fact, a function of both soil quality and quantity of NO_3 input in soil profiles. Accordingly, relationships were computed between NO_3-N content in groundwater and different soil properties analysed. Results showed significant positive correlation between NO_3-N content in groundwater and the total ($r = 0.574^*$) as well as NO_3-N ($r = 0.509^*$) (Figure 9.4) load in soil profiles. Obviously these profile N ultimately leached out of soil and accumulated in groundwater beneath. Clay content in soil profiles, on the other hand, had an inverse relationship ($r = -0.511^*$) with NO_3 content in groundwater (Figure 9.5). This was due to the impedance caused by high clay to NO_3-laden percolating water through soil profiles to underground aquifers. If the soil is clayey, such mobilization is restricted vis-à-vis a light textured loamy soil (Kar et al. 2003); while soils with high NO_3 load in profiles facilitate NO_3 enrichment in groundwater. The NO_3-N concentration in water also showed a significant negative correlation with pH (-0.22^*) and EC (-0.24^*) values of the water.

9.5 CONCLUSIONS

Results indicated that in spite of use of fairly high quantity of fertilizer (Indian standard) the NO_3-N content in groundwater of Hooghly district of West Bengal is well below the permissible limit of WHO (10 mg/L) for drinking purpose. There is, however, an indication of its possible build up in shallow aquifers where the soils are light in texture and shallow rooted crops (particularly potato) are grown with high nitrogenous fertilizer.

ACKNOWLEDGEMENTS

Authors are thankful to Indian Council of Agricultural Research, New Delhi for funding the research. We are extremely grateful to Dr. L.N. Mandal, former Professor of Soil Science, Bidhan

Chandra Krishi Viswavidyalaya, West Bengal for his valuable suggestions on this manuscript. The authors are grateful to Dr. A.B. Mukherjee and Dr. P. Bhattacharya for the editorial handling of the manuscript.

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CHAPTER 10

Pesticides as water pollutants

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ABSTRACT: Pesticides stand out as one of the major development of modern agriculture. They are widely used on crops to increase yields, save energy and labour, and make crop production efficient and profitable. However, their use and or misuse may lead to serious water quality problems that could impair the use of water for crop and animal production or even human consumption. The impact of agricultural chemicals on surface water and ground water quality has become an issue of national importance. Fish kills, reproductive failure of birds, and acute illness in people have been all attributed to the ingestion of pesticides or exposure to pesticides usually as the result of misapplication, careless storage and careless disposal of unused pesticides and pesticide containers. In addition to potential health and environmental threats, pesticide losses from fields and contamination of non-target sites (such as surface water and ground water) represent a monetary loss to farmers. This paper reports recent data on pesticide residues in surface water, ground water and wastewater.

10.1 INTRODUCTION

In recent decades there has been a radical evolution in agronomic practice in many regions of the world associated with (largely successful) attempts to increase agricultural productivity. The intensification of production from agricultural land is being sustained by the application of over-increasing quantities of inorganic fertilizers and a wide spectrum of synthetic pesticides.

Good quality water is a fundamental requirement for human health and survival. Various studies have indicated the deterioration of water quality over the years due to various human activities such as sewage effluents, municipal solid wastes, unplanned disposal of untreated industrial effluents etc. Quality of water used for human consumption is one of great importance as it significantly affects the human health and various constituents when present in more than certain specified limits have considerable health hazards. It is, therefore, imperative to assess the water pollution levels and pollution sources for sustainable development and conservation of this source.

Existing knowledge indicates that agricultural operations can contribute to water quality deterioration through the release of several materials into water: sediments, pesticides, animal manures, fertilizers and other sources of inorganic and organic matter. Many of these pollutants reach surface and groundwater resources through widespread runoff and percolation and, hence, are called “non-point” sources of pollution. Identification and quantification and control of non-point pollution remain relatively difficult tasks as compared to those of “point” sources of pollution.

A recent study by Baloch & Haseeb (1995) indicate that the total loss of crops due to the pests range up to 50% in Pakistan. In order to protect agricultural produce from the ravage of pests, the use of pesticides was thought to be the only solution. Hence, pesticide consumption have risen many fold during the two decades. The dependency on pesticides is evident from the increasing trend in consumption from 665 metric tons in 1980 to 48592 metric tons (as in 2004) mentioned in Agricultural Statistics of Pakistan (2003–2004). Due to such tremendous use pesticides can find their way into rivers, lakes and groundwater through runoff, run-in, leaching through macropores, direct spillage and wind drift causing environmental pollution and it also affect the ecology of man. Pesticide residues in drinking and groundwater of developed countries have been reported in

Agricultural Resources (1987). In Pakistan pesticide residues have been reported in cattle drinking water in Karachi by the authors Perveen & Masud (1988), shallow groundwater in Samundari, Faisalabad reported by Ali & Jabbar (1990), waste water samples from Lahore studied by Tropical Research Institute Annual Report (1993) and groundwater of Mardan mentioned by Ahad et al. (2000) and cotton-belt area of Multan Divisions studied by Ahad et al. (2001). But there is hardly any report on the surface and groundwater of rice and vegetable belts. The present research work was undertaken to analyze surface and groundwater collected from cotton, rice and vegetable belts and from different drainages.

10.2 MATERIALS AND METHODS

10.2.1 *Water sampling*

Surface (canal and field water channels) and groundwater (tube wells, hand pumps, wells) samples were collected in Pyrex glass bottles (capacity 1 L) randomly from cotton, rice and vegetable belts in triplicate and each sample was spiked with 1 mL of dichloromethane on-site in order to avoid biological degradation. The sealed bottles were stored in a container filled with ice and transported to the laboratory for further processing.

10.2.2 *Laboratory methods*

10.2.2.1 *Chemicals and reagents*

Reference pesticide standards were purchased from Dr. Ehrenstorfer, Germany through IAEA, Vienna, Austria. n-Hexane, acetone, methanol, ethyl acetate, sodium sulphate anhydrous, analytical grade or better, were obtained from Merck Marker, Lahore, Pakistan. Glass fibre filters (Whatman 934-AH) and Baker SPE-10 manifold (J.T. Baker, USA) were gifts of Dr. Charles S. Helling USDA, Beltsville MD, USA to the Pesticide Chemistry Lab., NIAB, Faisalabad. LiChrolute® C_{18} Si-60 cartridges (6 mL, packed with 500 mg) Merck, Germany were purchased from local market. Double distilled water was of Milli-Q grade (Millipore, Bedford, MA, USA). All solvents were freshly glass redistilled before use.

10.2.2.2 *Sample preparation*

All water and wastewater samples were filtered using glass fibre filters to remove the macro-particles and pesticide residues were extracted using solid-phase extraction technique used by Tanabe et al. (2000). The cartridges were conditioned with 5 mL of each n-hexane, acetone, methanol and bi-distilled water in sequence and then the filtered water samples (1000 mL) were passed under vacuum (10 mm Hg) at a flow rate of 5 mL/min. The cartridges were dried for one hour at maximum vacuum (30 mm Hg). The target compounds were eluted with acetone, n-hexane and ethyl acetate. The pooled volume of solvents was concentrated up to dryness under a stream of nitrogen gas and re-dissolved in n-hexane and stored at −4°C for further analyses.

10.2.2.3 *Gas chromatographic analysis*

A gas chromatograph, Fisons GC 8000 Series, Model GC 8160-00DPFC equipped with 63 Ni electron capture detector (ECD), splitless injector, capillary column SE-54 (methyl silicone, 25 m, 0.32 mm inner diameter (id), 2.0 mm film thickness) and Turbochrom hardware/software system was used for the analyses of chromatographic data on the pesticide residues in drinking water and waste water samples.

The gas chromatographic system was calibrated with reference pesticide standards to get optimum sensitivity of the electron capture detector. All samples extracted from solid-phase extraction cartridges were analyzed using a gas chromatograph equipped with 63 Ni electron capture detector, capillary column and Turbochrom hardware/software system. The solvent flush injection technique was used. The temperature programme was: detector temperature 300°C, oven temperature 80°C

(1 min), 20°C/min to 190°C and then 4°C/min to 280°C (15 min). The unknown components were identified on the basis of relative retention times and quantified on peak area basis using software reported by Miller & Miller (1980).

10.3 RESULTS AND DISCUSSION

The chromatogram of standard pesticides is given in Figure 10.1. Limit of detection (LOD) of the standard solutions was found in the range of 1 to 20 μg/L. Recovery of different insecticides at different spiking levels (0.05, 0.01 and 0.1 μg/L) is given in Table 10.1.

The data presented in Table 10.1 showed a linear behaviour to each standard compounds and the signal of detector showed least noise at high sensitivity. The regression coefficient was in the range of 0.989–0.995. The recovery percentage was acceptable and the behaviour of ODS cartridge for the retention and elution of pesticide indicated that the used material and amount is efficient to extract the pesticide residues from surface, ground and wastewater samples.

The recovery % results presented in Table 10.1 are much better than the reported results (7 and 11) because the author used dichloromethane as extraction solvent from water samples and recovery of the most studied insecticides was in the range of 11–88% at 1.0 mg/L spiking level. The samples collected from rice, cotton, vegetable belts and waste water from different waste channels showed the presence of multiresidues of organochlorines, organophosphates, carbamates and pyrethroids. Tables 10.2–10.5 summarize the results of the analyses of the pesticide residues detected with GC-ECD using capillary column.

Data presented in Tables 10.2–10.5, indicate that all sources of drinking water and wastewater were contaminated with variety of pesticide residues. The water samples of cotton belt contained higher concentration of pesticide residues as compared to the rice and vegetable belts. Our findings are in accordance with the reported results of Crutchfield et al. (1992) and Tian et al. (1994) and they had explained that intensive spray of insecticides in cotton area might contribute water quality problem due to run-off or leaching and pose health risk to the ultimate users.

The well water and canal water samples found more contaminated as compared to hand pump and tube well. Higher residue concentration for individual insecticides, especially α-Endosulfan, methamidophos and carbosulfan were found in canal and well waters as compared to tube well and hand pump. This may be due to the wind drift during application and run-off around the wells from where the samples were taken. Lindane, dimethoate, parathion-methyl, imidacloprid

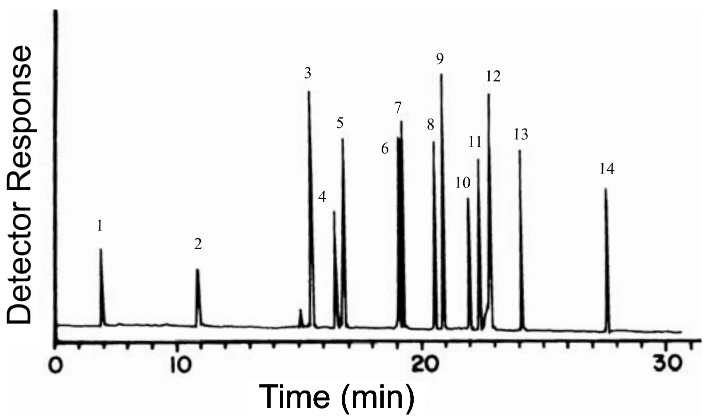

Figure 10.1. Chromatogram of standard pesticides (1 α-Endosulfan, 2 p, p′-DDT, 3 dieldrin, 4 methami dophos, 5 chlorpyriphos, 6 monocrotophos, 7 carbofuran, 8 oxamyl, 9 diuron, 10 cypermethrin, 11 deltamethrin, 12 thiophanate-methyl, 13 carbendazim, 14 imidacloprid).

Table 10.1. Recovery[1] of pesticides (in %) from double distilled water.

Pesticides	Spiking level 0.05 µg/L	0.01 µg/L	0.1 µg/L
α-Endosulfan	92	85	95
p, p′-DDT	89	90	94
Lindane	93	94	93
Methamidophos	94	99	96
Dimethoate	93	89	94
Monocrotophos	95	101	95
Parathion-methyl	96	98	94
Carbofuran	92	92	95
Methomyl	96	99	96
Carbosulfan	97	100	93
Cypermethrin	90	88	92
Deltamethrin	93	92	90
Cyhalothrin	91	90	95
Captan	91	96	91
Thiophanate-methyl	92	91	90
Carbendazim	94	95	93

[1] The reported values are mean of 9 replicates.

Table 10.2. Pesticide residues* (µg/L) in surface and ground waters collected from cotton belt.

Pesticides	Canal water	Hand pump water	Tube well water	Well water
α-Endosulfan	0.13–0.16	0.09–0.13	0.03–0.05	0.13–0.15
p, p′-DDT	0.07–0.18	0.09–0.16	0.04–0.05	0.06–0.10
Lindane	0.05–0.07	0.08–0.22	Traces	0.07–0.09
Methamidophos	0.013–0.025	0.011–0.016	0.009–0.012	0.10–0.15
Dimethoate	0.005–0.009	0.003–0.090	0.002–0.004	0.05–0.09
Monocrotophos	0.09–0.14	0.05–0.17	0.004–0.09	0.08–0.13
Parathion-methyl	0.002–0.025	0.004–0.007	Traces	0.007–0.012
Carbosulfan	0.07–0.11	0.10–0.20	0.009–0.05	0.16–0.29
Cypermethrin	0.009–0.015	0.01–0.016	Traces	0.01–0.017
Deltamethrin	0.008–0.012	0.003–0.009	ND	0.005–0.018
Cyhalothrin	0.008–0.012	0.012–0.018	0.003–0.006	0.014–0.024
Bifenthrin	0.010–0.016	0.012–0.018	Traces	0.012–0.017
Imidacloprid	0.009–0.020	0.016–0.035	0.004–0.005	0.014–0.025
Profenophos	0.010–0.016	0.009–0.080	0.004–0.008	0.020–0.050

* The values represent the mean of 5 replicates.

and profenophos were found only in samples of cotton belt whereas cartap, carbofuran, diazinon, carbaryl, chlorfenvinphos and decamethrin were detected in the water samples of rice belts and concentrations of these compounds were substantially higher in canal and well samples as compared to tube well and hand pump. Isoproturon, dichlorvos and carbendazim were found in the water samples collected from vegetable areas. All the canal and well water samples in vegetable areas contained higher concentrations as compared to tube well and hand pump samples. Individual insecticide residue concentration has not exceeding than the maximum concentration (0.1 µg/L) set by EC (14) but total insecticide concentrations exceed the MAC (0.5 µg/L). Organochlorine residues were also detected in surface and ground water samples and its concentration was higher

Table 10.3. Pesticide residues* (µg/L) in surface and ground waters collected from rice belt.

Pesticides	Canal water	Hand pump water	Tube well water	Well water
α-Endosulfan	0.02–0.08	0.01–0.05	Traces−0.02	0.03–0.09
p, p′-DDT	0.02–0.04	0.01–0.02	Traces−0.009	0.016–0.05
Carbofuran	0.18–0.36	0.110–0.170	0.007–0.11	0.09–0.12
Methamidophos	0.25–0.41	0.110–0.330	0.008–0.12	0.12–0.35
Cartap	0.12–0.32	0.09–0.18	0.020–0.11	0.125–0.325
Monocrotophos	0.14–0.28	0.10–0.15	0.005–0.06	0.15–0.29
Chlorpyriphos	0.02–0.038	0.01–0.021	Traces−0.012	0.02–0.045
Carbosulfan	0.07–0.09	0.02–0.07	Traces−0.01	0.075–0.10
Diazinon	0.018–0.035	0.01–0.02	0.008–0.01	0.02–0.04
Decamethrin	0.016–0.07	0.009–0.04	Traces−0.009	0.018–0.075
Cyhalothrin	0.05–0.08	0.01–0.03	Traces−0.010	0.055–0.090
Carbaryl	0.02–0.035	0.012–0.020	Traces−0.011	0.02–0.04
Chlorfenvinphos	0.016–0.03	0.012–0.025	0.004–0.01	0.015–0.035
Chlorothalonil	0.012–0.018	Traces−0.01	Traces−0.007	0.01–0.02

* The values represent mean of 5 replicates.

Table 10.4. Pesticide residues* (µg/L) in surface and ground waters collected from vegetable belt.

Pesticides	Canal water	Hand pump water	Tube well water	Well water
a-Endosulfan	0.037–0.047	0.011–0.016	0.007–0.009	0.038–0.05
p, p′-DDT	0.011–0.013	0.007–0.009	ND	0.012–0.014
Isoproturon	0.025–0.038	0.011–0.021	0.006–0.009	0.03–0.045
Methamidophos	0.25–0.45	0.180–0.230	0.065–0.09	0.26–0.475
Dichlorvos	0.064–0.082	0.021–0.029	0.011–0.016	0.065–0.09
Monocrotophos	0.21–0.32	0.11–0.16	0.04–0.07	0.215–0.34
Chlorpyriphos	0.018–0.035	0.008–0.016	ND	0.02–0.045
Carbosulfan	0.022–0.065	Traces	ND	0.035–0.09
Cypermethrin	0.017–0.012	0.007–0.012	Traces−0.007	0.02–0.035
Deltamethrin	0.045–0.065	0.009–0.012	ND	0.05–0.07
Cyhalothrin	0.040–0.064	0.024–0.027	0.01–0.015	0.065–0.085
Bifenthrin	0.032–0.058	0.016–0.025	0.007–0.009	0.045–0.065
Chlorothalonil	0.065–0.075	ND	ND	0.085–0.09
Carbendazim	0.026–0.045	Traces−0.009	ND	0.03–0.055

* The values represent mean of 5 replicates.

in the water samples belong to the cotton belt. It may be due to the repeated application of these compounds and got bound to the soil fraction and release slowly to the environmental samples. It is also notable that the half-life of these compounds are greater as compared to other classes of pesticides used for the protection of agricultural crops studied by Tomlin (1994). It is worthy to mention that the increase in residues in fresh water may be attributed due to the washing of agricultural appliance and containers on the bank of canals and near the wells.

In Pakistan, different variety of industry effluents are discharged into the drains with out any pre-treatment. Samples collected from different drains showed the presence of pesticide residues (Table 10.5) after extraction with SPE and analyses with GC-ECD. A total 224 wastewater samples were collected, out of which 175 were found contaminated with pesticide residues and the remaining 75 with out pesticide residues. Higher concentration of methamidophos, chlorpyriphos, monocrtophos, carbofuran, diuron, cypermethrin and deltamethrin were detected and these were in

Table 10.5. Pesticide residues (μg/L) in waste water samples.

Pesticides	Pesticide residues Present	Not detected
α-Endosulfan	12 (2–12)	4
p, p′-DDT	8 (1–5)	8
Dieldrin	13 (1–8)	3
Methamidophos	14 (20–50)	2
Chlorpyriphos	12 (10–50)	4
Monocrotophos	10 (20–60)	6
Carbofuran	10 (10–20)	6
Oxamyl	13 (2–50)	3
Diuron	11 (10–50)	5
Cypermethrin	14 (10–80)	2
Deltamethrin	12 (20–50)	4
Thiophanate-methyl	10 (1–20)	6
Carbendazim	6 (5–50)	10
Imidacloprid	4 (2–60)	12

the range of (10–80 μg/L). Imidacloprid and thiophanate-methyl were also found in the wastewater samples. This may be due to the spillage and washings of the industrial machinery that prepare pesticide formulations.

10.4 CONCLUSION

Pesticide residues found in the surface and ground water and waste water samples may be attributed due to leaching, run-off, spillage and fall of industrial effluents or whether due to irresponsible handling practices. The use of SPE for the extraction of multiresidues from water samples found efficient, reproducible and time saving technique. We also recommend that studies on monitoring of pesticide residues be carried out to evaluate the pathways of contamination in the surface and ground water sources and wastewater drains. Pesticide user education is essential to minimize the pollution levels in the aquifers as much practice was observed, i.e. washing and dumping of pesticide container in/on the canals and use of non-target pesticide formulations on crops.

ACKNOWLEDGEMENTS

Authors are grateful to International Atomic Energy Agency (IAEA) for providing the reference standards for the pesticides. We gratefully acknowledge Dr. Charles S. Helling, USA for providing the glass fibre filters (Whatman 934-AH) and Baker SPE-10 manifold as a gift to the Pesticide Chemistry Laboratory., NIAB, Faisalabad. The authors are grateful to Dr. P. Bhattacharya and Dr. Arun B. Mukherjee for their meticulous efforts on the suggested improvements of the earlier draft of this manuscript.

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CHAPTER 11

Immobilization of coagulant proteins for drinking water treatment

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ABSTRACT: More than 1 billion people in the developing countries do not have access to safe drinking water and as a consequence more than 2 million people, most of them children, die every year from water borne diseases. One of the reasons is the high cost of treatment chemicals and hence there is a need for low cost and locally available materials. The natural coagulants that can be used as a substitute for synthetic chemicals is a seed extract of Moringa oleifera. It contains an active coagulating protein that can be extracted for use in water treatment. This paper describes the purification and immobilization of the coagulant protein from M. oleifera seed. The coagulant protein from M. oleifera seeds was extracted with ammonium acetate buffer and its coagulation property for synthetic clay solution was studied. The crude extract has the same coagulation activity as alum. The active component was purified to homogeneity by ion exchange chromatography method using fast flow carboxymethyl Sepharose and the properties of the purified protein were characterized. The coagulant protein has also been purified using magnetic beads. The purified component is a cationic protein with a molecular mass less than 6.5 kDa (kilo Dalton). The magnetic beads have six times higher adsorption capacity for the crude extract compared to that of ion exchange matrix. The proteins immobilized to magnetic beads have coagulation property in synthetic clay suspension. The coagulated particles can be removed from immobilized protein by washing without disturbing the protein. The possibilities of using immobilized protein to magnetic beads are discussed. The immobilized beads or purified protein can be used in water treatment in developed and developing countries.

11.1 INTRODUCTION

The provision of safe drinking water has become a problem of central importance, both in developing and in developed countries. In developing countries, the quality of drinking water is often insufficient and does not meet the water quality standards. In developed countries, water purification processes use chemicals, despite the fact that their safety for health during long-term use and impact on the environment remain under question (Crapper et al. 1973, Martyn et al. 1989, Miller et al. 1984, Biosvert et al. 1997, Nalm et al. 1998). As a result, it is desirable to find sustainable alternatives that are friendly to human health and to the environment.

Studies have reported that biocoagulants can be extracted from microorganisms, animals and plants (Kawamura 1991, Lee et al. 1995, Ganjidoust et al. 1997). Among the natural coagulants, the seeds of a tropical plant, *Moringa oleifera* (MO) are of particular interest as they contain an active coagulant compound traditionally used for the purification of drinking water in rural areas of Sudan and Malawi (Jahn 1988, Noor et al. 2000, Folkard et al. 2001). These could be an environmental friendly alternative to alum and other coagulants because they are non-toxic and biodegradable. They produce less sludge volume than alum and the pH and conductivity of the treated water remains unaffected (Ndabigengesere et al. 1995). The sludge produced by coagulant protein is biodegradable and can possibly be used as fertilizer (Ndabigengesere & Narasiah 1998).

In addition, such seed extracts are capable of bacterial aggregation and removal; with efficiency similar to that of aluminum salts and other commonly used water treatment chemicals (Madsen et al. 1987, Ghebremichael et al. 2004). This suggests that seed extracts may be promising alternatives to the currently used water treatment chemicals. Therefore, natural coagulant proteins are attractive for water purification.

The use of MO seed extract (crude extract) in water treatment often results in the release of organic matter and nutrients to the water. The dissolved organic carbon (DOC) causes undesirable odour, colour and taste and also it becomes a precursor for disinfection by-product formation. It has been reported that crude extract increases organic, nitrate and phosphate contents of water while purified protein does not (Ndabigengesere & Narasiah 1998, Okuda et al. 2001, Ghebremichael et al. 2005). This renders crude extract difficult for use in large treatment systems since presence of organic matter would complicate the treatment process and adversely affect the water quality. In order to overcome the shortcomings the coagulant protein should be characterized and purified.

A number of studies have reported different active coagulant components from MO seed (Jahn 1988, Nkhata 2001, Ndabigengesere et al. 1995, Olsen 1997). Nkhata (2001) reported that the active components from water extract were dimeric cationic proteins with molecular weight of 12–14 kDa (kilo Dalton) and pI between 10 and 11. Ndabigengesere et al. (1995) indicated that the active component was water soluble protein with a net positive charge. Gassenschmidt et al. (1995) described it is a protein with molecular weight of 6.5 kDa and pI above 10. On the other hand, Okuda et al. (2001) reported that the active component of salt extract was neither protein, polysaccharide nor lipid but an organic polyelectrolyte with molecular weight of about 3.0 kDa. The nature and characteristics of the active components from MO seed are still unclear. It has been reported that a simple purification method that can be used for large scale purification could be applicable in countries where the materials are available at low cost (Ghebremichael et al. 2005).

Purification of the coagulant protein is of importance in order to use the natural products for water treatment. Use of purified protein often needs the source of the natural material all the time. When using natural coagulants such as MO seeds, the area needed for plantations and the method for harvesting and processing the seeds must also be considered. Hence it is important to find out the alternatives of using low cost natural material for water treatment. Therefore it would be useful to find a way of combining the desirable properties of natural coagulants with the possibility of large scale industrial production to design a product that is inexpensive, environmental friendly, reusable and effective in water treatment process.

Small particles have been utilized extensively in research applications for capture of cells and biomolecules. More specifically is the application of magnetic bead technology in the water treatment system. Magnetic separations in biology and biotechnology have diversified in recent years, leading to affinity mechanisms and processes (Frenzel et al. 2003, Altintaş et al. 2007). It employs magnetic particles for more conventional isolation and purification methods such as affinity and ion exchange (IEX) methods. Magnetic IEX (MIEX) is also commonly used for organic compounds removal in water and wastewater treatment (Zhang et al. 2006). Magnetic separations work due to an affinity group on the surface of the magnetic particle. Once the target binds to the affinity ligand, a magnet is used to immobilize the magnetic particles and then trap the target compound. The simplicity of the magnetic separations has recently resulted in much recent interest. Magnetic separation subject to very little mechanical stress compared to other methods, it is rapid, often readily scalable.

Previous studies in our group have shown that coagulant protein can be purified to homogeneity by CM Sepharose IEX matrix and characterized (Ghebremichael et al. 2005). The present study is focused on: 1) the purification of the coagulant protein with magnetic beads; 2) comparison of purification efficiency with CM Sepharose IEX method and 3) study the use of immobilized protein for the coagulation of clay suspension.

11.2 MATERIALS AND METHODS

11.2.1 *Extraction of coagulant protein*

Seeds were shelled and oil was extracted with 95% ethanol and the solids were dried at room temperature. Sample (5%, w/v) was prepared from the dried solids using distilled water or 10 mM ammonium acetate buffer (pH 6.7), stirred for 30 min and filtered through Whatman filter paper No 3 and 0.45 mm fiberglass. The filtrate is as termed crude extract. Alum solution was also prepared in 5% (w/v) solution.

11.2.2 *Coagulant activity assay*

To know if the extract has any coagulant activity, we tested the activity by coagulation assay method developed by Ghebremichael et al. (2005). The substrate was prepared by adding 10 g kaolin into 1 L tap water, stirred for 30 min and allowed to settle for 24 h for complete hydration. Desired turbidity was obtained by dilution.

Ten micro litre samples were taken from the extracts of the active component in a semi-micro plastic cuvette (10 × 4 × 45 mm, Sarsted Aktiengesellschaft & Co, Germany), (10 µL water as a control) and made up to 1 mL high turbidity clay suspension (250–300 NTU). The content was mixed well with the pipette and the initial absorbance was measured at 500 nm in the UV-Visible spectrophotometer (Cary 50 Bio). The reduction in absorbance relative to control defines coagulation activity.

11.2.3 *Protein estimation and molecular weight determination*

Protein content was estimated by dye-binding method (Bradfard 1976) with bovine serum albumin as a standard. The protein profile and the molecular weight of the partially purified protein were determined by 10% SDS-PAGE mini gels. The gels were stained with Coomassie brilliant blue and silver staining method depending on the concentration of the protein loaded on the gel.

11.2.4 *Desalting*

Purified protein eluted with NaCl may disturb purification and SDS-PAGE separation; the eluted proteins were desalted before electrophoresis by using a PD-10 column. The column was rinsed with milli-Q water to remove the ethanol solution and equilibrated the column with milli-Q water. One ml of sample was loaded; the fractions (1 mL) were collected and measured the absorbance of each fraction at 280 nm.

11.2.5 *Purification of coagulant protein*

11.2.5.1 *Carboxymethyl (CM) Sepharose*

Samples were selected based on the activity test for the purification of the active component. The selected samples were purified using high trap CM FF 1 mL column cation exchanger in Äkta explorer (Pharmacia Biotech). Purification of the coagulant protein by ion exchange method using CM Sepharose matrix is followed as described in Gebreamichael et al. (2005). A similar condition for purification was also tried with batch purification method.

11.2.5.2 *Magnetic beads*

The coagulant protein was purified using magnetic ion exchange beads. The magnetic beads are cation exchange beads (Europe Bioproducts Ltd) that can be used for separation and purification of neutral and basic proteins. The core consists of Iron Oxide particles (Fe_3O_4) and is covered with Carboxymethyl Cellulose. The beads are 1–10 µm size, stored in 20% ethanol and the stock solution has a concentration of 50 mg beads/mL suspension.

This method is based on the IEX method since according to the theory the magnetic beads has the same properties as that of IEX beads. This method has the advantage that it does not have to use any centrifugation to separate beads; instead one can use a magnetic stand to easily separate the beads from the solution.

A volume of 50 μL magnetic beads were washed (equilibrated) four times with 1 mL 10 mM ammonium acetate buffer (pH 6.7). The crude extract, dissolved in ammonium acetate buffer, was added to the beads and after 30 minutes of immobilization, the unbound protein was separated from the beads. The beads were washed three times with ammonium acetate buffer and then the bound protein was eluted with different concentrations of NaCl. Under immobilization, washing and elution steps, the beads were displaced from the magnetic stand to get an uniform distribution of beads.

11.2.6 *Beads capacity*

To evaluate the maximum adsorption capacity for the magnetic beads, series of tests were done by adding the same amount beads with different amounts of protein. The immobilization was performed as described above. After 30 minutes of immobilization the unbound protein was separated from the beads. The activity test was run for the samples. The adsorption capacity is calculated with Langmuir equilibrium adsorption model (Faust & Aly 1987).

$$\frac{Ce}{qe} = \frac{1}{b \times Xm} + \frac{Ce}{Xm} \tag{11.1}$$

where Ce is the amount of protein in solution at equilibrium (mg/L), qe is the amount of protein adsorbed per weight adsorbent (mg/g), b is a constant related to the heat of adsorption (mL/g) and Xm is the maximum adsorption capacity (mg/g). When the calculated Xm reaches the same value the bead capacity for our protein is found.

11.2.7 *Activity test for the immobilized magnetic beads*

Immobilised magnetic beads were used with a concentration of 6.54 mg protein/mL beads. The crude extract was added to the beads with ammonium acetate buffer and after 30 minutes of immobilization the unbound protein was separated from the beads. The beads were washed three times with ammonium acetate buffer. These immobilized beads were used for coagulation activity measurement. A volume of 10 μL of the immobilized beads was mixed with 990 μL 1% clay solution in an Eppendorf tube. The tubes were rotated with a rotating roller for 15 minutes to let the protein and clay particles come in contact. The solution was transferred to cuvette and the initial absorbance was measured at 500 nm with a spectrophotometer. The sample was let to settle and the absorbance was measured after 45, 110 and 150 minutes. The coagulation activity was calculated for the different samples. The non-immobilized beads, crude extract and buffer were used as controls.

11.2.8 *Reuse of immobilized beads*

The immobilized beads were cleaned from the clay particles by washing with different solvents (Table 11.1) without disturbing the bound protein to the beads. The supernatant was removed by placing the cuvette on the magnetic stand. A volume of 200 μL of the chosen solvent was poured to the beads and the mixture left for 10 minutes. The supernatant was taken away and the washing was repeated once again which followed by two washes with 1 mL of 10 mM ammonium acetate buffer. The beads were transferred to an Eppendorf tube and 990 μL of 1% clay solution was added. The procedure was repeated as described above. The absorbance was measured at the same times as before to be able to find out if there are any differences in activity.

Table 11.1. Solvents used for cleaning of beads.

Solvent	Concentration
NaCl	10–50 mM
NaOH	10–100 mM
SDS	0.01–10%
EDTA	50 mM
DMSO	0.5–1%
Ethanol	10–20%
Acetone	20%
Tween 20	0.05–0.1%

The reuse activity was calculated by comparing the coagulation activity before and after washing with solvent. The reuse activity shows how many percent the coagulation activity is retained.

$$reuse(\%) = \frac{use\ 2\ real\ activity}{use\ 1\ real\ activity} \times 100 \tag{11.2}$$

11.3 RESULTS AND DISCUSSION

The active coagulant protein was extracted both by water and buffer extraction methods and it was purified by using IEX and magnetic beads. The purification efficiency and adsorption capacity of the two matrices were compared. The immobilized beads were used to measure the coagulation activity.

11.3.1 *Coagulation activity*

The coagulation activity was determined both in the water and buffer extraction methods. The coagulant activity of the water extracts was higher compared to the buffer extracts, whereas the protein content was higher in buffer extracts. The possible reason might be the water has better environment for the coagulant that acts with clay solution. Since we have not performed any experiment on these aspects, we try to keep several controls in order to find out the true effect of coagulant protein. The controls were used in the presence of water and buffer solutions. The difference in control and coagulant protein is considered as a real activity. In our previous studies we have compared the water and salt extracts (Ghebremichael et al. 2005). The coagulation activity of both extract was similar to that of alum which was used as a positive control in order to find out the coagulant activity of the protein.

11.3.2 *Purification of coagulant protein*

The magnetic beads have similar property as that of CM Sepharose matrix; hence purification was made using same buffer and pH as in our previous study. Initial experiments with same elution parameters as that of CM Sepharose IEX method were not successful. Most of the bound protein was not able to elute with 0.6 M NaCl as used in the CM Sepharose IEX method. Hence it was necessary to optimize the conditions for eluting the bound protein. Once the protein adsorbed in the magnetic beads, different concentrations of NaCl were used to elute the protein (Fig. 11.1). The results indicated that higher concentration of NaCl was needed in order to elute all the bound protein from the magnetic beads. At 0.3 M and 0.6 M NaCl concentration (which were used in the case of CM Sepharose IEX purification), the protein was eluted at low concentrations; most of the protein is still bound to the magnetic beads. The bound protein was completely eluted at

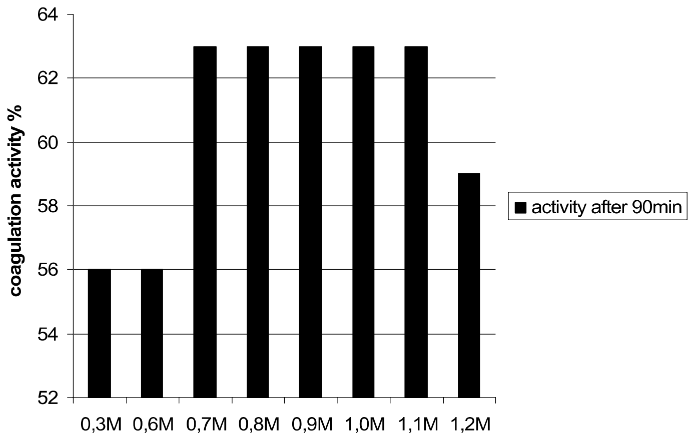

Figure 11.1. Coagulation activities to investigate the needed elution concentration by single elution. Coagulation activity values after 90 min settling time are shown in the figure.

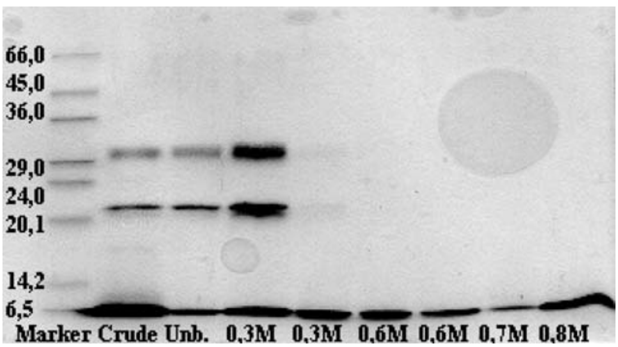

Figure 11.2. SDS-PAGE gel for step elution. Gel were stained with Coomassie brilliant blue and low range protein marker (Bio-Rad).

0.8 M NaCl concentration. The SDS-PAGE in Figure 11.2 shows the protein eluted with 0.3 M NaCl and 0.8 M NaCl (after elution by lower NaCl concentration). At 0.8 M NaCl elution step it has only one band which corresponds to the purified protein from CM Sepharose IEX. At low NaCl concentration other bound proteins were eluted as shown in Figure 11.2. In order to purify the protein we have used two step elution methods. This method depends on the application of coagulant protein. If one need the pure protein for further characterization studies and structure determination of coagulant protein, it is necessary to use two step elution methods. It was found in our previous study, that for water treatment application, one step elution would be sufficient in order to reduce organic loads in water treatment systems (Ghebremichael et al. 2005).

It was possible to purify the coagulant protein to homogeneity by both the methods. The salt concentrations needed for eluting the protein from magnetic beads were higher than the IEX. This might be due to high binding or affinity of coagulant protein to the magnetic beads.

11.3.3 *Reuse of magnetic beads for purification*

It is very important to know whether the magnetic beads can be reused for purification. We investigated the coagulation effectiveness of the regenerated magnetic beads. It was found that the regenerated beads had similar binding efficiencies as could be seen from the coagulation activity

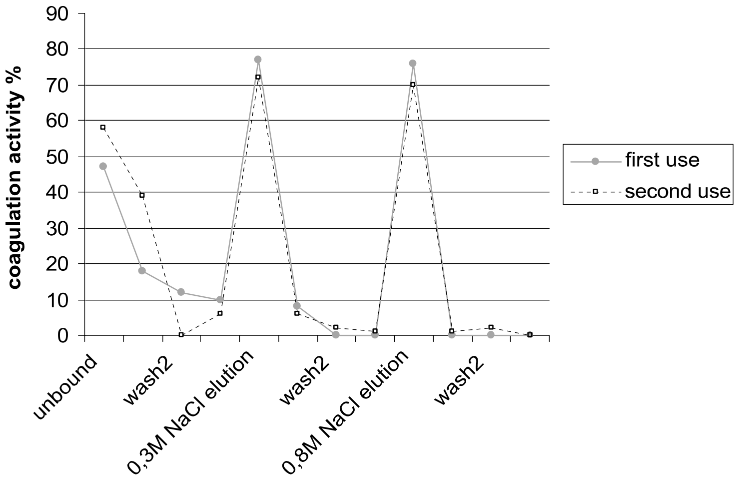

Figure 11.3. Comparison of coagulation activities for purification of crude extract with reuse of beads.

of the eluted protein with the original beads (Fig. 11.3). Hence it is possible to reuse the magnetic beads without changing the binding efficiency.

This has many advantages over other purification methods. The handling of the beads and purification can be done in a simple way only with magnetic stand without any need of sophisticated instruments. It is very fast and the results are reliable and consistent. The unbound protein showed coagulation activity, which could be due to the fact that higher concentration of protein was applied than the bead capacity or it might be due to binding competition with other proteins.

11.3.4 *Comparison of IEX and magnetic bead for purification*

This experiment was performed in order to find out if there are any differences in purification methods. Both IEX and magnetic beads can be used to purify the coagulant protein. Both method of purification have similar binding and elution characteristics (Fig. 11.4). Magnetic beads have better adsorption than IEX matrix when comparing the unbound fractions. Similarly magnetic beads have higher elution concentration with 0.8 M NaCl that all the adsorbed protein could be eluted. Even though all the bound protein eluted in both methods, magnetic beads has higher adsorption thereby the concentration of eluted protein is also increased.

11.3.5 *Bead capacity*

This is to compare the capacity of magnetic beads over CM Sepharose IEX matrix for the binding of coagulant protein. The magnetic beads have higher binding capacity compared to CM Sepharose IEX matrix. The capacity of CM Sepharose IEX matrix is 21 mg/g matrix for binding crude protein (Ghebremichael et al. 2005). This study showed that the magnetic beads have capacity of binding 131 mg protein/g beads. This reveals that the maximum adsorption of crude extract protein is six times higher for magnetic beads compared to CM Sepharose IEX matrix.

Since crude extract contains organic and inorganic substances, the adsorption (binding) capacity may be lower than using purified protein. The various compounds present in the crude extract may interfere with binding of the desired protein to the beads. There is also possibility of binding other proteins having similar charges. This can be seen by eluting protein with low salt concentration (0.3 M NaCl). The binding capacity of pure protein to IEX is three times higher than the crude extract (Ghebremichael et al. 2005). One can expect similar binding capacity for the magnetic

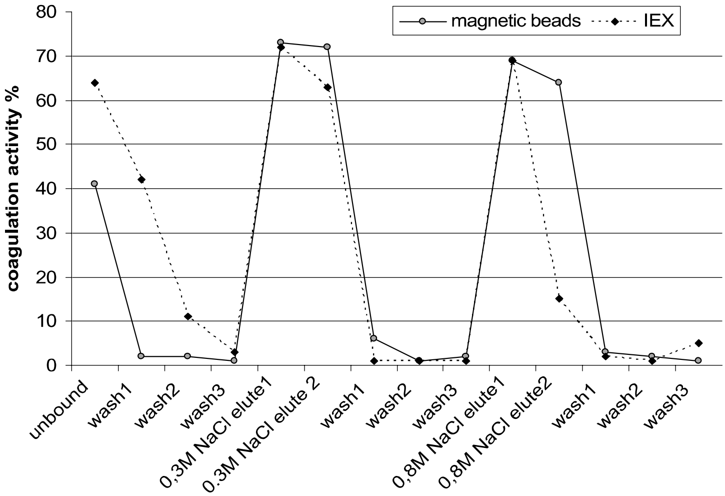

Figure 11.4. Comparison of coagulation activity of the purification products using IEX and magnetic beads.

beads. The binding capacity is six times higher than that of CM Sepharose IEX matrix which makes the use of magnetic beads attractive for the purification of coagulant protein.

11.3.6 *Coagulation activity with immobilized beads*

The immobilized beads were used to find out the coagulation activity of the protein when it was bound to the magnetic beads. The magnetic beads have no coagulation activity by itself. The immobilized beads have about 50% less activity than the crude extract in suspension. The coagulant activity does not increase with the concentration of protein bound to the beads. Results indicated that the coagulation does not increase markedly with the amount protein (Fig. 11.5). One reason could be that the protein may need longer time to bind to the clay particles when immobilized beads are used.

11.3.7 *Reuse of immobilized beads*

Once the particles are bound to the immobilized beads it is good to find out whether the immobilized beads could be used again for settling particles. In order to find out the capability of immobilized beads, we used different washing solutions to remove the attached clay particles without disturbing the bound protein. Immobilized beads cleaned with 20% ethanol had higher coagulation activity after washing out clay particles which was followed by Tween 20 and 1% DMSO (Fig. 11.6). Most of the cleaning solution used in this study had higher activity in the washed buffer solution. This implied that the protein might have detached from the beads there by loosing activity of the immobilized beads. It is advisable to use 20% ethanol for washing in order to use the bound particles without disturbing immobilized protein.

The activity test for washed buffer showed that there was some coagulation activity in the washing with each solvent and buffer, which mean that some of the immobilized protein was detached from the magnetic beads with the washed solvent. But the major reasons for low coagulation activity of the cleaned beads seem to depend on the fact that clay particle were not completely washed away from the beads. Whereas beads washed with ethanol and Tween 20 were showing coagulation activity and the washed solvent and buffer were also coagulation active. Thus, cleaning with these solvent may loose some coagulation active protein. On the other hand it also showed high coagulation activity after cleaning the beads, hence the low activity in wash solution were negligible. Further work is in progress to optimize the activity of the immobilized beads and regeneration of the active beads.

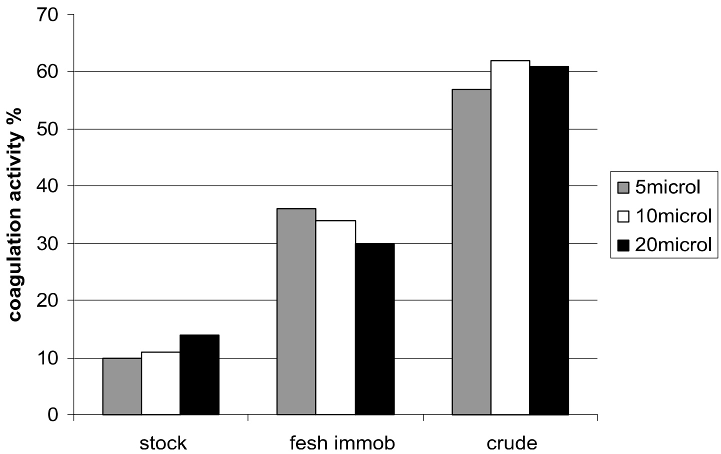

Figure 11.5. Coagulation activities after 205 minutes to compare the maximum settling.

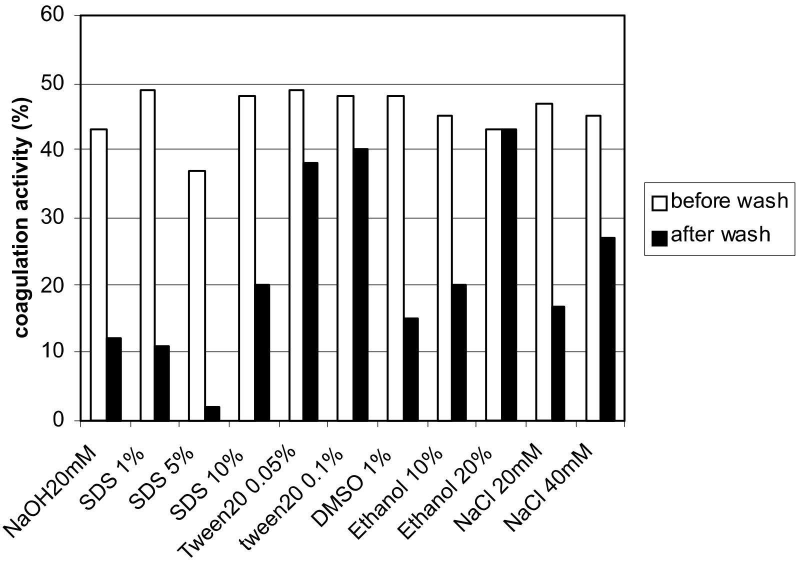

Figure 11.6. Coagulation activities for washing after clean of beads with different solvents.

11.4 CONCLUSIONS

Moringa oleifera seed extract has same coagulant activity as that of alum. A simple purification method was developed to purify the active component using magnetic beads with 0.8 M NaCl elution buffer. The protein could be purified to homogeneity using two step elution methods. The capacity of magnetic beads is much higher than carboxymethyl Sepharose ion exchange matrix and it is possible to reuse the beads for purification. Using protein immobilized magnetic beads as a coagulant for drinking water could be possible. If the purification is expensive, it can be feasible to use immobilized protein and therefore the protein can be reused for the treatment process. Further studies are needed in order to find out the possibility of stable binding of the coagulant protein to the beads and find a suitable cleaning method in order to use the immobilized beads in the drinking water systems to meet the economy in developing countries.

ACKNOWLEDGEMENTS

We are extremely grateful to Prof. Dr. Jan Hoinkis of the University of Applied Sciences, Karlsruhe for the critical comments on an earlier version of the manuscript. The authors are thankful to the organizers of the 2nd International Groundwater Conference and the School of Biotechnology for their encouragement and support for presenting this work. The authors are grateful to Dr. A.B. Mukherjee and P. Bhattacharya for their constructive criticisms and editorial handling of the manuscript.

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CHAPTER 12

Assessment of geochemical processes controlling groundwater chemistry in a river basin in Southern India

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ABSTRACT: A study was carried out to understand the geochemical processes and its contribution to groundwater chemistry in a region comprising of hard rock and sedimentary formations, south India. Results of this study indicate that groundwater occurring near the Palar river has high concentrations of major ions except calcium due to the absence of river flow, where as lower concentrations of major ions were observed in the central part of the study area (Hard rock formations) due to the recharge of fresh water from a number of reservoirs. The groundwater chemistry in this area is controlled by both mineral dissolution and anthropogenic activities, especially agricultural activities. The relative contributions of mineral dissolution and anthropogenic contamination are estimated by reaction stoichiometric approach, and it was found that mineral dissolution is the dominant processes in both the formations. Thus, groundwater chemistry of this region is largely influenced by mineral dissolution and anthropogenic activities.

12.1 INTRODUCTION

Groundwater chemistry is regulated by various factors, which include atmospheric input, mineral weathering through rock-water interaction and anthropogenic activities. Weathering of minerals generally exerts an important control on groundwater chemistry (Garrels & Mackenzie 1967, Kim 2002). Weathering of minerals is generally a dominant process controlling the concentration of the major cations (Ca, Mg, Na, K) in groundwater (Garrels & Mackenzie 1967). Additionally, intense agricultural activities have placed a high demand on groundwater resources in worldwide and also directly or indirectly play a major role in regulating groundwater chemistry (Chourasia & Tellam 1992, Kraft et al. 1999, Cardona et al. 2004). In the present study, an investigation was carried out to understand the geochemical process regulating groundwater chemistry in Palar and Cheyyar river basins, South India. Geologically, the study area is covered by both hard rock and sedimentary formations. Hence, this study concentrated the geochemical processes regulating the groundwater chemistry on both the formations. Further, as the groundwater is the only major source of water for agricultural and drinking purposes, it is important to know the effect of geological formations and agricultural activities on groundwater chemistry of the study area.

12.2 STUDY AREA

12.2.1 *General*

The study area is located in Kancheepuram District in Tamil Nadu State, India and it covers an area of 234 km^2 (Fig. 12.1). It forms a part of the Palar and Cheyyar River basins, and is located 70 km

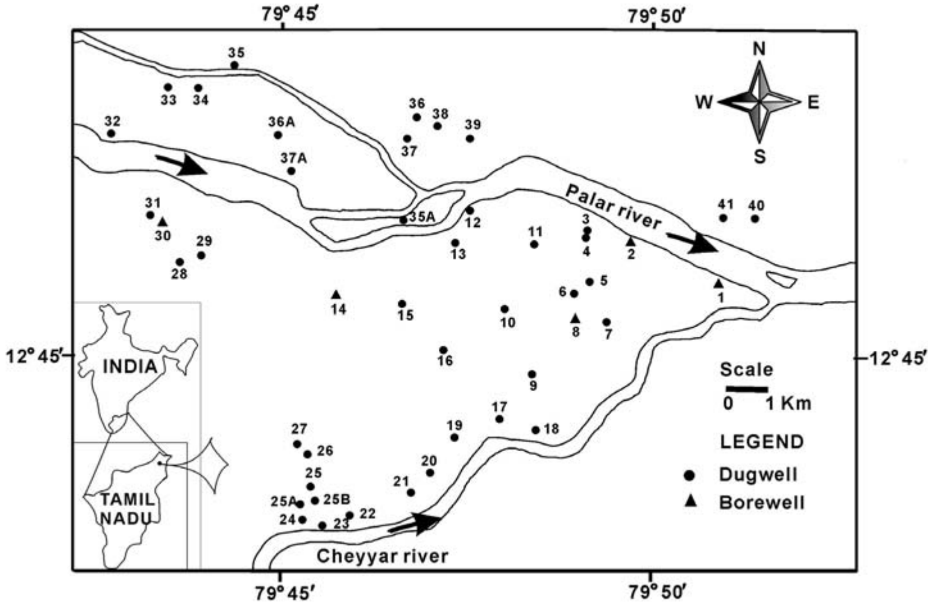

Figure 12.1. Location of monitoring wells in Palar and Cheyyar river basins, Southern India.

west of Chennai city. The main cropping season is from September through January, when paddy is grown. The second cropping season is from February through May, when paddy, vegetables, pulses and groundnut are grown. The study area has dry climatic condition with the maximum air temperature of 39°C during summer (April and May) and minimum air temperature of 21°C during winter (November and December). It receives an average annual rainfall of 1113 mm, of which 60% occur during winter (October through December by northeast (NE) monsoon); the rest by southwest (SW) monsoon (June to September). The rivers in this region flow only for a few days in a year after heavy rains during monsoon. The study area has number of reservoirs, which are the major source for recharge.

12.2.2 *Geology and hydrogeology*

The study area comprises partly sedimentary rocks (sandstone and shale) and partly crystalline rocks represented by charnockite, granitic gneiss and ultrabasics (Fig. 12.2). They are unconformably overlain by sandy and clayey soils of Recent to sub-Recent age. The crystalline charnockite and granitic gneiss of Archean age have been intruded by amphibolites, dykes of dolerite and occasionally by veins of quartz and pegmatites. The gneisses of this area have quartz, feldspars (potash feldspars and albite), hornblende, biotite etc. The acid charnockite of this area has quartz, K-feldspars, hypersthene and biotite minerals of coarse-grained nature. Alluvial deposits constitute the youngest formation consisting of sands and clays, and occurring along river courses. Groundwater occurs under water table conditions in weathered and fractured portions of crystalline rocks. The depth of wells in this region is generally up to 10 m and most of them are large diameter dug wells. Generally, the groundwater level in these crystalline formations is between 6 and 9 m below ground level. Alluvium deposit occurring on either sides of the river is essentially composed of sand with intercalated clay. The water level in these wells fluctuates between 5 and 20 m below ground level. In these formations, wells have been dug up to a depth of 23 m and most of the wells are bore and dug cum bore wells.

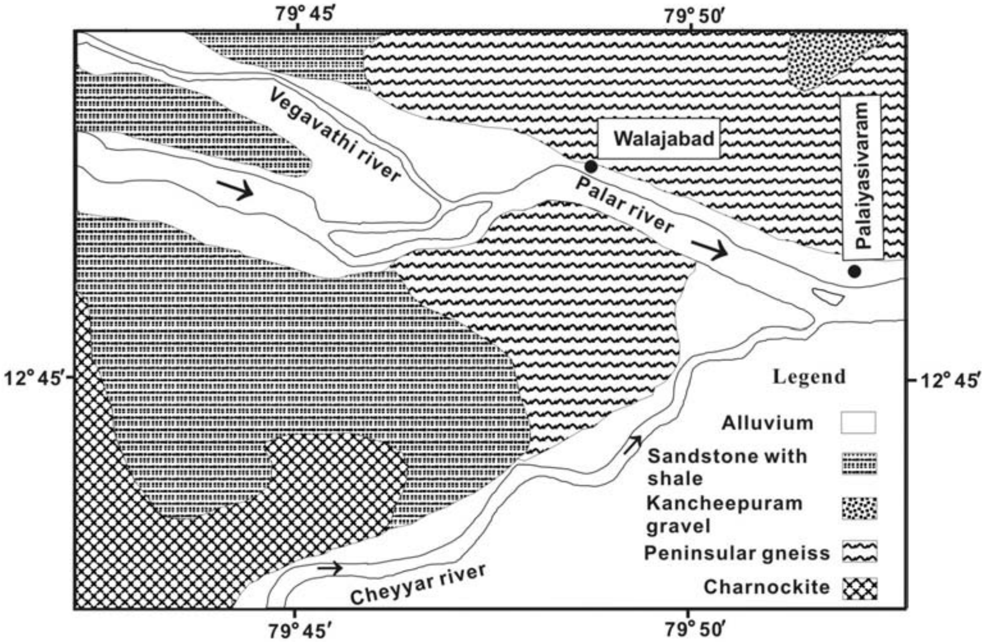

Figure 12.2. Geology of the study area in the part of Palar and Cheyyar river basins, Kancheepuram district, Tamil Nadu, India.

12.3 GROUNDWATER SAMPLING AND ANALYSIS

Groundwater samples were collected monthly from 43 sampling wells. In total, 641 groundwater samples were collected over the duration of the study. pH and electrical conductivity (EC) of groundwater samples were measured in the field using YSI 85 portable meters. Water level in the wells was recorded using a water level recorder (Solinst water level meter, model 101). Samples collected were transported to the laboratory on the same day, and filtered using 0.45 μm Millipore filter paper and acidified with ultra-pure nitric acid for cation analyses. For anion analyses, these samples were stored below 4°C. The samples were analysed for major cations (Na^+, Ca^{2+}, Mg^{2+}, K^+) and anions (Cl^-, SO_4^{2-}, HCO_3^-, CO_3^{2-}) as per the procedure given in APHA (1995). The analytical precision for the measurements of ions was determined by calculating the ionic balance error, which is generally within ±5%.

12.4 RESULTS AND DISCUSSION

12.4.1 *Major ion chemistry and geochemical process*

The results of the chemical analysis are given in Table 12.1. The pH of the groundwater is near-neutral to alkaline ranging between 7.0 and 8.0, except for a few samples with extreme values close to 6.3 and 8.8, and shows acidic to alkaline nature. The electrical conductivity (EC) of groundwater varies from 240 to 2700 μS/cm, with most of the samples less than 1000 μS/cm. Wells located near the Palar river have high EC values than the other wells. The general order of dominance of cations in the groundwater of the study area is $Na^+ \sim Ca^{2+} > Mg^{2+} > K^+$ and for anions $HCO_3^- > Cl^- > SO_4^{2-}$. The concentration of these ions in groundwater varied with respect to geological formations, i.e $Ca^{2+} \sim Na^+ > Mg^{2+} > K^+$ and $Na^+ > Ca^{2+} > Mg^{2+} > K^+$ is observed in hard rock and

Table 12.1. Descriptive statistics for the chemical analysis of groundwater samples.

Well location	pH	EC	HCO_3^-	Cl^-	SO_4^{2-}	Na^+	Ca^{2+}	K^+	Mg^{2+}
Hard rock formations (n = 356)									
Min	6.3	240	88.0	0.0	0.0	10.0	8.0	0.0	2.4
Max	9.5	2700	622	543	210	329	184	27.0	139
Mean	7.6	939	347	121	41.1	79.1	73.8	1.5	36.2
SD	0.5	473	101	112	35.3	58.6	37.2	4.3	20.0
Sedimentary formations (n = 285)									
Min	6.3	410	43.9	0.0	7.5	11.0	8.0	0.0	9.6
Max	8.3	2600	617	539	297	282	126	68.0	90.9
Mean	7.4	1064	333	125	54.5	106	58.2	7.1	34.4
SD	0.3	455	92.6	92.7	41.9	58.1	22.3	11.3	15.8
Total (n = 641)									
Min	6.3	240	43.9	0.0	0.0	10.0	8.0	0.0	2.4
Max	9.5	2700	622	543	297	329	184	68.0	139
Mean	7.5	995	341	123	47.1	90.9	66.8	4.0	35.4
SD	0.4	469	97.5	104	38.9	59.8	32.4	8.6	18.3

Unit: mg/L except EC (μS/cm) and pH. SD-Standard deviation. n-number of samples.

sedimentary formations, respectively. The concentrations of Na^+ in groundwater samples vary from 10 to 471 mg/L. Wells located in hard rock formations have comparatively low concentration of Na^+ (mean = 79.1 mg/L) than the wells located in sedimentary formations (mean = 105.6 mg/L) (Table 12.1).

The Ca^{2+} concentration of groundwater in the study area ranges from 8 to 323 mg/L. In contrary to Na^+, high concentration of Ca^{2+} is observed in groundwater of hard rock formations comparison with sedimentary formations (Table 12.1). High concentration of Ca^{2+} observed in the central part of the study area is mainly due to the cation exchange reactions as reported by Rajmohan & Elango (2004).

Like Na^+ sodium, high concentrations of K^+ are observed in sedimentary formations. But in hard rock formations, K^+ is below the detection limit in most of the wells, which seems to be due to the resistance to weathering of K^+ bearing minerals and its fixation in the formation of clay minerals (Rao 2002). Magnesium concentration varies from 2 to 152 mg/L and it is less than 72 mg/L in most of the samples and does not vary much between sedimentary and hard rock formations (Table 12.1). In general, Na^+ and K^+ concentrations are high in sedimentary formations, whereas Ca^{2+} concentration is high in hard rock formations (Table 12.1). Weathering of silicate minerals might be the major cause for high concentrations of major cations in groundwater of this region (Rajmohan et al. 2000). However, rainfall recharge, ion exchange processes and irrigation return flow may also be contributed in the major cations concentration in groundwater of this region (Rajmohan & Elango 2004).

Alkalinity concentration (as HCO_3^-) varies from 59 to 837 mg/L, and it contributes 55 to 60% (mean) to the total anions in groundwater of the study area. The variation in alkalinity between the wells in hard rock and sedimentary formations is not consistent. Alkalinity in groundwater is generally derived from the weathering of minerals such as silicates and carbonates (Stumm & Morgan 1996). It is well known that Cl^- concentration can be used as an indicator for contamination because Cl^- in the inland essentially originates from the surface sources such as domestic wastewaters, septic tanks, irrigation return flow and fertilizers (Andreasen & Fleck 1997, Lowrance et al. 1997). In the study region, the concentration of Cl^- in the groundwater ranges from 14 to 973 mg/L and suggests the contribution of anthropogenic contamination. As per the drinking water standard (250 mg/L; WHO 1984), 90 samples are not suitable for drinking

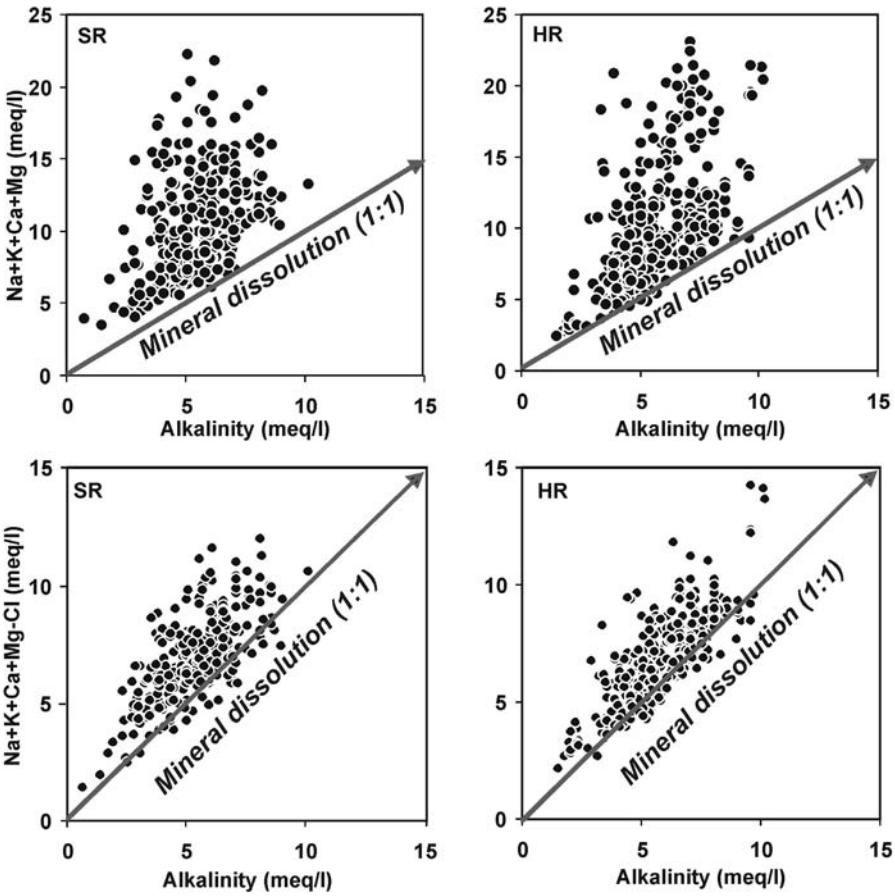

Figure 12.3. Plots of total cation and chloride corrected total cation as a function of alkalinity. Abbreviations SR: samples from wells in sedimentary formations; and HR: samples from wells in hard rock formations.

purposes. The concentration of SO_4^{2-} varies between 60 and 100 mg/L and it contributes only 10% to the major anions. Both Cl^- and SO_4^{2-} seem to be derived from irrigation return flow, synthetic fertilizers (potash, KCl), and gypsum ($CaSO_4 \cdot 2H_2O$).

12.4.2 *Assessment of geochemical processes*

As discussed earlier, the major ion chemistry of this region is mainly controlled by two processes: one is mineral dissolution and other is anthropogenic activities, especially, agricultural activities. As mentioned earlier, alkalinity contributes 60% in the total anion and indicates the dominance of mineral dissolution (Stumm & Morgan 1996). As the study area is mostly covered by agricultural lands, the major ion chemistry of groundwater seems to be controlled by irrigation return flow and application of synthetic fertilizers. To identify and estimate the relative contribution of both these processes, stoichiometric approach is used. Based on the stoichiometric analysis, Kim (2002) has suggested that the 1:1 ratio would be maintained between total cation and alkalinity when the dissolution of silicate and carbonate minerals is the major process affecting the groundwater chemistry.

Table 12.2. Contribution of mineral weathering and anthropogenic activities to the total cation concentrations (in meq/L). The numbers in the parentheses represent the relative contributions for the total concentration.

	Anthropogenic activities			
Well location	Chloride	Sulphate	Total	Mineral weathering
Hard rock formations	3.43 ± 3.16 (29 ± 16%)	0.86 ± 0.74 (8 ± 5%)	4.28 ± 3.73 (37 ± 17%)	5.69 ± 1.66 (63 ± 17%)
Sedimentary formations	3.53 ± 2.62 (31 ± 15%)	1.14 ± 0.87 (11 ± 5%)	4.67 ± 3.34 (42 ± 18%)	5.46 ± 1.52 (58 ± 18%)
Total	3.47 ± 2.93 (30 ± 16%)	0.98 ± 0.81 (9 ± 5%)	4.45 ± 3.57 (39 ± 18%)	5.59 ± 1.60 (61 ± 18%)

The plot of total cation versus alkalinity indicates that some of the samples plotted on or near the 1:1 line and most of the samples plotted above the 1:1 line (Fig. 12.3). Likewise, chloride corrected total cation is plotted against alkalinity (Fig. 12.3), which indicates that most of the samples plotted on or near the 1:1 line. Samples plotted on or near the 1:1 line suggest that mineral dissolution is the only process controlling its major ion chemistry. Samples plotted above the 1:1 line illustrate the contribution of other processes related anthropogenic activities. Therefore, we assumed that alkalinity is the representative of mineral dissolution, and amount of total cation balanced by the alkalinity is derived from the mineral dissolution. As per the geology of the study area, there is no known geological source for chloride and sulphate. Hence, both Cl^- and SO_4^{2-} might be derived from anthropogenic activities, especially agricultural activities. Based on this concept, the relative contribution of mineral dissolution and anthropogenic activities were calculated (Table 12.2).

Mineral dissolution contributes to nearly 61% (±18%) of the total cation dissolved in groundwater. Mineral dissolution is explained 63% (±17%) and 58% (±18%) in the total cation of groundwater in hard rock and sedimentary formations, respectively. Similarly, anthropogenic activities contributed 39% (±18%) in the total cation of groundwater. Like mineral dissolution, it is also varied with respect to geological formations. Major ion chemistry of groundwater is largely affected by anthropogenic activities in sedimentary formations (42 ± 18%) in comparison to hard rock formations (37 ± 17%). In over all, major ion chemistry of groundwater in the study area is regulated by both mineral dissolution and anthropogenic activities. Among these, the mineral dissolution is dominated in both the formations. However, the contribution of anthropogenic activities is slightly higher in sedimentary formations compare to hard rock formations (Table 12.2).

12.5 CONCLUSION

Hydrogeochemistry of the groundwater is regulated by both mineral dissolution and anthropogenic activities, especially agricultural activities. Results illustrate that groundwater occurring near the Palar river has high concentration of major ions except Ca^{2+} compared to central part of the study area, where recharge of fresh water occurs from a number of reservoirs. Results of the stoichiometric analysis indicate that mineral dissolution is the dominant processes in both hard rock and sedimentary formations. However, the contribution from anthropogenic activities is comparatively higher in the sedimentary formations. Hence, the major ion chemistry of groundwater in this region is largely influenced by mineral dissolution and anthropogenic activities.

ACKNOWLEDGEMENTS

We wish to thank the University Grants Commission, New Delhi for the financial grant.

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Section III
Groundwater modeling and its application in aquifer systems

CHAPTER 13

Numerical analysis of tide-aquifer interaction data for estimating aquifer parameters

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ABSTRACT: The analysis of tidal effects on aquifer systems plays an important role in coastal aquifer management. Adequate knowledge of the hydraulic properties of aquifer systems such as transmissivity (T) or hydraulic conductivity (K), storage coefficient (S) and/or hydraulic diffusivity (D) is crucial for almost all the studies concerning groundwater quantity and quality, including the modeling of subsurface flow and transport processes. This paper deals with the numerical analysis of tide-aquifer interaction data of unconfined and confined aquifers for determining aquifer parameters by the Levenberg-Marquardt technique considering two approaches: (i) without tidal data resolution (henceforth called "Approach I"), and (ii) with tidal data resolution (henceforth called "Approach II"). In addition, the effect of tide-aquifer interaction data corresponding to Spring and Neap tidal events on the parameter estimates has also been examined. The tide-aquifer interaction data for two unconfined sites and three confined sites located in two different coastal basins were used. The analysis of the results indicated that the aquifer parameters S and T, and hence hydraulic diffusivities (D) based on Approach II are more reliable and accurate for both the aquifers than those based on Approach I. Therefore, Approach II is strongly recommended for the determination of aquifer parameters by the tide-aquifer interaction technique. Furthermore, the Spring and Neap tide-aquifer interaction data significantly affected the parameter estimates; the use of such data should be avoided. It is concluded that a judicious use of the tide-aquifer interaction technique is essential for obtaining reliable estimates of aquifer parameters.

13.1 INTRODUCTION

Subsurface flow processes are concealed and very complex in nature. Just as atmospheric pressure changes produce variations of piezometric levels, so do tidal fluctuations by varying the load in confined aquifers extending under the ocean floor (Todd 1980, Brassington 1998). On the other hand, groundwater in coastal unconfined aquifers is affected by tides due to the direct propagation of tidal waves. The groundwater behavior in an unconfined aquifer with a mild sloping face in response to tides is influenced by the infiltration of seawater from the top of the beach-slope into the aquifer during high tide and tidal pumping with a free water table (Ashtiani et al. 1999). In coastal aquifers having hydraulic connection with the ocean, sinusoidal fluctuations of groundwater level occur in response to tidal events (Ferris 1951, Todd 1980). Philip (1973), and Smiles and

Stokes (1976) for the first time indicated that even for zero net groundwater discharge to the sea, the non-linear influence of sinusoidal tidal motion on a vertical shoreline will cause the water table to rise above mean sea level.

The analysis of tidal effects on aquifer systems plays an important role in coastal aquifer management. Adequate knowledge of the hydraulic properties of aquifer systems such as transmissivity (T) or hydraulic conductivity (K), storage coefficient (S) and/or hydraulic diffusivity (D) is essential for all the studies pertaining to groundwater quantity and quality, including the simulation modeling of subsurface flow and transport processes. There are several methods for determining the hydraulic parameters of aquifer systems such as pumping-test data analysis (e.g., Freeze & Cherry 1979, Todd 1980, Batu 1998), numerical modeling (e.g. Yeh 1986, Pandit et al. 1991, Sun 1994), floodwave-response technique (e.g. Pinder et al. 1969, Reynolds 1987, Jha et al. 2004), and tidal response technique (Ferris 1951, Carr & van Der Kamp 1969, Erskine 1991, Pandit et al. 1991, Millham & Howes 1995, Ching & Shih 1999, Fakir & Razak 2003), among others. Out of these methods, the pumping-test data analysis is very popular and is considered as the standard method to date, but it is very costly, time-consuming and the conventional analysis of pumping-test data is cumbersome and quite subjective. Also, pumping test is not always advisable for coastal aquifer systems because it may accelerate seawater intrusion and the pumping test data are most likely to be very noisy. Under such circumstances, the relationship describing the aquifer response to tidal fluctuations could be employed for estimating the important hydraulic parameters of coastal aquifer systems (Millham & Howes 1995).

Tide-aquifer interaction technique is a method of analyzing groundwater-level fluctuations in a well or piezometer in response to changes in the sea level caused by tides (Ferris 1951, Carr & van Der Kamp 1969). This paper deals with the optimization of aquifer parameters (T, S and hydraulic diffusivity) by Levenberg-Marquardt technique using the tide-aquifer interaction model. The tide-aquifer interaction data from both unconfined and confined coastal aquifers have been used to assess the efficacy of tide-aquifer interaction technique in estimating aquifer parameters.

13.2 STUDY SITES AND HYDROGEOLOGY

In the present study, tide-aquifer interaction data have been obtained from two different coastal groundwater basins because of the paucity of such field data. Two tide-affected sites I-2 (well depth = 25 m and screen length = 1 to 25 m) and H-5 (well depth = 15 m and screen length = 1.12 to 15 m) were selected from Konan Aquifer located in Kochi Prefecture, Japan (Jha et al. 1999) and three sites 1272/34, 1525/34, and 235/26 were selected from Dridrate Aquifer located in Qualidia Sahel, Morocco (Fakir & Razack 2003). Sites I-2 and H-5 are located at 350 and 500 m from the Pacific Ocean coast, respectively (Fig. 13.1) and the sites 1272/34, 1525/34, and 235/26 are located respectively at 2800, 2650 and 400 m from the coast of the Atlantic Ocean.

The Konan groundwater basin is bounded by the Monobe River (perennial) in the west and the Koso River (intermittent) in the east (Fig. 13.1). Mountains demarcate the northern boundary and the southern boundary is demarcated by the Pacific Ocean. Cold and dry winters, and warm and humid summers characterize the regional climate. The minimum temperature is −4°C in February and maximum is 37°C in August. The mean annual rainfall and evapotranspiration in the region are about 2600 mm and 800 mm, respectively. Unconfined aquifers comprising alluvial sand and gravel and/or diluvial silty sand and gravel are predominant over the Konan basin. The detailed hydrogeology of the Konan basin is given in Jha et al. (1999).

Dridrate Aquifer of Morocco is located along the Atlantic Ocean coast and is composed of sandy and dolomitic limestones, which is separated from Plioquaternary terrains by overlying reddish sandy and argillaceous deposits. The details about this aquifer system can be found in Fakir & Razack (2003).

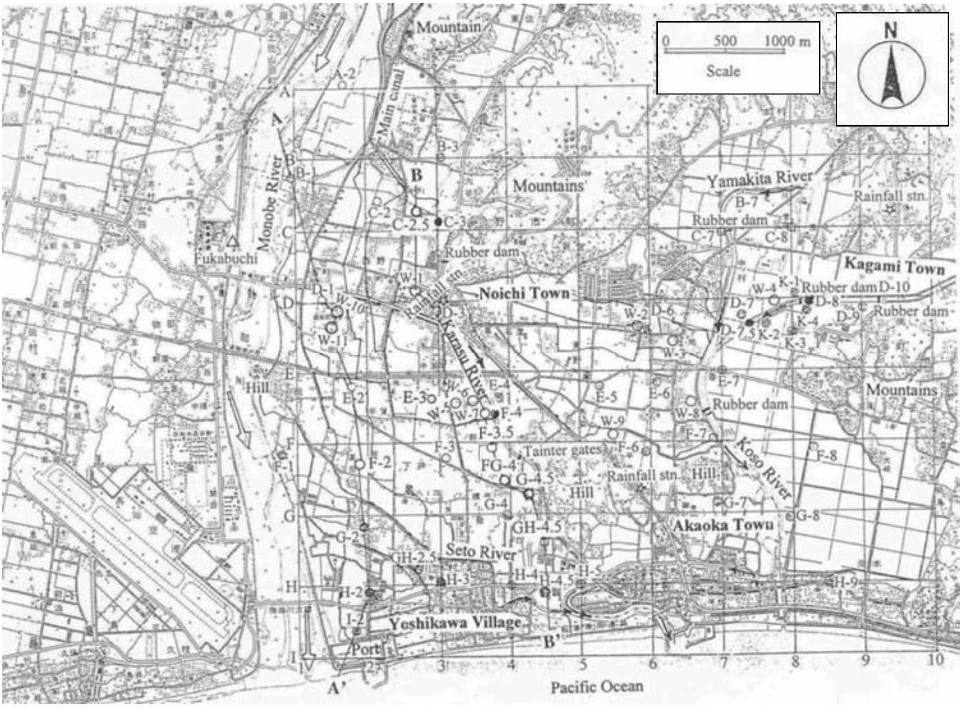

Figure 13.1. Map of the Konan groundwater basin in Kochi Prefecture, Japan.

13.3 METHODOLOGY

13.3.1 *Tide-aquifer interaction model*

In a homogeneous, isotropic, semi-infinite, confined coastal aquifer system, the one-dimensional transient propagation of sinusoidal oscillations of pressure due to tides into the aquifer system is described by (Ferris 1951, Todd 1980):

$$h(t) = h_0 \exp(-x\sqrt{\pi S/t_0 T}) \sin(2\pi t/t_0 - x\sqrt{\pi S/t_0 T}) \qquad (13.1)$$

where $h(t)$ = groundwater level from its mean at time t [L]; h_0 = amplitude of the tidal oscillation [L], and the remaining symbols have the same meaning as defined earlier.

It is apparent from eqn. (13.1) that the tidal oscillations remain sinusoidal in the aquifer with a time lag and the amplitude decreases exponentially with distance from the seashore. Besides several other simplifying assumptions, eqn. (13.1) assumes one-dimensional flow, i.e., no vertical flow or flow parallel to the seashore takes place. It is worth to mention that eqn. (13.1) was originally developed for confined coastal aquifer systems, but it can also be used for unconfined coastal aquifer systems, provided the ratio of maximum groundwater-level fluctuation to the saturated aquifer thickness is less than or equal to 0.02 (Roscoe 1990), and if the observation well is located far enough from the submarine outcrop of the aquifer to avoid vertical flow.

13.3.2 *Determination of S and T without tidal data resolution (Approach I)*

In this approach, the tidal data as such were considered as sinusoidal waves. Assuming initial guess values of S and T, the groundwater levels at a particular site and at different times due to tides were calculated using the tide-aquifer interaction model [eqn. (13.1)]. Now, using the computed and observed well hydrographs at a particular site for the given dataset, the root mean square error (RMSE) was computed. The values of S and T, which yielded a minimum value of RMSE, were considered as optimal S and T values. The Levenberg-Marquardt nonlinear optimization technique was used to simultaneously optimize S and T. This entire task was accomplished with the help of a computer program developed in C programming language.

The datasets used for this analysis were: (a) two sets of relatively long-duration (3 lunar days) hourly tide-aquifer interaction data (1–3 March 2000 and 22–24 February 2000) and one set of short-duration (1 lunar day) hourly tide-aquifer interaction data (22–23 February 1999) at the two unconfined sites corresponding to the normal tidal event, and (b) one set of half-hourly tide-aquifer interaction data of 150 hours at the three confined sites corresponding to a normal tidal event.

13.3.3 *Determination of S and T with tidal data resolution (Approach II)*

It should be mentioned that eqn. (13.1) is strictly applicable to perfect sinusoidal waves, though this equation is often used with the assumption that the tidal data themselves represent a sinusoidal wave. In fact, the observed tide level is composed of several simple harmonic components. Therefore, in Approach II, the observed tidal data were resolved into different simple harmonic components using the Fourier series which is expressed as (Kaplan 1973):

$$f(t) = \frac{a_0}{2} + \sum_{n=1}^{\infty}[a_n \cos\ nt + b_n \sin\ nt] \tag{13.2}$$

where, $a_0/2$ = neutral position, and a_n, b_n = fourier cosine and sine coefficients of the nth wave, respectively. These fourier coefficients are defined as follows:

$$a_n = \frac{1}{\pi}\int_{-\pi}^{\pi} f(t)\cos nt\, dt \quad (\text{For } n = 0, 1, 2, \ldots) \tag{13.3}$$

$$b_n = \frac{1}{\pi}\int_{-\pi}^{\pi} f(t)\sin nt\, dt \quad (\text{For } n = 1, 2, 3, \ldots) \tag{13.4}$$

Given the values of observed tide levels in the form of deviations from their mean, firstly Fourier coefficients were calculated using eqns. (13.3) and (13.4) and then the tidal components were calculated for all the tidal datasets under study. Now, assuming initial guess values of S and T, the groundwater levels at a particular site and time due to individual tidal components were computed using eqn. (13.1), and synthesized to obtain the time series of groundwater levels at the site (i.e. computed well hydrograph). Thereafter, the same procedure as mentioned in the previous section was followed to optimize S and T by the Levenberg-Marquardt technique. The tide-aquifer interaction datasets used for this analysis were the same as those used for the optimization of S and T following Approach I.

13.3.4 *Estimation of aquifer parameters using Spring and Neap tidal data*

The Spring and Neap tidal data were selected from the annual tidal data (year 2000) of the Pacific Ocean bordering the Konan groundwater basin, Japan, together with the corresponding groundwater-level data at the two tide-affected sites (I-2 and H-5). These tide-aquifer interaction datasets were used to determine aquifer parameters (S and T) following Approach II. The hourly tide-aquifer interaction datasets used for this analysis were of 14–16 September 2000 (Spring tide) and 21–23 January 2000 (Neap tide) for Sites I-2 and H-5 of Konan Aquifer. The results obtained were compared with those using normal tidal event data for evaluating the effect of Spring and Neap tidal data on the estimates of aquifer parameters.

13.4 RESULTS AND DISCUSSION

13.4.1 *Optimal S and T without tidal data resolution (Approach I)*

The optimum values of transmissivity (T), storage coefficient (S), and hence hydraulic diffusivity (T/S) were determined by the Levenberg-Marquardt technique following Approach I at the two sites of Konan Aquifer using three sets of the tide-aquifer interaction data, and at the three sites of Dridrate Aquifer using one set of tide-aquifer interaction data as shown in Table 13.1. Thus, the hydraulic diffusivities (D) based on the optimal S and T were found to range from 0.09 to 0.98 m^2/s at Site I-2 and 0.1 to 1.5 m^2/s at Site H-5, whereas these figures for the three confined sites were found to vary from 9.2 to 25.7 m^2/s.

Moreover, it should be noted from Table 13.1 that the values of root mean square error (RMSE) are very high—ranging from 0.071 to 0.547 m for the unconfined sites and 0.341 to 0.651 m for the confined sites. Relatively large RMSE values are attributed to the inherent nature of the conceptual tide-aquifer model used for optimizing aquifer parameters, i.e. tidal oscillations per se behave as sinusoidal waves.

13.4.2 *Optimal S and T with tidal data resolution (Approach II)*

The results of this analysis are summarized in Tables 13.2 and 13.3 respectively for the unconfined and confined sites. It is apparent from Table 13.2 that the values of RMSE vary from 0.013 to 0.038 m for Site I-2 and from 0.028 to 0.062 m for Site H-5, which are appreciably low. A short-duration dataset of 25 hours (i.e., 1 lunar day: 22–23 February 1999) was analyzed for optimizing S and T and the results were compared with that using relatively long-duration datasets of 75 hours (i.e. 3 lunar days: 22–24 February 2000 and 1–3 March 2000). It is obvious from Table 13.2 that the short-duration dataset yielded optimal S and T values with relatively low RMSE compared to those based on the long-duration datasets for both the unconfined sites. The values of optimal S and T

Table 13.1. Optimal aquifer parameters at the two unconfined and three confined sites following Approach I.

Site	Optimal S	Optimal T (m^2/s)	Optimal D (m^2/s)	RMSE (m)
A) Unconfined sites				
Dataset: 1–3 March 2000				
I-2	0.022	0.002	0.09	0.546615
H-5	0.039	0.06	1.54	0.544542
Dataset: 22–24 February 2000				
I-2	0.031	0.031	0.98	0.107916
H-5	0.01	0.009	0.90	0.526387
Dataset: 22–23 February 1999				
I-2	0.068	0.01	0.15	0.070822
H-5	0.159	0.016	0.10	0.371755
B) Confined sites				
1272/34	0.0002	0.0032	16.0	0.650835
1525/34	0.0035	0.09	25.7	0.340502
235/26	0.000087	0.0008	9.20	0.480897

Table 13.2. Optimal aquifer parameters at the two unconfined sites based on Approach II.

Site	Optimal S	Optimal T (m^2/s)	Optimal D (m^2/s)	RMSE (m)
Dataset: 1–3 March 2000				
I-2	0.02	0.05	2.50	0.034647
H-5	0.01	0.11	11.0	0.062492
Dataset: 22–24 February 2000				
I-2	0.29	0.69	2.38	0.037583
H-5	0.12	0.54	4.50	0.053821
Dataset: 22–23 February 1999				
I-2	0.13	0.37	2.85	0.012874
H-5	0.18	1.53	8.50	0.027782

Table 13.3. Optimal aquifer parameters at the three confined sites based on Approach II.

Site	Optimal S	Optimal T (m^2/s)	Optimal D (m^2/s)	RMSE (m)
1272/34	0.000079	0.013	164.6	0.028315
1525/34	0.0019	0.47	247.4	0.005502
235/26	0.000074	0.0004	5.41	0.034184

based on the short-duration dataset are also quite different from those based on the long-duration datasets; even the results of the two long-duration datasets are not comparable, though the RMSE values are approximately the same for both the sites. Thus, these findings suggest that professional judgment and a proper selection of the size of tide-aquifer interaction datasets is necessary for reliable parameter estimates.

On the other hand, the optimal S values and T values for Dridrate Aquifer are presented in Table 13.3. The RMSE values for three confined sites range from 0.006 to 0.034 m, which are considerably low compared to the corresponding RMSE values in the case of Approach I. As the adequate field data/information are not available for Dridrate Aquifer, no comments on the practical suitability of obtained optimal aquifer parameters could be made in this study.

Considering the results of Approach I and Approach II, it is evident that Approach II yielded both the aquifer parameters (S and T) at reasonably low RMSE values ranging between 0.013 and 0.062 m for the two unconfined sites of Konan Aquifer and between 0.006 and 0.034 m for the three confined sites of Dridrate Aquifer; these RMSE values are much lower than the corresponding RMSE values obtained using Approach I. Thus, relatively low and reasonable RMSE values as such indicate that the aquifer parameters (S, T and D) obtained by Approach II are more accurate and reliable compared to those obtained using Approach I. Therefore, Approach II should be adopted in future studies, which is not difficult to implement in the era of information technology.

13.4.3 *Matching between observed and computed well hydrographs*

While optimizing S and T, it was very difficult to match the calculated well hydrograph with the observed well hydrograph with correct amplitude and correct time lag for both the unconfined sites. The best match between the observed and calculated well hydrographs at Site I-2 for both the datasets was obtained at t (time step) = 1 h. However, the best match between the two well hydrographs at Site H-5 was obtained at t = 1 h for the 1–3 March 2000 dataset and at t = 1.5 h for the 22–23 February 1999 dataset. Similarly, for the three confined sites, the best match between

Table 13.4. Optimal aquifer parameters based on Spring and Neap tidal data II.

Site	Optimal S	Optimal T (m^2/s)	Optimal D (m^2/s)	RMSE (m)
Spring Tidal Dataset: 14–16 September 2000				
I-2	0.09	0.43	4.78	0.100083
H-5	0.011	0.12	10.9	0.122154
Neap Tidal Dataset: 21–23 January 2000				
I-2	0.10	1.02	10.2	0.117406
H-5	0.18	0.51	2.83	0.106441

the two well hydrographs was found at t = 2 h for Site 1272/34, and t = 2.5 h at Sites 1525/34 and 235/26. These findings clearly indicate that for the same amplitude, the observed speed of tidal wave propagation is faster than the model-predicted speed of tidal wave propagation for both unconfined and confined aquifer systems.

13.4.4 *Aquifer parameters based on Spring and Neap tidal data*

To explore the impact of Spring and Neap tidal data on the parameters estimates, the Spring tidal dataset (14–16 September 2000) and the Neap tidal dataset (21–23 January 2000) were also used for optimizing S and T at unconfined Sites I-2 and H-5 following Approach II. The results of this analysis are summarized in Table 13.4.

Clearly, the values of optimal storage coefficient (S) and optimal transmissivity (T) based on the Spring tidal dataset are lower than those obtained based on the Neap tidal dataset at both the sites (Table 13.4). However, the optimal hydraulic diffusivity based on the Spring tidal dataset was found lower (4.8 m^2/s) than that based on the Neap tidal dataset (10 m^2/s) at Site I-2, but it was higher (about 11 m^2/s) at Site H-5 compared to the Neap tidal dataset (2.8 m^2/s). It should be noted that the optimal S and T values based on the Spring and Neap tidal datasets were obtained with relatively high RMSE values compared to those based on the normal tidal dataset following Approach II (Tables 13.2 and 13.4). It is also clear from these tables that the optimal S and T values based on the normal tidal dataset are significantly different from those based on the Spring and Neap tidal datasets. Thus, it is concluded from these results that the estimates of aquifer parameters (S, T and D) are significantly affected by the Spring and Neap tidal datasets. Consequently, a proper selection of the tide-aquifer interaction dataset is indispensable for a reliable estimation of aquifer parameters by the tide-aquifer interaction technique. Based on the present results, the use of Spring and Neap tidal data is not encouraged.

13.5 SUMMARY AND CONCLUSIONS

Adequate knowledge of the hydraulic properties of aquifers viz., transmissivity (T) or hydraulic conductivity (K), storage coefficient (S) and hydraulic diffusivity (D) is essential for all the studies related to groundwater quantity and quality, including modeling. The main intent of this paper is to highlight the tide-aquifer interaction technique and to demonstrate its efficacy in estimating aquifer parameters using the tide-aquifer interaction datasets of two unconfined and three confined sites located in two different coastal groundwater basins. The Levenberg-Marquardt nonlinear optimization technique was used to optimize S and T considering two approaches of using tidal data for such an analysis.

Following Approach I (without tidal data resolution), the optimal storage coefficient (S) and the optimal transmissivity (T) values for the two unconfined sites were obtained. The optimal hydraulic diffusivities based on these optimal S and T values were found to vary from 0.09 to 0.98 m^2/s for

Site I-2 and from 0.1 to 1.5 m^2/s for Site H-5. The optimal S and T values were also determined for the three confined sites, and the optimal hydraulic diffusivities were found in the range of 9 to 26 m^2/s. Furthermore, aquifer parameters (S, T and D) for the two unconfined sites were also determined following Approach II (with tidal data resolution). In this case, the optimal hydraulic diffusivities based on the optimal S and T were found to range from 2.4 to 2.9 m^2/s for Site I-2, and from 4.5 to 11 m^2/s for Site H-5. Similarly, for the three confined sites, the optimal S and the optimal T values were found, which resulted in the optimal hydraulic diffusivity values ranging between 5.4 and 247.4 m^2/s.

A comparison of the optimization results based on Approach I and Approach II revealed that Approach II yielded aquifer parameters (S and T) with reasonably low RMSE values for both the unconfined and confined sites, which are significantly lower than the corresponding RMSE values in the case of Approach I. Thus, relatively low RMSE values suggest that the aquifer parameters (S, T and D) obtained using Approach II are more accurate and reliable for both the aquifer systems compared to those based on Approach I. Therefore, Approach II is strongly recommended for future studies.

Moreover, the tide-aquifer interaction datasets corresponding to Spring and Neap tidal events yielded completely different S and T values with relatively high RMSE at both the unconfined sites compared to the dataset of normal tidal events. Therefore, the tide-aquifer interaction data must be selected carefully for the estimation of aquifer parameters by the tide-aquifer interaction technique. Finally, it is concluded that a judicious use of the tide-aquifer interaction technique is indispensable for obtaining accurate and reliable hydraulic parameters of coastal aquifer systems.

ACKNOWLEDGEMENTS

The authors are very thankful to Dr. Y. Fakir at the Department of Geology, Semlalia Faculty of Sciences, Morocco for providing the tide-aquifer interaction data of a confined aquifer system. The first author gratefully acknowledges the research fellowship awarded by the Alexander von Humboldt Foundation, Germany. The authors appreciate the meticulous comments of Prof. D. Chandrashekharam on the manuscript. We are thankful to the organizers of the 2nd International Groundwater Conference and the editors of the volume especially Dr. Prosun Bhattacharya for the editorial assistance.

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CHAPTER 14

Stochastic modeling of groundwater discharge for hydrological drought forecasting

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ABSTRACT: The modeling and understanding of temporal variability of groundwater discharge are important with respect to efficient water resources management, especially in low flow seasons. Baseflow, which is a major component of streamflow, can be considered as groundwater inflow or discharge. In this study, two stochastic modeling approaches, seasonal autoregressive integrated moving average (ARIMA) and Thomas-Fiering (TF) models, are used to model monthly baseflow time series in six different basins in the southwest of Iran. One-step-ahead forecasts for the test portion of the time series are generated using the selected set of candidate models. The major objective of this research is to understand the stochastic behavior of baseflow time series and develop reliable forecasting models. The results show that the ARIMA (1,0,0)(0,1,1) model provides the most accurate forecasts for the particular basins and this model adequately describes the structure of the stochastic behavior of the monthly groundwater discharge time series.

14.1 INTRODUCTION

As demands for additional water continue to increase, the need for greater efficiency in the modeling and forecasting of water availability becomes increasingly important. The modeling and understanding of the temporal variability of groundwater discharge are important for achieving efficient water resources management, especially in low flow seasons. Stochastic models that have been developed for modeling and forecasting seasonal time series are very useful within the field of water resources planning in arid and semiarid regions. Time series analysis has been widely used in the field of hydrology and water resources for simulation and forecasting (Hipel et al. 1977a, Boes & Salas 1978, Cline 1981, Noakes et al. 1985, Hipel 1993, Hipel & McLeod 1994, Ahmad et al. 2001, Huang & Chan-Hilton 2004).

Baseflow, which is a major component of streamflow, can be considered as groundwater inflow or discharge. To model the temporal variability of groundwater discharge at the basin scale, baseflow data series can be studied. In previous hydrological studies, the recursive digital filter with a parameter value of 0.925 was used to generate monthly baseflow data series (Nathan & McMahon 1990, Ghanbarpour et al. 2007). The application of an automated recursive digital filter is convenient to use, accurate, consistent and does not need subjective judgment. This helps to ensure the applicability of the baseflow values for future operational hydrological drought forecasting.

In this study, two stochastic modeling approaches, seasonal autoregressive integrated moving average (ARIMA) and Thomas-Fiering (TF) models are used to model monthly baseflow time series in six different basins in the southwest of Iran. The background theory of these two techniques is briefly explained. For construction of the ARIMA models, the identification, estimation and diagnostic check stages of model development are followed (Hipel et al. 1977b). The Akaike information criteria (AIC) is utilized to select the most appropriate model from the candidate set of models (Akaike 1974). Each of the time series is split into two sections: one part is used for model calibration and the other for verification of the accuracy of the one-step-ahead forecasting results. The models are developed based on calibration data set. One-step-ahead forecasts for the verification part of data series are generated using the selected set of candidate models. The forecasts errors are examined using two error estimation criteria: the mean absolute error (MAE) and root mean square error (RMSE). One objective of this research is to gain a better understanding of the stochastic behavior of groundwater discharge in different basins. The methodology proposed in this paper can be used for forecasting of baseflow time series in similar basins during drought seasons. Improvement of the skill of baseflow forecasting, dramatically, affects water management efficiency, crop production, hydropower production, drought forecasting and environmental sustainability of arid regions.

14.2 DATA AND TIME SERIES MODELS

14.2.1 *Study area*

The Karun basin studied in this project is located in the southwest of Iran from 30°16′ to 32°39′N latitude and 49°33′ to 52°0′E longitude (Fig. 14.1). The basin covers an area of approximately 24,141 square kilometer, with elevations between 800 and 4400 m. The mean annual precipitation of the area is about 650 mm. The Poleshaloo hydrometric station, at the outlet of the basin with an elevation of 800 m, has recorded an average annual runoff of 341.9 m^3/s. The low flow season generally begins in July and ends in November. In this research, six basins with different characteristics in the study area are investigated. Details of the basins under study including average annual runoff, area, and the elevations of hydrometric stations, are listed in Table 14.1.

14.2.2 *Data*

The runoff data employed in this study consist of monthly flows, in cubic meter per second at the six sub-basins depicted in Figure 14.1, for a period of thirty years from 1969 to 1998. The

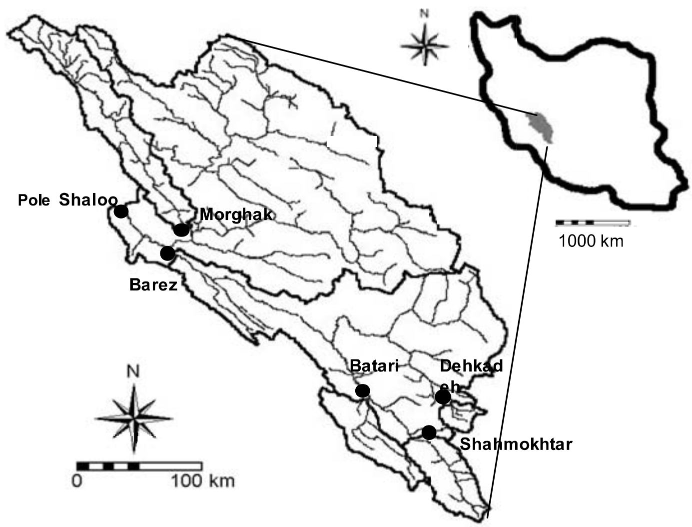

Figure 14.1. Location of the Karun Basin and its sub-basins in Iran.

technique for continuous baseflow separation using a recursive digital filter described by Nathan & McMahon (1990) was applied to compute the monthly baseflow time series in the study area (Ghanbarpour et al. 2007). The baseflow data were split into two sets, namely, the calibration and verification periods. The baseflow data from 1969 to 1993 are used for calibration and the data from 1994 to 1998 for verification of the models. Figure 14.2 shows the temporal variation of the monthly runoff and baseflow for the case of the Pole Shaloo hydrometric station for the 1979–80 water year.

14.2.3 *Time series analysis*

Time series models can be used to describe the stochastic structure of the time sequence of hydrological data. In this study, two stochastic modeling approaches, the ARIMA (Box & Jenkins 1976, Hipel & McLeod 1994) and TF (Thomas & Fiering 1962) models are used to model monthly baseflow time series. An ARIMA model can be constructed using a combination of moving average (MA) and autoregressive (AR) processes, after differencing the data. The MA operator is written as:

$$\theta(B) = 1 - \theta_1 B - \theta_2 B^2 - \cdots - \theta_q B^q \tag{14.1}$$

Table 14.1. Details of the sub-basins under study.

Basin	Area (km^2)	Average annual runoff (m^3/s)	Elevation (m)
Barez	8999	126.8	815
Batari	885	16.6	1560
Dehkadeh	200	5.3	2220
Morghak	2146	74.3	860
Pole Shaloo	24141	341.9	800
Shahmokhtar	1187	22.2	1730

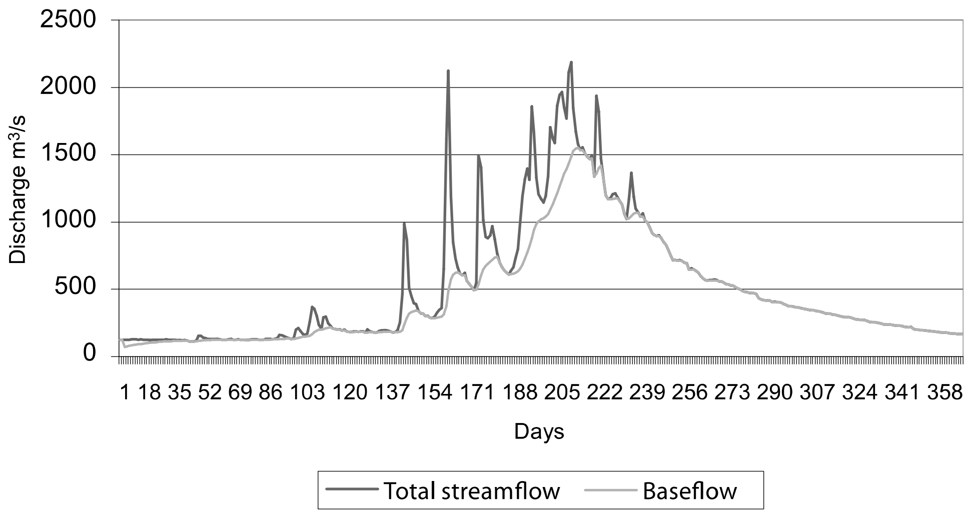

Figure 14.2. Temporal variation of the total streamflow and separated baseflow at the Pole Shaloo hydrometric station for the water year of 1979–80.

where q is the order of the MA operator, θ_j, $j = 1, 2, \ldots, q$ are the MA parameters, and B is the backward shift operator such that $BZ_t = Z_t - 1$. The AR operator is written as:

$$\phi(B) = 1 - \phi_1 B - \phi_2 B^2 - \cdots - \phi_p B^p \tag{14.2}$$

where p is the order of the AR operator, and ϕ_i, $i = 1, 2, \ldots, p$, are the nonseasonal AR parameters. In the practical hydrological applications, it is possible to include both AR and MA terms to obtain a parsimonious model. Box & Jenkins (1976) describe such a composite model by expressing the deviation of a variate from its mean as a finite weighted sum of previous deviations plus a finite weighted sum of random variables plus a random shock (Haan et al. 1982). The result of this combination is an autoregressive integrated moving average (ARIMA) model. The stationary nonseasonal ARIMA model for a set of equispaced measurements, $Z = \{Z_1, Z_2, \ldots, Z_n\}^T$, can be written as:

$$\phi(B)(1 - B)^d (Z_t - \mu) = \theta(B) a_t \tag{14.3}$$

where t is discrete time, and μ is the mean level of the process. Differencing removes nonstationarity in a time series. When differencing is not required, the model is referred to as an ARMA model. Construction of ARIMA models are conducted based on the three stages of model building: identification, estimation and diagnostic checking, which is discussed by Hipel et al. (1977b) as well as other authors. Moreover, one can incorporate seasonal MA and AR operators into the ARIMA model to handle the seasonal components of the model. Finally, both nonseasonal and seasonal differencing can be used to remove nonseasonal and seasonal nonstationarity, respectively.

Thomas & Fiering (1962) used the Markov chain model for generating monthly flows by taking into consideration the serial correlation of monthly flows. The model uses a month to month correlation structure. As Patra (1998) states, it can be assumed that the July flows are always dependent on June river flow values. A TF model accounts for the effects of seasonality on the variability of the data by considering month to month variation in the average values and coefficient of correlation between the data (Clarke 1984, Ahmad et al. 2001). The model uses the following equation, for example, for the month of January:

$$Q_{Jan} = Q_{av_{Jan}} + b_{D-J}(Q_{Dec} - Q_{av_{Dec}}) + t_i S_j (1 - r_{D-J}^2)^{0.5} \tag{14.4}$$

where,

Q_{Jan}, Q_{Dec}: discharge for January and December, respectively
$Q_{av_{Jan}}, Q_{av_{Jan}}$: mean monthly discharge for January and December, respectively
b_{D-J}, r_{D-J}: constant and correlation coefficient between December and January
t_i: the random independent variate with zero mean and unit variance in year i
S_j: standard deviation for January

The forecasting performance of the models are compared based on the MAE and RMSE criteria:

$$MAE = \frac{1}{n} \sum_{i=1}^{n} [z(x_i) - \hat{z}(x_i)] \tag{14.5}$$

$$RMSE = \sqrt{\frac{1}{n} \sum_{i=1}^{n} [z(x_i) - \hat{z}(x_i)]^2} \tag{14.6}$$

where,

n: number of data observations
$z(x_i)$: observed value and
$\hat{z}(x_i)$: estimated value.

14.3 RESULTS AND DISCUSSION

Three modeling stages consisting of identification, estimation of model parameters and diagnostic checking of the estimated residuals (Box & Jenkins 1976, Hipel et al. 1977b, Hipel & McLeod 1994) are followed for fitting ARIMA models to the time series of baseflow in the basins under study. To determine the possible persistence structure in the series, the autocorrelation function (ACF) and the partial autocorrelation function (PACF) are examined. The examination of these two function reveals that the data have seasonality, which requires one order of seasonal differencing for removal.

Figures 14.3 and 14.4 show the ACF and PACF, respectively for the seasonally differenced time series for the Batari basin. As can be seen in Figure 14.3, the ACF does not truncate but rather damps out, suggesting that a nonseasonal AR parameter is needed in the model. Because the PACF truncates after lag 1 (Fig. 14.4), one nonseasonal AR parameter should be included in the model. There is a significant value at lag 12 that indicates the presence of a seasonal MA term in the model. Similarly, all possible models are identified for the other time series. The approximate maximum likelihood estimates (MLEs) for the model parameters are obtained by employing the unconditional sum of squares method, suggested by Box & Jenkins (1976).

For checking the adequacy of the model fitted to the time series, the Q-statistics and residual autocorrelation function (RACF) tests (Hipel et al. 1977b) are employed. A plot of the RACF for the Batari basin is displayed in Figure 14.5, which reveals that the estimated values fall within the 5% significant interval. Therefore, the residuals are white noise. The Akaike Information Criteria (AIC) (Akaike 1974) is used to select the best fitted model out of the various competing models. In the study area, the ARIMA (1,0,0)(0,1,1) model adequately describes the structure of the stochastic behavior of the baseflow data series. In the notation for the ARIMA model, the three entries within the first set of brackets stand for the orders of the nonseasonal AR, differencing, and MA operators, respectively, while the three numbers contained inside the second set of brackets give the orders of the seasonal AR, differencing, and MA operators, respectively.

In this research, within an overall TF, a separate model is developed for each month for forecasting the baseflow time series. A sample calibrated TF model for the month of January in the Batari Basin can be represented as:

$$Q_{Jan} = \overline{Q}_{Jan} + 1.53(Q_{Dec} - \overline{Q}_{Dec}) + 15.95\sqrt{1 - (0.54^2)} \tag{14.7}$$

All of the time series of baseflow are split into two sections for use in calibration and verification of forecasting ability. The model parameters are estimated the calibration portion of the data sets. Baseflow data from 1969 to 1993 are used for calibration and data from 1994 to 1998 for forecasting

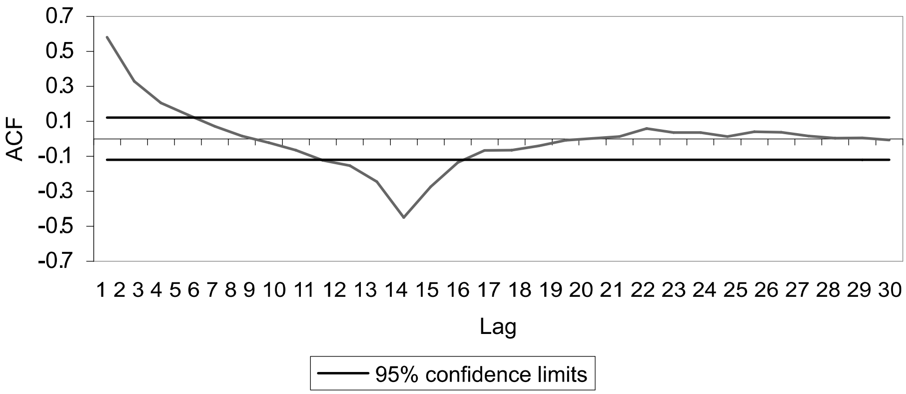

Figure 14.3. ACF for the seasonally differenced baseflow data series for the Batari Basin.

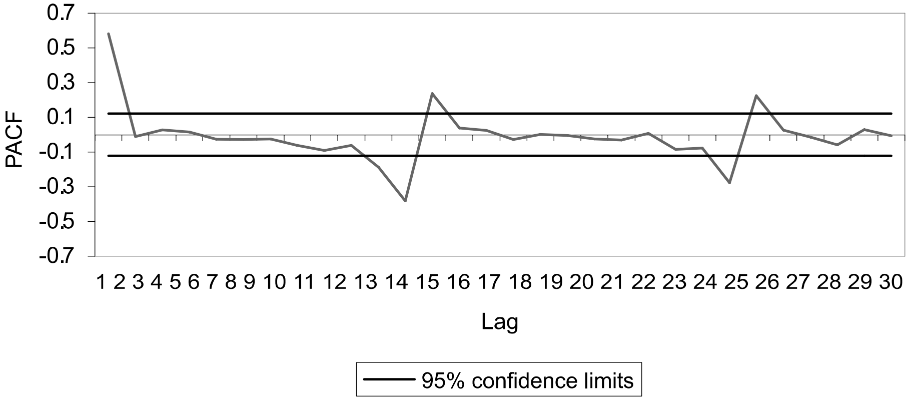

Figure 14.4. PACF for the seasonally differenced baseflow data series for the Batari Basin.

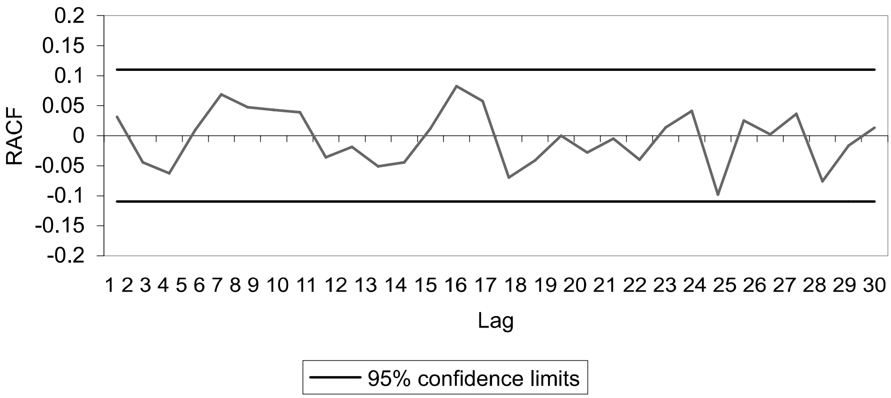

Figure 14.5. RACF for the best ARIMA (1,0,0)(0,1,1) model for the Batari Basin.

Table 14.2. Summary of the results of the forecasting performance for the baseflow forecasting.

	ARIMA		TF	
Basin	MAE	RMSE	MAE	RMSE
Barez	46.49	69.79	50.23	66.2
Batari	10.15	15.89	9.68	13.58
Dehkadeh	2.13	3.46	2.66	3.44
Morghak	18.66	30.14	30.36	40.75
Pole Shaloo	86.55	129.41	128.15	164.54
Shahmokhtar	13.78	22.99	13.44	18.84

verification of models. One-step-ahead forecasts for the verification part of the time series are generated using the selected set of candidate models.

As a comparative study, the MAE and RMSE are used to compare the performance of the ARIMA and TF models for forecasting purposes. Table 14.2 gives the error estimates obtained for

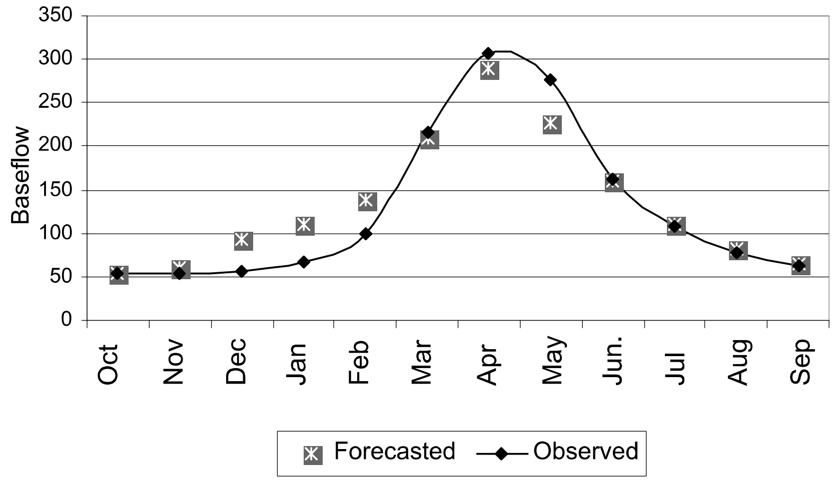

Figure 14.6. Monthly baseflow data series of the Barez Basin and the one-step ARIMA forecasts for the water year of 1995–96.

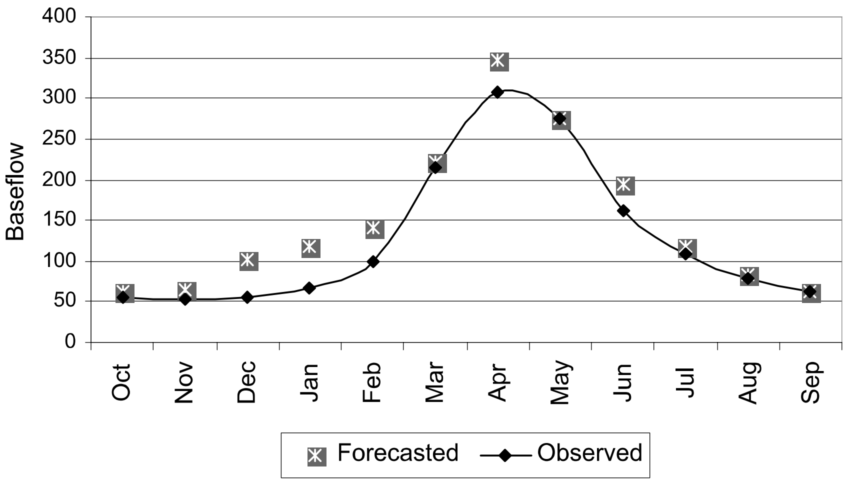

Figure 14.7. Monthly baseflow data series of the Barez Basin and the one-step TF forecasts for the water year of 1995–96.

each basin for each of the two models. As can be seen in Table 14.2, ARIMA models generate more accurate forecasts than TF models, in general. For the cases of the Barez, Dehkadeh, Morghak and Pole Shaloo Basins, ARIMA models provide more accurate forecasting results in terms of MAE and RMSE for the verification part of the time series. However, the forecasting results of the TF models are slightly better than the results for the ARIMA model for the Batari and Shahmokhtar Basins (Table 14.2). As a result, ARIMA models are recommended for application in baseflow or groundwater discharge time series forecasting in the study area.

Constructing a TF model requires the estimation of model parameters for each month. In particular, to build a TF model for a given month, the mean monthly discharge, correlation coefficient of the two proceeding months, and standard deviation for monthly discharges of that particular month, must be estimated. Operational hydrological forecasting systems should be based on more simple and realistic methods, which can be updated efficiently and quickly. Moreover, the simple models are more economical than more complicated and time consuming procedures. The foregoing are distinct advantages of an ARIMA model over a TF model for the case of the forecasting applications.

To show the forecasting ability of the ARIMA and TF models, the similarity of observed and forecasted values of baseflow can be graphically examined. Figures 14.6 and 14.7 show a time series plot of baseflow and their forecasts using ARIMA and TF models, respectively, for the water year (October to September) of 1995–96 for the Barez Basin. As can be seen in Figures 14.6 and 14.7, forecasting performance during the low flow seasons are reliable and accurate, especially for the ARIMA model. As depicted in Figure 14.6, the ARIMA (1,0,0)(0,1,1) model for that particular basin provides accurate forecasts during the low flow season, from June until December. In this case, the ARIMA model tends to estimate the groundwater discharge during peak flow season accurately, with slightly under-estimation, especially in May. An overestimation of groundwater discharge can be seen from December until February in this particular water year. For instance, an accurate forecast by a TF model can be seen during low flow season in the Barez Basin (Figure 14.7). In spite of the performance of the ARIMA model, the TF model in this case tends to slightly overestimate groundwater discharge during the spring season. An overestimation of groundwater discharge can be seen from December until February in this particular water year, which is similar to the ARIMA model performance.

14.4 CONCLUSIONS

Error estimates of forecasts from two different approaches, the ARIMA and TF models, are compared and it is found that ARIMA models provide satisfactory forecasting results for most of the basins under study. The ARIMA models, therefore, are recommended to forecast baseflow time series for employment in drought management and water resources planning during low flow seasons.

Operational hydrological forecasting systems should be based on simple and realistic models with efficient updating procedures. Therefore, the ARIMA model has the advantage over the TF model from the point of view of model parsimony. The ARIMA model allows the prediction of baseflow using the stochastic behavior of the historical time series, which eliminates the necessity of estimating the many parameters required in a TF model. Before fitting ARIMA and TF models, a recursive digital filter method is used to extract the baseflow time series. Application of an automated recursive digital filter is convenient to use in practice. Finally, the selected basins in this study have different physiographic characteristics and vary in flow magnitude and area from small to large river basins. The stochastic behavior of groundwater discharge or baseflow time series in the basins under study can be adequately modeled by ARIMA (1,0,0)(0,1,1) models from the large set of possible ARIMA models.

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CHAPTER 15

Influence of hysteresis in modeling of LNAPL migration through non-homogeneous binary porous media

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ABSTRACT: In this paper, the influence of soil heterogeneities and hydraulic Hysteresis as two natural phenomena on simulation of migration and redistribution of heptane's as a Lighter than water Non-Aqueous Phase Liquid (LNAPL) is demonstrated using a hysteretic three-phase Water-NAPL-Air numerical k-S-P model named "NAPL". The 2D simulations are conducted in a number of dissimilar granular porous media, which consist of a homogeneous medium, and a couple of non-homogeneous binary media that have different characteristics regarding the assignment of divergent soil hydraulic properties and spatial configuration of heterogeneities for each one. In all considered cases vertical and horizontal dimensions of the modeling domain are set respectively to the value of 0.7 m and 0.5 m where a constant spatial step of 1.25 cm used in spatial discretization of both directions. Boundary conditions are imposed specifically to investigate hysteretic k-S-P path behavior. Analysis of the acquired data shows that connivance of Hysteresis or soil heterogeneities through numerical modeling process may lead to significant discrepancies among output results, especially when phase imbibition-drainage history is more sophisticated, although hysteretic modeling of multiphase flow in porous media considering soil heterogeneities needs much more computational requirements and storage regarding the high degree of non-linearity which is accompanied with hysteresis.

15.1 INTRODUCTION

Over the past century, chemicals flow and transport through multiple phases phenomena, as one of challenges against engineering design of industrial processes, have been investigated by many researchers in particular for extraction of crude oil from ground reservoirs. Besides, during the recent three decades, there has been a dramatic increase in considerations towards the multi-phase flow processes with regard to ascending concerns about groundwater resources pollution with non-aqueous phase liquids. In general, a large variety of toxic contaminants, which threaten subsurface media, are highly volatile and immiscible in water, nevertheless, they show slight solubility in water that goes beyond water quality standards. Doubtlessly, implementation of effective refinement and recovery methods for restoring polluted aquifers depends upon correct understanding of such contaminants' characteristics and behavior in subsurface. According to reports from various researchers compromising either hysteresis phenomenon (Kool & Parker 1987) or soil heterogeneities (Keuper & Frind 1991, Miller et al. 1998), through simulations may have noticeable influence on intactness and accuracy of output results.

15.1.1 *Soil heterogeneities effects*

In groundwater literature, soil heterogeneities are verified as imperative factors having crucial influence on porous media characteristics through the earlier studies (Helmig 1997, Miller et al. 1998, de Nee 2000). Basically, regarding the effects of soil heterogeneities on migration of NAPLs fate and transport in subsurface, three major categories can be named as the following: a) magnification of the size of the contamination plume, b) increase in contaminant's infiltration depth, and c) making ineffective barriers from confining fine-grained soil layers or clay liners.

15.1.2 *Hydraulic hysteresis*

The magnitude of unsaturated soil hydraulic conductivity, which is dependent to phase effective saturations, and soil matric suction must be known in order to accurately simulate the NAPL migration process. The constitutive relations among matric suction-phase saturation-relative hydraulic conductivity terms are different for imbibition and drainage paths. These so-called differences are identified as hydraulic hysteresis phenomenon, and for the most of various soil types the soil retention curves show hysteretic characteristics.

15.2 MATHEMATICAL MODEL

A two-phase model that is described in this section is adapted for the three-phase case considering the idealizations that allows a three-phase model to be generated using two-phase data. Regarding the fact that phase saturation, pressure head, and relative permeability of each phase are the unknowns of the problem of flow in separate phases, we are bound to find six independent equations for each node inside the simulation domain. Considering the mass-balance criterion we have the following equations as follows:

$$\frac{\partial}{\partial t}(e\rho_\alpha S_\alpha) = \nabla \cdot \left[k\frac{\rho_\alpha k_{r\alpha}}{\mu_\alpha} \cdot (\nabla P_\alpha - \rho_\alpha g \nabla Z)\right] + Q_\alpha \tag{15.1}$$

$$S_W + S_{NW} = 1 \tag{15.2}$$

where P_α, S_α, and $k_{r\alpha}$ represent phase pressure, saturation, and relative permeability, respectively, k and e show the intrinsic permeability and effective porosity of the porous medium, and the fluid properties of present fluids are defined with each phase density; ρ_α and viscosity; μ_α, as well as g and Q_α that are gravity and sink/source terms, positive Z-axis lies along the downward vertical direction, and subscript α is representative of flow phases accepting values either N or NW in terms of wetting or non-wetting phases. Furthermore, two other equations can be obtained from constitutive interrelations among relative permeability-phase saturation-capillary pressure parameters using hysteretic relationship outlined by Luckner et al. (1989) based on Van Genuchten (1980) S-P_c equation, as the following:

$$h_{c(f)} = \left[(S_{e(f)})^{-\frac{1}{m}} - 1\right]^{1/\eta} \cdot (\alpha_{(f)})^{-1} \tag{15.3}$$

$$S_{e(f)} = \frac{S_W - S_{r(f)}}{S_{S(f)} - S_{r(f)}} \tag{15.4}$$

The parameter $S_{e(f)}$ in eqn. (15.3) shows the wetting phase effective saturation defined through eqn. (15.4), $h_{c(f)}$ represents the capillary head between the two phases of flow, α and η are Van Genuchten S-P curve scaling parameters, where $S_{S(f)}$ and $S_{r(f)}$ are indicative of phase maximum and residual saturations, the parameter m is defined to be equal to $1 - 1/\eta$, and the subscript f is set to define the soil retention curve type in concert with the flow-path history. The relative permeability-saturation relation based on the Mualem's statistical model (Mualem 1976) which relates the water phase relative permeability to water saturation at each node through a series of theoretical steps, is written as shown below:

$$k_{rW}(S_W) = (S_{eW})^{\zeta}\left\{1-\left[1-(S_{eW})^{\frac{1}{m}}\right]^m\right\}^2 \quad (15.5)$$

where the parameter ζ is a pore connectivity parameter for water-phase and is the same as defined in eqn. (15.4). Finally, assuming that the whole pore spaces are filled with the mobile wetting and non-wetting phases, the eqn. (15.5) can be modified as:

$$k_{rW} + k_{rNW} = 1 \quad (15.6)$$

As mentioned before, the so-called three-phase conceptual model used for simulations is generated based on the described two-phase model as discussed in the following section.

15.3 NUMERICAL MODEL

The NAPL Simulator, developed by the United States Environmental Protection Agency (US-EPA, 1997), is implemented in order to perform the simulations. It is capable of modeling the three-phase migration and distribution processes in 3D space accounting for both soil physical heterogeneities and hysteresis in terms of phase entrapment/release and contact angle (capillary) effects, using finite elements discretization approach with rectangular elements.

15.3.1 *Model verification*

In order to demonstrate the capability of NAPL simulator in modeling the hysteretic three-phase processes a laboratory experiment that is fully described by Van Geel & Sykes (1994) by means of which the authors tested their own developed hysteretic model is re-simulated. In addition, to verify the model applicability to heterogeneous porous medial the NAPL simulator is tested using a hypothetic scenario described by Ataie-Ashtiani et al. (2001). In the first case, an LNAPL spill is subjected to a 1.5 m in length and 1.145 m in height experimental box where 2 liters of n-heptane with density and viscosity of respectively 685.8 kg/m^3 and $4.09 * 10^{-4}$ Pa.s is allowed to infiltrate through a variably water-saturated well sorted silica sand medium under a 3.0 cm prescribed head boundary condition (equal to 2.052 cm H_2O) from 10 cm wide source area at the top center of cell during a 18.67 min period, after that injection stops and LNAPL plume is left to redistribute up to t = 3000 s, meanwhile lateral boundaries are subjected to hydrostatic water-phase pressure condition through which water-phase outflow is permitted. Hydraulic properties of sand #1 that constitutes the porous medium are presented in Table 15.1. Model output results together with Van Geel and Sykes' data are shown in Figure 15.1.

As a second case and to investigate the appropriateness of the model in simulating of heterogeneous media related problems, a hypothetical example by Ataie-Ashtiani et al. (2001) is modeled. Domain dimensions and ruling boundary conditions are as shown in Figure 15.2, where the designated patterns for distribution of heterogeneities can be seen, too. Sands #2 and #4 with the given properties in Table 15.1 are used as coarse- and fine-grained sand. The computational domain is discretized using nodal spacing of 1.0 and 0.5 cm, respectively, in horizontal and vertical directions; a maximum time step equal to 5 s is also used throughout simulations. PCE is released into initially water-saturated medium through a 10 cm wide area at the center of the top boundary with constant flux of $3.15 * 10^{-4}$ m/s. As shown in Figure 15.3, convincing agreement between acquired and existing data is obtained.

Table 15.1. Properties of the sandy media used during the simulations.

Property	Units	Sand #1 VG&S	Sand #2 #16 Silica	Sand #3 #25 Silica	Sand #4 #50 Ottawa
Intrinsic permeability k	m^2	$1.0 * 10^{-11}$	$5.04 * 10^{-10}$	$2.05 * 10^{-10}$	$5.0 * 10^{-11}$
Porosity n	–	0.374	0.4	0.39	0.4
Entry pressure P_d	m	–	0.0377	0.0443	0.135
Pore size distribution index λ	–	–	3.86	3.51	2.49
van Genuchten parameters					
η	–	6.49	6.3	6.2	4.5
α_d	m^{-1}	2.03	24	20	6.4
α_i	m^{-1}	2.71	40[A]	30[A]	–
Water-phase residual saturation S_{rW}	–	0.17	0.078	0.069	0.098
Oil-phase residual saturation as wetting-phase S_{rNW}	–	0.18	0.12[A]	0.11[A]	–
Oil-phase residual saturation as non-wetting-phase S_{rNNW}	–	0.205	0.12[A]	0.11[A]	–
Air-phase residual saturation S_{rA}	–	0.18	0.10[A]	0.10[A]	–

[A]Assumed values.

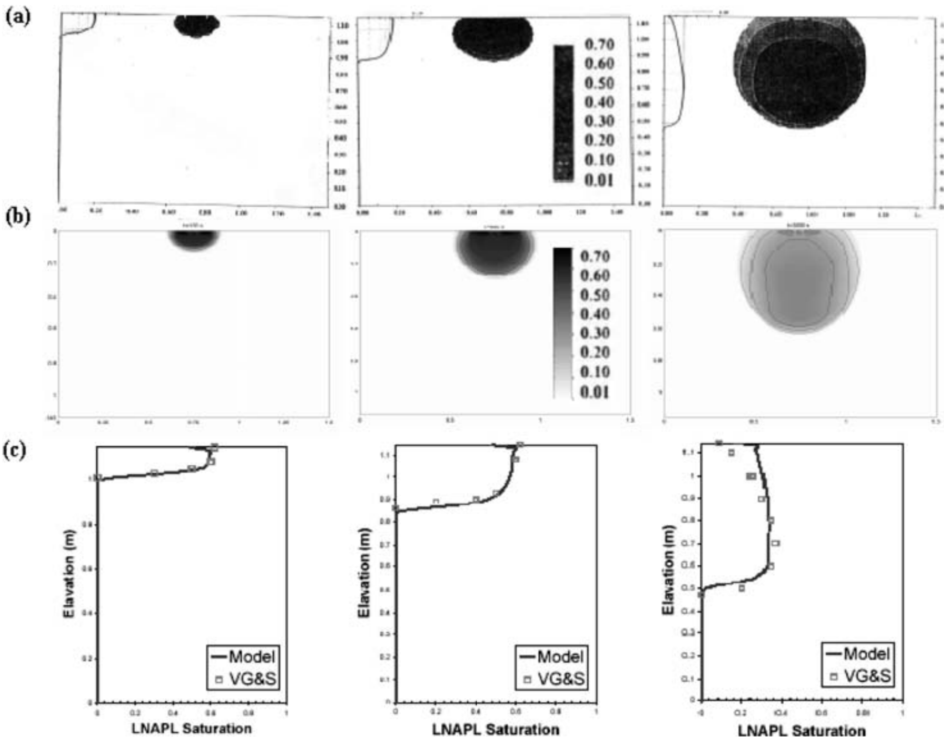

Figure 15.1. a) LNAPL saturation reported by Van Geel & Sykes (1994), b) simulated LNAPL distributions, and c) LNAPL saturation profiles along the vertical center-line of the simulation domain, after 120, 600, and 3000 s.

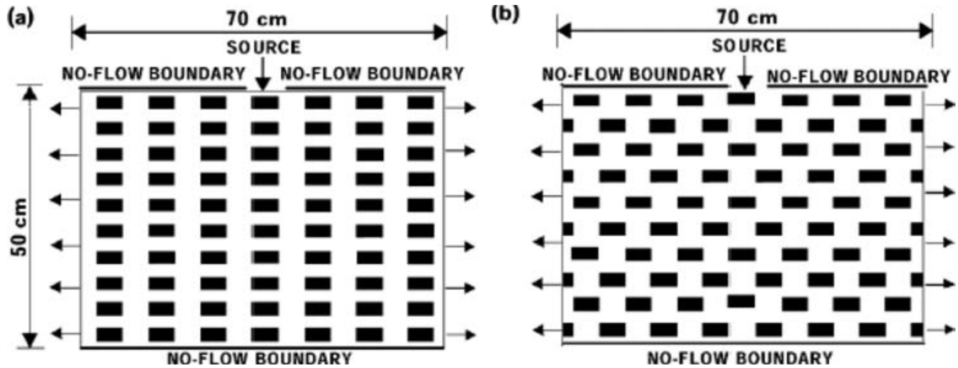

Figure 15.2. Two periodic patterns of heterogeneity. a) Straight P and b) Staggered P. The white background represents the coarse-grained sand while black areas designate fine-grained sand.

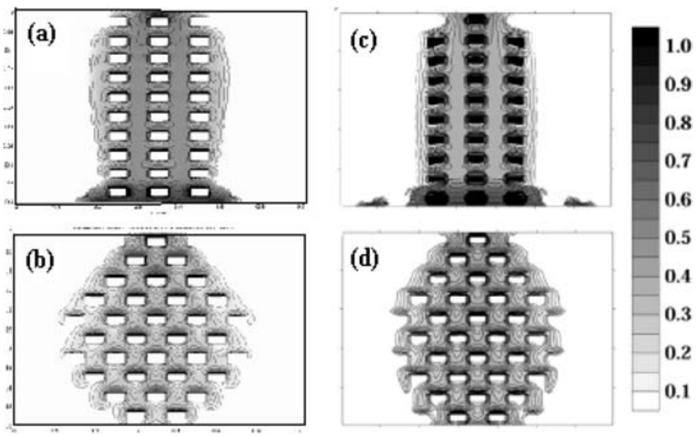

Figure 15.3. Model results a) & b), and existing results from Ataie et al. (2001), c) & d), for straight and staggered patterns at t = 500 s, respectively.

15.4 INFLUENCE OF HYSTERESIS AND SOIL HETEROGENEITIES

In this section we have used the porous media defined in the previous section (Fig. 15.2) to study the effects of physical non-homogeneities of soil on movement of heptane, as one aspect of lighter than water-NAPL species, and the sensitivity of model results towards the hysteresis phenomenon. Prior to release of LNAPL, simulations are started with two-phase flow of water and air where the air-phase penetrates into medium as a result of an imposed suction field caused by gradual falling of water table down to 35 cm below the top boundary, simulations continue until steady-state condition is attained. Afterwards, using the attained initial conditions 6.5 kg of heptane is injected under constant flux of $3.15 * 10^{-4}$ m/s into the porous medium during three identical periods of 100 s between each of which the infiltrated plume is allowed to redistribute for an interval which lasts for 100 s, and after the last injection period plume redistribution process will be simulated before it stops at t = 600 s. Sand #2 (Table 15.1) is used as the background coarse-grained sand while sand #3, which is slightly different in its characteristics from host sand, represents the fine-grained material. Letting heptane's properties be as defined in section 15.3.1, the phase pair surface tension values as reported by Adamson & Gast (1997) are considered to be 72.6, 50.2, and 19.7 dyn/cm for air-water, heptane-water, and air-heptane, respectively. A constant nodal spacing of 1.25 cm in both vertical and horizontal directions is used for spatial discretization of the simulation domain, while a dynamic time-step, is set to vary between $1.0 * 10^{-3}$ and 1.0 s. As indicated in Table 15.2, in order to investigate the consequences of implementation of monotonic or history-dependent soil retention curves and soil non-homogeneities into the modeling process, 6 distinct cases are considered.

The LNAPL distribution profiles obtained from the simulations are illustrated in Figures 15.4 to 15.6, where the left half of each figure shows the non-hysteretic results, and the hysteretic outputs are shown on the other half. From a brief look toward the model outputs resulted from the

Table 15.2. Summary of performed simulations.

Run#	Medium	Model
1	Homogeneous	Hysteretic
2	Straight P.	”
3	Staggered P.	”
4	Homogeneous	Non-hysteretin
5	Straight P.	”
6	Staggered P.	”

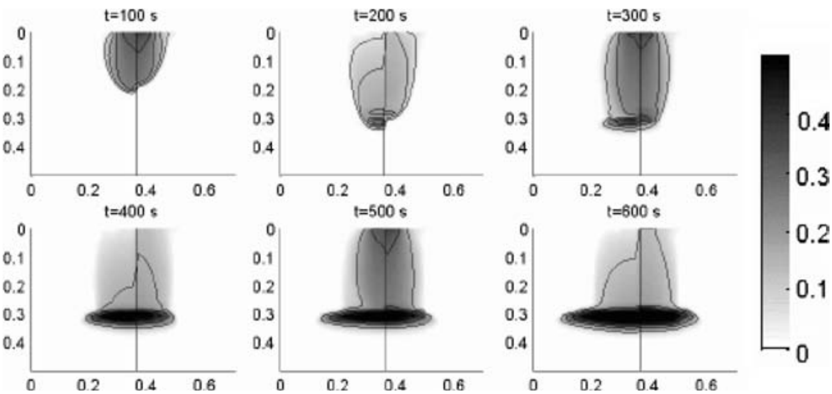

Figure 15.4. Predicted LNAPL saturation profiles for the homogeneous medium.

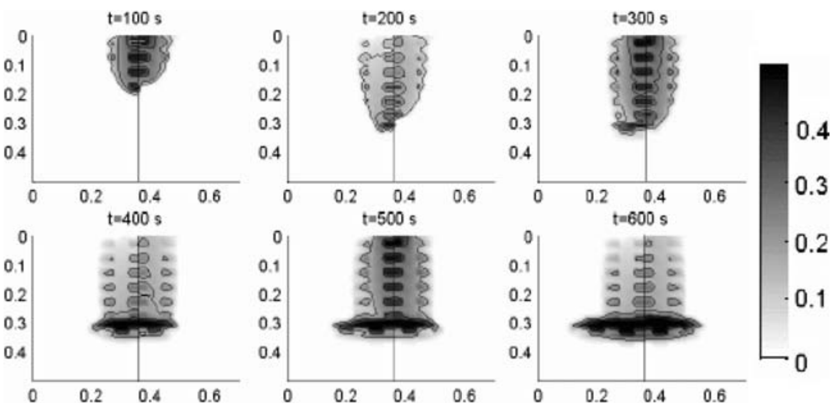

Figure 15.5. Predicted LNAPL saturation profiles for the non-homogeneous medium, with straight distribution pattern of heterogeneities.

non-hysteretic code, it seems that the non-wetting–phase migration through the media is more easily allowed during the early times of simulation and before the LNAPL plume reaches the capillary fringe after which slightly larger transverse distributions and pooling areas has been observed over the water table located at the depth of 35 cm below the top boundary. Besides, regarding the Figures 15.5–15.6, the heterogeneities wherever existed culminate in local pooling areas and act as small storage units of released hydrocarbon that persist as long-term contamination sources in the porous media. These observational differences between hysteretic and non-hysteretic results can be initially taken into account as a primary indication of outputs sensitivity in respect with the hysteresis phenomenon and soil heterogeneities that will be further discussed in the following section.

As shown in Figure 15.7, non-hysteretic model predicts deeper downward penetration magnitudes for the plume of contamination in both homogeneous and non-homogeneous media in comparison with the hysteretic code (see the red arrows), however, simulations results suggest even almost larger differences when heterogeneities are taken into account. Moreover, non-hysteretic model results that is distinguished by the bold green arrows is indicative of higher non-wetting-phase saturation above the water table, specifically during redistribution intervals.

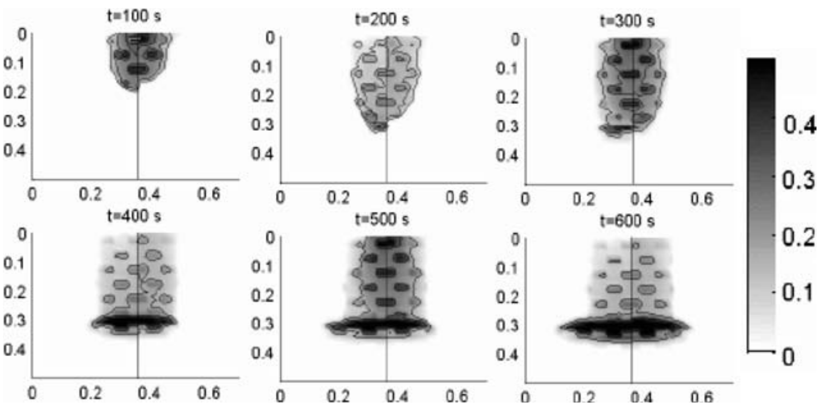

Figure 15.6. Predicted LNAPL saturation profiles for the non-homogeneous medium, with staggered distribution pattern of heterogeneities.

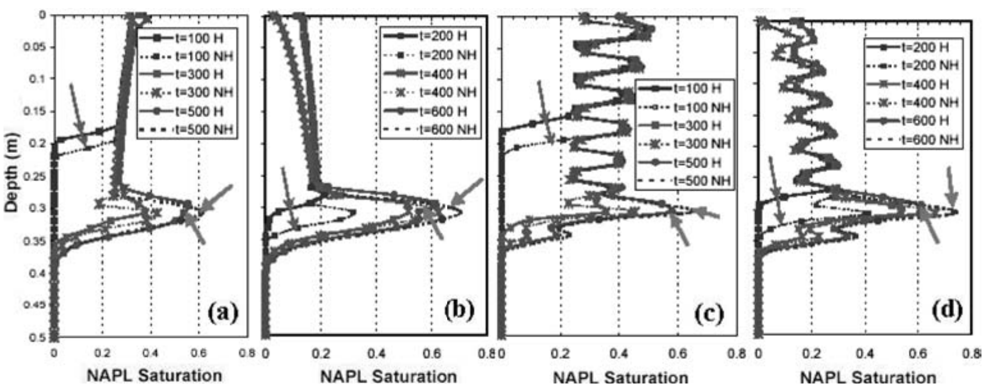

Figure 15.7. Vertical LNAPL saturation profiles along centerline of the domain, resembling hysteretic and non-hysteretic results for homogeneous medium at the end of injection periods (a), and the end of redistribution intervals (b), and as the same for heterogeneous straight medium (c) and (d).

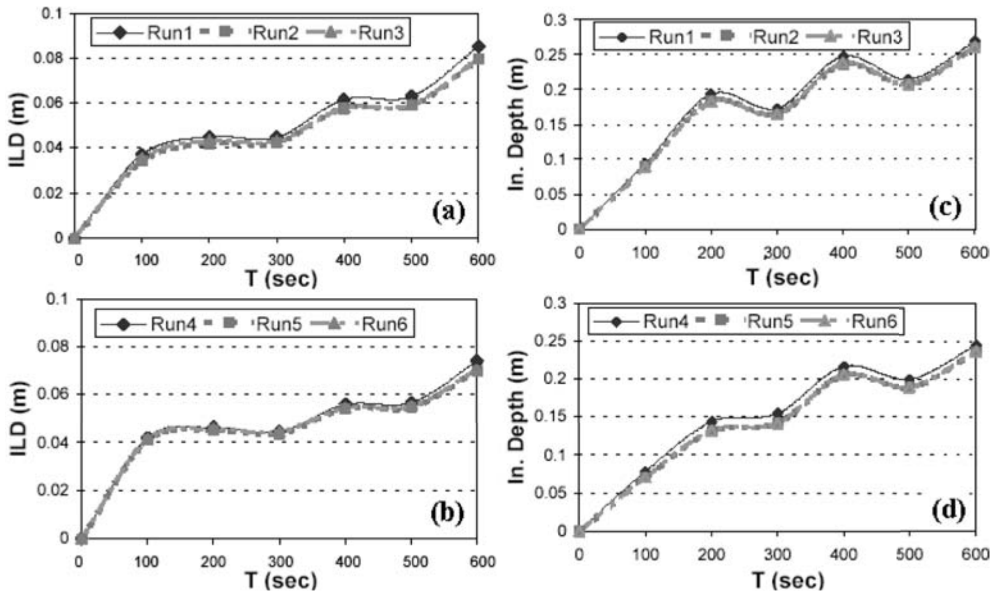

Figure 15.8. Variations of Index of Lateral Distribution (ILD) and vertical depth of the center of gravity of the infiltrated plume, respectively with time (T), gained from non-hysteretic results (a) and (c), and the hysteretic model (b) and (d).

Additionally, regarding the hysteresis, it is crystal-clear that there is a momentous distinction between model results at upper nodes of the domain especially for which are in the vicinity of the source boundary nodes, that is greater NAPL residual saturations are calculated by the hysteretic model at those areas. Another point that is implied from Figure 15.7, goes to the oscillations in NAPL saturation profiles on parts c and d, which are due to presence of fine-grained heterogeneities that are more likely inclined to retain the imbibed non-wetting-phase resembling the coarse-grained host sand.

Hereinafter, in favor of quantifying the transverse migration of the contamination plume, an area moment of inertia named Lateral Distribution Index (LDI), has been defined neglecting the trifling relative difference in the porosity magnitude between sands #2 and #3 that is about 2.5% according to Table 15.1. LDI has been calculated using the following relation:

$$LDI = \frac{\sum_{j=1}^{n_y} \sum_{i=1}^{n_x} \left| [(i-1) \times \Delta x] - \frac{L_x}{2} \right| \times S_{NW_{ij}}}{\sum_{j=1}^{n_y} \sum_{i=1}^{n_x} S_{NW_{ij}}} \tag{15.7}$$

$$= \frac{\sum_{j=1}^{42} \sum_{i=1}^{58} \left| [(i-1) \times 0.0125] - 0.35 \right| \times S_{NW_{ij}}}{\sum_{j=1}^{42} \sum_{i=1}^{58} S_{NW_{ij}}} \tag{15.8}$$

where n_x and n_y are number of nodes along horizontal and vertical directions, respectively, equal to 58 and 42, and L_x and Δx are representative of domain's horizontal length of 0.7 m and nodal spacing of 1.25 cm. In addition, the centroid of overall mass of contaminants that is computed with respect to the domain's top boundary is considered as an indication of LNAPL's Effective Infiltration Depth (EID) into the porous media. The results of the so-called analyses are illustrated through Figures 15.8 and 15.9.

According to additional performed analyses it can be uttered that, as shown in Figure 15.8, it is clearly obvious that ILD and EID parameters variation caused by heterogeneities are relatively

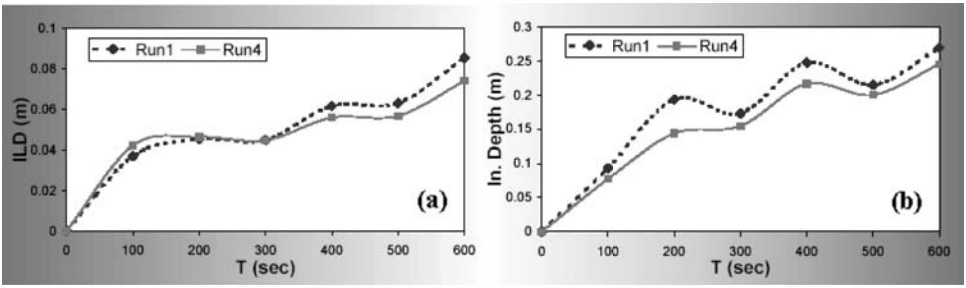

Figure 15.9. Variations of a) ILD, and b) vertical depth of the center of gravity of the infiltrated plume for homogeneous medium.

tiny, inasmuch as the maximum differences between homogenous and non-homogeneous media results are limited up to a value of 2%, yet, in fact, consideration of soil heterogeneities leads to smaller calculated amounts for both ILD and the EID of the plume no matter hysteretic code is used in simulations or not, although hysteretic model results indicate that the effective infiltration parameter shows more sensitivity toward soil non-homogeneities, while as for the non-hysteretic results, the effects of heterogeneities has been seen of higher significance over the index of lateral distribution where the more farther from the time origin we are, the more significant the disparities are expected to be. In addition, regarding the configuration of spatial distribution of heterogeneities, no imperative influence on the porous media behavior has been seen and each of ILD and EID are most likely the same for both straight and staggered media.

Regarding the two defined indices, so as to prevent the over-crowding and owing to relatively weak sensitivity of model results toward soil heterogeneities of a slightly different hydraulic nature, only the homogeneous medium data are used in order to demonstrate the influence of hysteresis on the model outputs (Fig. 15.9). Following the Fig. 15.9a, hysteretic model is indicative of larger quantities of lateral distribution during the first period of injection and redistribution, but as the simulation time goes on the Index of lateral distribution becomes gradually smaller in magnitude resembling that of the non-hysteretic model, such that the hysteretic ILD is about 15% smaller than the non-hysteretic one at the end of the simulation time. Despite what has been stated about the ILD, the hysteretic EIDs are smaller in comparison with the predicted non-hysteretic ones all along the simulation period, even though apparently these differences are to be ameliorated as the plume of contamination reaches the capillary fringe and forms the pooling area above the water table, another amazing fact which is observed is the differences becoming of more significance during the redistribution intervals where, in spite of the wetting process, the drainage of the NAPL phase, as the major wetting phase in presence of the air especially in near surface areas with relatively low water saturations, requires higher capillary pressure forces. Considering Fig. 15.9b, the largest difference between hysteretic and non-hysteretic effective infiltration depths returns to the end of the first redistribution interval at $t = 200$ s where the hysteretic EID is approximately about over 25% smaller than the non-hysteretic one showing that on no account the influence of hysteresis, especially when the non-wetting phase is allowed to redistribute, can be neglected.

15.5 SUMMARY AND CONCLUSIONS

In summary, we have demonstrated the influence of Hysteresis and Soil Heterogeneities as two common natural phenomena on modeling results of the migration of a petroleum-based lighter than water contaminant through a number of determinate granular porous media. A three-phase fully hysteretic code that has the ability to consider soil heterogeneities is used in order to perform the planned simulations. The heterogeneities that are introduced to the model are of a certain type of fairly fine-grained sand, which is slightly different in its hydraulic characteristics such as intrinsic permeability and soil/fluid-phase retention curves, in comparison with the host material constituting the main body of the simulated media. The results of the simulations suggest that with a maximum absolute difference ratio limited to 2%, small local variations in retention properties of the soil body do not have a significant influence on overall effective vertical or horizontal distributions of the introduced LNAPL, on the contrary to what expected for a two-phase system of water-DNAPL where even small amounts of non-homogeneities can dramatically impress the predicted profiles of the non-wetting-phase saturation and the shape of the infiltrated plume. On the other hand, when it comes to hysteresis, according to model outputs, noteworthy degrees of sensitivity is observed, specifically with regard to predicted effective infiltration depth of the introduced mass of contamination such that it becomes to about 20% during the redistribution intervals in the early times of simulation prior to formation of LNAPL body on top of the apparent water table. In addition, lateral distributions are also seem to be impressed spectacularly by NAPL-phase imbibition-drainage cycles getting more sophisticated. As a final point and instead of successive NAPL spills which complicates the internal imbibition-drainage cycles, the lateral hydraulic gradient fields and periodic seasonal water table fluctuations are to be considered for further studies as reported to have the potential to prominently affect the movement and distribution of insoluble contaminants in subsurface media.

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CHAPTER 16

Modelling of density-dependent flow systems: Sensitivity to spatial and temporal discretizations and numerical schemes using SEAWAT 2000

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ABSTRACT: Contaminant transport with significant density contrasts is of increasing interest in many subsurface-hydrology problems like seawater intrusion, saltwater up-coning in coastal aquifers, and dense contaminant migration. Several numerical codes have been developed to simulate these systems. Results from these codes are found to be sensitive to grid and time step sizes, and to the numerical schemes used to approximate the solution. Although the 3-D Finite Difference Method code, SEAWAT has been used over a wide range of problems, the sensitivity of the computed results to spatial and temporal discretization levels for different numerical schemes supported by the code has not been studied in detail. This study aims to: (1) provide an approximate guide for SEAWAT users to ensure proper selection of grid and time step sizes for a particular numerical scheme in order to minimize the numerical error (in 2-D) and (2) To investigate the ability of the SEAWAT code to simulate more complex 3-D systems. The Elder-Voss benchmark problem was selected for the study. Different levels of grid and time discretizations were found to produce significantly different results. A grid size with dx (0.38%) and dz each 0.6% of the total length and depth of the domain respectively is found to be fine enough to produce results with acceptable accuracy for most of the numerical schemes with Courant number (Cr) of 0.1. Some numerical schemes produce accurate results at coarser meshes compared with other schemes that produce similar accuracy at only an extremely fine mesh. For the frequently used C_r value of 1.0, the computed salinity-spread patterns varied significantly for different solution schemes compared with the case when C_r is 0.1 or 0.5. To ensure a high level of accuracy in the modelling results using SEAWAT, C_r should be $\leq$0.1 when the Peclet number is $\leq$ about 1. SEAWAT was able to capture the main physical features of the Elder-Voss convection pattern in 3-D at coarse mesh sizes compared with that produced by a Finite Element Method code "FEFLOW" at a very fine mesh. The 3-D modelling requires large numerical efforts compared with that of the 2-D case. Runtimes for the 2-D problem range from less than ½ hr for a coarser mesh to about 2.8 days in the case of a very fine mesh, whereas it takes around 6 days in the 3-D case with a relatively coarse resolution level. Similarity between 2-D and 3-D convection patterns was found to exist in terms of fingering, and up/down welling behaviours. Other factors like the level of accuracy required, computational expense and storage memory requirements should be considered along with the spatial and temporal resolution levels.

16.1 INTRODUCTION

In the last few decades, environmental concerns about groundwater contamination have increased substantially. Contaminant transport with significant density contrasts is of increasing interest in the sub-surface hydrology problems like seawater intrusion and dense contaminant migration. A small density contrast of (0.0008 g/cm^3) between the plume and the ambient fluid has been found to affect the flow and transport patterns significantly (Schincariol & Schwartz 1990). With variable density, the flow and transport patterns are very complex and unstable.

Table 16.1. Spatial resolution levels used in the study.

No	Resolution level	Resolution class	Grid size (dz, dx) m	No. of layers by columns	Resolution (No. of cells)
1	R1	coarse	5.556×13.636	27×44	1118
2	R2	fine	2.769×6.818	53×88	4664
3	R3	very fine	1.361×3.409	105×176	18,480
4	R4	extremely fine	0.917×2.273	157×264	41,448
5	R5	extremely fine	0.791×1.364	183×440	80,520

Due to the complex non-linearity of the density dependent problem, it becomes very difficult to derive analytical solutions. As a consequence, numerical methods are used and density is incorporated as a dependent variable of solute concentration and in some cases of temperature. Examples of such numerical solutions are SUTRA (Voss 1984), DSTRAM (Huyakorn et al. 1987), SHARP (Essaid 1990), 3D FEMFAT (Yeh et al. 1994), TOUGH2 (Oldenburg & Pruess 1995), FEFLOW (Kolditz et al. 1998), HST3D (Kipp 1997), MOCDENSE (Konikow et al. 1997), and SEAWAT (Guo & Langevin 2002). The accuracy of the density-dependent code convergence is usually tested by comparing its results with that of other codes, using concentration contours at different times for well-defined and studied benchmark problems. A well-studied unstable solute transport problem (Voss & Souza 1987) is based on the heat convection problem of Elder (1967), which will be used subsequently referred to as Elder-Voss-Souza (EVS) problem. The unstable EVS problem is particularly sensitive to choices in discretization and solvers with a major feature being the resulting variation in the direction of the flow and transport at the centre of the domain (up/down welling) (Diersch & Kolditz 1998, Diersch & Kolditz 2002, Woods et al. 2003). For instance, Oldenburg and Pruess (1995) showed that for a fine mesh there is an up-flow at the centre of the domain. This result is similar to that of Elder's original experimental results (Elder 1967) and has been obtained by e.g. Kolditz et al. (1998), Prasad (2000), and Oltean & Bues (2001). The central up-flow is considered the correct pattern (Woods et al. 2003) and will be considered in what follows. Using finer grids (R3-Table 16.1), Frolkovic & Schepper (2000) show a small central downwelling. The central down-welling is also observed with coarse meshes (i.e. R1, Table 16.1) in Voss & Souza (1987), Oldenburg & Pruess (1995) and Kolditz et al. (1998).

The finite-difference, variable density groundwater code SEAWAT has been validated and tested for various benchmark problems (Guo & Langevin 2002, Langevin et al. 2003, Bakker et al. 2004), however, a detailed study of its sensitivity to spatial and temporal discretizations has not yet been undertaken.

Given the increasing popularity of SEAWAT, it is appropriate to systematically explore the sensitivity of the code to spatio-temporal discretization, and choice of numerical transport solvers, for unstable dynamics as represented by the classic EVS problem. In this paper various mesh sizes are used (up to 82000 cells) with different numerical schemes and temporal discretizations to explore the sensitivity of the predicted solute concentration fields at different times. Solute transport and convective mixing are 3-D processes in real field situations (Schincariol 1989, Zhang 2000, Oswald & Kinzelbach 2004). The literature is deficient in modelling studies of 3-D variable density flow, an exception being described by Diersch & Kolditz (1998). The ability of the SEAWAT code to model the more complex 3-D systems will be explored. This will provide an approximate guide for SEAWAT users in their selection of grid and time steps sizes (for a given numerical scheme) for modelling problems, which feature unstable flows.

16.2 BASIC PRINCIPLES OF SEAWAT

SEAWAT couples the flow and transport equations of two widely accepted codes (MODFLOW & MT3DMS) with some modifications to include density effects. It reads and writes standard

MODFLOW and MT3DMS input and output files so that most existing pre- and post-processors for those packages can be used. The governing flow equation (cf. Guo & Langevin 2002) in SEAWAT is:

$$\frac{\partial}{\partial\alpha}\left[\rho K_{f\alpha}\left(\frac{\partial h_f}{\partial\alpha}+\frac{\rho-\rho_f}{\rho_f}\frac{\partial Z}{\partial\alpha}\right)\right]+\frac{\partial}{\partial\beta}\left[\rho K_{f\beta}\left(\frac{\partial h_f}{\partial\beta}+\frac{\rho-\rho_f}{\rho_f}\frac{\partial Z}{\partial\beta}\right)\right]$$
$$+\frac{\partial}{\partial\gamma}\left[\rho K_{f\gamma}\left(\frac{\partial h_f}{\partial\gamma}+\frac{\rho-\rho_f}{\rho_f}\frac{\partial Z}{\partial\gamma}\right)\right]=\rho S_f\frac{\partial h_f}{\partial t}+\theta\frac{\partial\rho}{\partial C}\frac{\partial C}{\partial t}-\rho_s q_s \tag{16.1}$$

where α, β, γ are orthogonal coordinate axes, aligned with the principal directions of permeability; K_f is equivalent freshwater hydraulic conductivity [LT^{-1}]; S_f is equivalent freshwater specific storage [L^{-1}]; t is time [T]; θ is effective porosity [dimensionless]; C is solute concentration [ML^{-3}]; ρ_s is fluid density of source or sink water [ML^{-3}]; and q_s is the volumetric flow rate of sources and sinks per unit volume of aquifer [T^{-1}]. The transport equation is:

$$\frac{\partial(\theta c^k)}{\partial t}=\frac{\partial}{\partial x_i}\left(\theta D_{ij}\frac{\partial c^k}{\partial x_j}\right)-\frac{\partial}{\partial x}(\theta v_i c^k)+q_s c_s^k+\sum R_n \tag{16.2}$$

where C^κ is dissolved concentration of species κ [ML^{-3}]; D_{ij} is the hydrodynamic dispersion tensor [L^2T^{-1}]; v_i is linear pore water velocity [LT^{-1}]; C_s^k is concentration of the source or sink flux for species κ [ML^{-3}]; and $\sum R_n$ is the chemical reaction term [$ML^{-3}T^{-1}$]. For details about the derivation and modification of the governing equations, readers are directed to the SEAWAT manual (Guo & Langevin 2002).

The SEAWAT code supports different numerical solvers of the solute transport equation. They are the Standard Finite Difference Methods (SFDM) (e.g. Upstream Weighting (UW), and Central in Space scheme (CIS)); Mixed Eulerian-Lagrangian Methods (MELM) (e.g. Method of Characteristics (MOC), Modified Method of Characteristics (MMOC), and Hybrid Method of Characteristics (HMOC)); and the 3rd-Order Time Variation-Diminishing Method (TVD3). For the flow equation, the recommended Preconditioned Conjugate Gradient Package 2 (PCG2) solver for water flow will be used for all the simulations in this study.

16.3 METHODOLOGY

16.3.1 *Elder-Voss-Souza (EVS) problem setup and spatial resolution*

For this study, the EVS problem is selected because it has been widely simulated using different codes and methods. Figure 16.1 depicts the set-up of the standard EVS problem with grid size of R1 as detailed in Table 16.1.

The mesh size (R1) will be refined successively to four finer resolutions as shown in Table 16.1 and the the parameters of the EVS problem are presented in Table 16.2. In all cases the dimensions of the source were kept constant (cf. Woods et al. 2003). Figure 16.2 presents the grid sizes (dx & dz) of the R1 to R5 meshes as a percentage of the total lengths (x and z) of the flow domain.

16.3.2 *Stability criteria*

Stability is usually characterized in terms of dimensionless parameters such as the Rayleigh Number (Ra), wavelength of perturbation (λ), Nusselt Number (Nu), and Peclet Number (Pe) (Wooding 1969, Schincariol et al. 1994, Schincariol et al. 1997, Diersch & Kolditz 1998, Woods et al. 2003, Weatherill et al. 2004 among others). Ra in this study is calculated to be 410. The Nusselt number Nu is calculated to be 41.6. Theoretically a stable system should have Nu $\sim$ 1 as all solute transport is by

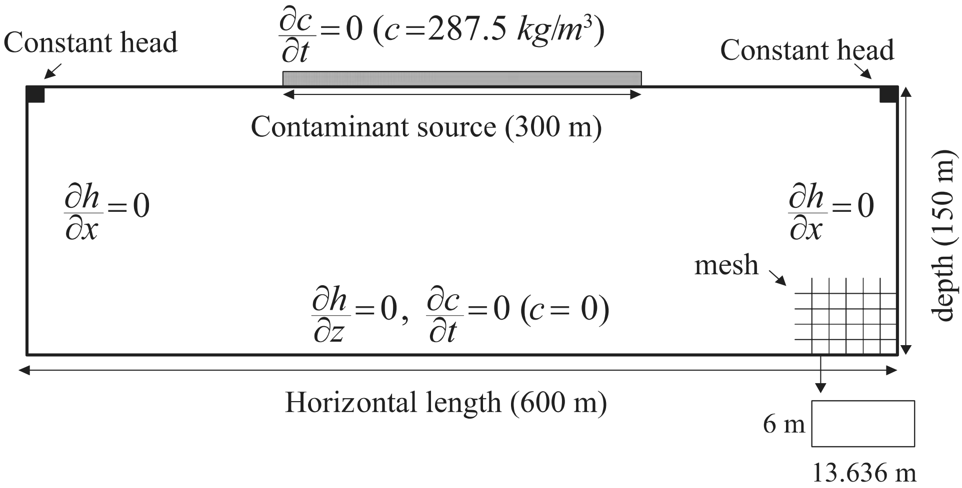

Figure 16.1. Sketch of Elder-Voss-Souza (EVS) benchmark problem.

Table 16.2. Parameters for the EVS problem.

Parameter	Value	Dimension
Porosity	0.1	–
Hydraulic conductivity (H & V)	0.411	m day^{-1}
Longitudinal & transversal dispersivity	0	m
Molecular dispersivity	0.308	m^2 day^{-1}
Density of fresh water	1000	Kg m^{-3}
Density of salt water at the source	1200	Kg m^{-3}
Concentration at the source	287.5	Kg m^{-3}
Courant number	0.1	–
Convergence criteria (γ)	1×10^{-5}	Kg m^{-3}

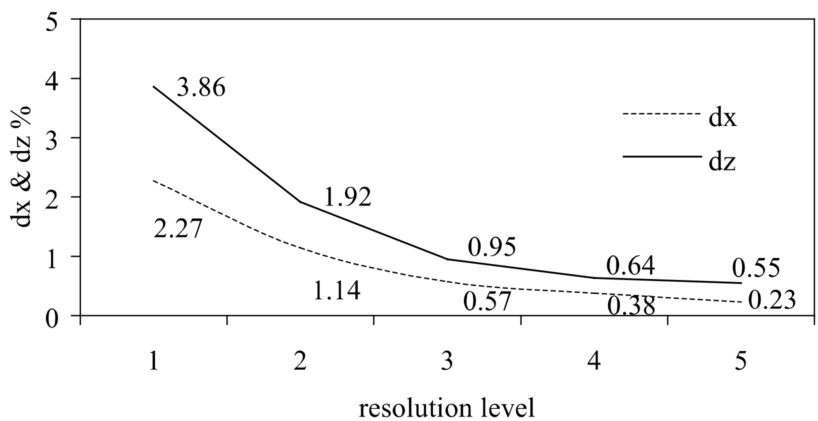

Figure 16.2. Grid sizes used for each resolution level shown as a percentage of the total length (dx) and depth (dz) of the flow domain.

diffusion (Weatherill et al. 2004). Nu > 1 indicates that the system is unstable and the solute transport takes place by convection along with diffusion. The Pe is calculated to be 8.03 (R1) to 0.82 (R5).

16.3.3 *Temporal discretization*

In SEAWAT, the temporal discretization scheme is a combination of that used in MODFLOW and MT3DMS. Unlike the implicit scheme with which the user assigns the time step, the time step is calculated by the program based on certain stability criteria when the explicit scheme is used (Guo & Langevin 2002).

The length of the transport step (TS) (which is a further division of the time step) is controlled by choosing the Courant number, C_r , in SEAWAT. C_r is defined as

$$C_r = \frac{v\Delta t}{\Delta x} \tag{16.3}$$

where v [LT^{-1}] stands for velocity, Δt [T] is time step, and Δx [L] is grid size in direction of v.

In this study, the effect of the TS size on the evolution of the salinity distribution is explored by setting different C_r numbers in the transport solver. The numerical schemes considered are: SFDM (UW), SFDM (CIS), MMOC, and TVD3. The grid size (R4) will be kept constant through all the runs to limit the effects on the results of the TS length. Table 16.3 below presents the numerical simulations for the temporal discretization part.

The range of C_r is selected based on the Diersh & Kolditz (1998) study using FEFLOW. They found that C_r of 1 is not small enough to ensure good results.

16.3.4 *The effect of choice of numerical transport schemes/solvers*

As mentioned above, MT3DMS allows the use of different numerical schemes to solve the advection term of the transport equation. To evaluate discrepancies in the results due to solver choice, the results from different numerical schemes will be studied for different resolution levels and C_r numbers described above in Tables 16.1 and 16.3.

16.3.5 *The 3-D EVS problem study*

Testing the ability and accuracy of SEAWAT for simulating 3-D problems is important to ensure its accuracy level when applying the code to a real field problem where the transport is three-dimensional. A 3-D EVS problem will be conceptualised and tested. The results will be compared with those obtained by Diersch & Kolditz (1998) using the FEFLOW code. The results will be analysed qualitatively to characterize the fingering process compared with that for the 2-D model.

16.3.6 *Measures for comparison*

Density dependent flow simulators are commonly tested qualitatively by comparing the code outputs (mainly isochlors, which means here contours of constant salinity) for a well-known benchmark

Table 16.3. The numerical simulations for the temporal discretization study.

No	C_r number	Grid size	Numerical scheme
1	1	R4	SFDM(UW), SFDM(CIS), MMOC, TVD3
2	0.75	R4	SFDM(UW), SFDM(CIS), MMOC, TVD3
3	0.5	R4	SFDM(UW), SFDM(CIS), MMOC, TVD3
4	0.1	R4	SFDM(UW), SFDM(CIS), MMOC, TVD3

problem (e.g. EVS problem) with those from other codes. The patterns of the concentration spread in terms of the overall shape and fingers produced are compared.

16.4 RESULTS AND DISCUSSION

16.4.1 *Spatial resolution effects*

Results are presented in Figure 16.3 to illustrate the differences between convection patterns for the 5 resolution levels and choices of numerical transport schemes. Figure 16.3 presents results of 0.2 (57.5 kg/m^3) and 0.6 (172.5 kg/m^3) iso-concentration lines at 7, and 20 yrs for SFDM (UW), MMOC, and TVD3 schemes (only part of the results). Results of the first three resolutions show significantly different convection patterns between the numerical schemes. These differences are reflected qualitatively in terms of the shape of the plotted isochlors.

During the simulation with grid R1, the number and shapes of evolving dense fingers change due to splitting and fusion processes. The SFDM (CIS), MMOC, and TVD3 schemes generate down-welling at the centre of the domain which agrees with that found by Voss & Souza (1987), Oldenburg & Pruess (1995), and to some extent that of Elder (1976) for an approximately similar mesh size (1100 cells) whereas the other schemes show an up-welling which agrees with Zhang

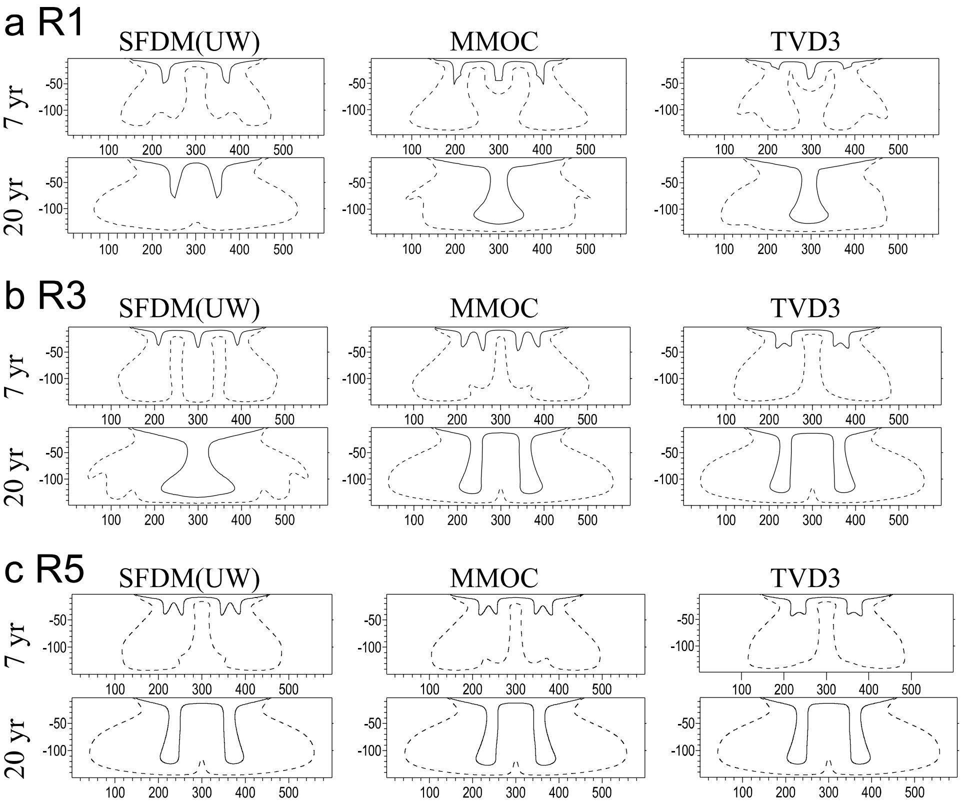

Figure 16.3. The 0.2 (dashed line) and 0.6 (solid line) isochlors for the EVS problem at time intervals of 7, and 20 yr for resolution levels of a) R1, b) R3, and c) R5. SFDM (UW), MMOC, and 3rd-TVD are the numerical schemes used during SEAWAT-2000 simulations.

(2000). The initial projection of the dense plumes is more pronounced in the plots with finer meshes where we expect less numerical dispersion.

The salinity evolution patterns are different in terms of upwelling and downwelling for the first three resolution levels and all the numerical schemes. For example SFDM (UW) shows upwelling after 7 yr for R1 and downwelling at R2. The Oldenburg & Pruess (1995) results for 3993 cells (for half of the domain since the problem is symmetrical) show an upwelling at the centre. Kolditz et al. (1998) were not successful in producing a central upwelling for a coarse mesh but did for 4539 resolution level. When MOC or HMOC schemes were used, SEAWAT produces an upwelling at the centre for R2 whereas other schemes show downwelling.

The isochlors match well at long times for all schemes after refining the mesh to R3 with a small discrepancy in the finger dimensions; an exception is the SFDM (UW) (Fig. 16.3). Likewise, using mesh R4 produces concentration patterns (i.e. isochlors) in good agreement for all selected numerical solutions and times, excluding SFDM (UW). Consistency in the results is assumed to be achieved at mesh resolutions R4 and R5 based on the above qualitative comparison, excluding the SFDM (UW), which agrees only at R5. This finding will be checked further using the quantitative indicators below.

The computational time for a run was found to vary from less than ½ hr for the coarse mesh (R1) to 2.8 days for R5 (MMOC & TVD3 schemes).

Despite the number of codes used to simulate the EVS problem, it is not clear-cut to decide the most accurate and efficient code. It seems that different codes may require different grid sizes to produce similar results. Figure 16.4 compares results of SUTRA (Voss & Souza 1987, Elder 1967), and SEAWAT whereas Figure 16.5 compares results from finite difference (SEAWAT) and finite element codes (FEFLOW & ROCKFLOW).

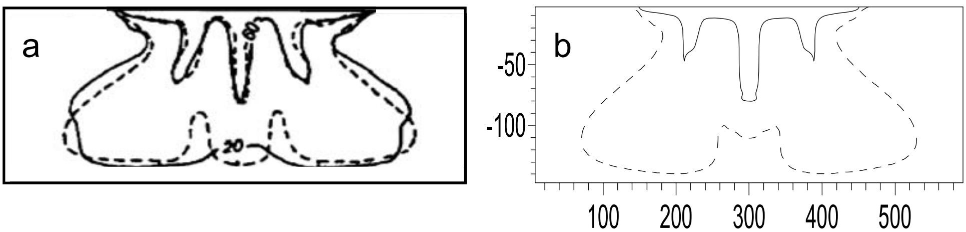

Figure 16.4. The 0.2 and 0.6 isochlors at l0-yrs for 1118 nodes mesh: a) Elder (1976)—solid line and—dashed line (Voss & Souz 1987). Adapted from Diersch & Kolditz et al. (1998); b) SEAWAT for grid size (where dashed and solid lines are the 0.2 & 0.6 isochlors).

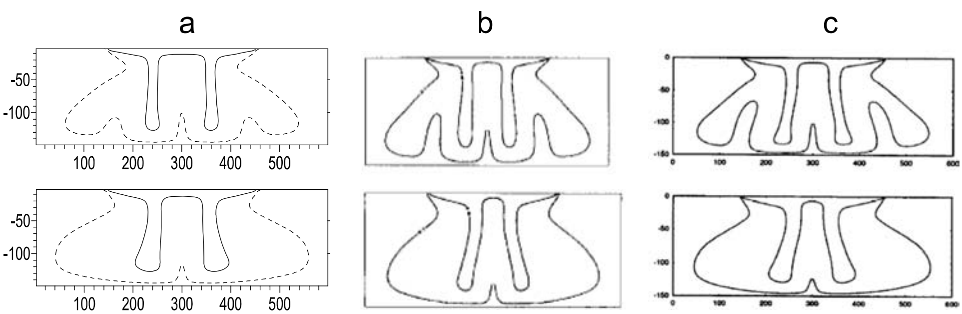

Figure 16.5. Comparison of results with other codes for 0.2 and 0.6 isochlors at 10, and 20 years. a) SEAWAT-MOC scheme (14,432 nodes); b) FELOW (10,108 nodes); c) ROCKFLOW (10,108 nodes). b & c adapted from Diersch & Kolditz et al. (1998).

16.4.2 *Temporal discretization effect*

The C_r number is found to affect the TS length. TS is reduced by about 90% when C_r is reduced by one order of magnitude (1 to 0.1) for all the used numerical schemes. Figure 16.6 shows the effect of the length of TS on the evolution of salinity distribution at time intervals of 5 and 20 yrs for SFDM (CIS), MMOC and TVD3 schemes. The choice of different C_r numbers produces different convection patterns between the numerical schemes. Choosing a C_r number of 0.1 (for R4) produces well-matched results for all schemes excluding SFDM (UW), which produces similar patterns only at R5.

16.4.3 *What numerical scheme to select?*

Since different numerical schemes produce different results at different discretization levels, care is required in selecting the numerical scheme and corresponding grid size that will give sufficiently accurate results with affordable computational expense.

The SFDM (UW) was found to deliver results that only match with other schemes at the highest resolution. The SFDM (CIS) suffers from artificial oscillation, which is reflected as a negative concentration for the first 3 grid sizes and disappears at resolutions R4 and R5. Thus, if the SFDM schemes are to be used, the regular cell size should be $\leq 0.23\%$ of the total domain length and the C_r

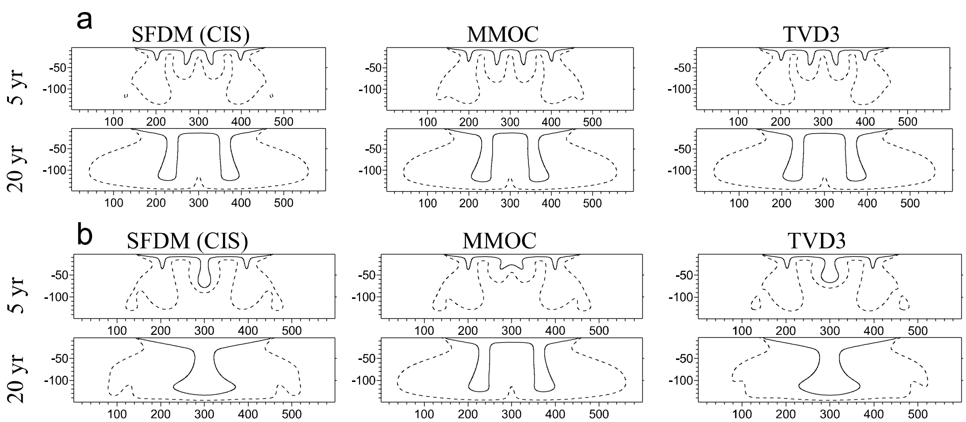

Figure 16.6. The 0.2 (dashed line) and 0.6 (solid line) isochlors for the EVS problem at elapsed time intervals of 5 & 20 yrs for SFDM (CIS), MMOC, and TVD3 numerical schemes using R4. a) C_r number 0.1; b) C_r number 1.

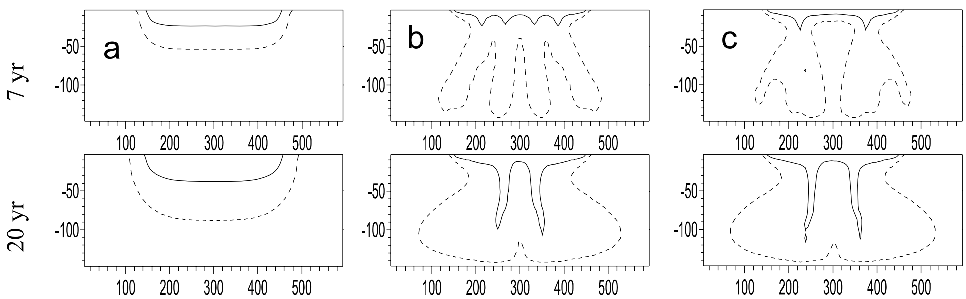

Figure 16.7. Contour lines of 0.2 & 0.6 isochlors for MOC at 7 yr, and 20 years. a) MAXPART 100,000; NPPLAN is 2; NPL is 0, NPH is 15; NPMIN is 0; and NPMAX is 15; b) MAXPART is 100,000; NPPLAN is 2; NPL is 20, NPH is 100; NPMIN is 10; and NPMAX is 200; c) MAXPART is 1,000,000; NPPLAN is 8; NPL is 80, NPH is 400; NPMIN is 40; and NPMAX is 800.

number taken as 0.1 to ensure that the results are free from artificial oscillations and the numerical error is minimized. SFDM schemes are found to be the cheapest in terms of computational expense.

The MELM should be used with caution especially for MOC and HMOC schemes. These methods were found to be sensitive to input solver parameters such as the maximum number of total moving particles (MAXPART) used in representing the concentration distribution, and other options like the pattern of initial placement (NPLANE), number of particles per cell (NPL), and (NPH), and minimum and maximum number of particles allowed per cell (NPMIN & NPMAX) (see Chiang & Kinzelbach 1998). Small NP causes SEAWAT either to stop or to deliver completely misleading results (smoothed isochors) (Fig. 16.7a). But when the NP is increased, the results improved to that of Figure 16.7c. At low NP (the default value), the model crashes and with increasing NP, the results improved. This might be because particles are widely dispersed to a level where there are not enough to approximate the concentration. It is difficult to tell what the optimal value of NP is, but what can be recommended is to start with NP of more than 100,000 particles keeping in mind the computational time and the required storage memory. The default NP was found to be insufficient to reproduce the free convection pattern in the EVS problem.

The NP does not affect the MMOC because only one particle is used to approximate the concentration interface between two nodes at each time step. Hence it is more efficient in terms of computational expense compared with MOC and HMOC. The MELM is expected to lead to a large mass balance discrepancy because it is not entirely based on the principle of mass conservation (Zheng & Bennett 1995). The MOC and HMOC were found to produce good results at intermediate mesh size when the NP is set high enough. Benson et al. (1998) suggested that the particle tracking method (MELM) algorithms create a non-uniform error vector within each numerical block when modelling highly variable velocity fields; and because the truncation error depends on the size of the grid, MELM methods may require an extremely fine mesh. The TVD3 scheme produces good results but is found to be computationally expensive.

16.5 TESTING SEAWAT FOR 3-D ELDER-VOSS-SOUZA PROBLEM

Most of the previous studies in variable density flow tackled the 2-D EVS problem. Solute transport and convective mixing are found to be 3-D processes in a real field situation (Schincariol 1989, Zhang 2000, Oswald & Kinzelbach 2004). Diersch & Kolditz (1998) found that modelling the 3-D EVS problem requires a large numerical effort and it is a time consuming task. The 2-D problem is in order of hours of CPU time whereas the 3-D problem can take days of runtime on a workstation.

In this study, a 3-D EVS problem has been modelled. The TVD3 scheme is selected based on the 2-D study above. The convergence tolerance used is 10^{-4} and Courant number of 0.1. The spatial resolution is ($27 \times 44 \times 44$) 52,275 cells (Fig. 16.8). The simulation cost is about 6 days CPU time.

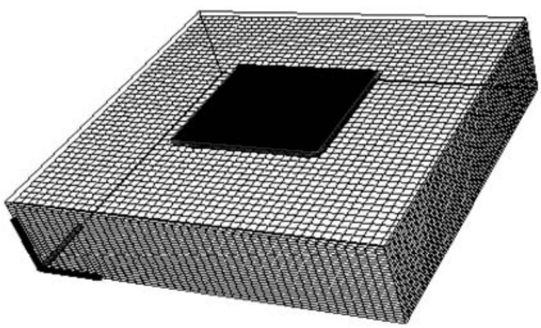

Figure 16.8. Grid discretization for 3-D EVS benchmark problem.

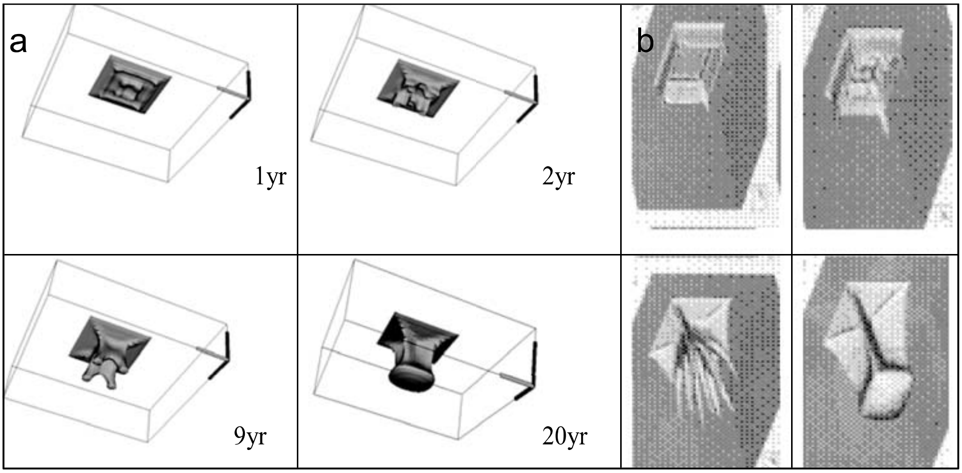

Figure 16.9. 3-D isosurface of 0.5 salinity pattern. a) SEWAT (52227 nodes) at 1, 2, 9, and 20 yr; b) FEFLOW results at 1, 2, 10, and 20 yr at 230000 nodes (adapted from Diersch & Kolditz 1998).

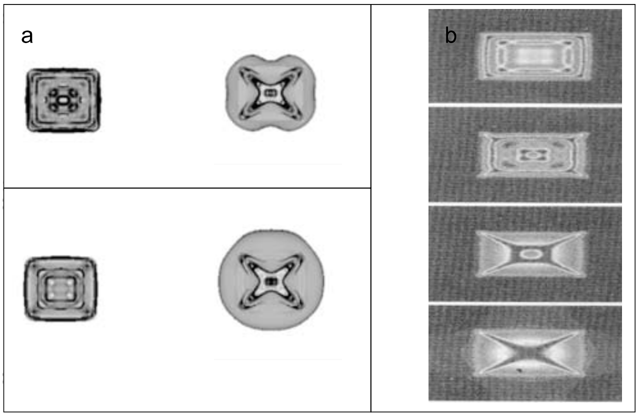

Figure 16.10. Upper Horizon at 0.9 depth (looking from the top) at 2, 4, 10, 20 yr. a) SEAWAT; b) Adapted from (Diersch & Kolditz 1998) for 2, 4, 10, and 20 yr.

Figure 16.9 shows that the "blobs" were developed at the corners and the middle of the source. With time, fingers develop and then fuse together resulting in one wide finger. Diersch & Kolditz (1998) presented results of 200000 cells (Fig. 16.9b). SEAWAT with 52227 cells was able to detect the essential features of EVS convection patterns. SEAWAT is expected to reveal the same at that level of resolution. The discrepancy between the results from SEAWAT and FEFLOW is expected due to the numerical dispersion because of the coarse mesh used for SEAWAT compared to that for FEFLOW as well as the difference in numerical schemes used to approximate the solution in each code.

The 2-D horizontal views at 0.9 d (15 m below the source) in Figure 16.10a also reveal the influence of 3-D convection. At the beginning, fingers appear around the border of the square source area and blobs grow down at the corners. After that, the pattern becomes more complicated with multi-cellular formations with time. As time passes, the complex multi-cellular structure

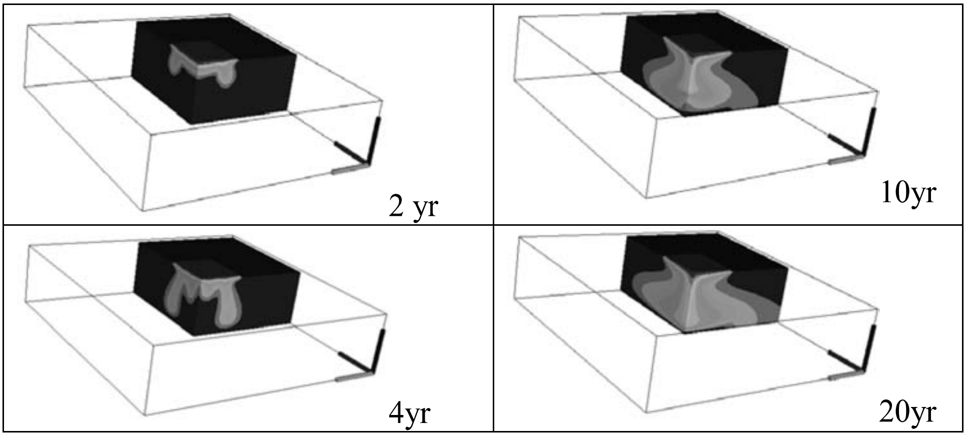

Figure 16.11. 3D view for the evolution of salinity patterns of the 3-D EVS problem at various times.

fuses and the convection pattern is totally restructured. After that, the pattern remains taking a star shape where the heads of the star are oriented to the corners of the square area of the source. Diersch & Kolditz (1998) mentioned that the star is a result of the source geometry of the intrusion area.

The 3-D view (Fig. 16.11) reveals the similarity between the 2-D and 3-D salinity growth pattern in terms of fingers shape and their development.

16.6 SUMMARY AND CONCLUSION

This study investigated the sensitivity of the finite difference code SEAWAT to spatial and temporal resolutions as well as solver choice for the unstable EVS problem in a 2-D and 3-D. For the 2-D model, the spatial resolution is found to affect the salinity distribution patterns. A grid size of about 0.38% (dx) and about 0.6% (dz) of the total length and depth of the domain respectively is found to be fine enough to produce results with acceptable convergence when C_r is 0.1 for all the selected schemes. The concentration-spread patterns behave consistently when the resolution level is $\leq$R4. SEAWAT delivers results showing a central up-welling pattern at a coarser mesh (R1) when MOC or HMOC schemes were used, where other codes like FEFLOW only do so at finer grids (R4).

Temporal discretization was also found to affect the evolving patterns of salinity distribution. At the default C_r number (1.0) there were significant differences in the results compared with the case when C_r is 0.1. In order to minimize the effect of temporal discretization on the concentration results from SEAWAT, C_r should be ≤ 0.1 when the Pe is ≤ 1 (R4 & R5).

Comparison between different variable density codes in the literature reveals variations among the results at different spatial resolutions. The numerical techniques used to approximate the solution for each code might be the reason. It follows that the effects of various features need to be properly evaluated for each simulator. These include spatial and temporal discretizations and types of solvers.

There are other factors that should be considered along with the spatial and temporal discretizations, like the level of accuracy required, and the computational expense. While the higher resolution is more accurate, the more expensive is the computation time. Hence, SEAWAT users should consider all of the relevant factors, and particularly the minimum requirements of the grid and time step sizes shown in this study.

SEAWAT was able to capture the main physical features (development of fingers and their fusion as time passes) of the 3-D EVS convection pattern at a relatively coarse mesh in comparison with the FEFLOW results at extremely fine mesh. The 3-D study requires a large numerical effort compared with that of the 2-D case. Runtimes for the 2-D model were found to range from less than ½ hr for a coarser mesh to about 2.8 days for R5 (for TVD3, and MMOC), whereas it takes around 6 days for the 3-D model with coarse mesh size. Similarity between 2-D and 3-D convection pattern was found to exist in terms of fingering, and up/down welling behaviours.

ACKNOWLEDGEMENTS

The first author would like to acknowledge the support of the Government of Oman and the University of Queensland.

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CHAPTER 17

Large scale modeling of nitrogen transformation in the unsaturated zone—A case study of Tehran City, Iran

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ABSTRACT: Since most of models for estimation of groundwater contamination are developed for agricultural applications and they are often sophisticated, in this paper a combined approach based on analytical methods that need limited input data has been developed. To estimate nitrate contamination of unsaturated zone in large areas, a Lumped Parameter Model (LPM) and the mass balance approach have been used together. In the case study, model results show an acceptable correlation with field measurements. In addition, the fate of nitrogen in the unsaturated zone is simply defined.

17.1 INTRODUCTION

In the recent decades, water shortage crisis is turned out to be an overwhelming problem especially in arid or semi-arid countries. Wastewater is one of the most significant anthropogenic sources of groundwater nitrate contamination especially in urban areas without sanitary disposal systems (Joekar-Niasar 2002).

Teseoriero et al. (1997) has concluded that nitrate is the most important organic pollutant in water bodies and therefore prediction of NO_3-N leaching in the unsaturated zone is essential for assessing groundwater contamination and ultimately in the development of a nutrient management protocol. High concentration of nitrate (more than 10 mg/L-N) in drinking water is the cause of concern. Ingestion of nitrate by infants can cause low oxygen level in blood, a potentially fatal condition (Spalding et al. 1993). Other negative health effects are potentially related to ingestion of nitrate in drinking water are spontaneous abortions and non-Hodgkin's lymphoma (Nolan 2001).

Since prediction of NO_3-N is very difficult due to the complex nature of soils and the processes of the fate of N in soils, researchers have developed several models for different applications that mostly are related to soil-plant applications. Antonopoulos & Wyseure (1998) developed WANISM model for predicting the transport and transformations of nitrogen resulting from agricultural applications. Kaluarachchi & Parker (1998) has also developed a three dimensional finite element model for simulation of transport and transformation of nitrogen in the unsaturated zone. Few models for non-agricultural applications have been developed yet. MacQuarrie & Sudicky (2001) developed a complicated model for simulation of the transport and transformation of nitrogen compounds in septic tank system, saturated and unsaturated zone. This model includes nitrogen and carbon transformations as well as geochemical, physical and biochemical processes (MacQuarrie et al. 2001).

Although a lot of sophisticated numerical models have been developed for nitrogen fate simulation, applying them is often difficult due to the limited availability of field information. Therefore analytical methods based on the Lumped Parameter Model (LPM) was developed for the estimation of nitrogen transformation in the unsaturated zone for a soil-plant system in first few inches of soil column. Lumped parameter models were introduced for the first time in 1950s and 1960s for interpretation of environmental radioisotope data in groundwater hydrology (Ling 1998). In the recent decade, a lot of analytical models have been developed using the first-order kinetic and linear isotherm equations to estimate pesticide contamination of groundwater (Ling & El-Kadi 1998). But none of these models have developed for large urban areas with continuous discharge of wastewater.

To predict the nitrate contamination in large areas, different methods such as vulnerability assessment index, mass balance and LPMs have been developed. DeSimone (1998) determined the percentages of different processes of nitrogen fate such as sorption, nitrification, volatilization, etc in a large area. Nolan (2001) has also studied on the relation between precipitation, population and nitrate concentration in groundwater in the study area by a lot of field tests. Furthermore Joekar-Niasar et al. (2003) has developed a vulnerability assessment model for urban area based on modified DRASTIC method that has shown effect of different factors such as wastewater discharge, population density and depth to water, etc on vulnerability index. Large scale studies on groundwater contamination cover qualitative aspects of problem (Osborn et al. 1998, Rupert 2001). Hence, to have a quantitative estimation for unsaturated zone contamination in a large area with lack of field data, a combined lumped parameter-mass balance model is developed in this research. Special specifications of this model can be summarized into large scale modelling with a relatively good quantitative estimation and limited field information as the following.

17.2 DESCRIPTION OF THE STUDY AREA

Tehran is located in a relatively semi-arid area in south of Alborz Mountains. The study area covers more than 700 km^2 and consists of 22 municipal districts (Fig. 17.1). The Greater Tehran, capital of Iran has a population of more than 6.8 million who rely on groundwater as the main resource of drinking water. 45% of water demand of Tehran is supplied by groundwater resource that is contaminated by individual traditional cesspools and septic tanks for many years (Mahmoudi 2001). General wastewater disposal except for some small areas is based on traditional cesspools and septic tanks from which wastewater is discharged directly into the ground and reaches groundwater table. Because of dense building structure, most of the area is covered by buildings, streets and impermeable surfaces except for parks. There is no agricultural land use in this study area and therefore major nitrogen contaminating sources can be summarized to wastewater discharges, parks

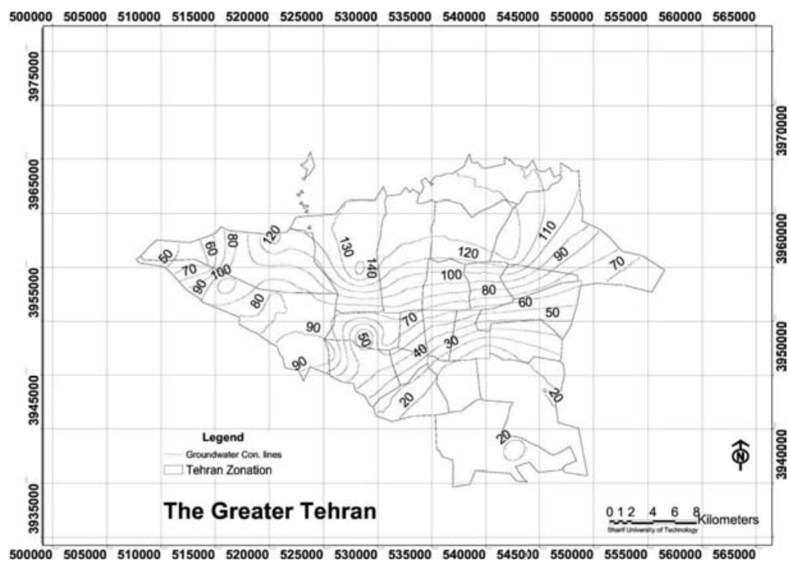

Figure 17.1. Map of greater Teheran and the depth to groundwater (m).

and atmospheric deposition. Depth to groundwater in different places varies from 10 to 140 m and mean annual precipitation in different areas varies between 250 to 350 mm.

17.3 MODEL DESCRIPTION

In the LPM, the dispersive flux is neglected and simplified equation of mass conservation for average concentration of contaminant in soil column is considered as follows (Fig. 17.2):

$$\frac{\partial(\theta c)}{\partial t} = -\frac{\partial(qc)}{\partial z} \pm (Sink/Source) \tag{17.1}$$

where θ = soil water content within depth; q = soil water flux; c = solute concentration in solution; and z = depth below the surface.

By integrating (17.1) over the depth we have

$$\int_0^L \frac{\partial(\theta c)}{\partial t} dz = -\int_0^L \frac{\partial(qc)}{\partial z} \pm \phi \tag{17.2}$$

in which ϕ = source sink term. It should be noted that the depth (L) is assumed to be independent of time. Based on groundwater table monitoring in different months, it has been shown that groundwater table changes are not considerable. In addition, because long term prediction in unsaturated zone is of interest, L can be assumed to be fixed.

Average values of the variables θ, c and (θc) along the depth at a given time are defined as follows:

$$E[\theta] = \frac{1}{L}\int_0^L \theta(z,t)dz \tag{17.3}$$

$$E[c] = \frac{1}{L}\int_0^L c(z,t)dz \tag{17.4}$$

$$E[\theta c] = \frac{1}{L}\int_0^L \theta(z,t)c(z,t)dz \tag{17.5}$$

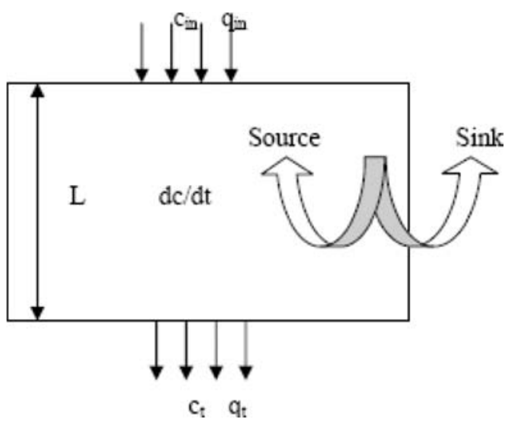

Figure 17.2. Schematic representation of simplified mass balance.

then (17.2) can be written as

$$LE\left[\frac{\partial(\theta c)}{\partial t}\right] = c_{in}q_{in} - c_t q_t \pm \phi \tag{17.6}$$

where c_t = concentration in solution below the depth of interest; c_{in} = dissolved concentration of contaminant applied to the top of depth of interest; q_{in} = recharge per unit area; q_t = outflow from the depth of interest per unit area; and ϕ = sink or source term over the depth.

The left side of (17.6) will be as follows:

$$LE\left[\frac{\partial(\theta c)}{\partial t}\right] = LE[\theta]\frac{\partial E[c]}{\partial t} + LE[c]\frac{\partial E[\theta]}{\partial t} \tag{17.7}$$

Where c and $\partial\theta/\partial t$ are independent and so θ and $\partial c/\partial t$. Consequently $E[v_1, v_2]$ can be approximately equal to $E[v_1]E[v_2]$ for two variables v_1 and v_2. This approximation can cause serious limitation.

Based on mass balance principle, change of the average water content equals the inflows minus outflows from the soil column.

$$L\frac{\partial E[\theta]}{\partial t} = q|_{z=0} - q|_{z=L} = P + I - R - q_t - ET \tag{17.8}$$

where P = precipitation; I = inflow rate; R = runoff rate; and ET = evapotranspiration rate.

By inserting equations (17.8) and (17.7) into (17.6), the solute conservation equation can be then written as follows:

$$(LE[\theta])\frac{dE[c]}{dt} + (P + I - R - ET)E[c] + (c_t - E[c])q_t = c_{in}q_{in} \pm \phi(t) \tag{17.9}$$

in which $\bar{\theta} = E[\theta]$ (average of θ). In addition, in (17.9), it is assumed that the average concentration in the soil column is equal to the concentration at the bottom of soil column and this makes limitations.

Since urea is hydrolyzed to ammonium rapidly in septic tanks and cesspools, the amount of urea in contrast to ammonium is negligible. Considered processes in model include transformation of ammonium to nitrate (nitrification) with k_1 transformation rate, transformation of nitrate to nitrogen (denitrification) with k_2 rate, soil adsorption and volatilization of ammonium to ammoniac gas (Fig. 17.3). Finally dominating equation in transformations can be defined as follows:

$$q_0 = P + I - R - ET \tag{17.10}$$

$$(L\bar{\theta})\frac{dA}{dt} + q_0 A = q_{in}A_{in} - L(\rho_b K_{dA} + \varepsilon K_H)\frac{dA}{dt} - k_1(L\bar{\theta}A) \tag{17.11}$$

$$(L\bar{\theta})\frac{dN}{dt} + q_0 N = q_{in}N_{in} + k_1(L\bar{\theta}A) - (L\rho_b K_{dN})\frac{dN}{dt} - k_2(L\bar{\theta}N) \tag{17.12}$$

Considering attenuation rates for the transport of ammonium and nitrate as λ_1 and λ_2 respectively in the following forms

$$\lambda_1 = \frac{1}{R_{FA}}\left(k_1 + \frac{q_{in}}{L\bar{\theta}}\right) \tag{17.13}$$

$$\lambda_2 = \frac{1}{R_{FN}}\left(k_2 + \frac{q_{in}}{L\bar{\theta}}\right) \tag{17.14}$$

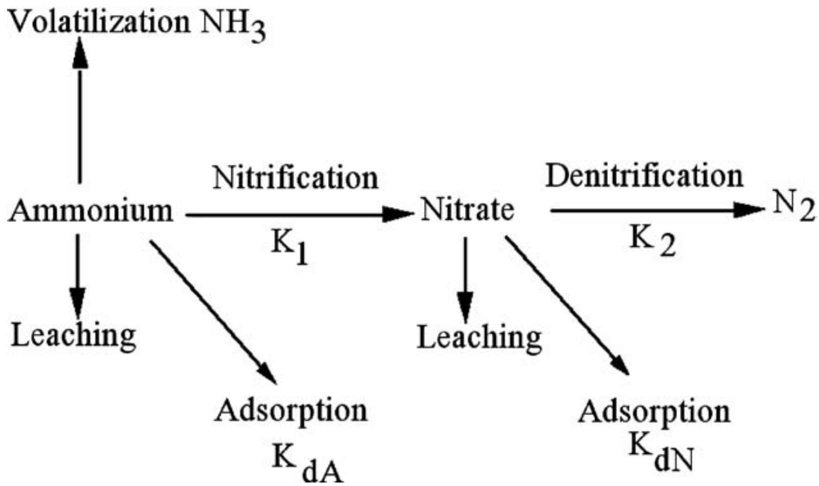

Figure 17.3. Schematic representation of nitrogen transformations in LPM and other processes in N cycle.

allows to rewrite (17.11) and (17.12) in a simpler form as follows:

$$A = \frac{q_{in}A_{in}}{LR_{FA}\bar{\theta}\lambda_1}(1 - e^{-\lambda_1 t}) + A_0 e^{-\lambda_1 t} \tag{17.15}$$

$$R_{FA} = 1 + \frac{\rho_b K_{dA}}{\bar{\theta}} + \frac{\varepsilon K_H}{\bar{\theta}} \tag{17.16}$$

$$R_{FN} = 1 + \frac{\rho_b K_{dN}}{\bar{\theta}} \tag{17.17}$$

where R_{FA} = retardation factor of ammonium; A_0 = initial concentration of ammonium in soil; and R_{FN} = retardation factor of nitrate.

Finally, by considering (17.15) to (17.17), final equation for nitrate will be as follows:

$$\begin{aligned} N = {} & \frac{q_{in}}{L\bar{\theta}\lambda_2 R_{FN}}\left(N_{in} + \frac{k_1 A_{in}}{R_{FA}\lambda_1}\right)(1 - e^{-\lambda_2 t}) + \frac{k_1}{R_{FN}(\lambda_2 - \lambda_1)}\left(A_0 - \frac{q_{in}A_{in}}{LR_{FA}\bar{\theta}\lambda_1}\right) \\ & \times (e^{-\lambda_1 t} - e^{-\lambda_2 t}) + N_0 e^{-\lambda_2 t} \end{aligned} \tag{17.18}$$

where N_0 = initial concentration of nitrate in soil; k_1, k_2 = first order transformation rate constants; t = time; and ρ_b = soil bulk density.

17.4 METHOD OF STUDY

After model development, in order to facilitate model calculations and data processing procedure, all input data and model were defined in GIS.

17.4.1 *Inflow data*

Three main sources of nitrogen in Tehran were considered namely, i) domestic wastewater; ii) atmospheric precipitation; and iii) parks and agricultural applications.

17.4.1.1 *Domestic wastewater*

Since there is no sanitary sewage system in Tehran, wastewater disposal in Tehran is done by traditional cesspools and sorption wells. To estimate nitrogen loading due to the domestic wastewater, it is necessary to estimate population density and wastewater information and composition in different urban districts. Then it will be possible to estimate nitrogen loading in different compositions as an annual mean value. Household water demand with 70% and public, commercial and industrial water demand with 30% are the two categories that are considered in water supply system. According to the statistics on water demand per capita in 2001, wastewater rate in Tehran is about 162 L/day per capita. Wastewater composition is determined based on inflow composition in some local wastewater treatment plants.

The latest official demographic statistics are related to 1996. To update these statistics, demographic statistics were multiplied by population increase rate of Tehran that equals 3.81%. Based on these data, annual nitrogen loading is estimated at about 30039.85 mg/m^2 which is about 18890 tons per year.

17.4.1.2 *Atmospheric precipitation*

Atmospheric precipitation is considered as an important source of nitrogen deposition. Measurement of wet and dry deposition of nitrogen involves complicated and time-consuming tasks that are expensive too. Therefore, it seems necessary to assess the role of this source of nitrogen in contrast to other sources. On this purpose, air quality and precipitation distribution maps of Tehran related to 2001 were prepared and average wet deposition of nitrogen was estimated to be about 50 kg/year-N. Since this value is ignorable in comparison with wastewater, it is expected that it will make no significant effect in calculations.

17.4.1.3 *Parks and agricultural applications*

According to the standards, normally nitrogen loading due to park applications is about 20000 kg/km^2/yr. Based on the area of parks in different districts, nitrogen loading in Tehran is estimated about 900 tons per year. According to the calculations 95.5% of nitrogen loading is produced by wastewater and the rest by park and air deposition.

17.4.2 *Outflow data*

The quality of surface of groundwater is the only necessary information that is considered as outflow. Groundwater quality is measured in 60 stations such as wells or piezometers all over the Tehran and groundwater quality maps are prepared for different nitrogen compounds.

17.4.3 *Model parameters and variables*

Model parameters are determined based on literature according to the conditions. Then calibration procedure was used based on the nitrate concentration in groundwater and models results.

It is deducted that precision in determination of model parameters such as k_1, k_H and ε have not significant effects on model results. But model is sensitive to the values of k_2 and θ. Therefore, except for k_1, k_2 all parameters have been determined based on literature. For soil water content various values have been mentioned in different references. MacQuarrie et al. (2001) has determined 0.4 for sandy aquifer under effect of septic tanks. Soil water content for gravel media is determined about 0.44. Since Tehran aquifer is under continuous recharge of wastewater for several decades, water content in soil media will be high equal to 0.40. ε and Henry coefficient are about 0.15 and 2, respectively. Ammonium and nitrate distribution coefficients are 3.5 and 0.001 respectively. In this model, concentration is calculated as a mean value in depth that is considered fixed within depth. Hence, with increase of depth, error occurrence possibility and sensitivity to k_1 and k_2 increases. To solve this problem a parametric model for these coefficients has been developed for different depths in that coefficients have been determined regarding to depth. These equations are derived regarding to ammonium and nitrate concentrations in groundwater in some sample points.

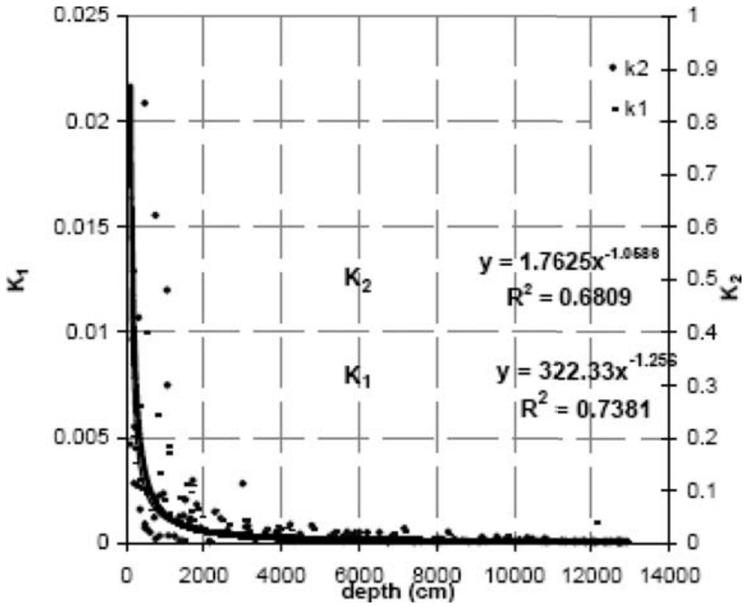

Figure 17.4. Relation between k_1 and k_2 and depth in LPM model.

Figure 17.4 shows trend of coefficients variation in different depths. This method for determination of coefficients can solve partially lack of field data related to these coefficients.

Since groundwater is contaminated for several years, groundwater contamination is relatively steady. For this reason in mass balance method calculations it is not necessary to estimate travel time and so inflow and outflow concentrations will be compared.

17.5 RESULTS

17.5.1 *LPM results*

After running LPM for more than 60 homogenous distributed points in Tehran, concentration of ammonium and nitrate as well as total nitrogen was plotted versus the depth (Fig. 17.5). As it is shown, total nitrogen concentration decreases with depth but in depth 20 m and more, the rate of this decrease is going to be fixed. In addition, the most nitrification occurs in depth 2 or 3 meters that has been verified in different researches. After 2 or 3 meters, nitrate concentration decreases and gradually becomes steady. But because of high adsorption rate of ammonium in soil, it has decreased rapidly. Hence, fate of nitrogen in unsaturated zone of Tehran aquifer can be simply defined by LPM.

In order to have an estimation of loading for nitrogen in groundwater and estimate the rate of different processes, mass loading of nitrogen compounds was considered. On this purpose, more than 60 points of groundwater were monitored three times within an eighteen month period and nitrate concentration in these 60 points was compared with model results that shows a meaningful correlation (more than 63%) between two data series (Fig. 17.8). Although there are some inaccuracies in some regions of study area (Figs. 17.6, 17.7), it is assessed that lack of geotechnical data has made serious errors in calculations in these regions.

17.5.2 *Mass balance calculations*

In mass balance calculations, sources of nitrogen that have been identified before, are considered as input sources. Nitrogen compounds measured in groundwater table were considered as output of

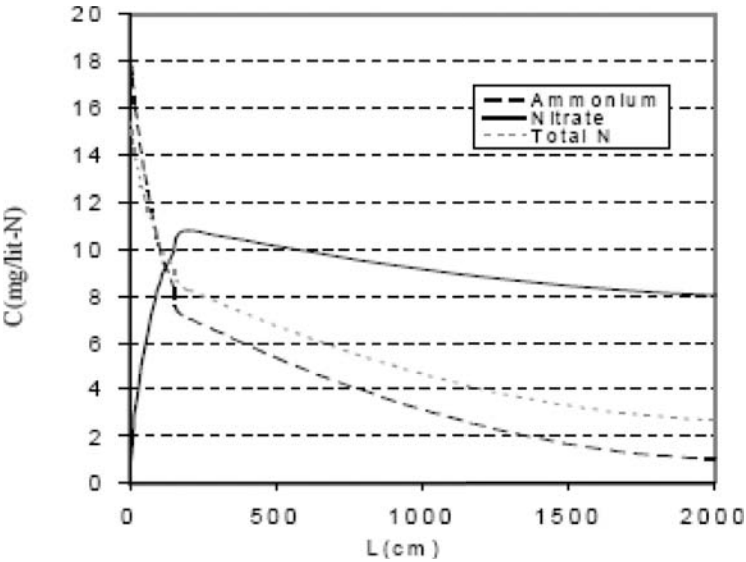

Figure 17.5. Ammonium and nitrate concentration in different depths.

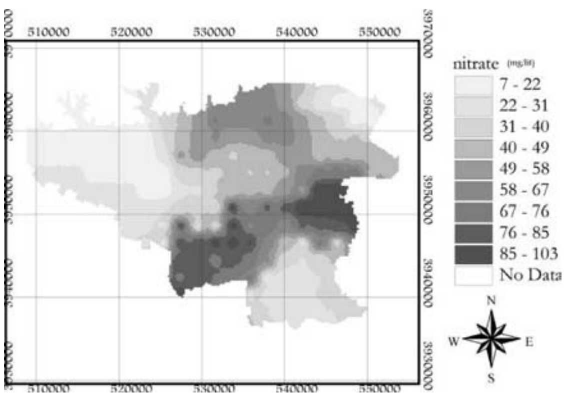

Figure 17.6. Nitrate distribution map based on model results.

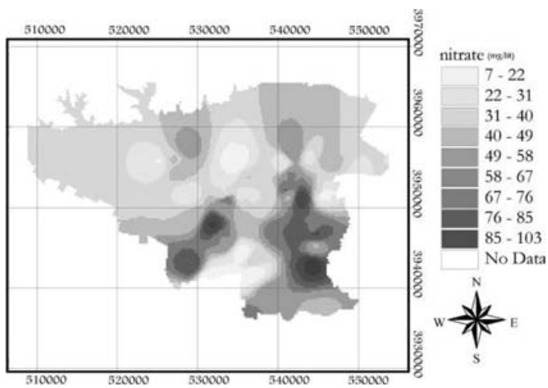

Figure 17.7. Nitrate distribution map based on measurements.

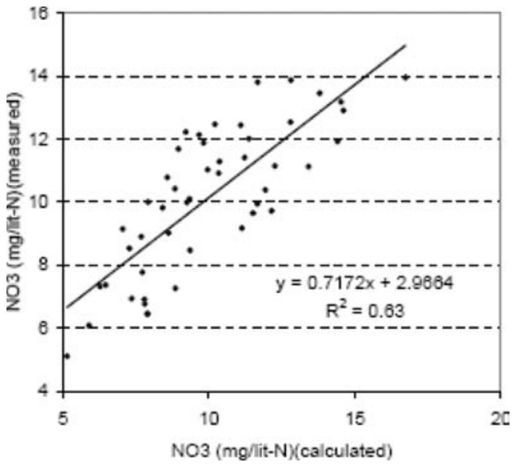

Figure 17.8. Correlation between measurements and LPM results.

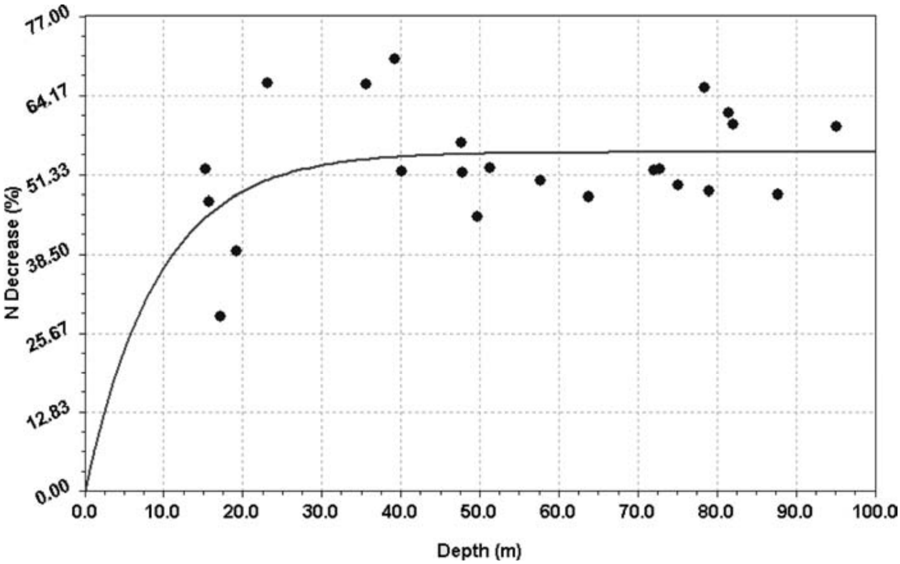

Figure 17.9. Nitrogen decrease percentage in different depths.

unsaturated zone. The differences between input and output of unsaturated zone make it possible to estimate nitrogen decrease as well as effects of different sources in groundwater contamination. Urban wastewater with 95.5% and agricultural application (parks) with 4.5% and air deposition with a very low percentage are the main sources of nitrogen in groundwater. As described before, it is not necessary to calculate mass balance in a specified time step.

In other words, input and output values can be considered in calculation without any travel time. Relatively steady state of Tehran groundwater contamination has been verified in field measurements too.

After comparison of input and output data in different urban regions, regional mean values of nitrogen removal have been shown for different depths. As shown in Figure 17.9, three different zones are distinguishable in nitrogen decrease trend in depth.

In depths less than 10 m, nitrogen removal occurs rapidly while in depths between 10 and 30 meters, rate of nitrogen removal decreases. In depths more than 30 meters, rate of nitrogen removal will be fixed to about 51%.

Based on modeling procedure, ammonium will be transformed rapidly to nitrate or volatizes or adsorbed in soil in first few meters. In depth between 2 or 3 meters, most nitrification rate occurs. Model results show that nitrogen removal happens mostly due to denitrification that needs high soil water content. In other words, initial assumption for relative moisture (0.4) can be correct. As a result of mass balance method, 45% of nitrogen is removed in unsaturated zone.

17.6 CONCLUSIONS

In large areas with limited information, it is too hard to use numerical models. Therefore a LPM using analytical equations was developed to estimate nitrogen transformation in the unsaturated zone. In addition by principle of mass balance, different nitrogen sources as well as their percentages in contamination were studies. Results show a relatively good correlation with field measurement, but in some areas serious errors have been observed that are caused by lack of geotechnical data such as soil water content.

Since, this LPM calculates mean concentration in depth; transformation coefficients can produce great error in results for high depths. Therefore, using parametric models for estimation of transformation coefficients was a successful experience in using LPMs for urban area. In unsaturated zone of Tehran, average nitrogen removal is estimated about 55% and removal rate increases by depths of up to 30 m and after 30 m it will be fixed. In this case study, the most nitrification rate occurs in depth of 30 m and after that nitrate concentration decreases due to denitrification and adsorption. In other words, it can be concluded that in depths less than 30 m, less nitrate contamination is predictable.

17.7 SYMBOLS AND ABBREVIATIONS

A_0: initial concentration of ammonium in soil (M/L^3)
c: solute concentration in solution (M/L^3)
c_{in}: solute concentration at the top of soil column (M/L^3)
c_t: concentration in solution below the depth (M/L^3)
ET: evapotranspiration (L)
k_1, k_2: first order transformation rate constants (1/T)
N_0: initial concentration of nitrate in soil (M/L^3)
P: precipitation (L)
q: soil water flux (L/T)
q_{in}: recharge per unit area (L/T)
q_t: outflow from the depth of interest per unit area (L/T)
R: runoff (L)
R_{FA}: retardation rate of ammonium (L^3/M)
R_{FN}: retardation rate of nitrate (L^3/M)
t: time (T)
z: depth below the surface (L)
ϕ: sink or source.
λ_1: attenuation rate for ammonium (M/M)
λ_2: attenuation rate for nitrate (M/M)
ρ_b: soil bulk density (M/L^3)
θ: soil water content (%)

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Section IV
Coastal groundwaters and impact of tsunami

CHAPTER 18

Characterization of groundwaters in Tiruvanmiyur coastal aquifer, Tamil Nadu, India

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ABSTRACT: Environmental isotope techniques have been employed along with hydrogeology & hydrogeochemistry for characterizing and identifying factors controlling groundwater quality in Tiruvanmiyur coastal aquifer system. Results indicate that the quality of groundwater is generally fresh except at a few locations. The brackish quality in top aquifer is due to the contribution from the Buckingham Canal and evaporated surface water. Leaching of salts and old seawater are responsible for brackishness & salinity of groundwaters in weathered & fractured aquifer. The saline water is about 7000a BP whereas the brackish groundwater is about 11,500a BP. The two aquifers are interconnected in the western part of the study area but chances of interconnections in the eastern part are remote. The unconfined aquifer is mainly recharged by local precipitation and surface water bodies whereas the semi-confined aquifer receives recharge from the precipitation collected in the depression on the western part of the study area and lateral flow.

18.1 INTRODUCTION

The Tiruvanmiyur coastal aquifer is located south of Chennai metropolitan City in southern India between longitudes 80°13′30″ and 80°16′30″E and latitudes 12°48′15″ and 12°59′15″N. The study area is bounded by Bay of Bengal in the East, River Adyar in the North, Kovalam creek in the South and about 3 km west of Buckingham canal (Fig. 18.1). The Buckingham canal extends across the study area from north to south parallel to the coast. This canal was used for navigation in the past whereas at present it is used as a channel for sewage disposal (Ballukraya et al. 1999).

This aquifer developed along the coast of Bay of Bengal to a length of 20 km and with 5 km width has been contributing significantly to the drinking water supply of Chennai City, the fourth largest city in India. Owing to the pressure of the population growth, increased urbanization and industrialization, the demand for the water supply to Chennai City is ever increasing and this aquifer is likely to be exploited by large scale pumping that may result in sea water intrusion.

In the present paper we present the results of a detailed hydrogeological, hydrochemical and environmental isotope investigation that has been carried out to assess the hydrogeological conditions, inter-relation between aquifers, source and origin of groundwater recharge and factors affecting quality of groundwater in the Tiruvanmiyur coastal aquifer. The hydrogeological methods were used to define the geometry of the aquifer system, which includes compilation of existing hydrogeological data as well as the data obtained from newly constructed piezometers to explore the subsurface hydrogeological and geochemical information to fill the data gaps. Temporal and spatial variations of groundwater regime were also studied.

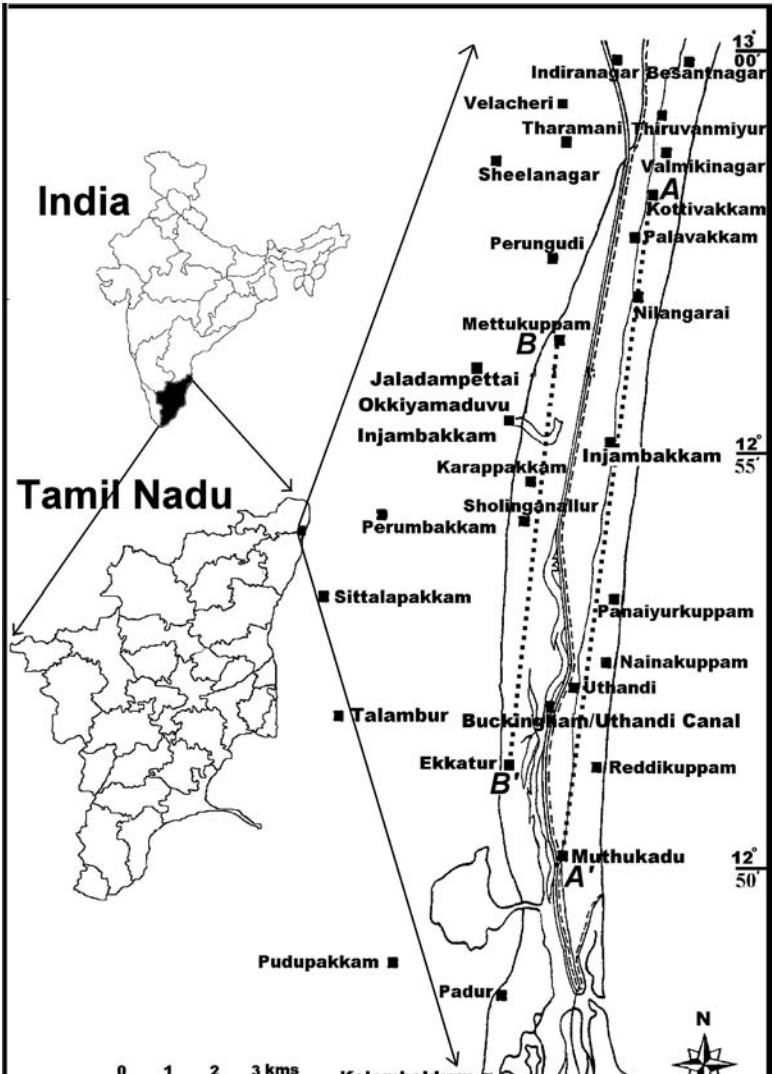

Figure 18.1. Location map of the study area with sampling points.

18.2 GEOLOGICAL AND GEOMORPHOLOGICAL FRAME WORK

This aquifer is characterized by sediments of Pliocene to Recent age deposited on Archaean basement consisting of charnockites and garnetiferous gneisses, and these sediments outcrop in the western margin of the study area. Litho-log data show that the weathering of the basement was extended up to 3–5 m and fractured zones were found further down to a depth of 10 m. Pliocene sediments overlying the weathered basement consist of unconsolidated coarse-grained sand & gravel. The thickness of this stratum varies from 3–5 m. About 5–10 m thick clayey sand zone acts as marker zone and separates the Quaternary sediments at the top and Pliocene sediments at the bottom. This marker zone, black in colour consists of fine to medium grained sand and clay. This zone is found to vary in thickness and it is conspicuously absent at Muthukadu, Ekkatur, Sholinganallur and Mettukuppam area. The Quaternary sediments consist of fine to medium grained sand with various shades of colour and its thickness varies from 5–10 m.

The geomorphologic features of the area show that it is characterized by undulating gentle slope towards east with sand dunes parallel to the coast. The general elevation of the terrain is in between 3 and 10 m above mean sea level (amsl). This area is studded with many surface water bodies of medium and minor types and has been broadly classified into different geomorphologic units (Srinivasan 1980): a) mud flats characterized by a highly saline barren tract ranging in width from 350–400 m close to Buckingham Canal comprising mostly silty and loamy soil. The general ground elevation varies from 1–2 m amsl; b) silty plain that extends from mud flat 350–400 m towards east and comprises silty soil. The general elevation varies from 2–3 m amsl. The plain gets groundwater seepage from western margin of terrace and gets collected in north–south trending ponds; c) beach terrace, extending from silty plain and its width varies from 300–450 m forming the western boundary. The general elevation is of the order of 5–10 m amsl. This zone is a good recharge area of water during the monsoon. Sand Dunes developed along the eastern margin of the terrace. The prevailing wind has modified the dunes, which trend sometimes parallel and perpendicular to the coastline; and d) inter-terrace area that is a low lying depression formed adjacent to the beach terrace with a width of 300–600 m and its elevation is about 3–4 m amsl. In the peak monsoon period it also forms as a zone of surface run off. First terrace is observed east of inter-terrace and its elevation is of the order of 4–6 m amsl with a width of 25–100 m. This zone is found parallel to the coast and is characterized by mangroves and fishermen's dwellings, which help in the protection of the coast. From this terrace, the sea waterfront is at a distance of about 50–80 m.

18.3 HYDROLOGICAL FEATURES

This area receives rain under the influence of southwest and northeast monsoons. The southwest monsoon stretches from June to September, while the northeast monsoon is active between October and December. The annual average rainfall of the area is $\sim$1450 mm, of which 34% is received from southwest monsoon and 59% from northeast monsoon. Rainfall received during winter (January–March) and summer (April–May) months account for 3% to 4% of the annual rainfall respectively.

The coastal aquifer comprises fine to coarse-grained sand, sand and clay admixtures and it can be divided into top unconfined sandy aquifer and weathered & fractured semi-confined aquifer system. The unconsolidated formation can be further segregated locally into layers comprising grey sand- clay intercalation (zone III) is overlaid by grey sand with argillaceous intercalations and characterized by shells (Gastropods and Lamellibranches-zone II) and brown/red sand (zone I). In addition to the existing data, 18 piezometers were constructed in the study area tapping the top sandy zone, clayey sand zone and weathered basement individually to study the behavior of groundwater system. Newly constructed piezometer's litholog data indicate that all sites do not have the three zones mentioned above indicating limited lateral extension of various zones. The clayey sand aquifer is not present in all the sites along the eastern boundary indicating pinching out of the zone. Similarly, the top brown sandy aquifer encountered in the eastern part of the area is conspicuously absent on the western part. It is found that sandy layer of zone-II sediments encountered in the piezometers at Kottivakkam, Nilangarai, Injambakkam, Uthandi and Muthukadu area are hydraulically connected with top zone-I and are unconfined in nature. Zone-III sandy layer has been encountered on the eastern side of the study area at Kottivakkam, Injambakkam and Uthandi has a limited lateral extension and it is also found as clay-sand intercalations. It is characterized by semi-confined nature. The same layer encountered on the western side of the study area at Ekkatur, Sholinganallur and Mettukuppam and it is unconfined in nature. The weathered and fractured basement aquifer (zone IV) has been encountered in all sites and it is semi-confined in nature.

Three-aquifer systems exists at the eastern side of the study area, viz., the zones I and II (combined) unconfined aquifer and zones III and IV semi-confined aquifers. At the western side, two-aquifer systems are present in the zone III and IV. The thickness of the zone-I and II varies from 3–15 m, the zone III varies from 2–6 m and the zone IV is encountered thereafter on the eastern side of the area. On the western side (west of Buckingham Canal), the thickness of the zone III varies from 5–10 m and the zone IV is encountered thereafter. The cross sections A–A′ on the eastern side and the section B–B′ on the western side of Buckingham canal have been drawn in the N–S direction and show the sub-surface disposition of the multi-aquifer system (Fig. 18.2A & B).

Bay of Bengal in the east and the Buckingham canal passing across the area pose important boundary conditions. Both, the sea and the canal will have their influence on the groundwater regime of the unconfined aquifer. The spatial variation of elevation of groundwater table is studied for the pre and post monsoon periods to generalize the flow pattern (Fig. 18.3A & B). The variable thickness of the top sandy aquifer and its potential has resulted in local exploitation and the groundwater flow

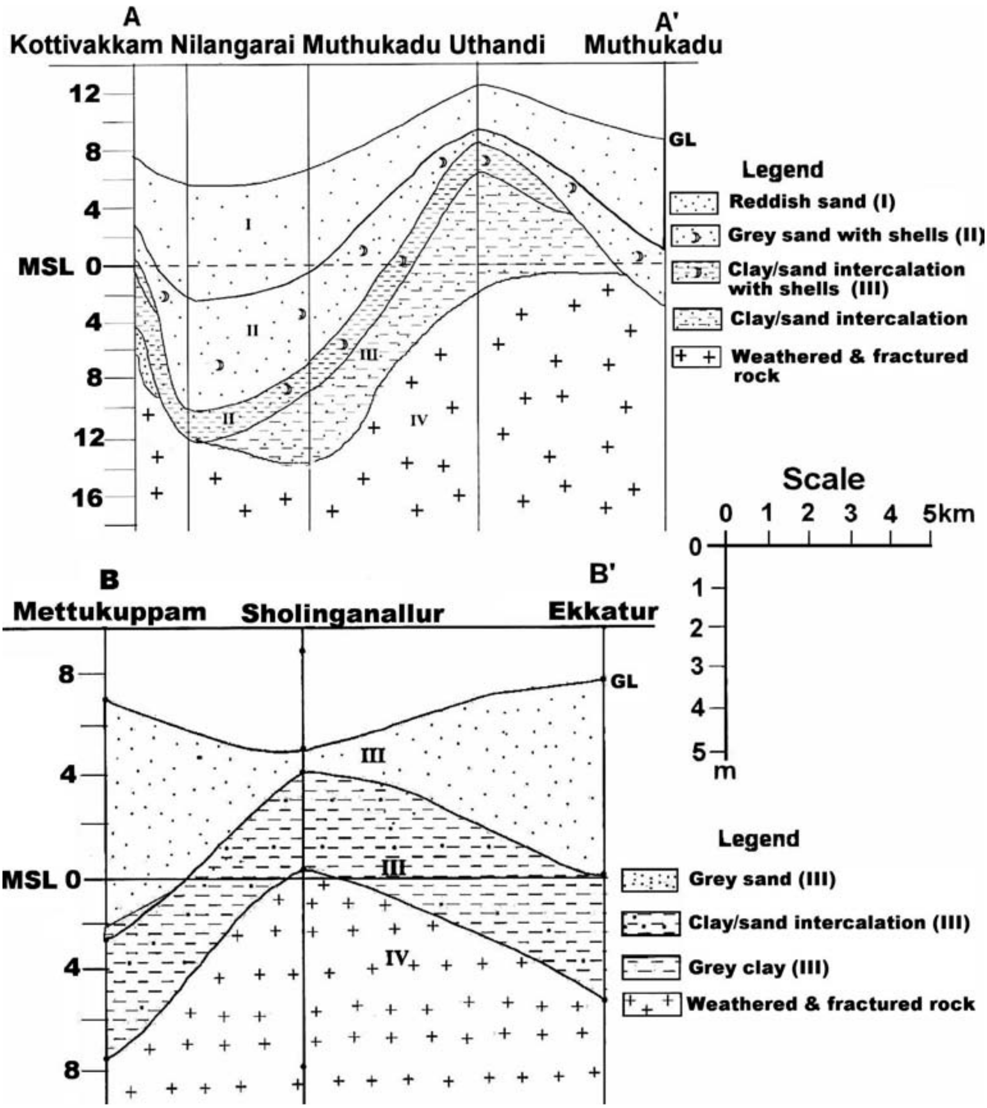

Figure 18.2. Subsurface geological cross sections (A–A′ and B–B′).

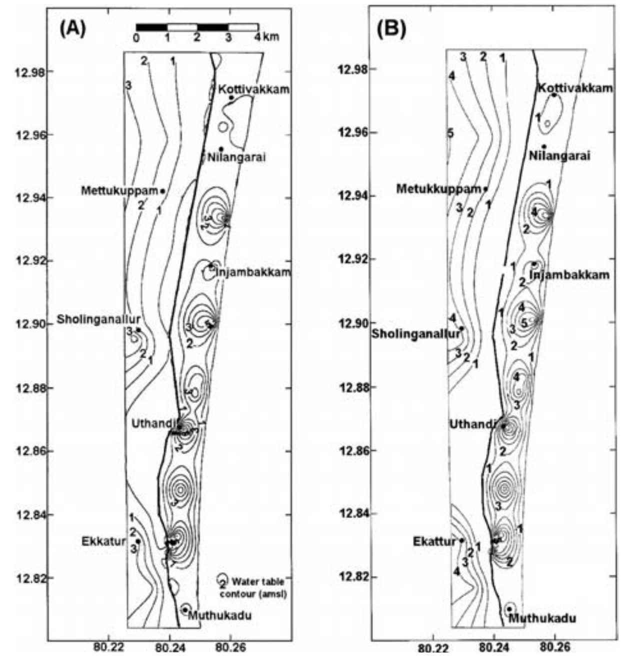

Figure 18.3. Water table elevation maps A) January 2001; B) May 2001.

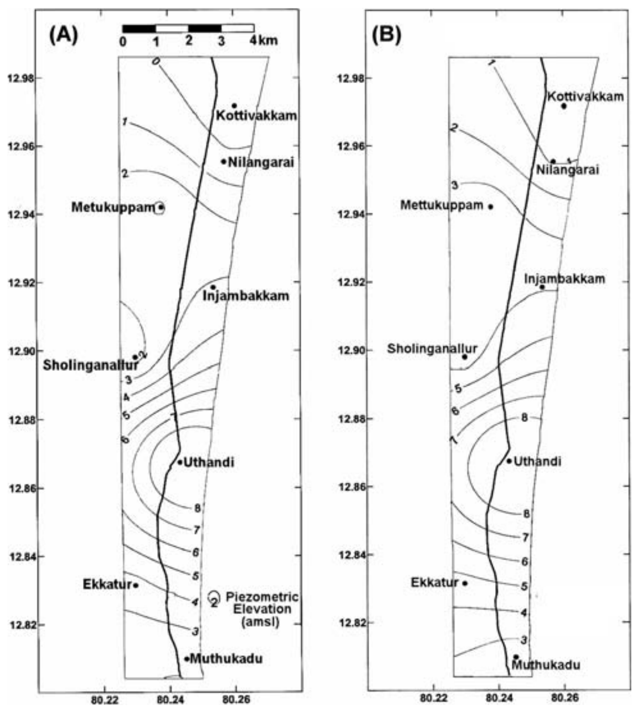

Figure 18.4. Piezometric elevation maps A) January 2001; B) May 2001.

pattern has been greatly affected. Groundwater contours run parallel to the coast and Buckingham canal with a flow pattern confirming the existence of groundwater draft in the area. The piezometric levels of pre-monsoon and post-monsoon periods (Fig. 18.4A & B) show the groundwater mound and radial outward flow pattern at Uthandi. In the northern part near Kottivakkam, a groundwater depression is noticed, indicating the over exploitation of the aquifer. The regional groundwater flow is from west to east direction.

18.4 SAMPLING AND ANALYTICAL METHODS

Water samples have been collected during June & December 2000 and in November 2001 from newly constructed piezometers, existing hand pumps, dug wells, surface water bodies, seawater and rainwater for the analyses of major and minor chemical species and environmental isotopes. Locations of sampling points are shown in Fig. 18.1. Samples collected for chemical analysis were filtered using 0.45 μm pore size membrane and stored in polyethylene bottles, which are initially washed with 10% HNO_3 and rinsed thoroughly with distilled water. A duplicate set is collected and acidified to pH < 2 by adding concentrated HNO_3 for cation, minor and trace metal analyses.

Physical parameters viz., pH, electrical conductance (EC) and temperature were measured in the field. Major, minor and trace chemical species were analyzed by Ion Chromatograph (Dionex-500) and Atomic Absorption Spectrophotometer. The accuracy of the measurement was verified by charge balance and the measurement error was in the range of -1.3 to $+4.1\%$.

For hydrogen and oxygen-18 isotope analyses, 25 mL of water samples were collected in airtight polyethylene bottles and the measurements were carried out using 602E VG ISOGAS mass spectrometer. The results are expressed in δ values, which is expressed in permil (‰) and written as

$$\delta^2\mathrm{H}(\text{or } \delta^{18}\mathrm{O}) = \left(\frac{R_{\text{sample}} - R_{\text{standard}}}{R_{\text{standard}}}\right) 10^3 \quad \text{where } R = \frac{{}^2\mathrm{H}}{{}^1\mathrm{H}} \text{ or } \frac{{}^{18}\mathrm{O}}{{}^{16}\mathrm{O}} \tag{18.1}$$

The precision of measurement for δ^2H and δ^{18}O is ±1.0‰ and ±0.2‰ respectively. For tritium measurement, 500 mL water samples were collected in airtight polyethylene bottle. 250 mL of distilled sample was electrolytically enriched at a low temperature of about 1 to 4°C and sample—scintillator mixture (8:12 mL) taken in a 20 mL polythene vial was counted in an ultra low background (0.5 cpm) liquid scintillation counter (Quantulus, Model 1220). The ^{3}H values are expressed in tritium unit (TU). One TU of sample has ^{3}H/^{1}H ratio equals to $1/10^{18}$, which corresponds to 0.12 Bq/kg of water. The minimum detection limit for this method is 0.5 TU (3σ) for 500 minutes counting. The counting efficiency and the calibration factor of the counter were about 25% and 70 TU/cpm respectively (Nair 1983).

For radiocarbon (Carbon-14) estimation, samples were collected in the form of $BaCO_3$ precipitate in the field. As the amount of carbonate required for ^{14}C estimation was equivalent to 1 gram of carbon, appropriate amount of saturated $BaCl_2$ solution was added to about 50–100 L of water sample (amount of sample depends on bicarbonate/carbonate concentration) taken in a polythene bag with pH raised to >9.0 by adding carbonate free NaOH solution. The formed $BaCO_3$ precipitate was settled by adding about 5 grams of $FeSO_4$ and 50 mL of polyacrylamidesolution. During precipitation and collection of precipitate, care has been taken to minimize contamination by modern carbon from atmosphere. The barium carbonate was acidified with concentrated H_3PO_4 and the evolved CO_2 after radon (Rn) decay was absorbed onto a mixture of carbosorb and scintillator in the ratio 11.5:11 mL. The sample was counted in an ultra—low background liquid scintillation counter Quantulus (background ~0.6 cpm). The ^{14}C activity is expressed in percent Modern Carbon (pMC). 100 pMC corresponds to 13.56 dpm/g of carbon (0.226 Bq/g of carbon). The experimental detection limit is 1.2 pMC for 1000 min. counting (3σ). The sensitivity factor and the counting efficiency are about 17 pMC/cpm and 43% respectively. The physical parameter data, major and minor chemical species, environmental stable and radio isotope results are presented in Tables 18.1 & 18.2.

Table 18.1. Physical parameters, major ions and environmental isotope data of water samples from the Tiruvanmiyur coastal aquifers.

S.No.	Location (zone)	Depth (m)	pH	Temp. (°C)	EC (μS/cm)	Na^{+} (mg/L)	K^{+} (mg/L)	Ca^{2+} (mg/L)	Mg^{2+} (mg/L)	Cl^{-} (mg/L)	SO_4^{2-} (mg/L)	CO_3^{2-} (mg/L)	HCO_3^{-} (mg/L)	NO_3^{-} (mg/L)	F^{-} (mg/L)	δ^2H (±1.0‰)	$\delta^{18}O$ (±0.1‰)	^3H (±0.5 TU)	^{14}C (a BP)
1	Kottivakkam (I & II)	8.5	7.8	30.0	785	74	31	112	5	50	259	0	195	5.6	0.6	−29.0	−4.8	7.4	–
2	Kottivakkam (III)	14.0	8.0	29.6	572	92	12	40	11	85	125	0	85	50	0.5	−32.6	−5.2	6.3	–
3	Kottivakkam (IV)	24.7	7.7	30.1	404	78	8	20	13	71	91	0	110	5	0.7	−34.2	−5.3	5.5	–
4	Nilangarai (I & II)	16.5	8.4	30.5	245	18	2	52	16	39	72	0	104	3	0.4	−33.7	−5.8	6.4	–
5	Nilangarai (IV)	27.9	9.0	30.4	2700	708	10	32	22	496	96	42	708	322	2.4	−36.5	−6.1	1.4	11600
6	Injambakkam (I & II)	14.4	8.0	30.0	370	28	2.4	56	19	28	120	0	116	0.6	0.3	−37.9	−6.0	8.2	–
7	Injambakkam (III)	20.7	8.7	30.6	1010	274	14	8	13	64	96	60	446	58	2.5	−35.3	−6.0	2.1	–
8	Uthandi (I & II)	0.6	7.0	28.0	1450	179	110	98	26	149	235	60	384	11.2	0.6	−31.5	−4.3	15.5	–
9	Uthandi (III)	13.9	8.5	29.0	1770	404	24	64	10	333	82	0	659	2.5	1.5	−35.2	−4.3	2.8	–
10	Uthandi (IV)	23.2	8.2	29.2	1655	400	10	44	10	198	163	66	513	61	2.6	−40.0	−5.0	1.3	11400
11	Muttukkadu (I & II)	13.0	8.2	33.0	198	44	2	24	7	18	2.4	18	116	43	0.9	−27.3	−5.4	9.5	–
12	Muttukkadu (IV)	20.6	8.9	32.5	1450	354	20	30	6	266	120	42	348	50	0.9	−32.6	−5.6	3.5	–
13	Ekkatur (III)	13.0	6.5	30.3	1170	138	4	122	44	369	91	0	128	81	0.3	−34.5	−6.1	6.9	–
14	Ekkatur (IV)	18.8	7.0	30.4	1214	147	2	102	46	379	144	0	92	6.8	0.7	−32.3	−5.8	3.8	–
15	Sholinganallur (III)	5.6	6.5	29.3	1050	62	43	148	41	340	19	0	281	3.7	0.8	−21.2	−3.4	16.2	–
16	Sholinganallur (IV)	15.7	6.1	29.4	875	138	4	74	43	149	240	0	232	12.4	1.0	−20.2	−3.3	15.9	–
17	Mettukuppam (III)	12.0	6.2	29.3	17420	4324	141	1202	1215	9926	3168	0	305	3.7	1.2	−34.9	−5.7	7.0	–
18	Mettukuppam (IV)	25.1	7.2	29.0	19230	5221	180	1283	1264	10493	4656	0	317	81	1.6	−38.2	−5.4	1.0	6900
19	Mettukuppam (DW)	–	6.4	29.0	1905	430	20	116	49	525	499	0	183	10	0.4	−34.5	−5.5	9.1	–
20	Mettukuppam (DW)	–	6.6	–	2170											−25.0	−4.5	11.0	–
21	Mettukuppam (ML)	–	8.5	–	3040											2.0	1.4	–	–
22	Muthukadu (BW)	–	9.6	31.0	20000	3887	219	152	292	9217	576	12	116	18.6	1.4	5.3	0.8	21	–
23	Taramani (HP)	–	7.2	30.0	1915	262	14	116	107	550	53	0	488	91	0.6	−12.3	−1.8	12.2	–
24	Sheelanagar (HP)	–	7.2	30.5	1635	166	8	60	83	319	250	0	128	43	0.4	−22.9	−3.9	11.6	–
25	Jaladampettai (HP)	–	7.2	31.6	890	106	4	36	36	89	226	0	122	11.2	0.7	−30.7	−5.4	6.0	–
26	Sitalapakkam (HP)	–	6.4	32.0	519	30	2	60	28	99	12	0	214	3.1	0.6	−32.5	−6.4	7.9	–
27	Talambur (HP)	–	7.2	31.0	1620	87	6	200	85	248	307	0	281	174	0.8	−28.3	−3.8	8.3	200
28	Uthandi Canal	–	8.4	28.6	3790	759	51	112	136	1276	432	48	250	19	0.6	− 8.7	−2.6	8.7	–
29	Seawater	–	8.4	27.3	20000	7671	301	204	559	16662	768	24	79	37	0.6	− 0.6	−1.3	4.0	–
30	Rain Water (January)					Samples not analysed for water quality parameters										−16.5	−4.0	21.7	
31	Rain water (February)															−35.5	−6.5	2.2	–
32	Rain water (July)															−22.2	−4.1	21.7	–
33	Rain water (August)															−35.3	−5.0	25.1	–
34	Rain water (October)															−40.0	−5.4	27.4	–
35	Rain water (November)															−30.9	−5.1	4.7	–
36	Rain Water (December)															−30.0	−5.8	2.2	

Note: I–Reddish sand zone , II–Grey sand & clay zone, III–clay sand intercalation zone, IV–weathered & fractured rock, DW–Dug Well, ML–Marshy Land, BW–Back Water, HP–Hand Pump & BDL–Below detection limit.

Table 18.2. Minor ion data of the groundwater samples from Tiruvanmiyur coastal aquifers.

	Ions	Kottivakkam			Uthandi		Muttukadu		Ekkattur		Mettukuppam		Nilangarai	
S.No.	(mg/L)	I & II	III	IV	I & II	IV	I & II	IV	III	IV	III	IV	I & II	IV
1	Li^{+}	<0.01	0.0	<0.01	0.0	<0.01	<0.01	bdl	0.0	0.0	0.1	0.1	bdl	bdl
2	Sr^{2+}	0.4	0.6	0.0	0.0	bdl	bdl	bdl	0.4	0.6	0.0	1.29	bdl	0.00
3	Br^{-}	0.0	0.1	0.1	bdl	0.4	0.0	3.6	0.0	1.1	24.4	33.1	0.00	3.2
4	PO_4^{3-}	3.2	0.1	1.4	4.6	6.7	bdl	0.0	0.8	3.2	14.0	16.4	2.40	8.2

18.5 RESULTS

18.5.1 *Hydrochemical characteristics*

The hydrochemical data show that the quality of groundwaters in zone I and II is generally fresh with EC <1500 μS/cm. These groundwaters are neutral to moderately alkaline with pH ranging from 7.5–8.7. Also in zone III, the quality of water is fresh except in Uthandi (brackish quality, EC: 1770 μS/cm) and Mettukuppam (saline, EC: 27100 μS/cm). Groundwaters in this zone are neutral to moderately alkaline with pH values in the range of 7.3–8.8. In zone IV, water is brackish at Taramani, Sheelanagar, Talambur and Nilangarai (EC 1600–2700 μS/cm respectively) and saline at Mettukuppam (EC 31000 μS/cm). These waters are neutral to alkaline and their pH values range from 7.6–9.1.

The overall hydrochemistry of unconfined and semi-confined aquifers in the study area, indicates that the groundwaters are dominated by cations Ca^{2+}, Mg^{2+} and Na^{+} and anions HCO_3^{-}, SO_4^{2-} & Cl^{-}. The saline groundwaters at Mettukuppam (zone III and IV) show high concentrations of Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, SO_4^{2-}, Cl^{-}, and NO_3^{-}. High NO_3^{-} content (50–322 mg/L) has also been observed in groundwaters (fresh as well as brackish) of other locations. Nitrate concentration in uncontaminated groundwater is generally below 2 mg/L (Edmunds et al. 1977) and concentration above this value points to anthropogenic pollution (Hitchon et al. 1999). Groundwaters in post-monsoon period show high nitrate concentration in Nilangarai, Mettukuppam, Taramani, Talambur and Ekkatur areas compared to pre-monsoon. This nitrate contamination has been noticed in wells located near the sewerage drains or latrine pits or cattle shed. This suggests that the possible source of NO_3^{-} is derived from solid and liquid wastes, which is evident from the rise in NO_3^{-} values in the post monsoon samples.

18.5.1.1 *Major ion trends—geochemical evolution*

Major ion chemical data has been plotted on Piper's diagram (Piper 1953) to depict the quality of groundwater as well as the possible pathways of fresh water and saline water movement and abstracted geochemical processes (Fig. 18.5). The saline groundwaters belong to the chemical facies Na-Mg-Cl- SO_4 type, brackish waters are of Na-HCO_3, Na-Mg-Ca-Cl-CO_3, Ca-Mg-Cl-SO_4-HCO_3 and Na-Mg-Cl-SO_4 types whereas fresh waters are of Ca-Na-SO_4-HCO_3, Ca-Mg -HCO_3-SO_4 and Ca-Mg-Na-SO_4-HCO_3 types. Zone wise similarity in chemical types is not seen in most of the locations except at Mettukuppam and Ekkatur suggesting remote possibility of interconnection between different zones. Water samples collected from different zones from northwest and southern parts of the study area show high Ca^{2+}, Mg^{2+} and SO_4^{2-} concentrations compared to the samples from eastern and northern parts indicating that garnetiferous granitic gneiss as a source.

Chloride is a conservative ion, as it does not react or enter into precipitation–dissolution processes except at brine concentrations and rarely enters into redox or adsorption reactions (Feth 1981). Therefore, correlations between major ion concentrations with chloride content help in understanding the possible water—rock interaction and mixing processes (Marie et al. 2001). The concentration of major chemical species versus chloride are plotted and shown in Fig. 18.6 (a–f).

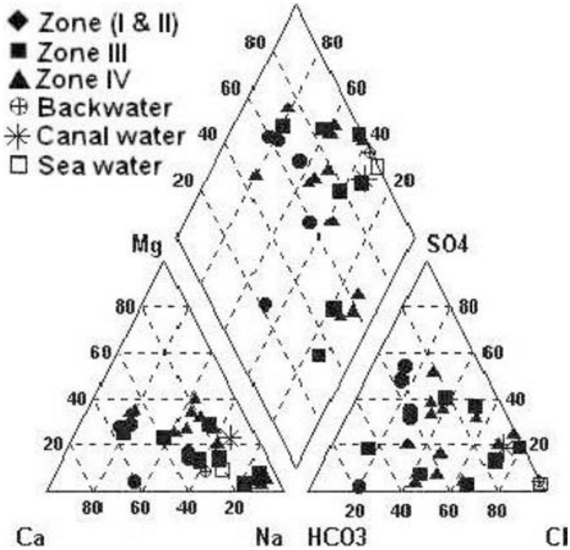

Figure 18.5. Major ion composition of the water samples plotted on a Piper's trilinear diagram.

In these plots, seawater collected near Peria Nilangarai and fresh water from Muthukadu are used as two end members to draw sea water—fresh water mixing line. In K^+, Ca^{2+}, Mg^{2+}, HCO_3^- & SO_4^{2-} versus Cl^- plots, saline groundwater samples lie above the mixing line, where as in Na^+ versus Cl^- plot, they lie on mixing line in between fresh water and seawater end members. Backwater sample lie on mixing line and the fresh water samples form a single cluster near freshwater end member except in Cl^- versus HCO_3^- plot. These plots suggest that the source of groundwater salinity in this aquifer is due to both seawater as well as leaching of salts from the formation. The high concentrations of K^+, Ca^{2+}, Mg^{2+}, HCO_3^- and SO_4^{2-} are acquired by water-rock interaction. Brackish samples fall above the mixing line near fresh water end member indicating that brackish quality of groundwater is due to mixing of evaporated surface water.

Many researchers used ionic ratios of $(Ca^{2+} + Mg^{2+})/(Na^+ + K^+)$ to distinguish between invading and retreating saline water bodies and also to verify the ion exchange processes (Mercado 1985, Tellam et al. 1986 & Said Ghabayen et al. 2006). In the present study, ionic ratio of $(Ca^{2+} + Mg^{2+})/(Na^+ + K^+)$ has been computed based on equivalent parts per million (epm) and used to confirm the possible seawater intrusion into the aquifer. Brackish samples of zone IV (Uthandi and Nilangarai) show ionic ratios 0.11 & 0.22 respectively. These values are less than the seawater value (0.43). Generally more diluted connate waters show lower ionic ratio compared to modern seawater (White 1957). A few brackish and saline samples of the same zone but different locations (Muthukadu, Taramani, Sheelanagar and Mettukuppam) show higher ionic ratios ranging from 0.73 to 1.32 compared to seawater value suggesting that the source of brackishness and groundwater salinity is not due to sea water.

18.5.1.2 *Minor and trace elements*

About five minor and trace elements (F^-, Br^-, PO_4^{3-}, Li^+ & Sr^{2+}) concentrations have been measured in a few selected samples and the data is presented in Table 18.1. The highest fluoride concentrations (1.5–2.5 mg/L) are encountered mainly in zone III and zone IV at Nilangarai, Injambakkam, Uthandi and Mettukuppam. In shallow zone (zones I and II), F^- concentration is <1 mg/L. A positive correlation between high bicarbonate and high fluoride indicates that leaching of fluoride from minerals is the process governing F^- contamination in these zones. Bromide content in groundwater samples from zone I & II and zone III are below the normal value and

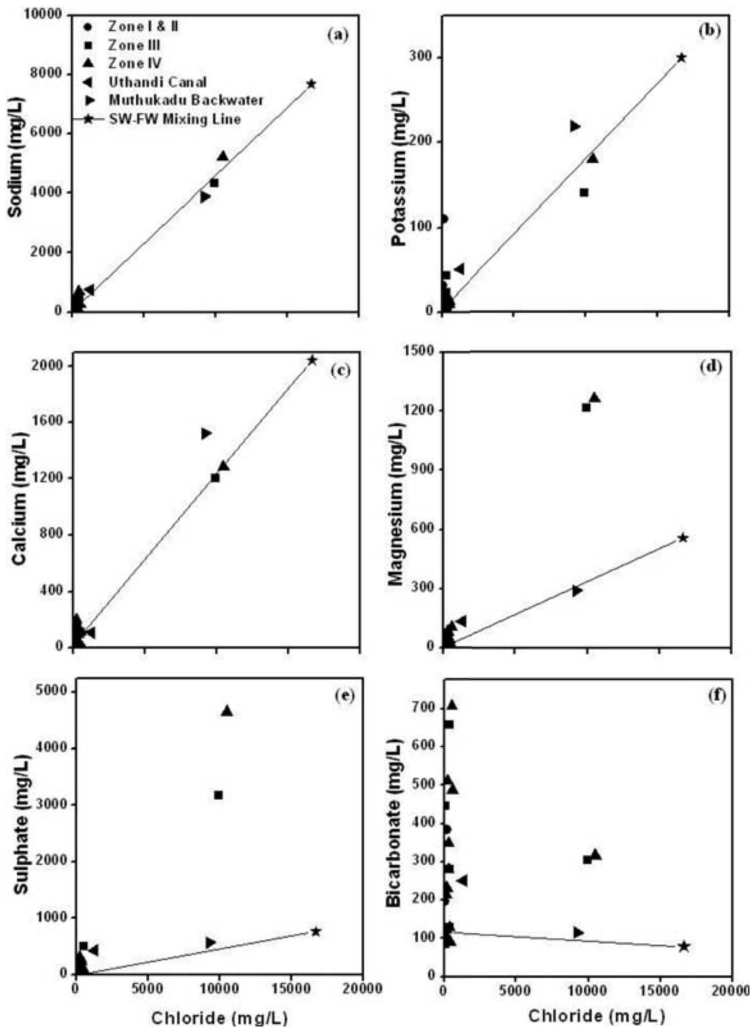

Figure 18.6(a–f). Correlation plots of major ion concentrations versus chloride concentrations.

in the Weathered and fractured aquifer, it is in the range of 0.1 to 3.6 mg/L. Saline groundwater in zones III and IV at Mettukuppam show high Br^- concentration (24 & 33 mg/L respectively) compared to other groundwaters. Sources of Br^- in groundwater are evaporated sea water or brine (Chave 1960) and marine organism present in the sediments. The brackish waters of Nilangarai and Muthukadu have Br^- concentrations 3.2 and 3.6 mg/L and very low Br^-/Cl^- ratios (0.19×10^{-3} and 0.16×10^{-3} respectively). This Br^- might have acquired from sediment. The observed high Br^- content (24 and 33 mg/L) and Br^-/Cl^- ratios 1.28×10^{-3} and 1.74×10^{-3} in zones III and IV at Mettukuppam are indicative of diluted seawater.

In fresh and brackish waters the Sr^{2+} concentration is found to be <0.6 mg/L. In contrast, saline water samples from zone IV of shows highest Sr^{2+} concentration (1.3 mg/L) indicating seawater contribution. Phosphate exhibits a similar distribution pattern like Sr^{2+}. In top sandy aquifer (zone I, II and III) samples show the phosphate contents <4.6 mg/L, while in the basement aquifer (zone-IV), the highest values are found and are in the in the range of 1.4–8.2 mg/L. Relatively high PO_4^{3-}

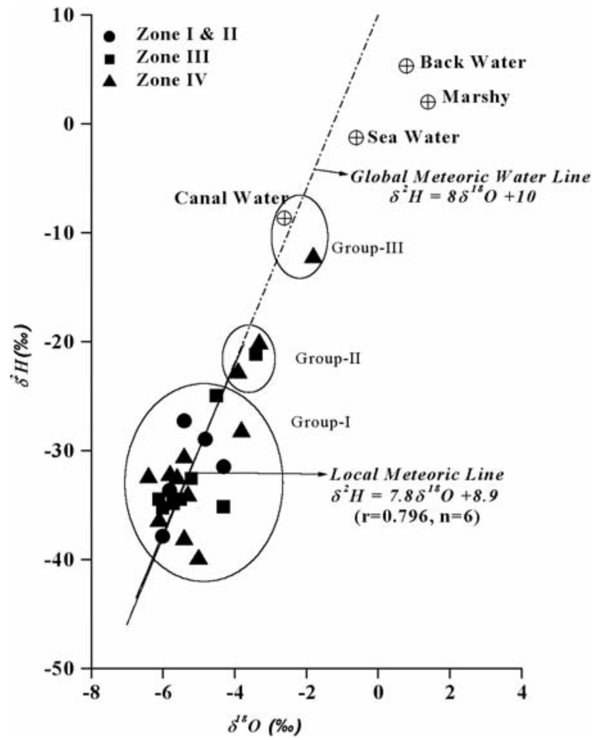

Figure 18.7. Plot of deuterium versus δ^{18}O.

concentrations are registered in saline samples (~15 mg/L). In general, deep zone groundwaters have high concentrations of phosphate compared to shallow groundwaters.

18.5.2 *Environmental isotope studies*

18.5.2.1 *Stable isotopes (^2H & ^{18}O)*

In the study area, local precipitation has δ^2H and δ^{18}O values are between −40 to −16.5‰ and −6.5 to −4‰ respectively. Surface waters (marshy land, backwater, Uthandi canal and seawater) have δ^2H and δ^{18}O values in the range of −8.8 to + 5.3‰ and −2.6 to +1.4‰ respectively. Groundwaters of zone-I, II & III have δ^2H and δ^{18}O values in between −37.9 to −21.2‰ & −6.1 to −3.4‰ respectively and in zone-IV, the values are in the range of −40 to −20.2‰ and −6.1 to −3.3‰ respectively.

In δ^2H vs. δ^{18}O plot, precipitation data for the year 2000 was used to construct local meteoric water line (LMWL) and the best fit equation is found to be $\delta^2\text{H} = 7.8\ \delta^{18}\text{O} + 8.9$ with regression coefficient of 0.79 (for n = 6) shown in Fig. 18.7. Global meteoric water line is also shown in the figure for reference. The backwater and marshy land samples are enriched in isotopic contents (δ^2 H: +2 to +5.3‰ and δ^{18}O: +0.8 to +1.4‰) compared to seawater indicating evaporation effect. The Uthandi canal sample and most of the fresh water samples of zones I, II, III and IV lie around LMWL indicating their meteoric origin without significant evaporation prior to infiltration. Water samples of Uthandi area (zones I, II and III) and Talambur hand pump (zone IV) show slightly enriched stable isotopic values compared to group-I samples and they lie away from the LMWL. The isotopic enrichment and brackish quality of these groundwaters are due to the contribution from evaporated surface water. Water samples of zone III and IV of Sholinganallur and Taramani

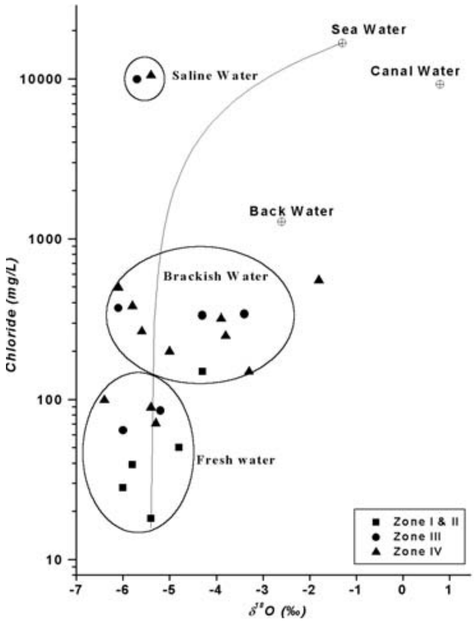

Figure 18.8. Plot of chloride versus δ^{18}O.

samples also lie away from the LMWL indicating evaporation effect. Muthukadu backwater and sample collected from marshy land near Mettukuppam are more enriched in stable isotopic content compared to seawater. The saline groundwaters of zones III and IV at Mettukuppam show a depleted δ^2H and δ^{18}O values compared to seawater. This is in contrary to the observed values of saline groundwaters from coastal aquifers (Shivanna et al. 1993). The positive correlation between the salinity of the waters and their δ^2H and δ^{18}O values in the other studies have been attributed to mixing between meteoric water and sea water or saline brines that have evolved through seawater modification or seawater intrusion. In the present study, the saline groundwaters show much depleted δ^2H and δ^{18}O values similar to that of local precipitation values.

In Cl vs. δ^{18}O plot, the fresh groundwater sample of Injambakkam having least chloride content and seawater collected at Peria Nilagarai were used as end members to construct seawater—fresh water dilution line, shown in Fig. 18.8. In this plot, all fresh water samples of zones I, II (except sample of Uthandi), as well as those from zones III and IV fall in one group. A few brackish water samples of zones III and IV are scattered on either side of the dilution line indicating the source of brackishness is evaporated surface water bodies such as canal, marshy land or leaching of salts from the formation. Saline samples lie left side and away from the dilution line indicating leaching of salts from formation in addition to seawater as the source of salinity. Backwater and canal water samples are also lie away from the dilution line indicating evaporation effect.

18.5.2.2 *Environmental tritium (^{3}H)*

In the study area, local precipitation has tritium content in the range of 2.2–27.4 TU. The rainwater samples collected during southwest monsoon show higher ^{3}H content (21.7–27.4 TU) than northeast monsoon rainwater samples (2.2–4.7 TU). Seawater has a ^{3}H value of 4.0 TU and in Muthukadu backwater, it is 20.9 TU. In groundwaters, ^{3}H content varies from 0.95–16.2 TU. In ^{3}H vs. δ^{18}O plot, water samples fall in three groups (Fig. 18.9). Most of the zone IV groundwater samples as well as the sample of Injambakkam (zone III) cluster in group (I) and their tritium content is <5 TU indicating old water. Water samples from zones I, II, III and zone IV at Kottivakkam, Jaladampettai and Sittalapakkam fall in-group (II) showing modern ^{3}H in the range of 5–10 TU. Apart from

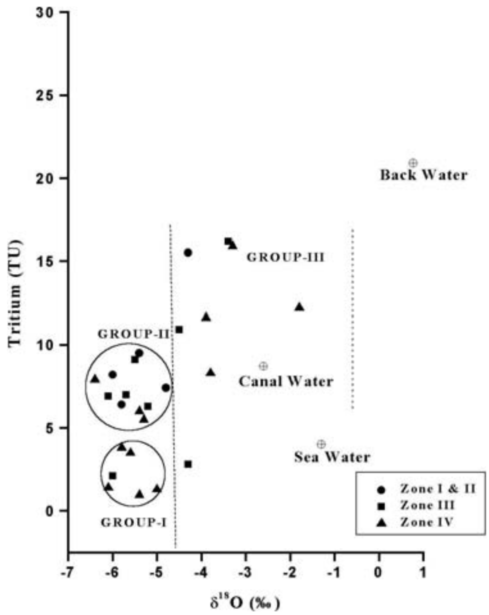

Figure 18.9. Plot of tritium versus δ^{18}O.

this, high ^{3}H content of 10 to 16 TU is noticed in zones III and IV at Sho linganallur, zone IV at Sheelanagar and Taramani as well as in the zones I and II at Uthandi [group (III)] and indicate relatively modern waters.

18.5.2.3 *Radiocarbon (^{14}C)*

Four carbon—14 samples were collected from locations where tritium content in groundwaters was less than 3TU. The ^{14}C content of these samples (Talambur, Nilangarai, Uthandi and Mettukuppam) ranges from 24.4–97.2 pMC. The uncorrected carbon—14 ages of these samples show that the saline groundwater at Mettukuppam is 6900a BP and brackish water at Uthandi and Nilangarai is 11,600a and 11,400a BP respectively. The brackish water at Talambur is modern.

18.6 DISCUSSION

The compilation of hydrogeological data reveals that the Tiruvanmiyur aquifer can be considered as two-aquifer system, viz., top sandy aquifer and weathered and fractured basement aquifer. The groundwater occurs under unconfined condition in top sandy aquifer and under semi-confined condition in the underlying basement. The groundwater contour maps of post and pre-monsoon period indicate that the contours are parallel to the coast with flow pattern confirming the existence of groundwater draft in the area. The elevation of piezometric surface during the same period shows that groundwater mound at Uthandi. In the northern part of the study area near Kottivakkam, a groundwater depression is noticed, indicating over exploitation of groundwater. The rainfall is the main source of recharge to the top sandy aquifer while the weathered and fractured basement aquifer gets the recharge from rainwater from the local depression on the western side and also receives recharge across the western boundary of the study area.

Depth wise similarity in the hydrochemical facies was not observed at many locations except at Mettukuppam and Ekkatur area, indicating the remote possibility of inter-connection between the different water bearing zones. Most of the fresh and brackish groundwaters lie in a single cluster near the fresh end member indicating a single source of recharge (Fig. 18.6) and these samples lie very close to LMWL in δ^2H vs. δ^{18}O plot (Fig. 18.7) except Sholinganallur and Uthandi, suggesting precipitation as the main source of recharge. Water samples from the top sandy aquifer at Uthandi, Sholinganallur and the basement aquifer samples from Uthandi, Talambur, Sholinganallur, Taramani area show deviation from the meteoric water line indicating an additional source of recharge from evaporated surface water bodies such as marshy land Buckingham Canal. As such, there is no indication of seawater intrusion observed in the top sandy aquifer even though northern part of the study area show groundwater levels close to mean sea level (MSL). This could be due to occurrence of fresh water lenses under sand dunes located in the western part of the study area, which arrest the seawater intrusion. A similar observation was made by Ballukraya et al. (1998) in Chennai City.

The low $(Ca^{2+} + Mg^{2+})/(Na^{+} + K^{+})$ ionic ratios compared to seawater, depleted hydrogen and oxygen values, low tritium and low carbon contents in brackish groundwaters of basement aquifer at Uthandi and Nilangarai suggests that these are diluted connate waters. The enriched ionic ratios, high stable isotopic values and tritium in brackish groundwaters of the top sandy aquifers at Uthandi, Sholinganallur, Mettukuppam, Taramani, Sheelanagar and Talambur point to the mixing of evaporated surface water to groundwater system. Brackish groundwaters at Muthukadu and Ekkatur show enriched ionic ratio, depleted stable isotopic values and modern tritium content indicating that brackish quality of groundwater is due to leaching of salts from the formation.

The saline groundwaters at Mettukuppam show high concentrations of Ca^{2+}, Mg^{2+} and SO_4^{2-} and they have different hydrochemical facies compared to other groundwaters. These samples lie separately on the dissolved chemical species vs. chloride plot (Fig. 18.6) and show enriched in $(Ca^{2+} + Mg^{2+})/(Na^{+} + K^{+})$ ratios compared to seawater. Further, the saline groundwaters lie on the meteoric water line in δ^2H and δ^{18}O plot (Fig. 18.7) and above the dilution line in Cl^- vs. δ^{18}O plot (Fig. 18.8) and their hydrogen and oxygen values are similar to that of local precipitation values. These samples have negligible tritium and low ^{14}C content. These findings demonstrate that evaporation and membrane filtration of meteoric water are not likely mechanisms for the origin of salinity. These saline groundwaters are derived from mixing seawater and fresh water with depleted δ^2H and δ^{18}O values. The mixing of highly depleted precipitation water (amount effect) can account for depleted δ^2H and δ^{18}O values in these saline groundwaters. Further, these saline groundwaters have been modified due to water-rock interaction. Thus the source of salinity in this groundwater is old seawater entrapped during Flandrian transgression in Holocene period. The host rock garnetiferous gneiss is probably the source for high magnesium, calcium, and sulphate contents in these groundwaters.

The observed high nitrate concentration in wells located close to latrine pits and sewage drains, its seasonal variation and presence of modern tritium content suggest the nitrate contamination from solid and liquid wastes. The high fluoride concentration in groundwaters at Nilangarai, Uthandi, Injambakkam and Mettukuppam are of Na-HCO_3 and Na-Mg-Cl type and there is a positive correlation between bicarbonate and fluoride in these groundwaters. High fluoride groundwaters have bicarbonate in the range of 300–700 mg/L and pH values are in the range of 7.0–9.0. This is in good agreement with the experiment done by Saxena et al. (2001) showing that the alkaline medium (pH 7.6–8.6), high HCO_3^- concentration (350–450 mg/L) and moderate EC were found to be favorable conditions for calcium fluoride dissolution in groundwater. Since the fluoride concentration is high in the basement aquifer compared to top sandy aquifer, the source of fluoride is geogenic.

High tritium content in precipitation during southwest monsoon period and in a few groundwater samples (group III, Fig. 18.9) in the study area could be due to the influence of releases from nuclear installation located at about 50 km south of Tiruvanmiyur aquifer. Similar values were observed in earlier studies (Shivanna et al. 1998). ^{14}C ages of groundwaters from Uthandi and Nilangarai are found to be about 11500 a BP indicating dissolution of aquifer material due to long residence time as the process for the observed brackishness in these groundwaters. Saline groundwaters at Mettukuppam show a ^{14}C age of 6900a BP indicating the source of salinity is seawater entrapped during Flandrian sea transgression in Holocene period.

18.7 CONCLUSIONS

The groundwater in this aquifer is generally fresh, except at a few locations. The top unconfined aquifer receives precipitation as the only source of recharge, except at Sholinganallur, where, recharge is from both surface water bodies and local precipitation. The weathered & fractured aquifers receive recharge from rainwater collected in the depression on the eastern side of canal as well as lateral flow across the study area.

The brackish groundwater at Uthandi and Nilangarai are old connate waters and their ^{14}C ages are 11,400 and 11,600a BP respectively. The brackish quality of groundwater in the top sandy aquifer at Uthandi, Sholinganallur and Mettukuppam dug well and in the basement aquifer at Taramani, Sheelanagar and Talambur is due to the mixing of evaporated surface water and leaching of salts from formation. The brackish quality of groundwaters in basement aquifer at Muthukadu and Ekkatur is due to leaching of salts from the formation and they are modern. The source of groundwater salinity at Mettukuppam is old seawater and salts acquired from the formation. The seawater is entrapped during Flandrian sea transgression in Holocene period and it is 6900a BP.

The high NO_3^- in groundwater is mainly due to anthropogenic activities whereas fluoride in the basement aquifer is geogenic.

There is no indication of modern seawater intrusion in the study area. But over exploitation of groundwater in this area may lead to seawater intrusion in intermediate zones of aquifer, as the active interconnections are not found among different zones.

ACKNOWLEDGEMENTS

This work is a part of a joint co-operation program between Isotope Applications Division, BARC, Central Groundwater Board, Chennai and M.S. Swaminathan Research Foundation (MSSRF), Chennai. Mr. M.B. Raju, Former Regional Director, CGWB and Dr. M.S. Swaminathan, MSSRF, were the main architects of the project. Dr. V. Venugopal, Director, Isotope Group and Dr. Gursharan Singh, Head, Isotope Applications Division, BARC, have offered their valuable support in the course of the project. Their contribution is sincerely thanked. The authors thank Dr. Ajay Kumar Parida, Programme Director-Biotechnology, MSSRF for his kind cooperation during the project period and all the technical staff for their support in carrying out the analytical work.

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CHAPTER 19

Evaluation of the coastal groundwater resources using limited hydrogeological data

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ABSTRACT: A conceptual model based on the sharp interface assumption was used to evaluate the sensitivity of saltwater intrusion to groundwater recharge. Sensitivity analysis shows that hydraulic conductivity and groundwater recharge are the main hydro-geological factors which affect the dynamics of freshwater-saltwater interface. The impacts of possible management scenarios on groundwater recharge have been investigated using different climate changes and land use pattern combinations. The combined climate and land use scenarios show that when aridity index is less than 60, the agricultural lands give higher groundwater recharge than other land use patterns. Relevant recharge values have been used to simulate the salinity intrusion and the freshwater loss in coastal aquifers. The combined effects of deforestation and aridity index on fresh groundwater loss show that deforestation causes the increase of the recharge and existing fresh groundwater resource in areas having less precipitation and high temperature.

19.1 INTRODUCTION

The availability of surface water is severely limited in many areas in the world. Groundwater acts as the largest available source of freshwater that is being extensively used to supplement the available surface water. While considering the water resource in areas bordering seas, coastal aquifers are very important resource of freshwater. The use of coastal aquifers as operational reservoirs in water resources systems requires the development of tools that make it possible to predict the behavior of the aquifer under different conditions. Since groundwater systems in coastal areas are in contact with saline water, one of the major problems is the prediction of the motion of the saltwater body in the aquifer. Already at this moment, many coastal aquifers in the world, especially shallow ones, experience an intensive salt water intrusion caused by both natural as well as anthropogenically induced processes. The quantitative understanding of the patterns of movement and mixing between freshwater and saltwater, and the factors that influence these processes, are necessary to manage the coastal groundwater resources.

The freshwater-saltwater interface seldom remains stationary. Large scale recharging into and withdrawals from the aquifer, result the movement of the interface from the steady position to another. The movement will be advancing or retreating depends on whether the freshwater flow through the aquifer is decreased or increased. The salinisation of coastal aquifers will accelerate due to the reduction of groundwater recharge. This could mean a reduction of fresh groundwater resources. Under such circumstances it is important to study the change in coastal fresh groundwater resources due to changes in recharge or discharge of groundwater and related activities such as climate changes and change in land use pattern.

Changes in climatic factors, such as precipitation and temperature and the land use change are very important part of the hydrologic balance. Land-use patterns are highly dynamic and rarely in a stable equilibrium. Land-use change may have regional or even global effects as a result of the accumulation of many small-scale, local changes. The extent and the temporal dynamics of the plant cover as well as the associated dominant crop characteristics have a strong impact on the potential evapotranspiration rate and the initiation of surface runoff. The main objectives of this study are to understand the effects of groundwater recharge on freshwater-saltwater interface and to evaluate the effects of climatic and land use changes on salinity intrusion. We try to evaluate the effects of changes in climatological factors such as precipitation and temperature, and land use on the groundwater recharge and the quantification of the groundwater recharge response to estimate the loss of fresh groundwater resources in coastal aquifers due to salinity intrusion.

19.2 MATERIALS AND METHODS

19.2.1 *Numerical modeling of the movement of freshwater—saltwater*

Many models have been developed to represent and to study the problem of saltwater intrusion. They range from relatively simple analytical solutions to complex numerical models. The first concept about freshwater saltwater interface, now widely cited as the Ghyben-Herzberg principle, is based on the hydrostatic equilibrium between fresh and saline water. After introducing Ghyben-Herzberg principle, several analytical solutions were published to describe various forms problems related to seawater intrusion in coastal aquifers (Glover 1959, Van Der Veer 1977, Bear 1979, 1999).

Recently the studies involving the movement of freshwater and saltwater in coastal aquifer systems are classically studied using two different approaches (Reilly & Godman 1985). In the first approach, freshwater and saltwater are assumed completely immiscible and a sharp interface exists between these two phases. In the other approach, the freshwater and saltwater are assumed to be in a dynamic equilibrium resulting from the flow and dispersion mechanisms within the aquifer.

Sharp interface models couple the freshwater and saltwater flow based on the continuity of flux and pressure. In this approach, together with Dupuit approximation for each flow domain, the equation of continuity may be integrated over vertical direction and come up with following system of differential equations (Bear 1979).

$$\frac{\partial}{\partial x}\left[K_{fx}(h^f - h^i)\frac{\partial h^f}{\partial x}\right] + \frac{\partial}{\partial y}\left[K_{fy}(h^f - h^i)\frac{\partial h^f}{\partial y}\right] + q_f = S_f\frac{\partial h^f}{\partial t} - \theta\left[(1+\delta)\frac{\partial h^s}{\partial t} - \delta\frac{\partial h^f}{\partial t}\right] + \alpha\theta\frac{\partial h^f}{\partial t} \tag{19.1}$$

$$\frac{\partial}{\partial x}\left[K_{sx}(h^i - z^b)\frac{\partial h^s}{\partial x}\right] + \frac{\partial}{\partial y}\left[K_{sy}(h^i - z^b)\frac{\partial h^s}{\partial y}\right] + q_s = S_s\frac{\partial h^s}{\partial t} + \theta\left[(1+\delta)\frac{\partial h^s}{\partial t} - \delta\frac{\partial h^f}{\partial t}\right] \tag{19.2}$$

The location of the interface elevation (h^i) is given by

$$h^i = \frac{\rho_s}{\rho_s - \rho_f}h^s - \frac{\rho_f}{\rho_s - \rho_f}h^f \tag{19.3}$$

where ρ_f and ρ_s are specific weight in fresh and salt water respectively, h^f and, h^s are the piezometric heads of freshwater and saltwater regions, q_f and q_s are flow rate in fresh and salt water respectively.

K_f and, K_s represent the hydraulic conductivity in fresh and salt water regions. Storage coefficients in fresh and salt water regions are given by S_f and S_s respectively. δ is defined as $\delta = \rho_f/(\rho_s - \rho_f)$ and θ is the porosity of the aquifer media. $\alpha = 1$ for unconfined aquifer and $\alpha = 0$ for confined aquifer.

Except for very simple systems, analytical solutions of those two coupled non linear partial differential equations are rarely possible. Various numerical methods must be employed to obtain approximate solutions. The sharp interface models which solve the coupled freshwater and saltwater flow equations have been developed with different numerical techniques (Shamir & Dagan 1971, Vappicha & Nagaraja 1976, Wilson & Sa Da Costa 1982, Polo & Ramis 1983, Essaid 1986, 1990).

From equation (19.1) and (19.2), it is possible to derive a numerical model using implicit finite difference techniques. The continuous system described by above two equations are replaced by finite set of discrete points in space and time, and the partial derivatives are replaced by terms calculated from the differences in both freshwater and saltwater head values at these points. Spatial discretization is achieved using a block entered finite difference grid which allows for variable grid spacing. To solve the two simultaneous linear algebraic difference equations, the Strongly Implicit Procedure -SIP (Remson et al. 1971) was used as a suitable numerical technique. Empirical evidence suggests that for cases of flow in heterogeneous or anisotropic media, the strongly implicit procedure is much faster than the other methods. Also the strongly implicit method does not depend upon the complexity of the problem (Essaid 1986).

19.2.2 *Sensitivity analysis: factors affecting salinity intrusion*

Specific storage, porosity and hydraulic conductivity were considered to investigate the effect of hydro geologic factors on the dynamics of the freshwater-saltwater flow systems. A horizontal strip through an unconfined aquifer has been simulated by changing the hydro geological properties while observing the system's transient responses. The effect of the specific storage was evaluated by increasing the storage coefficient by orders of magnitude. The change in storage coefficient does not affect the location of the interface. The system responds in almost same manner for different specific storage values because most of the water to fulfill the changes in storage is coming from the drainage of water table rather than elastic storage. The other factor which illustrates the storage of the aquifer is the porosity. To investigate the effect of porosity on the behavior of the flow system, the porosity was changed from 0.1 to 0.4 in increments of 0.1. The change in the porosity does not lead to change in the position of the interface. It leads the change in the time period to achieve the steady state of the interface. Theoretically it can be explained as the freshwater heads fall to steady state more rapidly since less water must drain from the pores and the interface change more rapidly.

Hydraulic conductivity is another factor affecting the change of the position of freshwater-saltwater interface. The hydraulic conductivity was changed over the range of 10^{-3} m/s to 10^{-4} m/s. Those values are in the hydraulic conductivity range for clean sand or basalt aquifers (Freeze & Cherry 1979). Figure 19.1 explains that the changes in hydraulic conductivity have quite an impact on the steady position of the interface. The change in hydraulic conductivity makes the changes in the transmissivity and it affects the head gradients necessary to maintain the freshwater flux. This process showed that the model is more sensitive with respect to changes in hydraulic conductivity than other hydro-geological factors.

The effect of groundwater recharge on the dynamics of the freshwater-saltwater interface can be understood most readily by considering a simple, finite ground water flow system in which the only source of recharge is from precipitation and all discharge is to the ocean. To represent the effect of groundwater recharge, the simulation runs were conducted with realistic annual recharge values between 50 mm/year and 100 mm/year with constant hydro-geological properties. A set of interface profiles for various recharge rates were obtained. Location of freshwater-saltwater interface profiles for above recharge rates are shown in Figure 19.2. It shows that high recharge can reduce saltwater intrusion effectively. For the estimation of fresh groundwater loss in coastal

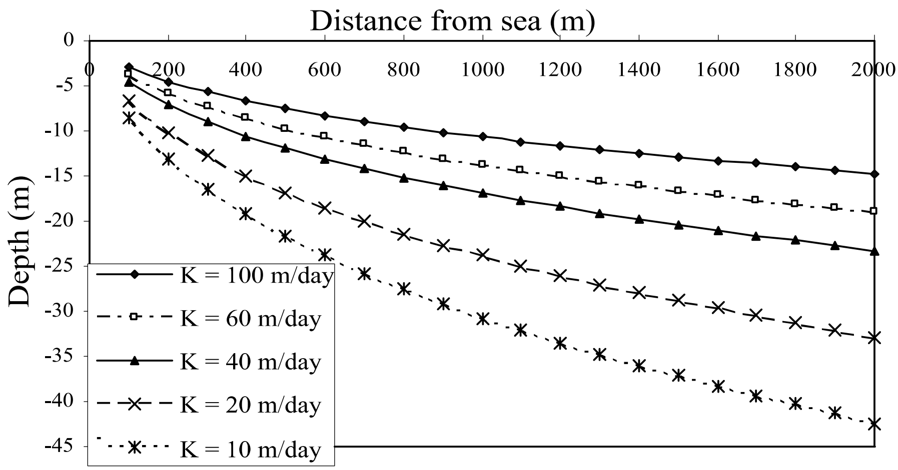

Figure 19.1. Change in the interface with hydraulic conductivity.

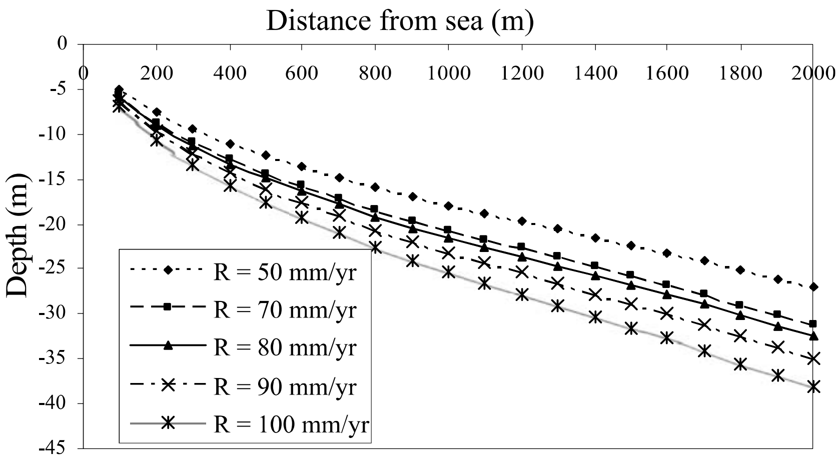

Figure 19.2. Change of interface with groundwater recharge.

aquifers, the effect of groundwater recharge in the watershed area which is illustrated by the climatic changes and land use change are important.

19.3 QUANTIFICATION OF GROUNDWATER RECHARGE USING LIMITED HYDROLOGICAL DATA

The possible changes in groundwater recharge due to changes in climatic conditions such as precipitation and temperature and the land use change were aimed to evaluate. Even though groundwater recharge is the major source of freshwater across much of the aquifers, particularly in arid and semi-arid regions, but there has been very little research on the potential effects of climate change on groundwater recharge. Changing land-use and land-management practices are also altering the hydrological system. The concept of water balance is useful for assessing how changes in catchment conditions can alter the partitioning of rainfall into different components. The simple water balance for any catchment can be written as:

$$\mathrm{R} = \mathrm{P} - \mathrm{ET} - \mathrm{RO} \tag{19.4}$$

where R is the groundwater recharge, P is the precipitation, RO is the surface runoff and ET is the evapotranspiration. All variables have dimensions [L/T].

In this equation, recharge is estimating as precipitation minus evapotranspiration minus runoff, which is very sensitive to measurement errors due to the involvement of large number of observation parameters. The data requirement to estimate the groundwater recharge using water balance technique is large. If the groundwater recharge can be represented as a function of limited available parameters such as annual precipitation, mean annual temperature and land use pattern, it is possible to reduce the errors in estimation. Hence, a methodology for estimation of the long-term average spatial patterns of actual evapotranspiration, surface runoff and groundwater recharge has to be developed. Such kind of methodologies can be used in areas where lack of meteorological data and also in ungauged basins.

19.3.1 *Evapotranspiration*

The effect of the land use on evapotranspiration has to be mainly evaluated to estimate the groundwater recharge with different land use patterns. The crop evapotranspiration is a simple representation of the physical and physiological factors governing the evapotranspiration process, taking vegetation parameters into account. The crop evapotranspiration can be estimated as a multiplication of reference crop evapotranspiration (ET_0) and crop coefficient (K_c).

$$ET_{crop} = ET_0 \times K_c \tag{19.5}$$

Several methods exist for the empirical estimation of reference evapotranspiration (ET_0). These include temperature based methods, pan evaporation, radiation and combination methods, referring to the data requirements of each method. In the most of those methods, the large amount of parameters to be considered in estimation of evapotranspiration accumulates measurement errors and the lack of data for the necessary parameters leads the final estimation of groundwater recharge erroneous. Most of the areas with limited meteorological data sources, the estimation of groundwater recharge became practically impossible. When considering data requirements of each estimation method, the only alternative available for the consumptive use operation is the temperature based method. A widely used temperature based theoretical method to calculate reference crop evapotranspiration is SCS Blaney Criddle method (Shuttleworth 1992). This method presents the temperature as the main physical factor governing the evapotranspiration process, together with annual percentage of monthly sunshine hours.

SCS Blaney Criddle method gives

$$ET_0 = K_t \times \left(T \times \frac{p}{100}\right) \tag{19.6}$$

$$K_t = 0.0173T - 0.314 \tag{19.7}$$

where T is the mean monthly temperature (°F) and p is percentage of daylight of the year occurring during a particular month.

19.3.2 *Surface runoff*

The surface runoff was estimated using the Soil Conservation Service Curve Number (SCS-CN) method developed by United States Department of Agriculture (USDA). It computes direct runoff through an empirical equation that requires the rainfall and a watershed coefficient as inputs. The watershed coefficient is called the curve number (CN), which represents the runoff potential of the land cover and soil complex. The standard SCS CN method is based on the following relationship between rainfall depth, P, and runoff depth RO;

$$RO = \frac{(P - 0.2\,S)^2}{(P + 0.8\,S)} \tag{19.8}$$

where P is the precipitation and the S represents the potential maximum retention after runoff begins. The retention factor is related to the soil and land use condition of watershed through the curve number and it is determined by;

$$S = \left(\frac{1000}{CN} - 10\right) \times 25.4 \tag{19.9}$$

where CN is the curve number and S is in millimeters.

S is expressed in terms of CN, which is a dimensionless watershed parameter ranging from 0 to 100. The SCS has developed tables of initial curve number (CN) values as a function of the watershed soil type, land use condition and antecedent moisture condition (AMC). The list of soils is prepared by the SCS and soils are classified in one of four different categories, ranked A to D on the basis of their runoff potential.

19.3.3 *Estimation of groundwater recharge*

There are several assumptions; the annual precipitation is uniformly distributed over the year considering equal monthly precipitation for each month and the mean temperature is uniform over the year. In the estimation of evapotranspiration using SCS Blaney-Criddle method, the reference evapotranspiration (ET_0) varies with the daily sun shine hours. Daily sunshine hours vary with the location of the interested area, mainly with the latitude of the area. Considering the pattern of the change in mean sunshine hours, the average pattern for the percentage of monthly sunshine hours was selected for the estimation of evapotranspiration. In the estimation of runoff, it is assumed that the monthly rainfall is a single storm event in the particular month.

19.4 RESULTS AND DISCUSSION

19.4.1 *Groundwater recharge and aridity index*

The estimated groundwater recharge for each land use pattern mainly depends on two climatic effects; precipitation and temperature. With the change of precipitation and temperature, the groundwater recharge can be graphically represented using the climatic indexes such as aridity index. Aridity indexes are quantitative indicators of the degree of water deficiency present at a given location. A variety of aridity indexes have been formulated. Aridity Index was a ratio between mean annual precipitation and mean annual temperature (Lang's index) and a modified version done by E. de Martonne in 1925 is widely used because their data requirements were minimal (Oliver & Fairbridge 1987).

$$AI = \frac{P}{T + 10} \tag{19.10}$$

where T is the mean yearly temperature (°C) and P is the mean yearly precipitation (mm).

The estimation of the groundwater recharge was carried out considering annual precipitation range from 500 mm to 3000 mm and mean annual temperature range from 5°C to 20°C.

The combinations of precipitation and temperature are used to estimate the aridity index and the estimated annual groundwater recharge is presented as a function of aridity index, land use (Fig. 19.3). Using these graphs, the annual groundwater recharge can be estimated as a function of annual precipitation, mean annual temperature, land use and hydrologic soil condition for any watershed.

Figure 19.3 shows that when the aridity index is less (less than 60), the contribution to groundwater recharge is higher from agricultural lands. When the aridity index is less (less precipitation and high temperature), the effect of surface runoff is less due to less precipitation and the evapotranspiration is the leading factor to decide the amount of groundwater recharge. The crop lands have less evapotranspiration and it gives larger recharge while the forest with higher evapotranspiration gives less recharge. For higher aridity indexes (over 60), the forest has more contribution to recharge than other crops. This is due to the high precipitation which leads to higher runoff in agricultural lands than in forests. In such climates, the evapotranspiration has less impact than surface runoff for groundwater recharge. Agricultural lands which have low evapotranspiration lead to a relatively higher recharge and the forest areas which have higher evapotranspiration gives low groundwater recharge. The introduction of forests in catchments increases the evapotranspiration, whereas agricultural lands lead to reduce evapotranspiration. Aridity index is higher in humid area whereas it is smaller in arid and semi arid areas. With respect to effective groundwater recharge, agricultural crops are the best vegetation cover in arid and semi arid areas.

19.4.2 *Simulation of the effects of land use and climate changes on salinity intrusion*

It is important to understand the effects of the combination of climatic changes and land use change on the movement of the saltwater wedge through the changes in groundwater recharge. Those effects can be understood most readily by considering a simple groundwater flow system and changing the groundwater recharge according to the different combinations of climatic factors and land use scenarios.

Simulation runs were conducted with selected annual groundwater recharge values, within the range of estimated recharge and hydraulic conductivity values in the range of 10^{-2} m/s and 10^{-4} m/s. Depth to the freshwater-saltwater interface at one kilometer distance from coastline, for different annual recharge rates and hydraulic conductivities are shown in Figure 19.4. It shows the relationship between groundwater recharge rate and saltwater intrusion in coastal aquifers with

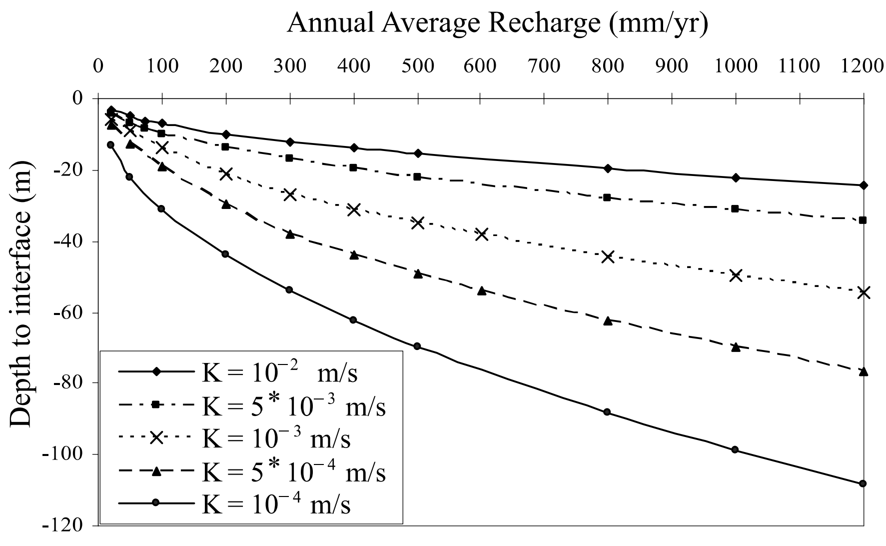

Figure 19.3. Relation between aridity index and groundwater recharge for land use patterns.

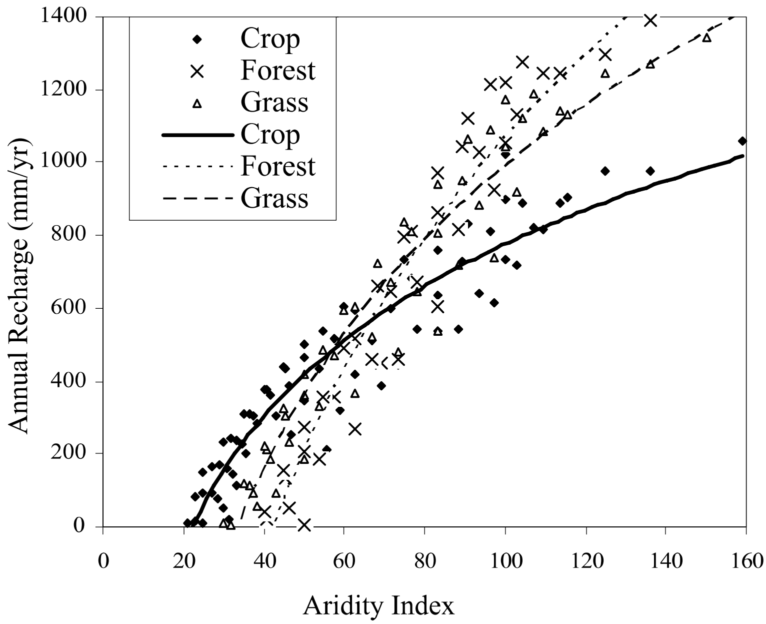

Figure 19.4. Variation of depth to salinity interface with groundwater recharge rate at 1 km away from the coastline.

typical range of hydraulic conductivities for sandy coastal aquifers. This result can be linked with the graphs shown in Figure 19.3, in such a away the annual groundwater recharge estimated reference to the aridity index and land use pattern can be used to find salinity interface in the coastal aquifer together with hydraulic conductivity.

19.4.3 *Effects of deforestation and climatic changes on fresh water loss in coastal aquifers*

19.4.3.1 *Fresh groundwater loss due to salinisation*

The concept of interface between freshwater and saltwater can be used to estimate the amount of fresh groundwater resources in coastal aquifers. The movement of salinity interface due to the changes in recharge/discharge leads to change in available fresh groundwater resources in the aquifer. As illustrated in Figure 19.5, when the aquifer is totally filled with freshwater (interface 1), the freshwater loss can be considered as zero and the movement of salinity interface landward, leads to reduce the freshwater amount in the aquifer. When the salinity interface coincides with piezometric head (whole aquifer fills with saltwater), the freshwater loss will be 100%. If the groundwater recharge is zero, then the whole catchment trends to fill with saltwater and freshwater loss will be 100%. The impact of the reduction in groundwater recharges due to change in climate and land use for the loss of freshwater resource was aimed to estimate. To evaluate the effects of land use change for the loss of fresh groundwater resource, the deforestation concept have been taken into account.

19.4.3.2 *Deforestation*

Here the deforestation is defined based on the availability of forests and agricultural lands in the catchment. If the whole catchment is covered with forests, the deforestation is defined as zero whereas the whole catchment is covered with agricultural crop land, the deforestation is defined as 100%. Intermediate values are defined based on the percentage of forest cover and the agricultural land cover in the catchment. For example, if the 75% of the catchment area is covered with forests

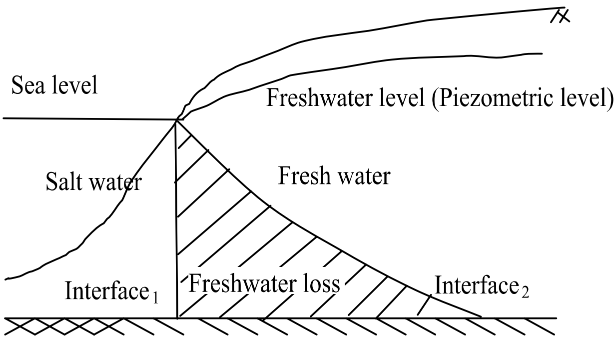

Figure 19.5. Loss of fresh groundwater resource due to salinisation.

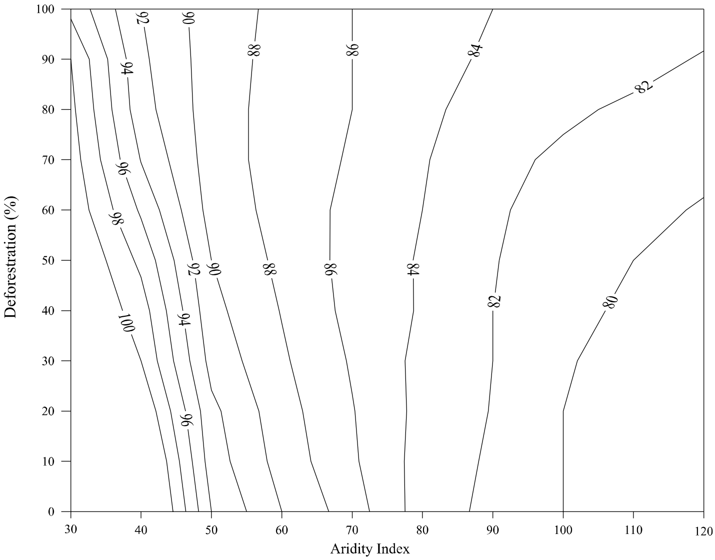

Figure 19.6. Change in fresh groundwater loss in coastal aquifers with deforestation and climatic change (aridity index) for hydraulic conductivity of 10^{-2} m/s.

and 25% of the area with agricultural crops, then the deforestation is defined as 25% for that catchment.

19.4.3.3 *Combination of deforestation and climatic changes*

Groundwater recharge was estimated for different deforestation ratios for the range of aridity index. The estimated recharge was applied to simulate the freshwater-saltwater interface with two hydraulic conductivity values; upper and lower boundary of the hydraulic conductivity for sandy/basalt coastal aquifers. The simulated salinity interface profiles were used to estimate the available freshwater resource of aquifers and the relative change of the salinity interface profiles were used to calculate the percentage loss of the fresh groundwater resources in the aquifer. Percentage loss of fresh groundwater resource due to the movement of salinity interface for the combinations of deforestation and climatic changes for hydraulic conductivity of 10^{-2} m/s is shown in Figure 19.6. From the above results it indicate that deforestation leads to a reduction in fresh groundwater loss in many areas which have low aridity index (arid and semi arid climates).

When hydraulic conductivity of the aquifer is higher, the freshwater loss is relatively higher, because the high hydraulic conductivity allows freshwater to flow to the sea quickly.

In many coastal aquifers in the world, sufficient amount of recharge is available during periods of high precipitation. Although some water is captured during these periods and stored in surface reservoirs, very little water is recharged groundwater reservoirs for use in the drought periods. This extra water which is wasted to the ocean could be used to replenish the aquifer, build up groundwater levels, and repel the saltwater intrusion. In many instances the recharge region of the principal water supply aquifer is far away from the coast. In these regions, it is possible to change the land use patterns in such a way to increase the recharge of the aquifer far from the shoreline and prevent saltwater intrusion.

19.5 CONCLUSIONS

Fresh groundwater supplies in many coastal aquifers throughout the world are threatened by saltwater intrusion. Saltwater intrusion can lead to a severe deterioration of the quality of existing fresh groundwater resource. Relative reduction of groundwater recharges due to climatic and land use changes will lead to accelerate the salt water intrusion. Numerical modeling can be considered to be a tool to enhance the knowledge of the saltwater intrusion process. In this study, saltwater intrusion was simulated through a numerical model based on sharp interface approach. The model further simulates the loss of fresh groundwater resource in coastal aquifers with respect to climate and land use changes.

The sensitivity analyses highlight that the model is very sensitive with respect to changes in hydraulic conductivity and groundwater recharge. To evaluate the factors affecting the groundwater recharge, the water balance technique has been employed in order to establish the groundwater recharge as a function of annual precipitation, mean annual temperature, land use pattern. The aridity index has been introduced to represent the variations in precipitation and temperature scenarios. Aridity index is higher in humid areas whereas it is lower in arid areas. Results show that, when aridity index is less (less than 60), the agricultural lands give high groundwater recharge whereas the forests give low groundwater recharge. With respect to groundwater recharge, agricultural lands are the best land use pattern in arid and semi arid areas. The combined effects of deforestation and aridity index on fresh groundwater loss conclude that, deforestation causes the increase of the recharge and existing fresh groundwater resource in areas having less precipitation and high temperature (arid climates).

The developed methodology will be useful in areas with limited hydrological data. The results from this study would assist the planners and decision-makers to come up with better land use and water resources management concepts ensuring its long term sustainability for natural and anthropogenic impacts.

ACKNOWLEDGEMENTS

The authors would like to acknowledge the Grant-in-Aid for Scientific Research by Prof. Nobuo Mimura, Ibaraki University, Japan. The authors would also like to acknowledge the efforts of editors and reviewers comments to improve the paper.

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CHAPTER 20

Tsunami impacts on shallow groundwater and associated water supplies on the east coast of Sri Lanka

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ABSTRACT: In order to assess the immediate and intermediate impacts of the December 26, 2004 tsunami on groundwater and associated water supply on the east coast of Sri Lanka, a monitoring program, focusing on domestic drinking water wells, was conducted in three study areas in the period March to July, 2005. The areas investigated were overlaying shallow, unconfined sandy aquifers, and the topography was flat. A total of 150 wells were monitored, covering both affected (inundated by tsunami water) and unaffected wells. Results indicate that wells were affected by salinity intrusion to various degree between the sites and within sites, up to 1.5 km inland. Thirty nine percent of all monitored wells had been flooded within a distance of 2 km from the coastline. Salinity levels after seven months after the tsunami were above a defined drinking water acceptability criterion (1000 μS/cm) in the majority of the affected, tsunami-flooded wells (91%). Excess salinity in wells is expected to persist for at least one more monsoon season.

20.1 INTRODUCTION

The December 26, 2004 tsunami that hit the Indian Ocean had devastating effects on water resources and water supply in affected coastal areas. In Sri Lanka, where about 75 percent of the coastline was affected by the tsunami, physical destruction and salinization of water supply schemes, either based on smaller decentralized, waterborne systems, or individual wells raised an immediate concern for short as well as long term rehabilitation of safe water access to the large population living in or displaced from affected areas. More than 50,000 water supply wells were estimated to have been affected in Sri Lanka alone (ADB, JBIC & WB 2005, UNEP 2005).

In this study, the focus was on the east coast of Sri Lanka because here the tsunami devastation was among the highest in the country (ADB, JBIC & WB 2005). Also, these areas are generally less developed, partly because of longstanding civil unrest. Finally, groundwater plays a critical role in supplying drinking water and irrigation water in the coastal areas of eastern Sri Lanka. This is because the aquifers generally provide a reliable and good quality water source, readily available on the spot and on demand from traditional shallow open dug wells. Furthermore, alternatives, in the form of surface water, are not locally available and have to be transferred from inland reservoirs. The tsunami accentuated this dependence and the implied need to protect these aquifers, not just after the tsunami but in a general sense.

The coastal sand aquifers, dominating on the east coast where they stretch up to 8 km inland, consist primarily of spits and bars, coastal dunes and raised beaches and palaeo-beach deposits (Panabokke & Perera 2005). The total area of these aquifers has been estimated to be 200,000 hectares, or ca. 3% of the area of Sri Lanka. Mostly, they consist of sand and constitute shallow, unconfined, limited, and vulnerable freshwater aquifers because of their inherent close proximity to the sea and due to their open exposure to contamination from superficial contamination sources.

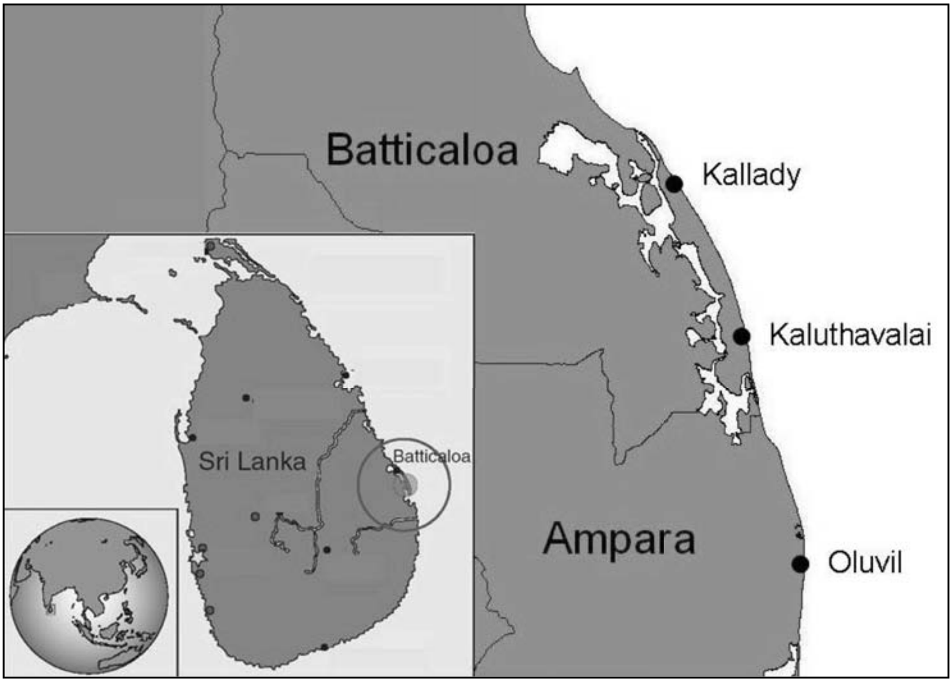

Figure 20.1. The three monitoring sites on the east coast of Sri Lanka.

In many places, the sandy aquifers are bordered to the inland side by coastal freshwater or brackish lagoons (Fig. 20.1). This configuration further limits the local fresh groundwater resources and increases the vulnerability which needs to be taken into account in water resources assessment and management.

After the tsunami, no data existed on the extent of the salinization problems and there was an urgent need to initiate systematic monitoring and assessment of the immediate as well as longer term impacts that could lead to appropriate rehabilitation methods and the protection of the groundwater resources for future water supply.

In order to support the rehabilitation of the water supply on the east coast following the tsunami and to understand the extent, magnitude and the duration of the problems of salinization due to the tsunami a project was initiated by IWMI (International Water Management Institute). The project focused on groundwater and consisted of two parts, one of setting up a monitoring program on salinity and one on guiding the local efforts on the cleaning and use of wells in the wake of the tsunami and general dissemination of information related to saltwater-freshwater relations in coastal groundwater systems. In this paper the first part of the project will be presented. For a more thorough and complete report of the project, reference is given to Villholth et al. (2005).

20.2 MATERIALS AND METHODS

20.2.1 *Field sites*

Three sites on the east coast were selected for the monitoring program after discussion with the stakeholders and authorities with knowledge of the local conditions: Kallady, Kaluthavalai and Oluvil, which belong to the districts of Batticaloa and Ampara (Fig. 20.1). The sites were chosen to be representative of some of the general characteristics on the east coast with respect to

Table 20.1. Dates (2005) for the monitoring trips.

Trip no.	Dates	No. of months after the tsunami
1	March 8–13	2.4
2	March 28–April 2	3.1
3	May 3–9	4.3
4	June 10–15	5.5
5	July 15–20	6.7

physiography, demography, land and water use. Also, areas that were devastated by the tsunami were chosen, as these areas were expected to suffer most from salinization.

Each study area stretched approximately 1 km along the coast and 2 km inland. The topography in the areas was relatively flat, with highest points of elevation of 15 m. The first two sites, Kallady and Kaluthavalai, were bordered to the west by lagoons, at approximately 2 and 3 km from the seashore. The land use varied from residential semi-urban areas with quite high population density (up to 3000 persons per km^2) to more rural areas with vegetable cultivation, paddy fields and open land. Assuming that each family of five had their own well, the maximum well density would be more than 600 wells per km^2.

20.2.2 *Monitoring program*

The field monitoring program was initiated in March 2005 (2.5 month after the tsunami). A total of approximately 150 existing wells were selected, covering both affected and non-affected wells, approximately 50 wells at each site. Because of the large density of existing wells not all wells were selected, and a pattern of four transects in each sites, with wells on lines perpendicular to the coast line were selected, with a distance between adjacent monitoring wells of 100–150 m. However, as the study progressed other wells were added to the monitoring program, to include the diversity of the types of wells that were encountered.

Salinity, groundwater level, turbidity, and temperature were monitored on a regular basis, at 20 to 40 day intervals. Five field trips each of 5 days duration were required for the collection of information, with the last one occurring in July 2005, 6.7 months after the tsunami (Table 20.1). The monitoring period commenced at the tail end of the rainy season, and followed through the dry season. No significant rain fell in the areas during the period February to August, 2005.

Groundwater levels in the wells were monitored by a measuring tape. Salinity, temperature, and turbidity were monitored by a 4.5 cm diameter stainless steel Troll 9000 monitoring probe from In-Situ Inc. (In-situ Inc., 2006). Individual sensors registered the salinity (here reported as EC, specific electrical conductivity), turbidity, temperature, and the hydraulic pressure at the water level of the sampling point. The probe automatically compensated the EC measurements for temperature.

Because the probe was measuring the parameters in-situ, i.e. directly inside the well or whatever other water body, and because it was connected directly to a data logger, there was no need to take samples and either measure them in the field with a portable monitoring probe or to bring them to a laboratory for analysis. Also, the monitoring of salinity at different depths of a well could easily be achieved by sinking in the probe to the various levels and logging the results directly.

In addition, the physical dimensions of the wells, the type of well (tube well or open dug well), the primary function of the wells (whether domestic or agricultural), the actual use of the wells (for drinking, bathing/washing, irrigation and whether in use at present), the water lifting system, the ownership of the wells (whether public or private), and whether the wells had been flooded by the tsunami was registered. At each monitoring visit, it was also recorded whether the wells were in actual use and whether any cleaning of the wells had occurred. The location of the wells and other measurement points were recorded with a GPS. Besides the wells, lagoon water, seawater, canal water and tank water were monitored in a few locations. Rainfall data from existing rainfall stations close to the sites were collected from the Sri Lankan Meteorological Department.

20.3 RESULTS AND DISCUSSION

20.3.1 *Wells sampled*

The characteristics of the wells monitored at the three sites are given in Table 20.2. The majority of wells (91%) were open, dug, shallow, private domestic wells, with an average depth of 3.4 m and an average diameter of 1.4 m. The other categories of wells consisted of deeper, smaller diameter tube wells, with average depth of 5.7 m. They were mainly used for irrigation and public water supply. About one third of the wells had a mechanized pumping system whereas the rest relied on some simple, manual lifting techniques, like a bucket or a pulley.

20.3.2 *Spatial pattern of salinity impacts*

Overall, 39 percent of the monitored wells were reported to have been flooded by the tsunami. In general, and obviously, the flooded wells were situated closer to the sea and exhibited the highest

Table 20.2. Characteristics of wells in the monitoring program.

Parameters/Locations	Kallady	Kaluthavalai	Oluvil	Total
No. of domestic wells	40	33	53	126
No. of agro-wells	0	13	3	16
No. of public wells	6	2	7	15
No. of wells with mechanized pumps	16	19	13	48
No. of tube wells	0	11	3	14
No. of open dug wells	43	38	53	134
Average depth of tube wells	–	5.9 m	4.7 m	5.7 m
Average depth of open dug wells	3.3 m	3.9 m	3.2 m	3.4 m
Average diameter of tube wells	–	0.2 m	0.2 m	0.2 m
Average diameter of open dug wells	1.7 m	1.3 m	1.1 m	1.4 m

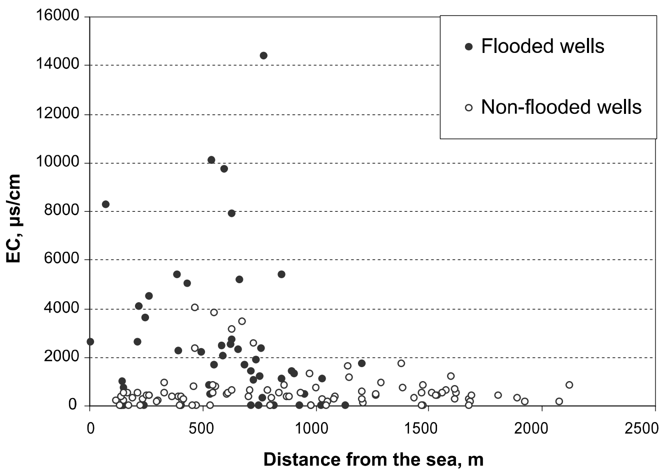

Figure 20.2. Salinity levels of flooded and non-flooded wells as a function of distance from the coast during trip 1 (2.4 months after the tsunami).

post-tsunami salinity (Fig. 20.2). Based on the identification of flooded vs. non-flooded wells, a flood line, demarcating the distance to where the tsunami waves reached and flooded the wells within the areas, was derived. The maximum observed inundation distance of flooded wells was 1.5 km. It is seen from Figure 20.2 that there is no clear correlation between the distance from the sea and the salinity levels, neither for the flooded, nor the un-flooded wells.

The sites were impacted to various degree. This is seen from the average salinity of all wells (affected and unaffected) at the three sites during the first field trip as well as from the percentage of flooded wells, the maximum inundation distance, and the flood line location. Kaluthavalai was hit the hardest and Oluvil was the site least impacted (Table 20.3).

The variability between sites is mostly related to the various factors that influenced the tsunami flooding event, like the number of waves, the wave height and angle, the nature of the seashore, the topography and any protective and wave-breaking features at the sea bottom as well as the coast itself. In Kallady and Kaluthavalai, the flood line was related to the topography and this could explain some of the individual patterns observed (Villholth et al. 2005). However, for Oluvil at least, other geomorphologic factors governing the entry of the waves must have been responsible for the marked differences between this and the other sites.

20.3.3 *Recovery of wells after the tsunami*

The salinity of the flooded wells decreased after the tsunami due to mixing with resident water and incoming rainfall just after the tsunami and due to diffusion and dissipation into surrounding soil and groundwater. Saltwater entering wells and aquifers would initially be primarily present as a layer of denser fluid on top of a less dense fluid (pre-tsunami freshwater). This unstable configuration would tend towards a more stable situation in which the denser, salty water would be below the freshwater by a relatively fast overturning of the waters or the fingering of the salty water through the freshwater (Wooding et al. 1997a, b). These mechanisms were in effect or most efficient just after the tsunami (say first couple of weeks), when the concentration gradients were greater and a lot of rainfall fell in the eastern coastal areas. Later, the restoring of freshwater conditions in the wells was much slower, because of the slower mixing processes and the much less rainfall that occurred in the areas.

At the time of the start of the regular monitoring program almost two and a half months after the tsunami, the salinity levels in affected wells had decreased significantly, to average levels of 3200 μS/cm (Fig. 20.3).

The flooded wells remained more saline than the background, non-flooded wells throughout the monitoring period. This means that after seven months of the tsunami, towards the end of the dry season, the wells still had not totally recovered to pre-tsunami conditions and that at least one more rainy reason would be required to leach out the residual excess salinity, if possible. At this point in time, the flooded wells had an average salinity of 2600 μS/cm, compared to the non-flooded wells of 1084 μS/cm. Since wells close to the coast prior to the tsunami, represented by the un-flooded, did not appear more salty than more inland wells (Fig. 20.2) it can be stated that the flooded wells in general had not recovered to pre-tsunami-levels. The variability of the salinity of the flooded wells decreased significantly during the monitoring period, indicating a leveling out

Table 20.3. Salinity impacts in the three study sites 2.4 months after the tsunami.

	Kallady	Kaluthavalai	Oluvil	Total
No. of wells monitored	43	49	56	148
No. of wells flooded (%)	21 (49)	24 (49)	12 (21)	57 (39)
Max. distance of flooded wells (km)	1.4	1.5	0.8	1.5
Average salinity, ± std.dev. (μS/cm) Trip 1	2033 ± 2847	2275 ± 2366	560 ± 636	1568 ± 2237

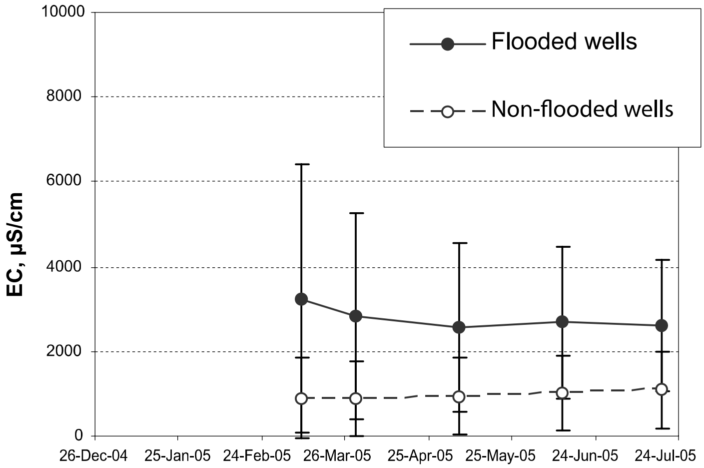

Figure 20.3. Average well salinity with time after the tsunami for flooded and non-flooded wells. The error bars indicate ± one standard deviation.

of the salinity differences within the aquifers and the wells, from natural dissipation mechanisms and from pumping/cleaning.

Of the flooded wells, 91 percent were still unfit for drinking at this point in time, based on a drinking water acceptability criterion developed as part of the study (max. EC of 1000 µS/cm). An analysis of the salinity of wells being actually used for drinking at the time of monitoring showed that the average salinity level of water used for drinking water was approximately 1000 µS/cm (Villholth et al. 2005). This value, however, may not be universal and unrelated to people's perceptions, based on their immediate access to water sources. It was hypothesized, but not verified in this study, that people living in the affected areas could have been used to better quality drinking water (from a salinity perspective), relative to pre-tsunami conditions, because of the alternative water sources brought in as part of the relief efforts. This is supported by data on pre-tsunami salinity in the studied areas indicating that even before the tsunami the salinity of the wells could exceed this threshold level (Jeyakumar et al. 2002, Vaheesar et al. 2000), and with less access to alternative sources, except bottled water, the population most likely were accustomed to drinking water with higher salinity than the criterion put forward here. Nevertheless, the results indicate that the salinity remained higher than background levels and people did not like to drink it.

For the non-flooded wells, there was a slight increase in salinity over the dry season, from an average of 890 to 1084 µS/cm. This increase is comparable to pre-tsunami observations mentioned above, and probably is a result of the high evapotranspiration rates and the general drying out of the areas during the dry season. Hence, there was no indication that the unaffected areas were being encroached by salinity from the affected areas.

20.3.4 *Variability within sites*

Well water salinity varied significantly within the flooded areas at the start of the monitoring period (Figs. 20.2 & 20.4). This can be explained by various factors:

- Local differences in the flooding pattern. This is related to how much water was standing on the surface during the inundation and the duration of the flooding over the sites.
- Soil and aquifer characteristics and conditions (like permeability, and initial groundwater level) and micro-topography. This influences the amount of tsunami water that infiltrated during and after the tsunami.

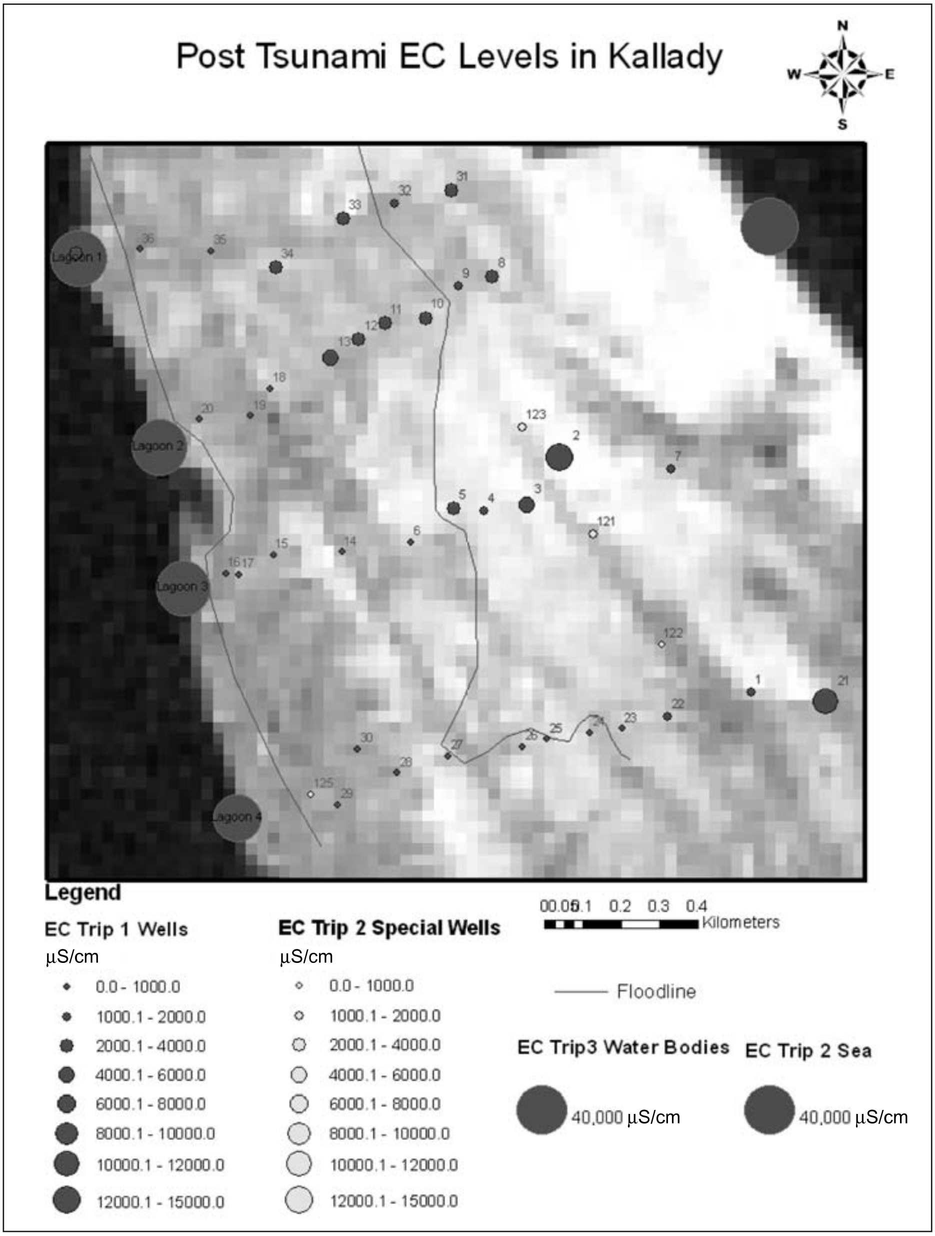

Figure 20.4. Salinity levels at the Kallady site, during the beginning of the monitoring period. The flood line is indicated. Note the two lines because water also inundated land from the lagoon side.

- Well characteristics. Some wells were closed, like the tube wells, impeding a direct influx of the tsunami wave into the wells.
- Post-tsunami pumping and cleaning impacts. Individual wells were operated and treated differently after the tsunami, which could influence the local salinity levels.

After the tsunami, widespread efforts of cleaning the wells were initiated. Wells were pumped out to remove salinity and debris and chlorination took place, often repeatedly in the same wells.

The general pattern of improving groundwater quality, in terms of salinity, in flooded wells as seen in Figure 20.3, also masks a lot of spatial variability. In Figure 20.5, the changes in salinity over the monitoring period in individual wells in the Kallady site are shown. It is seen that flooded wells with highest salinity decreased more rapidly than wells with less initial salinity, and that in some cases two adjacent flooded wells showed opposite trends, i.e. one well was improving and the other was deteriorating in terms of salinity (e.g. wells marked with a circle in Fig. 20.5).

These observations are consistent with a picture of the well water salinity differences leveling out with time, with the initially highly saline wells improving while the initially less saline wells decreasing less rapidly or in fact increasing temporarily in salinity. Hence, it appears that the cleaning and pumping of the wells which occurred in all wells did not unitarily improve the salinity levels, though the data set collected in the study is not sufficiently detailed to infer very local impacts.

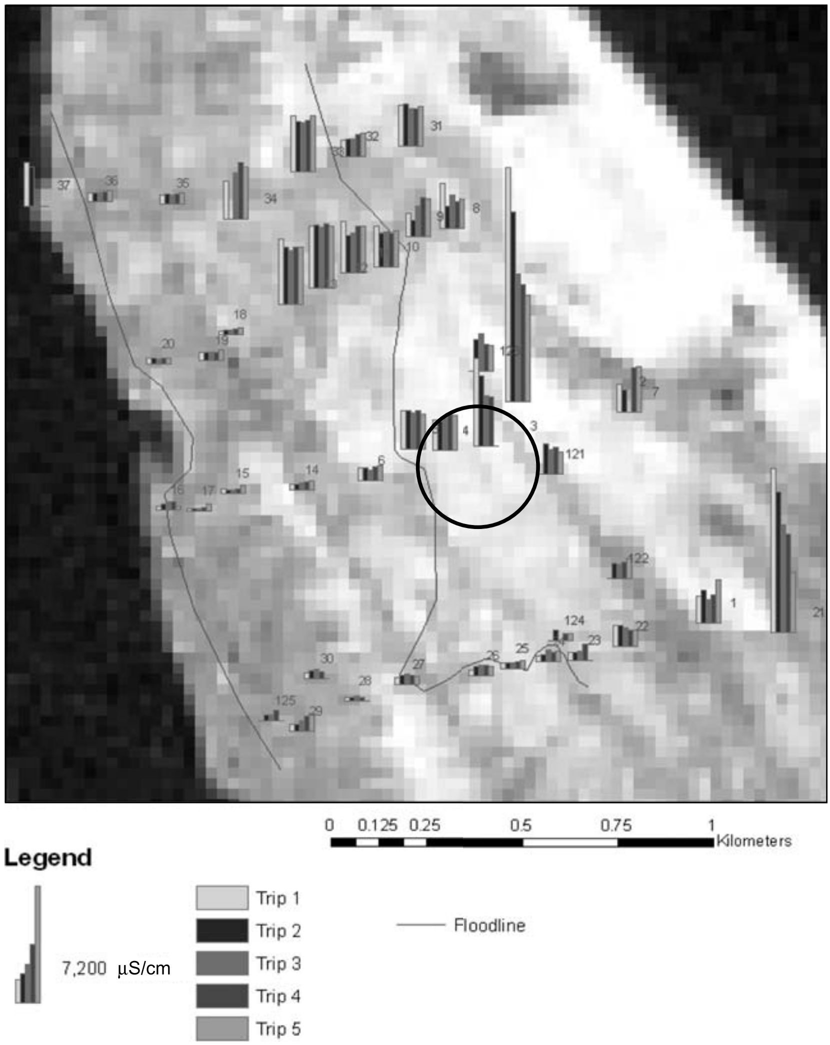

Figure 20.5. Salinity changes in individual wells in the Kallady site over the study period.

20.3.5 *Salinity with depth*

The salinity was practically uniform with depth in all the wells (except one, see below) during the monitoring period (data not shown). This is in contrast to observations done during a reconnaissance trip shortly after the tsunami, in end of January, when there was a significant stratification in salinity, with water significantly more saline (some >20,000 μS/cm) at the bottom of the wells (Villholth et al., 2005). There could be three major reasons for this. Firstly, between the initial sampling and the start of the monitoring period (March, 2005), the wells could have had time to equilibrate and even out the vertical concentration gradients, also by vertical overturning due to the density instability phenomenon. Secondly, heavy rainfall occurred just after the tsunami (see Section 20.3.6), partly explaining that freshwater was at the top of the wells. Thirdly, most likely all the wells at the time of the first field trip would have been pumped, either due to use or due to cleaning, or both. The pumping of the wells would tend to smooth out the differences in salinity with depth due to mechanical disturbance and mixing of the water column. It appears that the continuous usage of the wells was not required to maintain a uniform salinity profiles in the wells because all the wells showed the same smooth picture throughout the monitoring period irrespective of whether they were used or not. This supports the explanation of the rapid overturning and sinking of the denser, highly saline water overlying freshwater.

As the only exception, the deepest of the monitored wells, well 51 in Kaluthavalai, which was located 0.8 km from the coast, showed a distinct increase in salinity with depth (Fig. 20.6). At approximately 8.5 m depth, the salinity increased abruptly from a steady background level of 800 μS/cm to max. 5000 μS/cm at the bottom at 10.1 m. This is interesting because it could be due to seawater intrusion.

In Fig. 20.6, well 52, which is only 50 m away from well 51, is shown as well. It is seen that this well is distinctively and consistently more saline than well 51. However, there is no significant salinity increase at the bottom of this well, which could be explained by the fact that this well was not as deep as well 51 and hence the bottom of well 52 did not reach into the highly saline zone

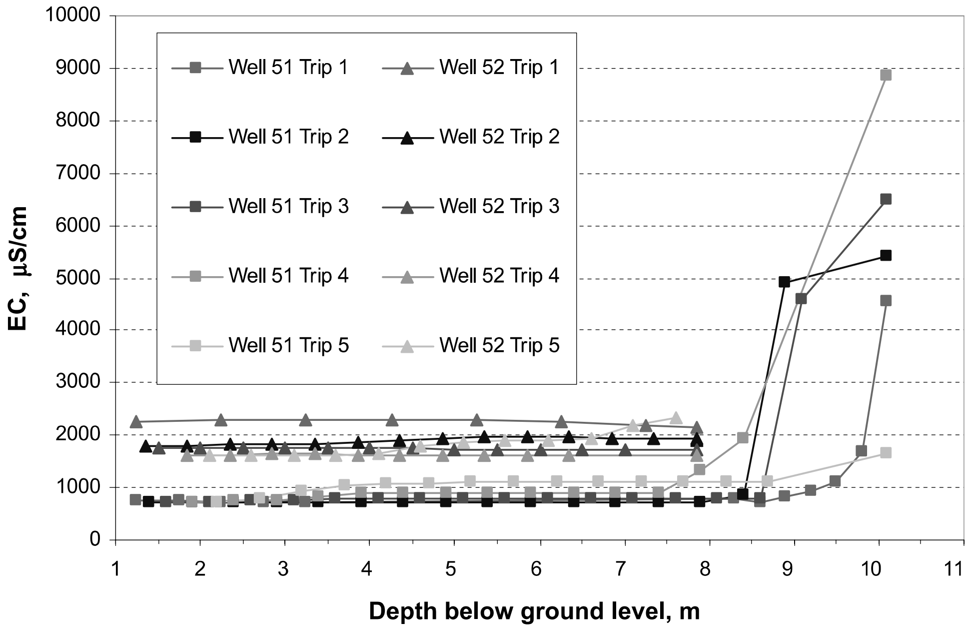

Figure 20.6. Salinity profiles of two wells (wells 51 and 52) located 50 m apart, in Kaluthavalai. Values are given for the five field trips.

deeper down. There were no indications that well 51 was pumped more heavily than well 52 (data not shown), which could have explained a higher salinity due to salinity intrusion from below. The interpretation of the results is that well 51 reaches the underlying interface between fresh and saltwater. Whether this is an intermittent layer of saltwater that is due to the infiltration of saltwater from the tsunami or whether this is a more permanent transition to the saline water below cannot be inferred from the present results.

20.3.6 *Rainfall*

The rainfall pre- and post-tsunami was analyzed in order to understand the degree to which the pre- and post-tsunami rainfall conditions were particularly dry or wet compared to a 'normal' year. As rainfall is the primary agent for restoring the freshwater conditions in the aquifers the amount and timing of rainfall would give an indication of whether the salinity problems were representing a relatively bad or good situation.

From the rainfall data it was found that the areas were very wet before and just after the tsunami. Later on the situation was relatively dry. On average, approximately 75 percent of the 1000–1700 mm annual rainfall falls in the months from October to February. This is the rainy season. Hence, the tsunami hit towards the end of the rainy season.

The rainfall prior to the tsunami was significantly higher than 'normal' and the rainfall after the tsunami was significantly less than 'normal'. This is seen from Fig. 20.7 giving the monthly rainfall for Oluvil as an example. The other areas had similar patterns. The 'normal' was estimated from the monthly averages of rainfall for the total length of the data series available for the rainfall stations. The rainfall after the tsunami (in the months of January to and including September 2005 (or August in the case of Kallady)) was only 83, 65 and 40 percent of the normal rainfall in Kallady, Kaluthavalai and Oluvil, respectively. The rainfall before the tsunami (in the months of August (or September in the case of Kallady) to and including November 2004) was 201, 161 and 170 percent of the normal rainfall in Kallady, Kaluthavalai and Oluvil, respectively. In the month of December

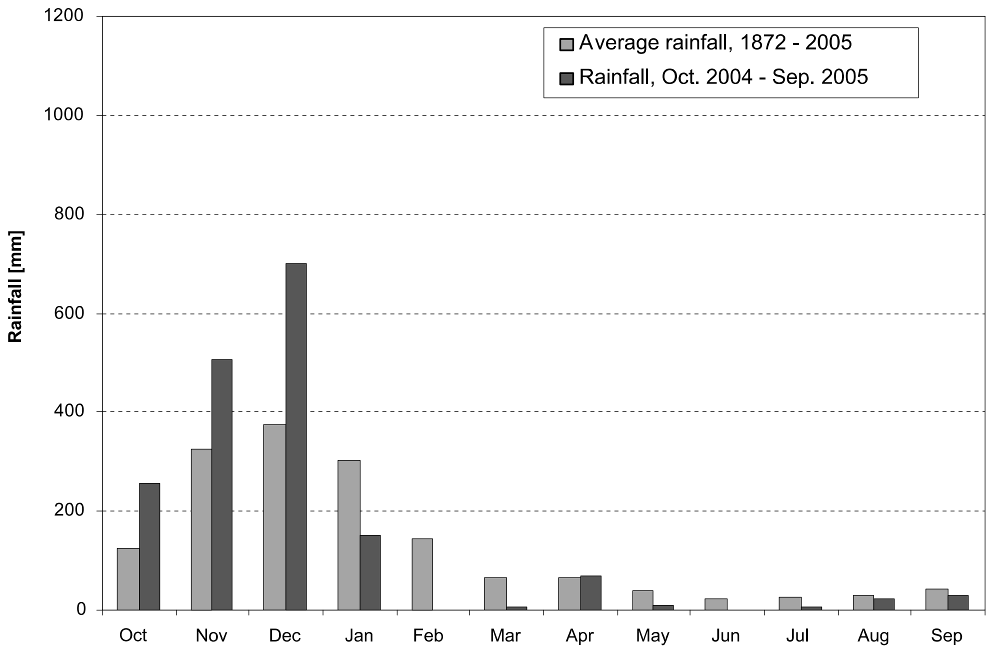

Figure 20.7. Rainfall in Oluvil before and after the tsunami, compared to a 'normal' year.

when the tsunami struck, a disproportionately higher amount of rainfall occurred after the tsunami: 37, 34, and 52 percent in the three cases during the last 6 days, representing only 20 % of the time of that month.

Firstly, at the time of the tsunami, the soil was wet and the groundwater levels must have been high within the soil profile. This is considered a favorable condition as the unsaturated zone was small leaving little room for entry of saline water from the entering seawater. Also ponds and depressions would already be filled with freshwater, and incoming seawater would not fill but rather be mixed with this resident water.

Secondly, the heavy rains that followed just after the tsunami caused an increased influx of water aggravating the flooding situation and maybe spreading the saltwater more, but it also helped to flush out and dilute the saltwater accumulated in ponds, wells, soils and groundwater.

Thirdly, the ensuing low rainfall after the tsunami meant that the aquifer systems were not replenished as much and the aquifer restoration was less than would be expected in a normal year. This is to be considered an unfortunate condition.

In conclusion, the rainfall pattern and amounts observed in the areas indicate that the impact of the tsunami in terms of salinity was relatively benign, representing rather a best case scenario.

20.4 CONCLUSIONS

The monitoring program carried out as part of this study showed that wells were affected by salinization due to the tsunami up to 1.5 km inland, and that 39 percent of the monitored wells within 2 km from the coast were flooded by the tsunami. The three study sites were impacted to various extent, in terms of number of flooded wells and the distance to which the waves reached inland (Kaluthavalai $=$ Kallady $\geq$ Oluvil). The topography could explain some of the variability of impact between sites. However, other factors such as bathymetry, number of waves, wave height and angle and wave braking features on the coast were probably equally important. Well water salinity also varied significantly within areas flooded by the tsunami, probably due to different flooding patterns, soil and well characteristics and possibly post-tsunami pumping and cleaning impacts.

The average well water salinity of flooded wells decreased rapidly within the first few months after the tsunami, but excess residual salinity persisted throughout the dry season following the tsunami. Hence, the average well water salinity in flooded wells remained higher than the salinity of non-flooded wells throughout the monitoring period, indicating that the affected wells had not recovered and that the salinity impacts persisted after seven months after the tsunami. Recovery of the flooded wells and restoring freshwater conditions in the affected shallow aquifer required at least one more monsoon season. Recovery from rainfall recharge and potentially additional recharge from natural or constructed ponds are the primary means of flushing and restoring the aquifers. The very wet conditions before the tsunami and just after probably meant that less saltwater entered the aquifers than what would occur during a normal year. Similarly, rapid dilution of salinity took place. On the other hand, the relatively dry conditions after the tsunami resulted in prolonged salinization of the systems.

The average salinity of non-flooded wells increased slightly throughout the monitoring period. The rate of increase and the levels observed were comparable to pre-tsunami observations, indicating that the shallow aquifers in the non-flooded areas were not affected by the tsunami and that the increase observed was a normal process due to the drying out of the areas. This implies that wells in non-flooded areas could be used, with caution, to augment or substitute local water supply in flooded areas.

The majority of wells in the flooded areas were unfit for drinking seven months after the tsunami. The estimation was based on a drinking water acceptability criterion based on the actual use of the well water after the tsunami. This criterion may however be stricter than under normal, pre-tsunami times, because people were getting accustomed to better drinking water from the relief supply.

Highly saline water was consistently encountered at approximately 10 m depth below the ground, at a distance of 0.8 km from the coast. Whether this was tsunami water still sinking into the aquifer or pre-tsunami saltwater at the bottom of the freshwater lens could not be inferred in the study.

From the above conclusions it can be understood that the well cleaning efforts alone, taking place as part of the relief work, did not recover the flooded wells. It was not possible to detect whether the cleaning initiatives were in fact predominantly ameliorating or aggravating the salinity in the wells. Also, it is not clear whether the present pumping patterns are threatening the groundwater salinity.

In general, the tsunami highlighted the need for increased focus and awareness on the protection of the local groundwater resources for the establishment of long-term solutions for water supply in these regions.

ACKNOWLEDGEMENTS

The project was funded by CARE. We gratefully acknowledge the constructive comments from the two anonymous reviewers that helped to improve the manuscript. We are also thankful to Dr. AL. Ramanathan and Dr. Prosun Bhattacharya for the editorial comments and updates prior to the publication of this manuscript.

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CHAPTER 21

Status of a tsunami affected coastal aquifer along the east coast of Sri Lanka

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ABSTRACT: Highly productive unconfined aquifers with transmissibility of around 2500 m^2/day, and electrical conductivities around 200 μS/cm where static water level is of about 2.5 m in the east coast of Sri Lanka are affected by the tsunami on December, 2004. A hydro geological and geophysical survey carried out 10 months later, in the Batticaloa District, shows a gradual lateral and downward movement of saline water plume towards the sea and the lagoon. In addition, the slow vertical sinking of the highest saline zone of the plume has reached a depth of about 20 m at present. The water quality of the uppermost horizons of the aquifer is recovering slowly. Spatial distribution of Electrical conductivity of the dug well water too, indicates the lateral retrieval of saline water front towards the sea. It is expected that infiltration of precipitation of the oncoming monsoon rains will further improve the quality of water.

21.1 INTRODUCTION

The December 26th, 2004 tsunami waves hit the east cost of Sri Lanka very badly even flowing several kilometers inland at some places. Shallow sea and only slightly raised beaches without elevated headlands and low elevated flat topography at most places facilitated the tsunami water to flow inland easily. Although the bulk of seawater retrieved within few hours some water stagnated in the surface depressions and in open dug wells while a considerable amount has absorbed and infiltrated into the sandy soils of the area. As a result, and due to mixing of thus infiltrated and directly hit saline water, the groundwater in the vast majority of the wells and groundwater in most parts of the shallow aquifers along the coastline became unusable for domestic and irrigation purposes. The purpose of the present study was to investigate and understand the status of the salinity problem caused by the tsunami in the coastal aquifer in the east coast of Sri Lanka.

21.2 LOCATION AND HYDROGEOLOGICAL SETTING OF THE STUDY AREA

21.2.1 *Location of the study area*

The present study was carried out in a tsunami affected coastal village in east cost of Batticaloa District (Fig. 21.1). This area extends as a narrow strip of land about two kilometers in width and running approximately in NS direction between the sea in the East and a lagoon in the West. The ground surface is almost flat and the elevation in the middle part between the sea and the lagoon is about 3 to 4 meters above mean sea level. The geological formation from ground level down to about 9 m depth is medium to coarse sand with a small fraction of clay and shell fragments. A clay and silty clay layer of about five to 10 m thick is found below the sand layer, which is then followed

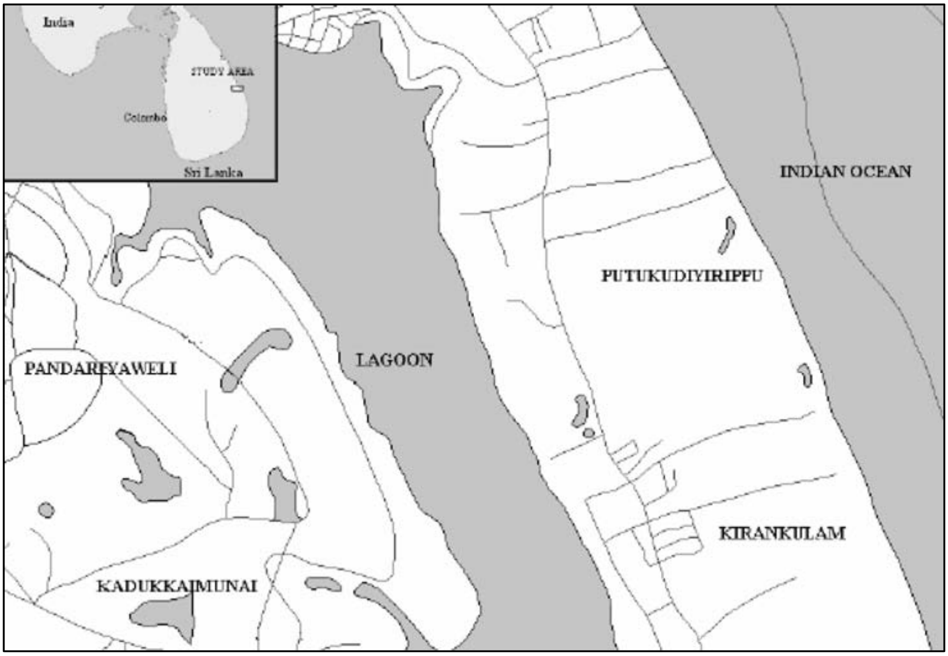

Figure 21.1. Index map showing the location of the study area.

by a thin coarse sand layer of about 2 m thick. Thickness of the sandy formations increases towards sea whereas thickness of clayey layers increases towards the lagoon.

Groundwater occurs in the unconsolidated sandy aquifer with water level at a depth of between 2.0 to 3.0 m from surface. Part of groundwater which flow towards the sea appears to seep out of the surface along the beach forming shallow (0.25 to 0.30 m deep) fresh/brackish water pools parallel to the coastline. The distance to sea from these pools is about 10 m and the water level stands at about 1.0 to 2.0 m above the sea level. A generalized cross section of the area is shown in Figure 21.2.

21.2.2 *Aquifer conditions in the eastern coastal areas*

Groundwater resources across most of the Sri Lankan coast are dominated by the coastal sand aquifers, which consist primarily of spits and bars, coastal dunes, raised beaches and paleobeach deposits (Panabokke 2001). Windblown accumulations of recent sands forming dunes occur along the Eastern coast line of Sri Lanka (Cooray 1965). These together with Pleistocene and Holocene deposits of sand have created sufficiently thick (up to 25 m) local and discontinuous highly productive aquifers (Survey Department, Sri Lanka 1988). However, the studies by Panabokke & Perera (2004) has revealed that the east coast of Sri Lanka consists of regosols overlying directly the unconfined sandy aquifers ranging from fine to moderately coarse structure-less sands. The total area of these aquifers has been estimated to be 1,800 km^2 (Somasiri 2001) and are tapped by shallow open dug wells and occasionally by deeper wells termed as tube wells. These unconfined, and occasionally semi confined aquifers are found along most parts of the eastern coast from south to north. Landward extent of these formations is about 3 km at maximum. The transmissivites of these aquifers are high and, values up to 2500 m^2/day (Wickremaratne 2004) have been observed. Good quality groundwater occurs in the aquifers and the groundwater table is present at a depth between 2 to 6 m from the surface. Data on groundwater quality from coastal sand aquifers have shown that salinity (as measured by specific conductance) in the coastal sand aquifers can range

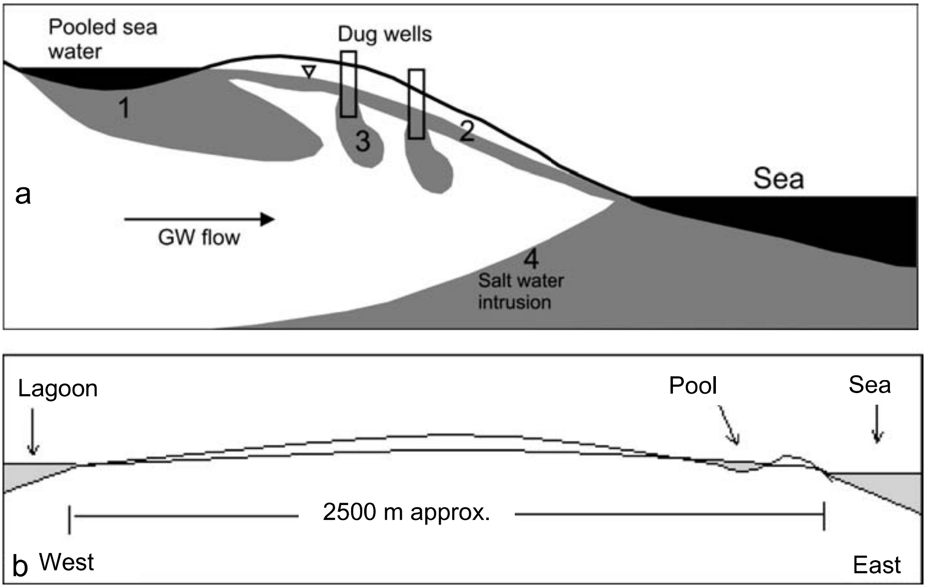

Figure 21.2. a) The preliminary conceptual model of the contamination source of the tsunami (modified after Villholth et al. 2005); b) a generalized morphological cross section of the study area.

Table 21.1. Chemical quality of groundwater in the coastal aquifers along the east coast at Kaththankudi.

Parameter	KKT W01	KKT W02	KKT W03	KKT W04	KKT W05	KKT W06
pH	7.2	7.2	7.3	7.3	7.3	7.4
EC (μS/cm)	619	662	354	346	442	505
TDS (mg/L)	412	441	236	230	294	336
Total. Hardness (mg/L)	220	216	136	136	152	160
Chloride (mg/L)	50	54	24	24	24	16
Total Iron (mg/L)	1.0	0.6	0.5	0.7	1.2	0.9

(Source: Wickremaratne 2004).

from a minimum of 400 μS/cm to as high as that of seawater (Panabokke 2001). However, most wells used for drinking water do not exceed 2000 μS/cm. While there are no strict health based drinking water quality standards for salinity, levels below 1000 μS/cm are considered to be palatable, that is, not giving objectionable taste (Illangasekare et al. 2006). According to Panabokke et al. (2002) only two agricultural wells out of 25 from the Trincomalee District (located on the north east coast of the country) had EC values less than 500 μS/cm; most were above 1000 μS/cm while and some were as high as 2500 μS/cm. The general chemical quality of groundwater from few shallow tube wells on a sandy aquifer in the east coast is given in Table 21.1.

Generally in these aquifers, fresh water floats on the saline water at fresh water—saline water interface on the seaward side of the aquifers, describe as barrier islands. Beneath this island fresh groundwater may occur as a thin lenticular shaped body referred to as fresh water lens that is separated from the underlying seawater by a zone of mixed water commonly called transition zone (Underwood et al. 1992). Excessive pumping often causes saline water intrusion. Recharge to the aquifers occurs as direct infiltration from surface. These aquifers are highly vulnerable to contamination where direct infiltration of contaminants is common from agricultural activities and onsite waste disposal in these areas. At some areas, high nitrate has been a common water quality problem.

21.2.3 *Inundation by the tsunami*

More than 1 km inland was inundated (Villholth et al. 2005) and the height of the tsunami waves in Sri Lanka ranged from 3–8 m, with run up heights as much as ≥15 m (Liu et al. 2005) during the December 26th, 2004 tsunami (Fig. 21.2). The height of the tsunami wave was several meters at the beach and the maximum run-up height within the village, situated about 500 m inland from the beach was about 2 m. The run-up height gradually decreased towards inland and disappeared at a distance of about 1.5 km from the sea. The sources of contamination (Fig. 21.2 above) by the tsunami have been discussed by Illangasekare et al. (2006). About two third of the wells in the village submerged in the tsunami water whereas another about one third were not submerged or filled with saline water because of the parapet walls around those wells. Another few wells that were situated away from the reach of tsunami remained unaffected. At the time of the present investigation, all affected wells had been cleaned by removing the debris and by pumping out water for several times. Except the few wells that were not affected, all other wells are not in use for drinking purposes at present due to the (slightly) salty taste of water which people believed far inferior quality compared to what they were used to obtain from the wells prior to tsunami.

21.3 FIELD INVESTIGATIONS

Depth to water table and the electrical conductivity (EC) of water were measured in all wells in the village. A 400 m long continuous vertical Electrical sounding (CVES) profile was done approximately in West to East direction, starting in the unaffected area, crossing the tsunami flow boundary

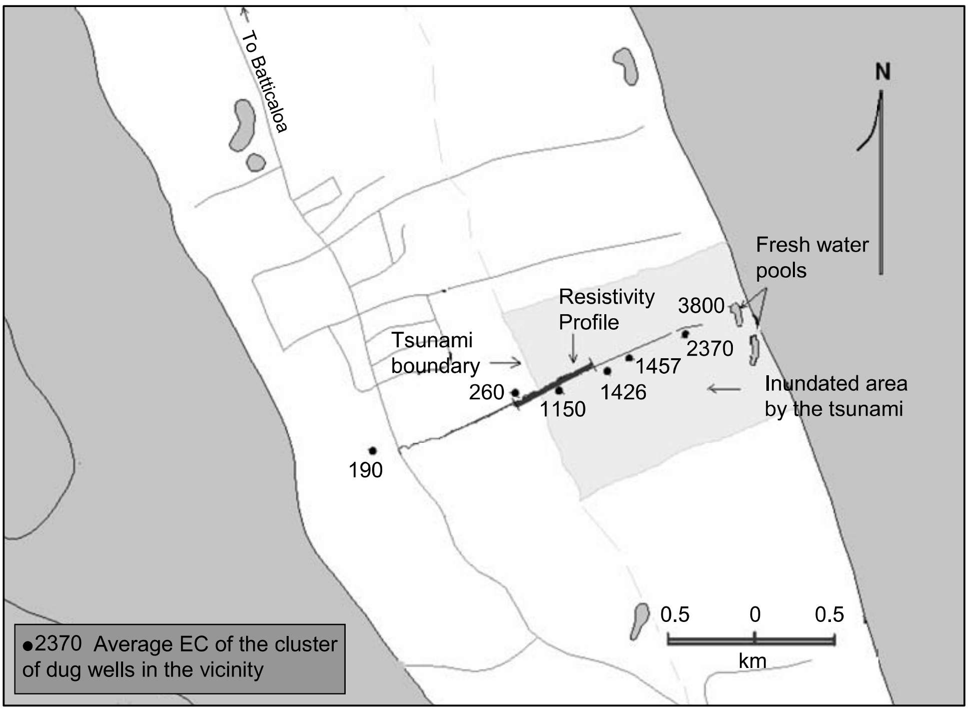

Figure 21.3. Map showing the tsunami boundary, location of the representative dug wells and the layout of the resistivity profile in the study area.

and then into the affected area. Wenner electrode configuration was used with a 10 m electrode spacing. For the purpose of obtaining a better resolution and verification of results, the measurements in the second half of the profile was repeated with 5.0 m electrode spacing. Resistivity data were interpreted to produce subsurface resistivity image using Geo electrical imaging software (Geotomo Software USA). Location of some of the wells and the layout of the resistivity profile are shown in Figure 21.3.

21.4 RESULTS AND DISCUSSION

21.4.1 *Results from the dug wells*

Under the pre-tsunami conditions, the salinity of the shallow groundwater has been very low and the representative electrical conductivity values of the water have been in the range of 200 μS/cm. The electrical conductivities of the dug well water are shown in Table 21.2. The EC of the dug wells varies between 170 and 3200 μS/cm. The remarkable feature observed was that the groundwater immediately beyond tsunami inflow boundary has extremely low electrical conductivity values compared to that of the affected wells. It is also noted that there is a general increasing trend of EC (therefore the salinity of the groundwater) towards the sea.

As indicated earlier, a part of groundwater which flows towards sea appears to seep out of the surface along the beach into the shallow pools. The pools are situated very close to the sea on a line parallel to the coast and the electrical conductivities of the water of three of such pools were measured and found to be 3800, 8800 and 16000 μS/cm. The latter two high values appear to be a result of mixing with sea water that occasionally overflows the sand bar in high tidal condition. Distribution of the observed EC in groundwater in relation to the distance from the sea and tsunami flow area is shown in Figure 21.4.

It is evident from the results that infiltrated saline water plume and its effect on groundwater has decreased with increasing distance from the sea and with the tsunami run-up height. Also, a part of saline water is appears moving slowly towards the sea with the lateral groundwater flow and seeps out at the near-beach fresh water pools.

21.4.2 *Results from resistivity imaging profiles*

Resistivity image in Figure 21.5 clearly indicates that that there are four distinct vertical resistivity horizons. The upper high resistivity (>1000 Ωm) zone extends from ground level to about 20 m depth. This zone is followed by a moderate (100–1000 Ωm) resistivity zone with a thickness of few meters. Below this occurs a thick and very low resistivity zone and at some places extending

Table 21.2. Electrical conductivity (EC) of the groundwater in village dug wells and in three shallow pools near the sea.

Well no.	W01	W02	W03	W04	W05	W06	W07	W08	W09	W10
EC (μS/cm)	1180	1500	1200	2300	3000	1600	2300	2800	2500	2300
Well no.	W11	W12	W13	W14	W15	W16	W17	W18	W19	W20
EC (μS/cm)	2600	3200	2900	1900	2300	1400	1900	1600	1500	1700
Well no.	W21	W22	W23	W24	W25	W26	W27	W28	W29	W30
EC (μS/cm)	1200	1200	1300	1500	1800	1100	1200	1400	1500	2100
Well no.	W31	W32	W33	W34	W35	W36	W37	W38	W39	W40
EC (μS/cm)	1300	1500	1200	800	1400	1700	2300	1400	700	1000
Well no.	W41	W42	W43	W44	W45	W46	W47	W48	W49	W50
EC (μS/cm)	1400	1500	2100	1300	1200	1100	1200	900	1000	1000
Well no.	W51	W52	W53	W54	W55	W56	W57	POOL1	POOL2	POOL3
EC (μS/cm)	600	800	800	260	210	190	170	3800	8800	16000

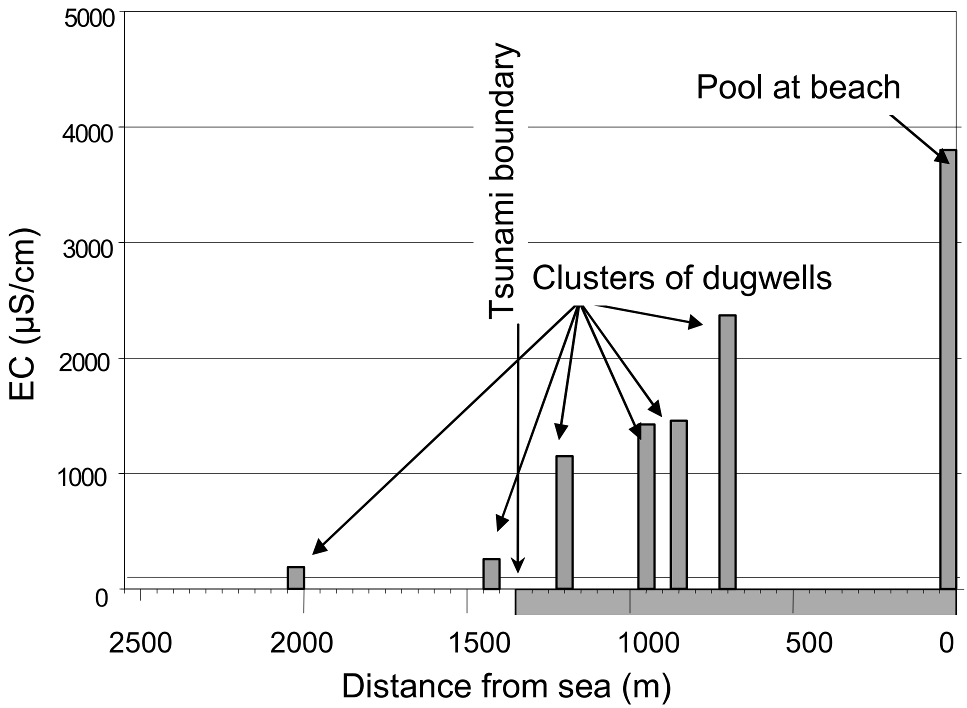

Figure 21.4. Variation of average electrical conductivity of groundwater in clusters of wells in relation to distance from the sea.

down to about 40 m depth. Lowest resistivity values appear to concentrate at a depth between 25 to 30 m. Presence of lowest and comparatively high resistivity zone probably indicate the crystalline bedrock below about 40 m depth. Surface morphology was not considered for the image since the ground surface can be considered as almost flat.

The groundwater table lies within a depth between 2 to 3 m in most parts of the area but has not apparently seen as a resistivity contrast in the image probably due to masking effect by the high resistivity of the upper sandy formation. The middle Moderate and low resistivity zone appears to be due to saline water where the lowest values are concentrated as a gently dipping layer at a depth between about 20 and 30 m. Further, it appears that a groundwater divide (at least for the saline water) is situated towards the eastern part of the profile and (saline) groundwater flows both towards West and East, away from this divide. The low resistivity zone however, is about 20 m thick in average, nearly horizontal and therefore could also be due to presence of clayey horizon at that depth in addition to the effect of saline water.

In order to clarify this situation, the resistivity image was further simulated assigning small resistivity variations between resistivity contours and narrowing down the extremely high resistivity values that are unrealistic for saline water bearing formations. Also, the model calculation was allowed to extend the calculated model towards either side of the profile. Thus observed results are shown in Figure 21.6.

The distribution of the saline water plume in the extended model is shown in Figure 21.6. It also indicated that the saline plume while descending, slowly mobilize towards the lagoon through the unaffected area in the west. At present the main saline water front is well below the depth of traditional dug wells in area and therefore will not pose a threat to the wells under normal (low discharge) water use conditions. A considerable number of deep water supply boreholes have been constructed after the tsunami in the unaffected part of the area to cater to the water supply needs

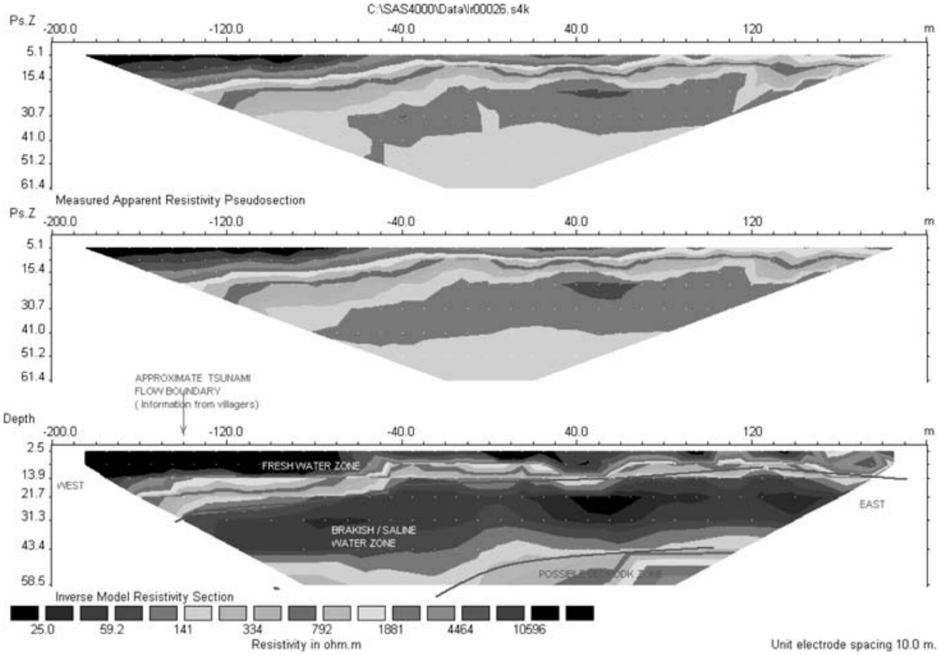

Figure 21.5. Subsurface resistivity image along the resistivity profile.

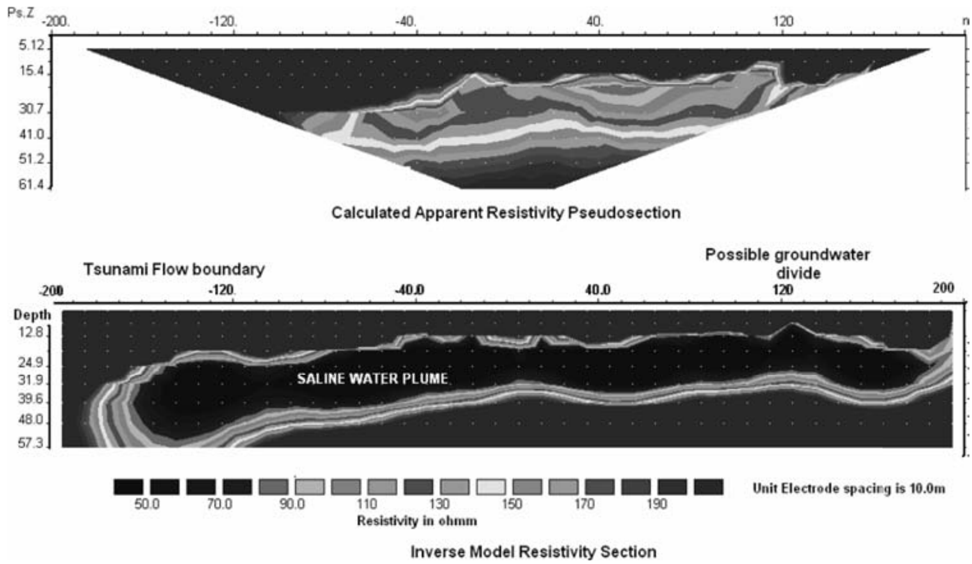

Figure 21.6. The distribution of the saline water plume.

in the affected parts. Westerly moving saline water plume can cause problems in these boreholes in future due to direct mixing or up coning of saline water plume.

Surface inundation with salt water due to the tsunami has increased the soil salinity. Especially in the areas where paddy cultivation has taken place the soil is clayey, which has high adsorption capacity the slats from the salt can be attached on to the clay surfaces by ion exchange and surface complexation processes. In those areas the effect of tsunami can be prolonged than the sandy aquifer system and that can reduce agricultural productivity. However, in the sandy aquifer systems the effect of tsunami will disappear rapidly than other aquifer systems due to high hydraulic conductivity in the system and the density driven effect.

21.5 CONCLUSIONS

The results of present investigation reveal that the amount of salinity of shallow groundwater in the coastal sandy aquifer has a distinct correlation with the tsunami run up height since most of the other physical properties of the aquifer are almost uniform through out the area. The infiltrated saline water plume in the sandy coastal aquifer is gradually sinking down while moving laterally towards the sea in the east and the lagoon in the west. The highest concentration part of the plume has descended well below the depths of the existing dug wells in the area and therefore, a gradual improvement of the water quality will take place without posing a threat to the shallow dug wells to become saline again under normal well use patterns. Deep water supply bore holes in the unaffected area in the west however are at risk due to oncoming saline water plume with groundwater flow towards east. This indicates the necessity of protective measures before the bore holes become contaminated. Infiltration of precipitation of the annual and regular monsoon rains is expected to improve the quality of water through flushing and dilution in the uppermost horizons of the aquifer.

Further investigations are in progress for imaging the entire length of the land between the sea and the lagoon covering the whole aquifer profile and verifying the ground truth through test drilling logging and water quality analysis.

ACKNOWLEDGEMENTS

We would like to thank the editors of the book for their kind encouragement. Valuable comments given by the reviewers significantly improved the manuscript quality.

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CHAPTER 22

Interactions between saline and fresh water in coastal region of northwestern Sri Lanka

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ABSTRACT: The Puttalam lagoon, situated in the northwestern region of the Republic of Sri Lanka is an area where groundwater is exploited from the Quaternary and Miocene aquifers. Three types of favorable saline intrusion aquifer systems were identified in the study area. Analyzing the forming of seawater intrusion to water intakes with the pumping and without pumping, considering different densities of salt water, on the basis of analytical solutions. Using the analytical method determine the length of salt water "lens" considering different values of saline and fresh water density. The calculations Indicate, for the definite groundwater gradients being polluted due to sea water intrusion. The analytical solutions can be used for forecasting the seawater Intrusion in the coastal aquifer areas if correct hydrogeological information being available.

22.1 INTRODUCTION

Groundwater in the coastal regions there is some direct or indirect interference with sea water (saline water). It can be caused due to infiltration of sea water into coastal aquifers by fresh groundwater exploration. Under natural conditions, in coastal areas an equilibrium exists between seawater and freshwater which depends on the geological and hydrogeological conditions of the region. Due to high density of seawater, it tends to force its way underneath fresh water (Gavich 1980, Goldberg 1984, Todd 1959, Xue 1993). However, the piezometric head of the fresh water is higher than that of seawater, the fresh water continually discharges to the sea. The fresh water discharges opposite the inland movement of seawater, and thereby equilibrium is established. Once the groundwater level equilibrium has been established, water level will stabilize and only fluctuates annually when there are seasonal changes and other natural causes.

Groundwater salinity can occur by following three types of processes:

- salinity zones of natural and tidal origin
- an advancing seawater front intruding inland as a result of over-pumping
- zones of connate saline water remaining from geological or ancient marine conditions

Groundwater salinity intrusion is brought about by man and the remaining two are natural phenomena for which allowance has to be made in groundwater development planning. If water resources are overdeveloped and abstraction rates exceed the average recharge, groundwater levels will decline. This causes shallow wells and springs to dry up and increases the cost of pumping from deeper tube wells. Otherwise seawater intrusion occurs when fresh water is withdrawn. Eventually, groundwater will no longer be available for abstraction. The withdrawal changes the equilibrium between the fresh and seawater. Other serious problems may develop particularly by deterioration of the water quality caused by such factors as seawater intrusion into coastal aquifers. Hence, the fresh water piezometric head is decreased. The decreased in the fresh water piezometric head allows seawater to move further inland. Sea water (saline) and fresh water actually miscible fluids; therefore the zone of contents between them take the form of a transition zone caused by hydrodynamic dispersion.

In the current century, fresh groundwater uses have been increased enormously, resulting in step declines in the water table and even reversal of the fresh water gradient. This activity has resulted in high-density seawater intrusion in the coastal aquifers. Identification of coastal aquifers is very important in studying the problems of salt water intrusion, helps in finding out the features of the salt water and fresh water interface, it is also very essential to study the pattern of its movements and the hydrochemical characteristics of the sea water intrusion area.

22.2 AREA OF STUDY

22.2.1 *Location*

Puttalam town area is in Northwestern coastal region of Sri Lanka (Fig. 22.1). The western part of the area forms extremely flat coastal plain, but to the east and northeastern direction is isolated rounded and isolatly elongated ridges reaching a maximum elevation of about 60 m above mean sea level. Puttalam is the largest town in the study area, near its eastern margin of lagoon and is 132 km away from capital of Sri Lanka, Colombo. Puttalam lies between the latitude 77°50′17″ to 80°5′38″N and by longitude 79°55′17″E and on the west by the Indian Ocean (Puttalam lagoon).

The climate of Sri Lanka in general, and of the area examined in particular was a typical tropical type. North western region that experiences an average annual rainfall between 1000 to 1250 mm range and the annual rainfall for Puttalam is comparatively about 1167 mm. The average relative humidity of the Puttalam area 79% and temperature varies from 23°C to 32°C.

In the coastal strip of the study area, there are a few man-made lakes and a number of natural lakes (*villus*) and mashes, because of the flatness of the area, very slow drainage take place through the meandering streams and the *villus* and the marshes into the lagoon or sea. The north of the Putalam town area is drained by Mi Oya river system. Mi-oya flows from east to west and discharge its water into Puttalam Lagoon. In considering Mi Oya delta, there are many tributaries and during the heavy rainy sessions serious flooding effect the Mi Oya delta.

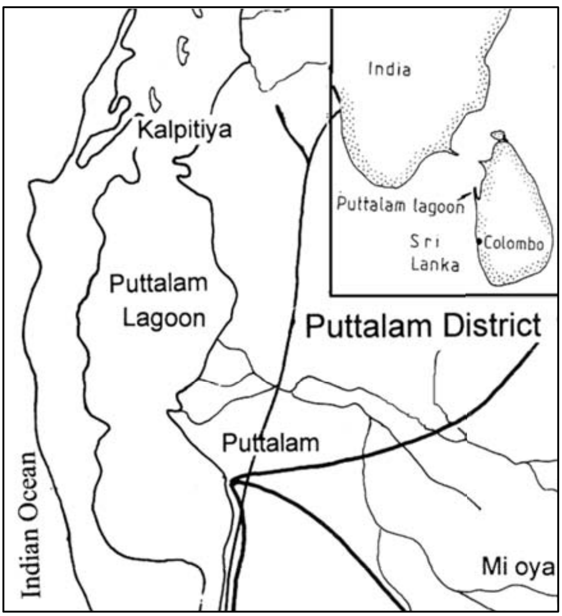


Figure 22.1. Location map of northwestern region of Sri Lanka.

22.2.2 *Geology*

The geology of the north-western region of Sri Lanka was studied by Coorey (1984, 1995), Wadia (1942). The western part of the study area is underlain by sedimentary (Tertiary) Miocene limestones, which are generally fossiliferous. This limestone is covered by sedimentary deposits of Quarternary age. East of these geological formations, the basement consists of Proterozoic metamorphic rocks of the Wanni complex (formerly the west Vijayan complex).

The underlying crystalline basement rocks and the overlying Quaternary deposits are the unconformable with the Miocene succession. Miocene sequence is represented by the Mannar sandstone (Lower Miocene) which dips to the west where it attains a maximum thickness of about 40 meters. Lower part of the Mannar sandstone formation becomes more silty and the upper part gradationally changes to limestone. The Mannar sandstone consisting of soft, yellow coralline limestone passes upwards into a series of hard siliceous limestone, with clay bands containing a lagoonal fauna of gastropods and lamellibranches. Upper portion of the Miocene succession is the Vanathvillu Limestone (Upper Miocene) which has a gradational contact with the underlying clayey quaternary sediments or confined limestone and overlying with Moongail Aru (Central Miocene) thin layer of alluvial clay sediments or argillic limestone and shales.

22.2.3 *Hydrogeology*

The aquifer of modern alluvial depositions is distributed on flood plain terraces and mostly they are mainly on flood plain of the river basin of Mi-oya (Figs. 22.2 and 22.3). Water bearing sands in top of a section are more often fine and in lower sections are usually coarse sand with gravel. Sometimes, in a lower section of the aquifer consists of gravel and bolder beds. The recharge of the aquifer takes place mainly by precipitation and flood water. The maximum values of a discharge rate in dug and tube wells compute to 2–5 liters/second. Coefficient of transmissivity of the aquifer in the study area change up to 20 m/day and the average value is 3–7 m/day.

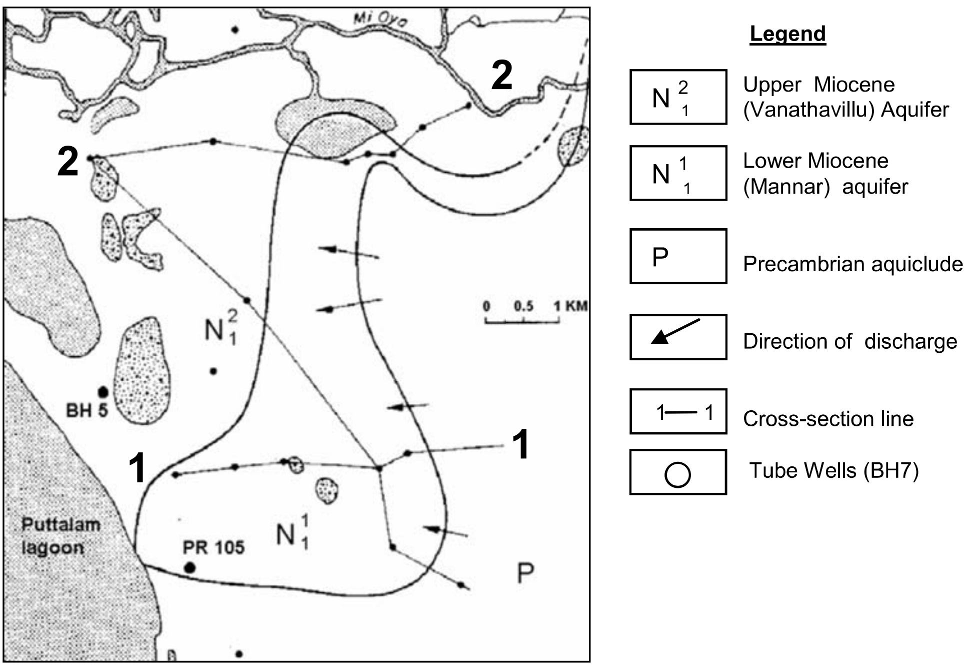


Figure 22.2. Hydrogeological map of the Puttalam area at the northwestern coast of Sri Lanka.

Lagoon water bearing aquifer lies in low and flat area in parallel to 1–2 km from lagoon and near a delta of the river Mi-oya. The thickness of aquifer varies in between (the available information) from 5 to 10 m. Water bearing formations are the sand, sandy clay, alleviates, coarse sand with low contents of gravel. The groundwater level within the aquifer change in-between 1–2 meters below the ground level. The recharge is mainly receiving from the precipitation and during the dry seasons by the lagoon saline water. In groundwater close to lagoon area chloride content is basically high and wells drilled on lagoon aquifer which is 0.5 kms from city center the chloride content is 750 mg/L.

The old alluvial sand aquifer extends in high terraces and in the alongshore lagoon Puttalam and Delta of Mi-oya. Water bearing soils are sand, sandy clay loams and clay. The groundwater level occurs in the depth of 0–2 m from ground surface and in more sandy clay areas water level is 2–5 m. Recharge of the aquifer on the high terrace basically receiving from the atmospheric precipitation. The aquifer has a thickness of 10–15 m and characterized by a high permeability and good quality water.

The Quaternary deposits in this area (Fig. 22.3a, b) mainly consists with clay and small quantity of gravel and is an aquiclude (impermeable). The thickness of the aquiclude is 10–15 m. Upper Miocene aquifer is under the quaternary aquiclude and manly consists with limestone and less often marls and chalkstones. In the western part of the area close to the lagoon, the aquifer consists of low permeable chalkstones. This aquifer basically in this area is not used for water supply and therefore, there is no adequate information in this regard.

Central Miocene aquiclude (Fig. 22.3b) mainly consists of argillic Limestone, shales and Fossil clay. It occurs in the northwestern part of study region and thickness varies from 5 to 40 m. In eastern and southern parts of region, the aquiclude is not available. In northeast part the thickness of aquiclude consists of 25 m.

Water bearing soils in the lower Miocene aquifer consist mainly with sand stone and sandy limestone. In eastern part of the region sandstones are available more than western part. In the western part of the region, lower Miocene aquifer thickness consists by 100–120 m and northern part 50–60 m. The pumping tests have shown that the production rate of the wells varies within the limit of 1500–2500 m^3/d.

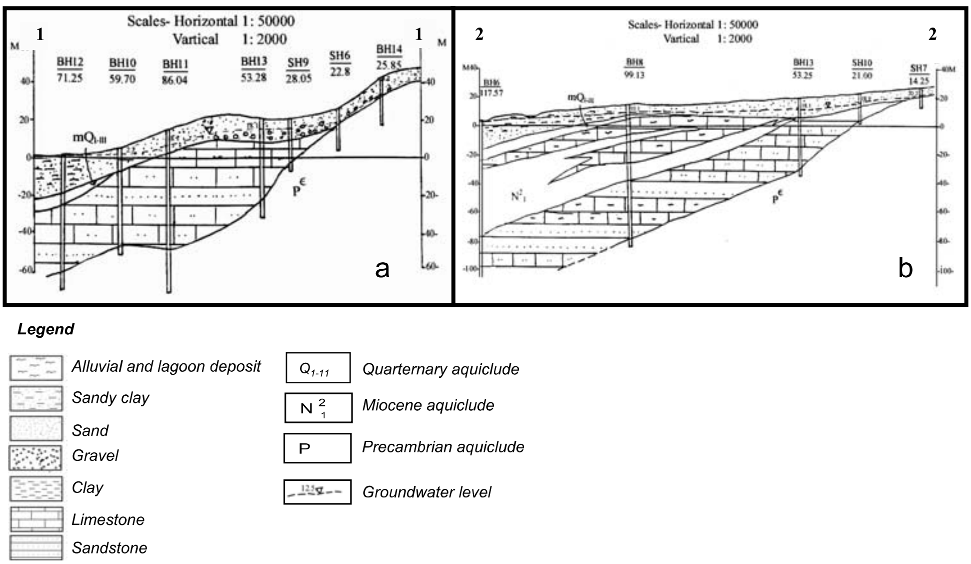


Figure 22.3. Hydrogeological cross sections along the transects: (a) 1—1; (b) 2—2. See Figure 22.1 for the location of the transect lines.

22.3 METHODOLOGY

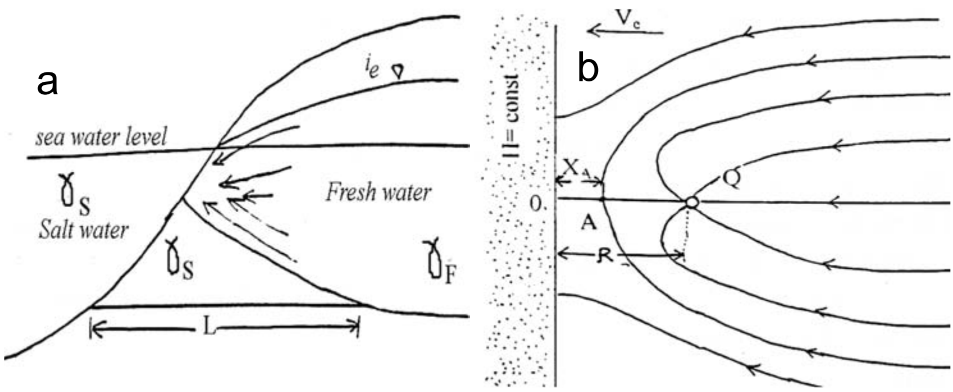
The particular features of fresh groundwater quality in coastal regions are indicating due to direct or indirect interference with seawater. It based on infiltration of seawater into coastal fresh aquifers or sea water intrusion in to groundwater due to groundwater exploration in the coastal aquifers. Filtering fresh and salt water in coastal regions system composed, in shared, from two heterogeneous fluids of characteristic on the densities and viscosities. Two different densities of fluids form the interface in between the fluids and the forming of the saline water "lens" in the coastal aquifer areas. The saline Intrusion "lenses" more of a heavy liquid on a base surface of a layer. If the initial margin between fresh and salt water is vertical and further this margin becomes inclined because more high-density pressed salt water than light density fresh water. The intrusion of seawater into aquifer mainly occurs due to groundwater exploration in the coastal regions. The approximated expression for an estimation of final length (L) of a wedge of salt water in aquifer in conditions of hydrodynamic equal balance fresh and salt water in Equation 22.1(Fig. 22.4a) (Todd, 1959).

$$L = Tm(\gamma_0 - 1)/q = m(\gamma_0 - 1)i_e \tag{22.1}$$

where i_e gradient of the groundwater, $\gamma_0 = \gamma_s/\gamma_f$ relation of densities, saline γ_s and fresh water γ_f, T = transmissibility (m^2/day), m = aquifer thickness, q = specific yield per unit length of aquifer (m^2/day).

According to the equation for the high values of the natural groundwater flow length of the seawater lens is decreasing. With variation of a gradient of a groundwater changed the length of seawater intrusion lens. For the low groundwater gradient values seawater lens length is increasing.

The estimated the seawater intrusion lens in the coastal areas with constant natural groundwater head (Goldberg, 1984). The water intake is located R distance from coastal line of the sea and received the constant natural groundwater head. The Y-axis is located in the coastal line (Fig. 22.4b). In the coastal area exploration well working with discharge of "Q" the natural groundwater flow discharge toward to the coastal line. In this situation the water divider in-between salt water and fresh water formed in coastal area. The closest point of the water diviner is located from coastal



γ_f - density of fresh water; γ_s - density of salt (sea) water; ***L*** *- Length of the sea water lens;*
X_A *- length of the sea water front from coastal line;* ***Q*** *- Discharge of the tube well*
R *- Distance between costal line and tube well;* ***i_e*** *- Fresh water hydraulic gradient*

Figure 22.4. (a) Schematic diagram showing the formation of salt-water lens along a coastal line; (b) pattern of groundwater flow in the coastal belt with exploration tube well.

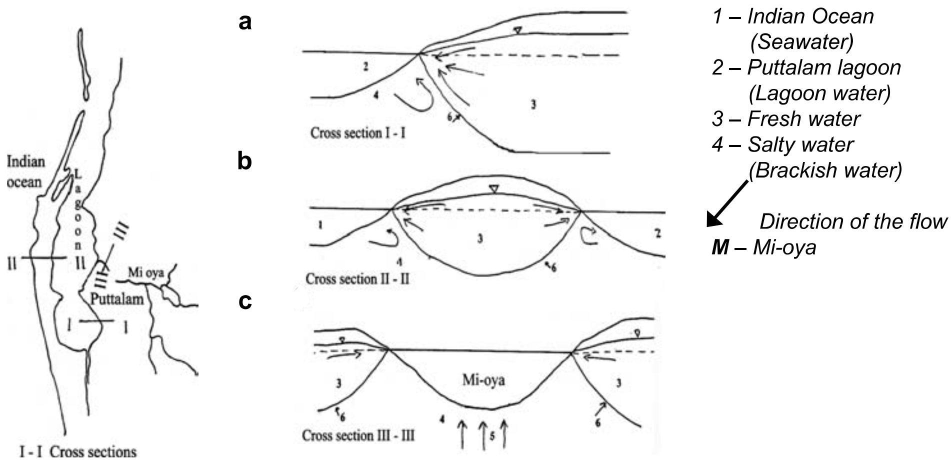


Figure 22.5. Coastal aquifer systems favorable for sea water Intrusion.

line is "A" in X-axis (Fig. 22.5b). Distance between coastal line and point "A" can be calculated from Equation 22.2.

$$X_A = \sqrt{R^2 - (qR/\pi V_e)} = \sqrt{R^2 - (QR/\pi mki_e)} \tag{22.2}$$

where q = production rate of a water intake referred to seam thickness, V_e = velocity of a natural flow of groundwater, k = coefficient permeability of groundwater; i_e, gradient of the natural groundwater flow; m = Thickens of the aquifer.

According to the above equation, criteria for possibility of seawater intrusion in to the coastal wells are following;

$$q/(\pi V_e R) \geq 1 \tag{22.3}$$

The value of $q(V_e R) < 1$ the saline water intrusion in to the exploration well is not possible and if $q(V_e R) > 1$ saline water intrusion possible. Therefore the Equation 22.3 defining the water divider point is available in coastal area.

22.4 SALINE WATER INTRUSION IN NORTHWESTERN REGION

The several analyses were conducted surrounding River delta Mi-Oya and River water to identify the saline intrusion. The quaternary unconfined aquifer distributed in the study area directly contact with saline water bodies such as Indian Ocean or Puttalam lagoon and therefore more possibility is available for saline water intrusion to the aquifers. In the study area identified three types of aquifers favorable for the seawater intrusion and mainly these aquifers are consisted with lagoonal sandy alluvium, and old alluvium formations (Fig. 22.5).

First aquifer system is distributed along the coastal line of Puttalam town and the aquifers are represented by the lagoonal deposits (Fig. 22.3). In this area unconfined lagoonal aquifer directly contacts with Puttalam lagoon water and the wells are distributed close to shoreline. Second possibility of seawater intrusion of aquifer system is distributed in between the Indian Ocean and Puttalam lagoon and it's identified as Island type unconfined aquifer. In this area recharge receiving only from atmospheric precipitation. Third type of aquifer system is available in the Mi-oya river delta area and aquifer is consisting with alluvium and lagoon soils.

Table 22.1. Chloride concentration in the unconfined aquifer.

	River water near SH3	Groundwater in SH4	River water near SH4	Groundwater in SH5	River water near SH5
Chloride (mg/L)	510	1625	510	Upper layer 462 Lower layer 1196	510

Table 22.2. Calculated values for identify saline water front in to the coastal aquifers.

I = gradient of natural groundwater flow	0.011	0.013	0.015	0.017	0.019	0.021	0.0236
V_e = Velocity of natural ground water flow	0.033	0.039	0.045	0.051	0.057	0.063	0.069
$Q/\pi V_eR$	0.89	0.75	0.65	0.58	0.51	0.47	0.42

The preliminary results revealed that the chloride concentration is high in the study area and it increased with depth. The quality of the groundwater in the unconfined aquifer is analyzed and chloride concentrations are in the Table 22.1.

According to the information's groundwater in the unconfined aquifer is highly polluted due to lagoonal water and tidal waves. But water available in the river water is not so polluted due to inland river runoff. Concentration of the chloride in the groundwater values place to place difference and some times its value takes 462 to 1196 mg/L. In the depth 2.1 m chloride concentration is 1196 mg/L. Extent of groundwater and lagoon water quality depends on the aquifer characteristics.

22.5 GROUNDWATER QUALITY DISTRIBUTION IN COASTAL AQUIFERS

To identify the length of the sea water intrusion "lens" in the coastal region of Puttalam due to groundwater exploration use the analytical method discussed in section 22.3. Most possible seawater intrusion can occur in the Puttalam town area and in this area top unconfined aquifer is directly contact with lagoon. Two tube wells located in the western region of Puttalam town area are selected to identify saline intrusion due to groundwater exploration. Two tube wells PR105 and BH5 (Fig. 22.2) constructed in lagoonal unconfined aquifer to identify saline intrusion by analytical solutions. Two exploration tube wells are located 312 m (PR105) and 520 m (BH5) from coastal shore line of Puttalam lagoon with constant natural water head. Natural groundwater flow is parallel (X axis) and perpendicular to lagoon shoreline. Pumping rates of the two wells are simultaneously 1300 and 430 m^3/day. Calculated gradient of a natural flow in region of well PR105 varies within the limits 0.011–0.022 and in region of well BH5 within the limits of 0.0001–0.0095. Thickens of the aquifer in the well PR105 is 45 m and coefficient permeability of the aquifer is 3 m/day. Thickens of the aquifer in the tube well BH5 is 25 m, and coefficient of permeability of the aquifer is 3 m/day.

22.5.1 *Calculation of salt water intrusion "lens" to tube wells without considering density of fresh and salt water*

Exploration well working in the natural groundwater flow developing the area of recharge surrounding the well, If the coastal area natural water divider is calculating from Equation 22.2. Position of the water divider point X_A calculated with deferent hydraulic gradients in between 0.011–0.022 for tube well PR 105 (Table 22.2).

Within the calculated hydrologic gradient range the value $Q/\pi V_eR < 1$ and saline water front not reach to the exploration well. According to the natural flow of the area calculated the values of X_A (Distance between shore line and water divider) (Table 22.3).

Table 22.3. Calculated the values of the X_A according to the natural groundwater flow.

I = gradient of natural ground water flow	0.011	0.013	0.015	0.017	0.019	0.021	0.0236
V_e = Velocity of natural ground water flow	0.033	0.039	0.045	0.051	0.057	0.063	0.069
X_A, m	102.3	153.2	183.2	202.7	216.1	227.1	236.5

Table 22.4. Calculated condition for saline water intrusion is in the different gradients.

I = gradient of natural groundwater flow	0.0001	0.0002	0.0003	0.0004
V_e = Velocity of natural groundwater flow	0.0003	0.0006	0.0009	0.0012
$Q/\pi V_e R$	35.1	17.6	11.07	8.8

The results reveled that the calculated length of the saline water "lens" is less than distance between Puttalam lagoon and Exploration tube well (PR105). Therefore saline water intrusion (front) is not developing within the conceded area. According to the hydraulic parameters calculated the saline water front in the well BH5 region (Table 22.4).

Within the calculated hydrologic gradient range the value $Q/\pi V_e R > 1$ and saline water front is reach to the tube well BH5. Therefore the results of calculations demonstrate, that $X_A > R$ and saline water front is advanced into the exploration BH5 well.

22.6 CONCLUSION

As per the hydrogeological conditions of the North western region of Sri Lanka identified three different aquifer systems favorable for the sea water intrusion. Presently Puttalam lagoon area and delta of the Mi-oya area unconfined aquifers polluted due to sea water intrusion.

The results from the present study reveals saline water intrusion due to hydraulic gradient equal to 0.011 in the area of PR105 tube well. If the hydraulic gradient above 0.011 the saline water intrusion seems unlikely to occur in the area of PR105. The calculated hydraulic gradient in the area around BH5 indicate that sea instrusion has occurred.

Therefore, calculations the ranges of gradients of a current obtained on simulation utilized. The executed analytical calculations demonstrate, that the intrusion of tube wells of salt water to development wells PR105 and BH5 is possible in considered ranges of a gradient of a natural flow, specially accounting with the different densities of the gravity saline and fresh water.

ACKNOWLEDGEMENTS

The author appreciates the many helpful suggestions given by Mr. A. Hapugaskumbura of Water Resources Board Colombo Sri Lanka. The comments from the two anonymous reviewers have helped to the quality of the manuscript. Dr. P. Bhattacharya and Dr. AL. Ramanathan are specially acknowledged for the editorial comments.

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CHAPTER 23

Groundwater flow model in a mangrove forest

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ABSTRACT: The ability to predict groundwater fluxes with minimum efforts and measurements is an important objective. Numerical modeling is one approach to obtain such prediction. Predictions of groundwater fluxes can be used to determine fluxes of other materials such as salts and nutrients. In this paper an analytical model is developed to predict the flow of groundwater from mangrove forest to the creek. The model uses the geometry and hydraulic conductivity of the mangrove forest sediment, which is inundated by tidal water from day zero to day five, with the flux ranged from 0.026 to 0.007 $m^3/(m^2$ day) with an average error of about 10%. The solution for the groundwater flow is written in terms of an analytic series solution, based on two dimensional potential flows. The approach is basically to solve the hydraulic potential flow for steady state conditions using the Laplace Equation. The advantages of this method are that it is simple yet accurate, and the error in the computation can be readily calculated. The result of this model are then compared to the result of the field measurement from also day zero to day five after inundation, which ranged from 0.030 to 0.013 $m^3/(m^2$ day) with the average error about 40%. The results reveal that the series solution model can be used to calculate the flux of the groundwater, especially in the mangrove forest area.

23.1 INTRODUCTION

The flux of groundwater flow in the mangrove forest sediment is very important. This groundwater flow can reduce the concentration of salt in the sediment. One of the methods for calculating the flux is by investigating the hydraulic conductivity of the sediment. The determination of hydraulic conductivity of sediment can be conducted by in-situ and laboratory measurement. The hydraulic conductivity of the sediment, the mangrove forest is influenced by burrows occupying the sediment. Susilo & Ridd (2005) and Susilo (2005) found that by pumping out of the water in the burrows, where one burrow system is intermingled with each other, the hydraulic conductivity ranges from 1 to 10 m/day. The estimated surface hydraulic conductivity of intertidal zone sediments from laboratory measurements was estimated to be approximately 0.01 m/day (Hughes et al. 1998). Because of the burrows on the sediment, they estimated that the soil hydraulic conductivity would increase by factor 1 or 2 orders of magnitude. This hydraulic conductivity will determine the flux of the groundwater flowing in the mangrove sediment which will be able to flush the solutes, such as salt, nutrient and toxins (Heron 2001).

Another way to determine flux of the groundwater flowing in the mangrove forest sediment is by installing an array of piezometers. Susilo et al. (2005) found that the flux ranges from 0.03 to 0.013 $m^3/(m^2$ day) from the measurement using piezometers. The highest flux was found in the area closest to the creek and smaller at the area farther away from the creek.

The sediment in the mangrove forest is composed of two layers, the upper layer of which has a very high hydraulic conductivity due to the presence of animal burrows. The lower layer is effectively impermeable. The presence of a relatively impermeable salt flat adjacent to the mangrove swamp is also taken into account by forcing a no flow boundary condition at this interface.

In this paper, an analytical model is developed to predict the flow of groundwater from the mangrove forest to the creek. The model uses the geometry and hydraulic conductivity of the mangrove sediment. The solution for model utilizes similar to the methodology used by Gill & Read (1996). It is written in terms of an analytic series solution, based on two dimensional potential flows. The model will be validated by comparing the model with data collected from the field experiment (Susilo 2004, Susilo & Ridd 2005).

23.2 METHOD

23.2.1 *Laplace equation*

The general Equation of steady state flow in the groundwater flow is (Rushton & Redshaw 1979):

$$\frac{\partial}{\partial x}\left(k_x\frac{\partial\Phi}{\partial x}\right)+\frac{\partial}{\partial y}\left(k_y\frac{\partial\Phi}{\partial y}\right)+\frac{\partial}{\partial z}\left(k_z\frac{\partial\Phi}{\partial z}\right)=0 \tag{23.1}$$

where k_x, k_y, k_z are hydraulic conductivities of the sediment to the x, y and z directions. Φ is hydraulic potential of the groundwater, which is the groundwater level (Wilson 1994). If the soil of the sediment is homogeneous and isotropic, Equation (23.1) can be written as:

$$\frac{\partial^2\Phi}{\partial x^2}+\frac{\partial^2\Phi}{\partial y^2}+\frac{\partial^2\Phi}{\partial z^2}=0 \tag{23.2}$$

Equation (23.2) is well known for steady state flow, and is called the Laplace Equation. If the geometry is two dimensional vertically with flow only in the x and y direction, Equation (23.2) becomes (Strack 1989, Bouwer 1978):

$$\frac{\partial^2\Phi}{\partial x^2}+\frac{\partial^2\Phi}{\partial y^2}=0 \quad \text{or } \nabla^2\Phi=0 \tag{23.3}$$

Equation 23.3 will be used to solve the problem of the groundwater flow from the mangrove forest sediment to the creek.

23.2.2 *Boundary conditions*

The situation of the groundwater seepage is modeled as illustrated in Figure 23.1. In order to solve the Laplace Equation, the following boundary conditions are applied:

1. No flow across the vertical boundaries at $x = 0$ and $x = s$.

$$\left.\frac{\partial\Phi(x,y)}{\partial x}\right|_{x=0}=0, \quad \left.\frac{\partial\Phi(x,y)}{\partial x}\right|_{x=s}=0 \tag{23.4}$$

   The above Equations are velocity potential to the x direction.

2. No flow into the lower impermeable layer at $y = 0$.

$$\left.\frac{\partial\Phi(x,y)}{\partial y}\right|_{y=0}=0 \tag{23.5}$$

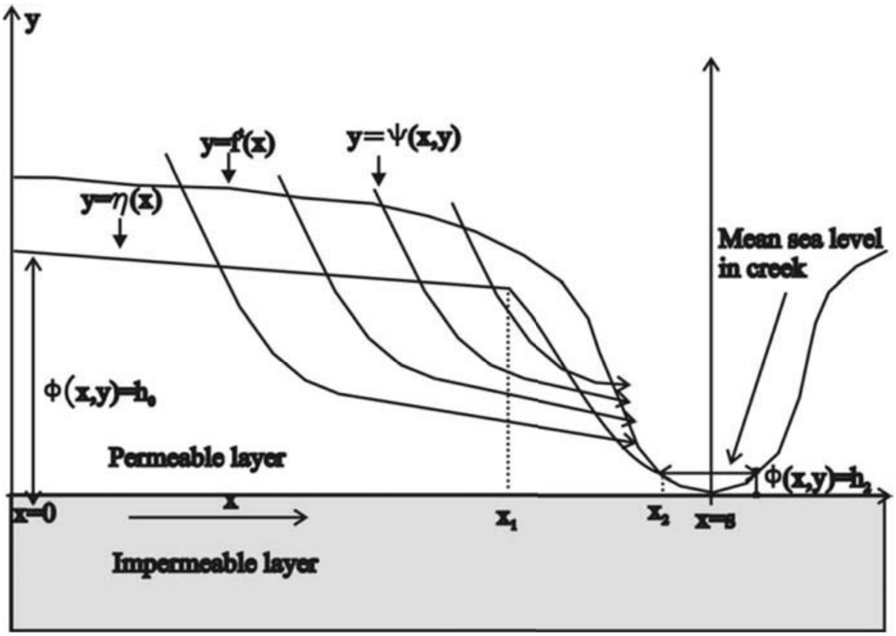

Figure 23.1. Model of the groundwater seepage to the creek. In this model, h_2 is 0 m. $\eta(x)$ represents the groundwater free surface, $f^t(x)$ the sediment surface, $\Psi(x, y)$ is the stream function of the groundwater.

3. The water level in the creek takes its tidally averaged value, i.e.

$$\Phi(x, y) = h_2 = 0, \quad x_2 < x \leq s \tag{23.6}$$

23.2.3 *Solution for the hydraulic potential, $\Phi(x, y)$, of the groundwater*

The method used to solve Laplace Equation is separation of variables i.e the hydraulic potential is written in the form (Gill & Read 1996, Liggett & Philip 1983):

$$\Phi(x, y) = X(x)\,Y(y) \tag{23.7}$$

Using the Laplace Equation, Equation (23.7) can be written as:

$$X''Y + XY'' = 0 \tag{23.8}$$

The solution of Equation (23.8) is:

$$\frac{X''}{X} + \frac{Y''}{Y} = 0 \quad \text{then } \frac{X''}{X} = -\frac{Y''}{Y} = -\sigma^2 \tag{23.9}$$

Equation (23.9) can be rewritten as:

$$X''(x) - \sigma^2 X(x) = 0 \tag{23.10a}$$

$$Y''(y) - \sigma^2 Y(y) = 0 \tag{23.10b}$$

Applying the boundary conditions into Equation (23.4), the solution of Equation (23.10a) becomes:

$$X_n(x) = \cos(n\pi x/s), \quad \text{and } n = 0, 1, 2, \ldots \tag{23.11}$$

Applying the bottom boundary condition (Equation (23.5)), the solution of Equation (23.10b) is:

$$Y_n(y) = \begin{cases} \alpha_0, & n = 0 \\ \alpha_n \cos\mathrm{h}(n\pi y/s), & n \geq 1 \end{cases} \tag{23.12}$$

The general Equation can be written as (Gill & Read 1996):

$$\Phi(x,y) = \sum_{n=0}^{n=\infty} [\alpha_n \, u_n(x,y)] \tag{23.13}$$

where

$$u_n(x,y) = \cos\mathrm{h}(n\pi y/s)\cos(n\pi x/s) \tag{23.14}$$

Equation (23.14) is a time independent function, and represents the steady hydraulic head (or hydraulic potential) throughout the saturated part of the soil beneath the mangrove forest.

23.2.4 *The stream function, $\Psi(x,y)$, of the groundwater*

In this model, where the sediment is homogeneous and isotropic, the direction of the flow will be normal to the equipotentials (Bouwer 1978), and therefore the streamlines of the groundwater flow must be orthogonal to the equipotential lines or water table. The stream function $\Psi(x,y)$ is thus defined as:

$$v_x = -\frac{\partial \Psi}{\partial y}, \quad v_y = \frac{\partial \Psi}{\partial x} \tag{23.15}$$

where v_x and v_y are velocities of flow (Rushton & Redshaw 1979) in the x and y directions.

The equation of continuity is defined as:

$$\frac{\partial v_x}{\partial x} + \frac{\partial v_y}{\partial y} = 0 \tag{23.16}$$

Substituting Equation (23.15) to (23.16) we have

$$\frac{\partial^2 \Psi}{\partial x \partial y} - \frac{\partial^2 \Psi}{\partial y \partial x} = 0 \tag{23.17}$$

In a homogeneous isotropic soil, where the hydraulic conductivity of the sediment is constant, v_x and v_y are (Rushton & Redshaw 1979):

$$v_x = -\frac{\partial \Phi}{\partial x}, \quad v_y = -\frac{\partial \Phi}{\partial y} \tag{23.18}$$

Therefore, the relationship between the hydraulic potential and stream function (Strack 1989 p. 222) is:

$$\frac{\partial \Phi}{\partial x} = \frac{\partial \Psi}{\partial y}, \quad \frac{\partial \Phi}{\partial y} = -\frac{\partial \Psi}{\partial x} \tag{23.19}$$

Equation (23.19) is well-known as the Chauchy-Riemann Equation. Using Equation (23.19), which is applied to Equation (23.14), the stream function of the groundwater will be:

$$\Psi(x,y) = c + \sum_{n=1}^{n=\infty} \alpha_n \sin\text{h}\left(\frac{n\pi y}{s}\right) \sin\left(\frac{n\pi x}{s}\right) \tag{23.20}$$

where c is an arbitrary constant. In this model, c is zero along the impermeable boundary. The dimension of the stream function is $[\mathrm{L}^2\mathrm{T}^{-1}]$.

23.2.5 *Solving for $\Psi(x,y)$, calculation of α_n coefficient*

When the sediment is water saturated, the free water table will be the sediment surface itself,

$$\eta(x) = f^t(x) \tag{23.21}$$

If variable y in Equation (23.14) is replaced by the function of the sediment surface, $f^t(x)$, then the equation will only have one variable, i.e. x. The hydraulic potential, $\Phi(x,y)$, will also only vary in x. $\Phi^t(x)$ is introduced as a hydraulic potential, Equation (23.14) becomes:

$$\Phi^t(x) = \sum_{n=0}^{n=\infty} \alpha_n \cos\text{h}\left(\frac{n\pi f^t(x)}{s}\right) \cos\left(\frac{n\pi x}{s}\right) \tag{23.22}$$

To solve Equation (23.22), it is needed to calculate the coefficient α_n. It will use the least square method to calculate the A_n, the least square approximation to α_n. The series after $n = N - 1$ terms needs to be truncated, so Equation (23.22) is rewritten as:

$$\Phi^t_N(x) = \sum_{0}^{N-1} A_n u^t_n(x), \tag{23.23}$$

where

$$u^t_n(x) = \cos\text{h}\left(\frac{n\pi f^t(x)}{s}\right) \cos\left(\frac{n\,\pi x}{s}\right)$$

The hydraulic potential along the upper saturation boundary is:

$$h^t(x) = \begin{cases} \eta(x), & 0 \le x \le x_2 \\ h_2, & x_2 < x \le s \end{cases} \tag{23.24}$$

Equation (23.24) assumes that the location of the water table, $\eta(x)$, is known. Therefore

$$h^t(x) \approx \sum_{n=0}^{n=N-1} A_n u^t_n(x)$$

The hydraulic potential along the saturated boundary must satisfy:

$$h^t(x) \approx \sum_{n=0}^{n=N-1} A_n u_n^t(x) \tag{23.25}$$

The least square method is used to calculate the coefficient A_n from Equation (23.25),

$$u^t \underset{\sim}{a} = \underset{\sim}{h}^t \tag{23.26}$$

$$[u^t]_{ij} = \langle u_i, u_j \rangle, [\underset{\sim}{u}^t]_i = \langle u_i, h^t \rangle \tag{23.27}$$

Note that $\langle u_i, u_j \rangle \neq \|u_i\|^2 \delta_{ij}$
The first step in this process is to calculate:

$$[u^t]_{ij} = \langle u_i^t, u_j^t \rangle = \int_0^s u_i^t(x)\, u_j^t(x) dx \tag{23.28}$$

where,

$$u_i^t(x) = \cos\mathrm{h}\left(\frac{i\pi f^t(x)}{s}\right) \cos\left(\frac{i\pi x}{s}\right) \tag{23.29}$$

$$u_j^t(x) = \cos\mathrm{h}\left(\frac{j\pi f^t(x)}{s}\right) \cos\left(\frac{j\pi x}{s}\right) \tag{23.30}$$

Note that i and j are integers, which are 1, 2, 3, ..., n.
In the second step

$$[\underset{\sim}{u}^t]_i = \langle u_i^t, h^t \rangle = \int_0^s u_i^t(x) h^t(x) dx \tag{23.31}$$

is calculated, where $u_i^t(x)$ is calculated from Equation (23.29) and $h^t(x)$ is taken from Equation (23.26). The values of A_n are obtained by solving numerically the matrix Equation (23.27). These values are then used to calculate hydraulic potential (Equation (23.14)) and stream function (Equation (23.20)).

23.2.6 *Error analysis*

The root-mean-square error (rms error) is defined by the error between the approximate measurement of A_i and the true value of T_i (Scheid 1968). For discrete data, the rms error is defined as:

$$\varepsilon = \left[\frac{1}{N} \sum_{i=0}^{N} (T_i - A_i)^2\right]^{\frac{1}{2}} \tag{23.32}$$

where T_i are known values and A_i are approximate values. In this model, both the fixed value of hydraulic potential $h^t(x)$ and the approximate value of hydraulic potential $\Phi(x, y)$ resulted from the calculation of the series solution are continuous. Therefore, the rms error is given by:

$$\varepsilon = \left[\frac{1}{s} \int_0^s (f(x) - \hat{f}_N(x))^2\right]^{\frac{1}{2}} \tag{23.33}$$

where $f(x)$ is a true value and $\hat{f}_N(x)$ is a calculated value. With larger values of n, the error tends to zero (Scheid 1968).

23.2.7 *Calculation of the flux (q)*

The flux per unit length normal to the plane of flow of the groundwater can be calculated using the result of the stream function (Rushton & Redshaw 1979), i.e.

$$q = \int_{y_1}^{y_2} v_x \, dy \tag{23.34a}$$

$$= \int_{\Psi_1}^{\Psi_2} d\Psi = \Psi_2 - \Psi_1 \tag{23.34b}$$

From Equations (23.20) and (23.34b), the flux between two points of the stream values can be calculated.

23.2.8 *Programming, determination of the sediment surface, water table and free surface water*

The upper saturated soil boundary consists of the soil surface when the aquifer is fully saturated, and the water table when it is partially saturated. The approximation of the saturated boundary is a series of three saline segments. The creek bank is approximated using a cubic, and a linear function is used for the upstream section. The function from the creek bank to the (linear) forest sediment is approximated using a quadratic. In this model, the water level in the creek is assumed to be zero or $h_2 = 0$.

23.2.9 *Steps in the program*

The programming tool used in this modeling is MATLAB®. The following steps are used in the model. First, three polynomials are used to represent the true value of sediment surface as measured with field data. The same approximation is used for the field data, when a water table is present. A_n was calculated from:

- Calculation of $\int_0^s u_i^t(x) u_j^t(x) dx$ (from Equation (23.28))
- Calculation of $\int_0^s u_i(x) h^t(x) dx$ (from Equation (23.31))
- Calculation of A_n, $u^t \underset{\sim}{a} = \underset{\sim}{h}^t$.

when A_n has been found, the calculation of $\Phi(x, y)$ and $\Psi(x, y)$ proceeds.

Both the water table, $h^t(x)$, and hydraulic potential of the groundwater, $\Phi(x, y)$, are plotted together along the saturated boundary. Basically, both functions must have similar values. However, because the value of n for the calculation is not infinite, there must be an error for this calculation. N is chosen, so that the rms error is less than 10^{-3}. This ensures three figures accuracy in the solution for the stream function. The flux of the groundwater of this model was calculated using Equation (23.34b). The results of this model calculation are compared to the flux of the groundwater from the field measurements.

23.3 RESULTS

The model was applied to calculate the groundwater flow at the field area. The distance between the impermeable salt flat to the creek is 11 m, and the slope of the creek bank is 45°. The elevation of the sediment at the salt flat is 1.1 m and 1 m at the creek bank. The slope of the mangrove sediment model is 0.6° or 0.01. Calculations were performed for neap tides i.e. when there was no

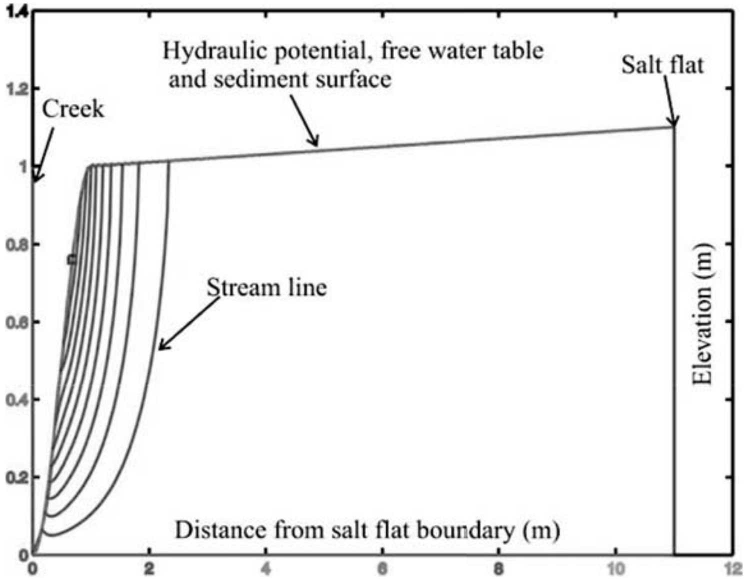

Figure 23.2. The condition of the sediment, free water table, and stream line when the sediment is water saturated or day zero.

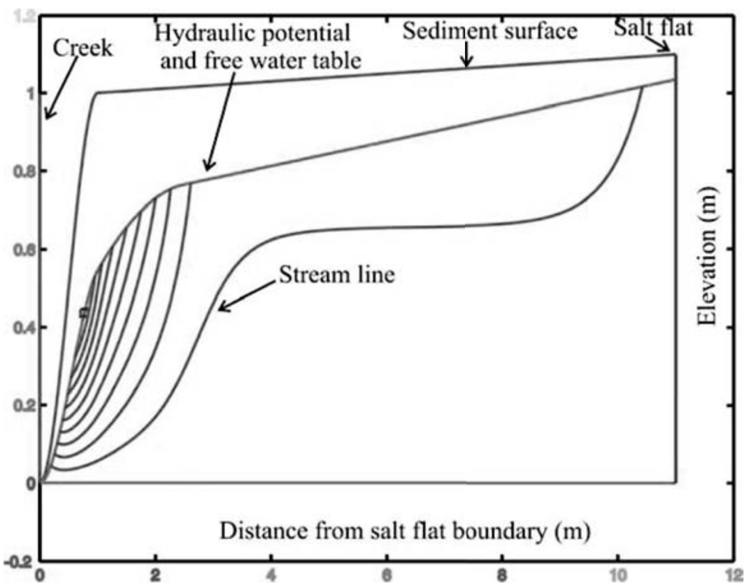

Figure 23.3. The condition of the sediment, free water table and stream line, when one day after inundation.

tidal inundation of the swamps. From the piezometer data, the position of the free water table is known over this period and was used as an input to the model.

Since the model is not time dependent it is not possible to calculate the evolution of the free water table and water fluxes over the period of the neap tides. It is however possible to calculate the fluxes at particular times, provided that the position of the water table is known. The neap tide period, which will be defined as the period when no inundation of the mangroves occurs, usually

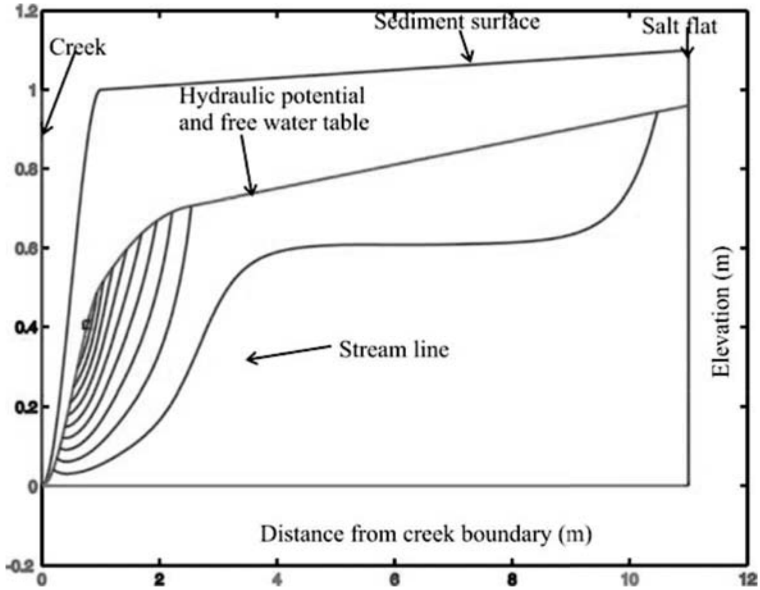

Figure 23.4. The condition of the sediment, free water table and stream line, on two days after inundation.

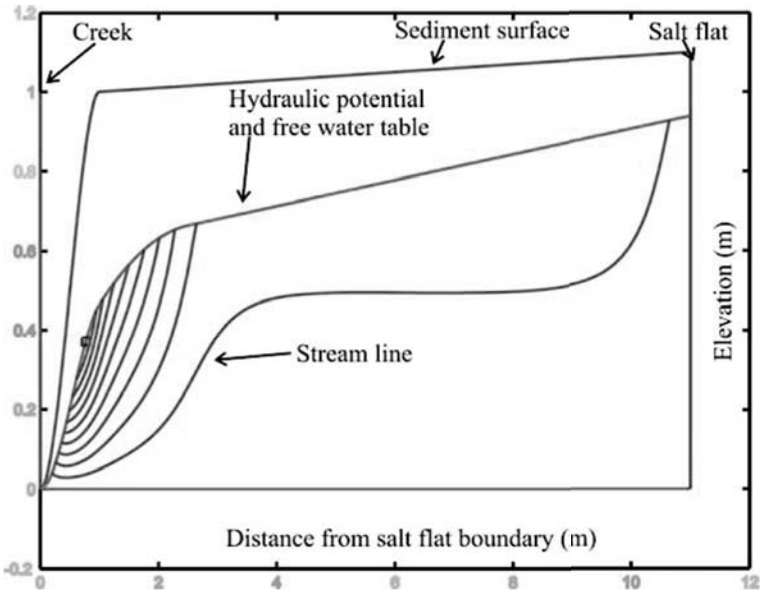

Figure 23.5. The condition of the sediment, free water table and stream line three days after inundation.

lasts for 5 days. In the analysis the fluxes are calculated on each of the 5-day neap tide period. Since the water table during spring tides is effectively at the sediment surface, the model can also predict the spring tide situation. The net flux of groundwater from the sediment is calculated by numerically integrating the flux of water passing into the creek. Therefore, the model of six days after inundation is similar to the day zero, i.e. the sediment is water saturated.

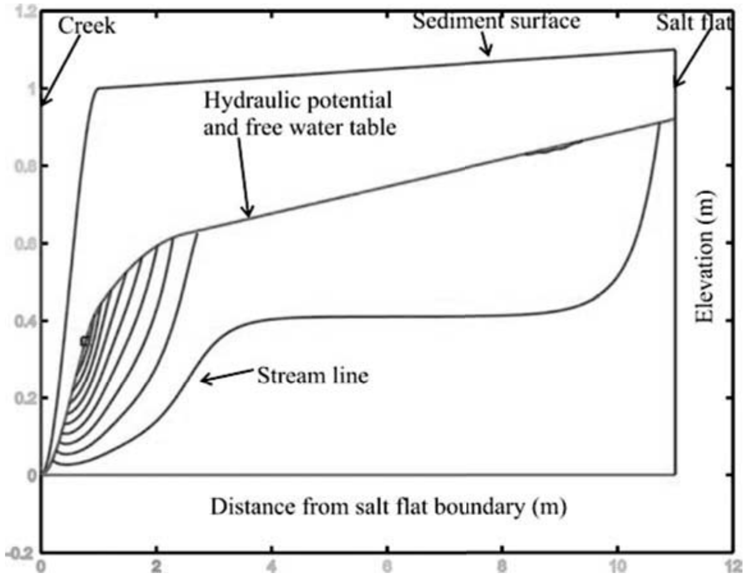

Figure 23.6. The condition of the sediment, free water table and stream line four days after inundation.

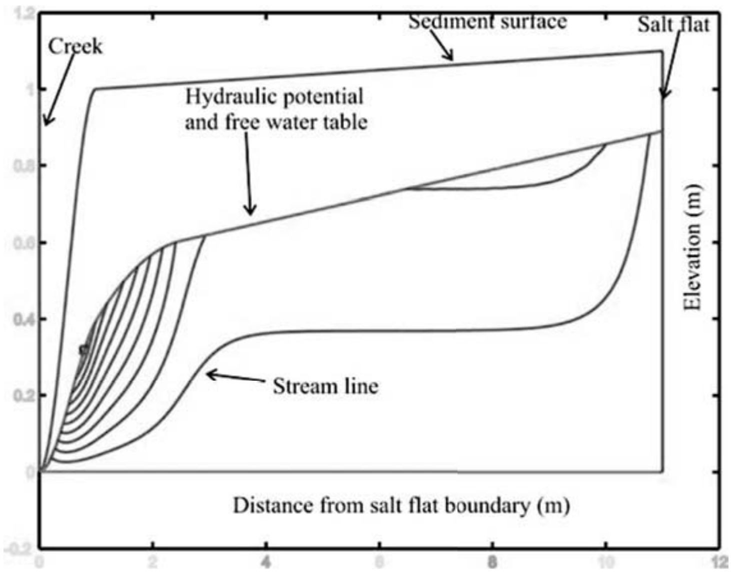

Figure 23.7. The condition of the sediment, free water table and stream line, on five days after inundation.

Figure 23.2 shows the results of the stream function immediately after the last tidal inundation of the spring tides, day 0, i.e. when the free water table was at the sediment surface. It should be noted that the results shown in Figure 23.2 are also applicable for spring tides as the time of inundation is generally only a few hours per day even during spring tides. The water table for the rest of the period is at the sediment surface. Using piezometer data from successive days, the stream lines were calculated from day 1 to 5, after the last tidal inundation. These results are shown in Figures 23.3 to 23.7.

Table 23.1. The comparison between the results of the flux calculated from the model (stream function) and those from the field measurements.

Day	Flux calculated by model	Flux inferred from piezometer data
0	0.026 ± 0.002	0.030 ± 0.021
1	0.029 ± 0.002	0.028 ± 0.017
2	0.021 ± 0.002	0.026 ± 0.013
3	0.016 ± 0.001	0.021 ± 0.010
4	0.012 ± 0.001	0.016 ± 0.008
5	0.007 ± 0.001	0.013 ± 0.005

Figure 23.3 shows the condition of the model one day after inundation, where the slope of the water table increases from 0.01 on the day zero to 0.015 on the day one. The water table level is down by 0.065 m at the beginning of the sediment to 1.035 m at the salt flat.

Figure 23.4 shows the condition of the model two days after inundation (day two), where the slope of the water table increases from 0.015 on day one to 0.02 on day two. The water table level is down by 0.14 m from day zero at the beginning of the sediment to 0.96 m at the salt flat.

Figure 23.5 shows the condition of the model three days after inundation (day three), where the slope of the water table increases from 0.02 on day two to 0.025 on day three. The water table level is down by 0.16 m from day zero at the beginning of the sediment to 0.94 m at the salt flat.

The condition of the model after four days of inundation is shown in Figure 23.6, where the slope of the water table increases from 0.025 on day three to 0.03 on day four. The water table level is reduced by 0.19 m from day zero at the beginning of the sediment to 0.91 m at the salt flat.

Figure 23.7 shows the condition of the model five days after inundation (day five), where the slope of the water table increases from 0.03 on day three to 0.04 on day five. The water table level is down by 0.26 m from day zero at the beginning of the sediment, which is 0.84 m.

It was found from another experiment (Susilo 2004) that the porosity of the sediment was *ca*. 0.45 (45%), and the water that filled the burrows was *ca.* 0.1 (10%) from the total volume of the sediment (Stieglitz et al. 2000). Therefore, the water volume was 55% of the total volume of the sediment. The calculation of the fluxes for all conditions is multiplied by 0.55. The fluxes calculated for each of the 6 days are shown in Table 23.1. It can be seen from Table 23.1 that the model gives a reliable estimate of the groundwater flux.

23.4 CONCLUSION

The success of the model is in part due to the extensive use of high quality water table data. Although in many applications such data may not be available, due to the absence of piezometers, calculations can still be performed in periods when the tides inundate the creek, as this is when the water table is similar to the sediment surface. It is a relatively simple matter to survey the height of the sediment surface, and thus the model can be applied to many locations with a minimum of effort.

The model is applicable to spring tides as well as neap tides in many locations. In most places, because the average period of inundation during the spring tides is only a few hours per day. For this short time groundwater flow will be zero as there will be no significant pressure gradient. For the rest of the day the water table will be at the sediment surface and the results for Figure 23.2 apply.

It should be noted that this model is limited because it is not a genuine time dependent model, however useful results of groundwater fluxes can still be obtained and used in a variety of applications.

ACKNOWLEDGEMENTS

The author acknowledges the thoughtful comments of the anonymous reviewers and especially Dr. A.B. Mukherjee and Dr. AL. Ramanathan for the final editorial handling of the manuscript.

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Section V
Arsenic and fluoride in groundwater

CHAPTER 24

Groundwater arsenic contamination and its health effects in the Ganga-Meghna-Brahmaputra plain

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ABSTRACT: From last 20 years survey in the Ganga-Meghna-Brahamaputra plain (area 569749 sq km; population >500 million) analyzing around 2,11,000 water samples we have found, a good portion of the states and countries (Uttar Pradesh, Bihar, Jharkhand, West Bengal, Assam and Bangladesh) is arsenic affected (>50 µg/L). Screening more than 1,50,000 people from affected villages about 15000 patients were registered with different kinds of arsenical skin lesions. Arsenic neuropathy as well as adverse pregnancy outcomes such as spontaneous abortion, still birth, preterm birth and low birth weight was recorded. Infants and children drinking arsenic contaminated water are at high risk. About 50,000 biological samples analysis from arsenic affected areas showed elevated level of arsenic in both patients and non-patients indicating many are sub clinically affected. Proper watershed management, economical utilization of available surface water, and the participation of all who live in or have influence over the region are needed for mitigating the situation.

24.1 INTRODUCTION

In recent past years there is an emerging threat of groundwater arsenic contamination in Asian countries (China Conference 2004). In 1976 arsenic contamination in Chandigarh was reported (Datta et al. 1976). In 1984 the groundwater arsenic contamination in lower Ganga Plain of West Bengal was first reported (Garai et al. 1984). In 1992 we identified arsenic groundwater contamination in Padma-Meghna-Brahmaputra (PMB) plain of Bangladesh where people were drinking arsenic contaminated water and suffering from arsenical skin lesions (Dhar et al. 1997). Only in 1995 the arsenic situation in West Bengal and consequent suffering of people came to limelight (Kolkata Arsenic Conference 1995). In 2001 groundwater arsenic contamination in the Terai region of Nepal was revealed (Shrestha et al. 2003, Bhattacharya et al. 2003, Tandukar et al. 2006). In June 2002 we discovered arsenic contamination in Bihar in middle Ganga plain and apprehended contamination in Uttar Pradesh lying in middle and upper Ganga plain (Chakraborti et al. 2003). Between October 2003 and December 2003, we identified 25 arsenic affected villages of Ballia district in UP and people suffering from skin lesions. Further during our surveys between December 2003 and

Table 24.1. Important years related to groundwater arsenic contamination incidents in GMB plain.

Year	Place	Reference
1976	Chandigarh, North-India	Datta et al. 1976
1984	West Bengal, India	Garai et al. 1984
1992	Bangladesh	Dhar et al. 1997
1995	West Bengal incident came to limelight	International Conference 1995
1998	Bangladesh incident came to limelight	International Conference 1998
2001	Nepal	Shrestha et al. 2003
2002	Bihar, India	Chakraborti et al. 2003
Oct 2003–Dec 2003	Uttar Pradesh, India	Chakraborti et al. 2004
Dec 2003–Jan 2004	Jharkhand, India	Chakraborti et al. 2004
Jan 2004–Feb 2004	Assam, India	Chakraborti et al. 2004
March 2006	Manipur, India	Singh et al. 2007

Table 24.2. Groundwater arsenic contamination in states and countries of the GMB plain.

Parameters	West Bengal	Bangladesh	Bihar	Uttar Pradesh	Jharkhand	Assam
Area (km^2)	38,861	147620	94163	238,000	75,834	78,438
Population (million)	80.1	201	83	166	27	26.6
Total arsenic affected districts	9	50	12	3	1	2
Total arsenic affected blocks/P.S.	108	178	32	9	4	2
No. of villages where groundwater contains arsenic >50 µg/L	3500	2000	107	91	7	27
Total hand tubewell samples analyzed	1,40,150	50808	14530	4780	1024	241
% samples having arsenic >10 µg/L	48.7	43	32.9	46.0	30	42.3
% of samples having arsenic >50 µg/L	23.8	31	18.6	27.7	19.4	19.1
Total number of biological samples analyzed	30000	10000	920	178	–	*
Total people screened by medical group of SOES-JU	96000	18841	4513	989	212	*
People registered with arsenical skin lesions	10000	3725	525	153	87	*
People may drink contaminated water >10 µg/L (million)	9	52	**	**	**	**
People may drink contaminated water >50 µg/L (million)	7	32	**	**	**	**

* Not done yet. ** Not yet estimated.

January 2004, we found groundwater arsenic contamination in 698 hand tubewells from 17 villages of the Sahibgunj district of Jharkhand state, India in the middle Ganga plain and consequent suffering of hundreds of people. Again a preliminary survey during January–February 2004 in Assam showed 26% of 137 hand tubewells analyzed in 2 districts had arsenic concentration above 50 µg/L. The important incidents related to groundwater arsenic situation in Ganga-Meghna-Brahmaputra (GMB) Plain are shown in Table 24.1.

According to our latest estimates, a good portion of all the states and countries in the Ganga-Meghna-Brahmaputra (GMB) plain may be at risk from groundwater arsenic contamination

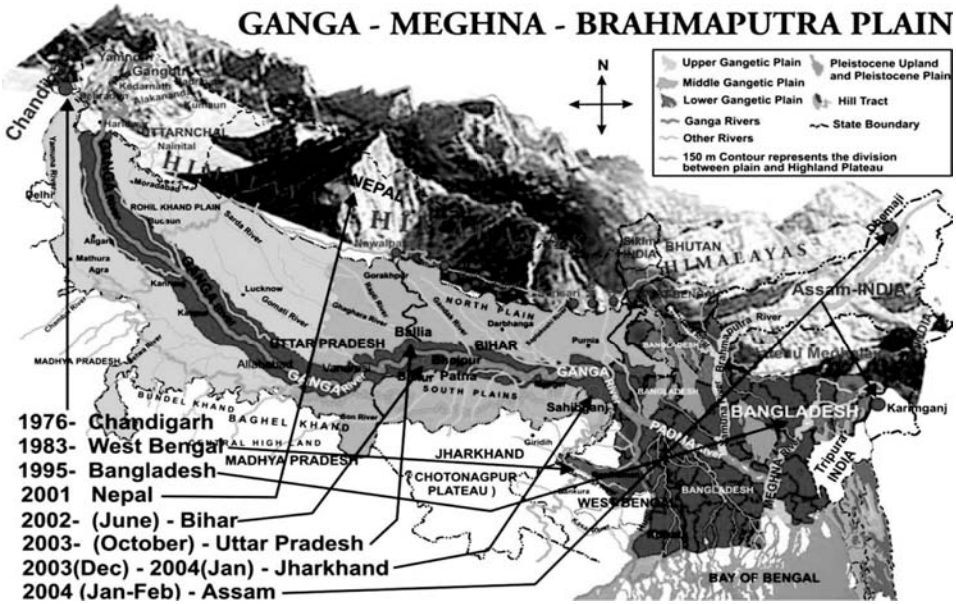

Figure 24.1. Groundwater arsenic contamination in the GMB plain.

(Chakraborti et al. 2004). Table 24.2 and Figure 24.1 show the groundwater arsenic contamination status in different states and countries in GMB Plain.

24.2 GROUNDWATER ARSENIC CONTAMINATION IN STATES AND COUNTRIES OF THE GMB PLAIN

24.2.1 *West Bengal, India*

We began our survey of the villages of West Bengal in 1988. At that time we knew of only 22 affected villages in 12 blocks of 5 districts. In the subsequent years, with every additional survey, we found an increasing number of contaminated villages and affected people. The contamination scenario would be evident from Table 24.2. Even after 18 years of survey we feel we have seen the tip of the iceberg and with every new survey we are discovering newer affected villages.

West Bengal has total 19 districts. We have analyzed 1,40,150 hand tubewell water samples from the nine arsenic affected districts of west Bengal. Besides the nine arsenic affected districts of West Bengal including Kolkata we have also analyzed 2923 hand tubewell water samples from the districts situated in the northern part of the West Bengal as East and West Dinajpur, Jalpaiguri, Darjeeling and Cooch Bihar. In these districts use of groundwater for irrigation and drinking has started recently. The analysis results of water samples from northern part of West Bengal do not show arsenic above 50 µg/L (except a few) but 10–15% of the samples contain arsenic between 10 and 50 µg/L indicating the possibility of future arsenic contamination. We have also analyzed 1123 hand tubewells from the districts situated in the western and southwestern parts of West Bengal-Birbhum, Bankura, Purulia and East and West Medinipur districts. They do not show arsenic contamination in groundwater above 3 µg/L. But in some parts of these districts, fluoride contamination much above the permissible limit has been detected in groundwater and patients have already been identified suffering from dental and skeletal fluorosis. There is no universal guideline value for fluoride in drinking water, but WHO set up a permissible upper limit at 1.5 mg/L (WHO 2006). For India the permissible upper limit was lowered from 1.5 mg/L to 1.0 mg/L (BIS 1990).

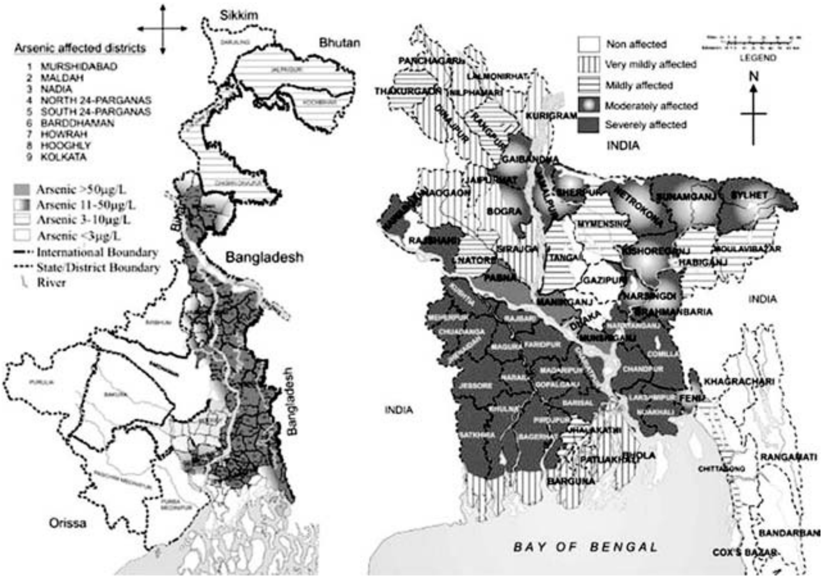

Figure 24.2. Groundwater arsenic contamination status in West Bengal-India and Bangladesh.

The incidence of fluoride contamination and its consequential health effects in India have been reported (UNICEF 1999).

To know the magnitude of groundwater arsenic contamination and health effects in West Bengal, we have studied district Murshidabad in detail for five years (Rahman et al. 2005a, Mukherjee et al. 2005). We have also studied one block (Rahman et al. 2005b) and carried out semi micro level and micro level study in one gram Panchayet (Rahman et al. 2005c) and one village of Murshidabad (Rahman et al. 2005d).

24.2.2 *Bangladesh*

In our last eight years field survey in Bangladesh (Table 24.2), we had collected and analyzed 50808 hand tube-well water samples from all 64 Districts. We had collected hand tube-well water samples covering four principal geo-morphological regions of Bangladesh: Table land, Flood plain, Deltaic region and the Hill tract. From our results, it appears that Hill Tract region and Table land are arsenic contamination free. The areas of Flood Plain and Deltaic region including coastal region are highly arsenic contaminated. During our study we noticed that in fringe area of Table Land with Flood Plain; Hill Tract with Flood Plain and if rivers of Flood Plain have eroded Table land and Hill tract areas, some contamination are there (Chakraborti et al. 1999). Deposition of Holocene sediments in the Bengal delta is considered to be source of arsenic.

24.2.3 *Bihar, India*

In June 2002, we had reported on groundwater arsenic contamination in the Bhojpur district of Bihar in the Middle Ganga Plain but it was refuted. With ongoing study we are finding more and more contaminated districts (Table 24.2). A detailed study is needed to understand the arsenic contamination situation in groundwater of Bihar. We predict from our up-to-date preliminary survey from Bihar that the districts lying in the area where Ganga and other tributaries originating from the Himalaya shifted in course of time, would be arsenic contaminated. The areas of Bihar, adjacent to arsenic contaminated Terai region, Nepal may also be affected.

24.2.4 *Uttar Pradesh (UP), India*

We apprehended groundwater arsenic contamination in UP in 2003 (Chakraborti et al. 2003). Later with our continuing study we are finding newer contaminated sites from UP. We have already identified contamination in three districts of UP: Ballia, Gazipur and Varanasi. Here also arsenic contamination was denied by the administration (Gupta 2004). A thorough survey is required to understand the magnitude of arsenic contamination in UP. We fear the areas of UP adjacent to arsenic contaminated Terai region, Nepal may also be affected.

24.2.5 *Jharkhand, India*

During Dec 2003–January 2004 we have found groundwater arsenic contamination in the Sahibganj district of the Jharkhand state, in the middle Ganga plain (Chakraborti et al. 2004). We noticed arsenic contamination is close to the Ganga river or in those areas from where the Ganga river shifted during recent past. We also expect arsenic groundwater contamination in the areas of Jharkhand adjacent to arsenic affected Malda district of West Bengal like Taljhari, Borio in Sahibgunj district. We have also noticed many dug-well and hand tube-wells away from the Ganga plain being contaminated with fluoride as in Berhait block of Sahibgunj district.

24.2.6 *Assam, India*

We have already reported that the old Brahmaputra plain of Bangladesh is arsenic contaminated. However in January 2004, for the first time we also detected arsenic contamination in the groundwater of the upper Brahmaputra plain in Assam (Chakraborti et al. 2004). We have analyzed 137 hand tubewell water samples from Dhemaji and Karimganj districts of Assam. The results show that 43% of the samples contain arsenic above 10 μg/L and 26% above 50 μg/L. The maximum concentration detected in a hand tubewell was 490 μg/L. We may also expect groundwater arsenic contamination in these North eastern hill states where the rivers originating from the Himalayas are flowing.

24.3 IRON IN GROUNDWATER IN THE GMB PLAIN

Iron concentration in groundwater in all the states and countries in the GMB plain is quite high ($n = 21045$, mean $= 3751.8$ μg/L, median $= 3110$ μg/L, mode $= 1180$ μg/L minimum 40 μg/L and maximum 80,346 μg/L). However Assam appears to be most contaminated with respect to iron.

24.4 CLINICAL EFFECTS OF GROUNDWATER ARSENIC CONTAMINATION IN THE GMB PLAIN

24.4.1 *Arsenical skin lesions*

Over many years of study in this region we have conducted preliminary surveys to establish how many people have arsenical skin lesions in the affected villages. We observed various types of skin manifestations such as melanosis, leucomelanosis, keratosis, hyperkeratosis, dorsal keratosis, non-pitting oedema (Rahman et al. 2001, Chakraborti et al. 2003, Chakraborti et al. 2004). We also noticed gangrene, Bowens and skin cancer (Squamous and Basal cell carcinoma) among those having arsenical skin lesions. Internal cancers such as bladder cancer, lung cancer, liver cancer etc were also noticed among those who had arsenical skin lesions. We do not expect such a high percentage of arsenocosis patients in all the arsenic affected districts (Table 24.2). The large number of people showing arsenical skin-lesions is due to the fact that we had examined villagers from only those villages, which are highly arsenic contaminated and we had prior information of the presence of arsenic patients. Undoubtedly the overall percentage of arsenic affected people is expected to be lower in the less contaminated areas. Figure 24.3 shows arsenic affected people from each arsenic affected state from India (except Assam) and Bangladesh.

24.4.2 *Arsenical neuropathy*

Neurological examination was generally undertaken for arsenocosis patients whose skin lesions were already diagnosed by experienced dermatologist. The neurological part was conducted by the same experienced neurologist to obviate inter-observer variability for each patient of arsenocosis so tested. Observations were recorded for items considered consistent with peripheral motor and sensory neuropathy and for other neurologic abnormalities as well. Pain history and pain-specific sensory examination were stressed. The items included to characterize neuropathy were (i) pain and paraesthesias (e.g., burning) in a stocking and glove distribution, (ii) numbness, (iii) hyperpathia/allodynia, (iv) distal hypesthesias (reduced perception of sensation to pinprick/reduced or absent vibratory perception/affected joint-position sensation/affected touch sensation), (v) calf tenderness, (vi) weakness/atrophy of distal limb muscles or gait disorder, and (vii) reduction or absence of tendonflexes. Overall prevalence of clinical neuropathy were noted in our studies (Rahman et al. 2001, Mukherjee et al. 2003, Chakraborti et al. 2003, Chakraborti et al. 2004) in populations of Murshidabad and Nadia districts of West Bengal, Bhojpur district of Bihar and Ballia district of UP, India or Comilla district of Bangladesh (Ahamed et al. 2005 communicated).

24.4.3 *Arsenic in drinking water and obstetric outcome*

Arsenic exposure during pregnancy can adversely affect several reproductive endpoints. Several studies have examined the association between arsenic exposure and adverse pregnancy outcome, including spontaneous abortion, preterm birth, stillbirths, low birth weight and neonatal and

Figure 24.3. Differing kinds of arsenical skin lesions.

perinatal mortality (Chakraborti et al. 2003, 2004). All these parameters were compared to those observed in the control women group from a non-arsenic exposed district (Medinipur) of West Bengal. Adverse obstetric effects were observed in our studies with village women from Murshidabad district, West Bengal (Mukherjee et al. 2005), Bhojpur district of Bihar (Chakraborti et al. 2003), Ballia district UP or Comilla district, Bangladesh.

24.5 SUBCLINICAL EFFECTS OF ARSENIC

Arsenic concentration in hair and nail plays an important role in evaluating the arsenic body burden (NRC 1993). We measured inorganic arsenic and its metabolites in urine and total arsenic in hair, nail and skin scale. High arsenic concentration in urine is indicator of recent arsenic exposure. In arsenic affected villages of Bihar, UP, Jharkhand high arsenic in urine of both patients and non patients indicate they were drinking contaminated water (Chowdhury et al. 2003, Chakraborti et al. 2004). It appears from the analyses that many villagers may not be suffering from arsenical skin lesions but have elevated levels of arsenic in their hair and nail, and thus may be sub-clinically affected. The high arsenic content in the biological samples from the affected regions also reveals continued exposure to arsenic contaminated drinking water in vast population of the GMB plain.

24.6 ARSENIC AFFECTED CHILDREN

Infants and children are considered to be more susceptible to the adverse effect of toxic substances than adults. Our last 18 years field experience in West Bengal and 8 years in Bangladesh, we observed that normally children under 11 years of age do not show arsenical skin lesions although their biological samples contain high level of arsenic. However, we have observed exceptions as when (i) arsenic content in water consumed by children is very high ($\geq$1000 μg/L) and (ii) arsenic content in drinking water is not so high (around 500 μg/L) but the children's nutrition is poor (Rahman et al. 2001). High arsenic content in their biological samples prove that children in the arsenic affected areas of the GMB plain have a higher body burden, though dermatological manifestations are few (Chakraborti et al. 2004).

Figure 24.4. Future generation at risk? A group of arsenic affected children (60% of them had arsenical skin lesions) from Madanpur village, GP Akhriganj, Block Bhagabangola, District Murshidabad, West Bengal, India.

Table 24.3. Arsenic concentration in rice from different countries.

Country	No. of samples	Mean Arsenic concentration (µg/gm)	References
Bangladesh	15	0.13	Williams (2005)
China	11	0.49	Xia (1998)
Taiwan	280	0.10	Lin (2004)
United States	7	0.26	Williams (2005)
Vietnam	31	0.21	Phuong (1999)
India	6	0.21	Roy Chowdhury (2002)
West Bengal, India (from contaminated area)	100	0.18	Our unpublished data
West Bengal, India (from uncontaminated area)	55	0.08	Our unpublished data

Table 24.4. Arsenic concentration in vegetables from India (Chowdhury 2001) and Bangladesh (Huq 2006).

	Arsenic concentration in different countries (µg/kg)			
	Bangladesh		India	
Vegetables	Uncontaminated	Contaminated	Uncontaminated*	Contaminated
Green papaya	221–460	40–2220	32	117*
Arum	77–387	130–153200	60	692 ± 26
Bean	92	130–1160	9	87*
Long bean	300	370–2830		
Potato	620	710–2430	45	294*
Bitter ground	1560	2120		19 ± 2
Aubergine	230	2300		
Chilli	410	1520		
Egg plant			33	443 ± 33
Ladies finger			61	65 ± 8

* Our unpublished data.

24.7 ARSENIC IN THE FOOD CHAIN

Arsenic contaminated hand tubewells are used by the villagers not only for drinking and cooking but also for the irrigation of their fields. In arsenic affected areas ground water is the major source of water for agriculture and many of such sources are arsenic contaminated. The total arsenic concentration in rice and vegetables from some countries are presented in Table 24.3 and Table 24.4 respectively.

24.8 SOURCE AND MECHANISM OF ARSENIC CONTAMINATION

From the arsenic contamination scenario in Asia it appears that the flood plains of many rivers originating from the Himalayan Mountains and the Tibetan plateau are affected (Chakraborti et al. 2004). On this basis we noticed arsenic contamination in West Bengal, Bihar, Jharkhand, UP in the Gangetic plain, Brahmaputra plain in Assam and PMB (Padma-Meghna-Brahmaputra) plain in Bangladesh (see Fig. 24.5). The source of arsenic is geogenic. Various theories have been postulated on sources of arsenic and mechanism of mobilization from the source (Das et al. 1996, Bhattacharya et al. 1997, Nickson et al. 1998, Roy Chowdhury et al. 1999, Harvey et al. 2002, Akai et al. 2004, Islam et al. 2004). The exact nature of mobilization process is still unknown.

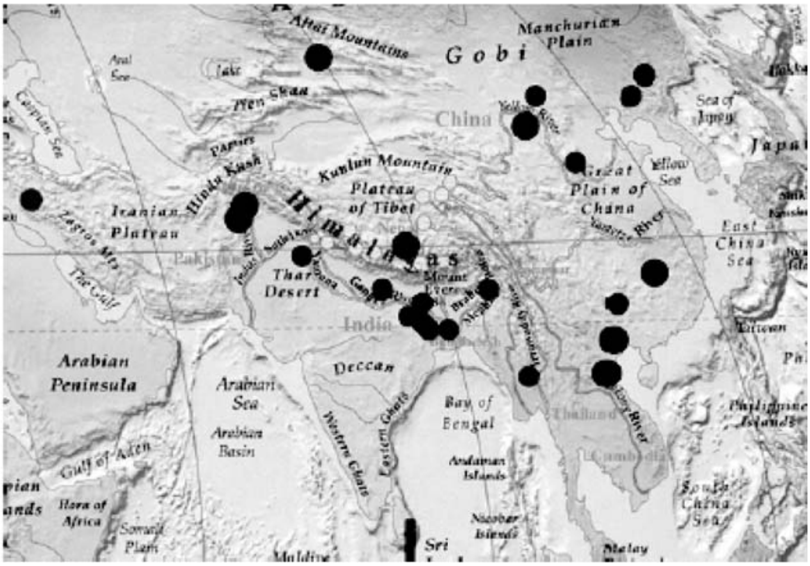

Figure 24.5. Arsenic affected floodplains of the rivers originating from Himalaya and Tibet plateau.

24.9 AN APPROACH TO SOLVE GROUNDWATER ARSENIC PROBLEM

From our 20 years long field experience in arsenic affected areas of the GMB plain we have realized that arsenic mitigation strategy should be location specific. A method suitable for a specific area may not be generalized for the other affected regions due to a) geographical and geomorphological variations, b) differing socio-economic and literacy conditions of people. But whatever be the approach, for success at field level we need awareness amongst the people and their wholehearted participation. To combat the present arsenic crisis in the GMB basin we urgently need to consider the following actions.

It is observed that in the Gangetic plain arsenic contamination in hand tube wells has been observed to decrease after a certain depth but in unconfined aquifers there appears to be no depth guarantee, even if the construction of tube well is done properly. Based on our eighteen year long study over different parts of the GMB plain on groundwater arsenic contamination analyzing more than 2000 deep tubewells (>150 m), (Chakraborti et al. 1999) we observed that deep tubewells (>150 m) could be contamination free (as per WHO recommended value).

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Out of 1538 tubewells studied at two different time points in 75 villages of West Bengal, 1031 (67%) showed increase in arsenic concentration with time (4–10 years). A certain portion of tubewells in the arsenic affected regions may undergo temporal increase in concentrations. Therefore regular monitoring of arsenic concentrations in “safe” tubewells becomes imperative.

Analyzing 50,808 hand tubewell water samples from four geomorphological regions of Bangladesh we observed that Hill Tract region and Table land are arsenic contamination free

Figure 24.6. Examples of safe water options (surface water and properly managed dugwell) in GMB plain.

and can be safe sources. Alternative safe water sources such as abundant surface water in the GMB plain (wetland, oxbow lakes, and flooded river basins), arsenic-safe dugwells (Fig. 24.6), rainwater harvesting after proper treatment against waterborne bacterial contaminants and other toxins may serve as alternate safe water options. Community participation in ensuing proper running and maintenance of the arsenic removal plants (ARP) in the affected areas may be another option for providing arsenic safe drinking water.

The villagers in the affected areas need to be educated about i) the existence of the problem and ii) the signs and symptoms of arsenic toxicity. There is no medicine to cure chronic arsenic toxicity. Safe water, nutritious foods are only the proven measures to fight against chronic arsenic toxicity. Better nutrition may be obtained from low cost seasonal fruits and vegetables which are available aplenty in the GMB plain. Importance of community involvement with special emphasis to women should be stressed in developing any strategy to fight the menace.

24.10 CONCLUSION

Though first case of arsenocosis was revealed in West Bengal in early 1980s the widespread contamination was not recognized until 1995. Similar pattern followed in the late recognition of groundwater arsenic contamination of Bangladesh. In Bihar, till date we found 12 districts by the side of Ganga arsenic contaminated and in 6 districts identified subjects with arsenical skin lesions since the discovery of arsenic contamination back in 2002 and more are coming to fore with the continuing surveys. We predict from our up-to-date preliminary survey from UP and Bihar that the districts lying in the area where Ganga and other tributaries originating from the Himalaya shifted in course of time, would be arsenic contaminated. The areas of UP and Bihar, adjacent to arsenic contaminated Terai region, Nepal may also be affected.

In India before arsenic contamination problem surfaced in 1983, we knew about fluoride contamination in groundwater from 1937. At present only in India 62 million people are suffering from fluorosis, a crippling disease. The presence of uranium, boron, and manganese in groundwater of Bangladesh above WHO prescribed limiting values has already been reported (van Geen et al. 2005, BGS/DPHE 2001). Though small amount of boron is beneficial for persons with certain conditions such as arthritis, excessive consumption (>15 mg B/kg body wt daily) can adversely affect growth, reproduction or survival (EPD 2003). Manganese may cause adverse neurological effect due to extended exposure to very high level in drinking water (WHO 2006a). There is enough scientific evidence about the occurrence of nephritis (Hurth and Spoor 1973, WHO 2006b) due to excess amount of uranium exposure. Unless immediate measures for detailed water analysis are undertaken and awareness about contaminants in drinking water is generated, toxins of higher toxicity may affect in course of time.

In Bangladesh and West Bengal, at present less people are drinking arsenic contaminated water due to growing awareness and access to arsenic safe water. But in Bihar, UP, Jharkhand, and Assam still the villagers are drinking contaminated water owing to non recognition of arsenic contamination as a problem requiring urgent action. The blunder committed in West Bengal and Bangladesh before should not be repeated.

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CHAPTER 25

Initial data on arsenic in groundwater and development of a state action plan, Uttar Pradesh, India

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ABSTRACT: Several studies have revealed presence of arsenic in groundwater in parts of Uttar Pradesh in India where this problem was hitherto unknown. In particular, this paper reports results of screening testing from two districts; Ballia and Lakhimpur Kheri. This has revealed that 350 villages in 18 blocks are affected by arsenic in groundwater above the prevailing upper limit of 50 µg/L. Blanket testing in the affected areas and mapping of the arsenic distribution with Geographic Information System software is being undertaken to further clarify the scale of the problem. A communication effort will be undertaken in parallel with blanket testing to inform communities living in affected areas of the implications of test results and potential mitigation strategies. Well-switching, i.e. use of arsenic-safe handpumps in preference to arsenic-contaminated ones, is one option available. Source-switching to alternative water sources such rainwater, deep groundwater and very shallow groundwater will be piloted in some areas. Several deep handpumps have already been installed on a trial basis and preliminary data suggests these will provide drinking water with substantially reduced arsenic content. The full range of water quality parameters will be tested to determine the potential for wider use of deeper groundwater for drinking in this area. The results of screening testing in these two districts have informed development of a large scale project to screen for arsenic in priority blocks of the state which is also detailed here.

25.1 INTRODUCTION

Although exact published figures are not available, in Uttar Pradesh (UP) in India, the predominant water source used for drinking and cooking is undoubtedly groundwater. The 2005 'Rapid Assessment of Rural Water Supply and Sanitation Sector in UP' conducted by the Centre for Symbiosis of Technology, Environment and Management (STEM) indicates that 83.9% of rural households are using either 'tubewell', 'handpump' or 'well' water (i.e. groundwater sources) and 15.8% are using 'taps' where the water source is unspecified, although likely to be largely groundwater based. One percent is reported to be using 'other' sources described as 'tubewell, tank, pond, lake, river, canal spring, etc' i.e. largely surface water sources.

Following identification of cases of 'arsenicosis' in Chandigarh (Datta 1976) a limited number of drinking water samples (20) from Uttar Pradesh were tested for arsenic by Datta and Kaul (1976). Four samples from 'wells' were found to contain arsenic <50 µg/L, however, of 16 samples from 'handpumps' 8 were found to contain arsenic >50 µg/L and 3 contained arsenic >100 µg/L. A maximum concentration of 545 µg/L was recorded from UP. The location of the sampling sites is given only as 'Meerut district'. Unfortunately these findings were not taken further by the administration of the time and to our knowledge no further investigations were taken up.

Following reports of arsenic in bordering districts of Nepal arsenic testing was started in UP by the Shriram Institute of Industrial Research (SIIR) in Delhi with assistance from UNICEF in 2003. 3390 samples were taken from 10 districts and analyzed for arsenic by Atomic Absorption

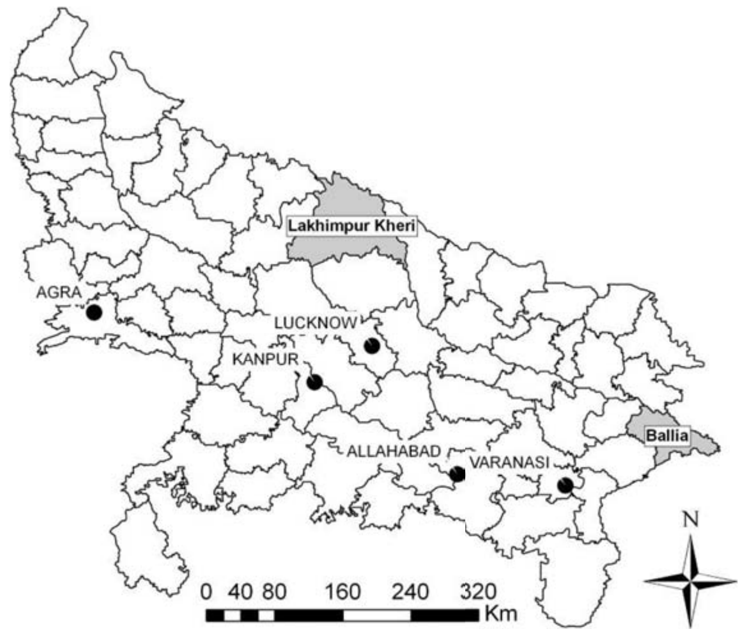

Figure 25.1. Location of Ballia and Lakhimpur Kheri districts in state of Uttar Pradesh in India.

Spectrophotometry (AAS). 4.3% of the samples collected in this study were >10 μg/L and only 0.1% (3 samples) >50 μg/L (SIIR 2004). In 2004 researchers from the School of Environmental Studies-Jadavpur University, Kolkata (SOES-JU) visited Ballia district in the east of the state and took 914 samples from handpumps in 25 villages in 3 blocks of the district. After analysis by AAS they reported that 56% of these samples contained arsenic in concentrations >10 μg/L and 43% of the samples with >50 μg/L (SOES-JU 2004). UP Jal Nigam subsequently analyzed 52 samples from the affected areas and found 3 sources >50 μg/L with a maximum arsenic concentration of 102 μg/L. Continued investigation by SOES-JU in Ballia and the neighbouring districts of Gazipur and Varanasi have found that out of 3901 samples analysed 46.6% contain arsenic in concentrations >10 μg/L and 30.5% of the samples have >50 μg/L. 68 villages in 9 blocks of these 3 districts were identified as affected with arsenic in drinking water at levels >50 μg/L (SOES-JU 2005).

Upon reporting of the results of several studies in 2004 it was decided that more detailed investigation of the occurrence of arsenic in groundwater in Uttar Pradesh was required. Initially UP Jal Nigam with the assistance of the United Nations Children's Fund (UNICEF) begun a project in two districts, Ballia and Lakhimpur Kheri (Fig. 25.1), to in investigate and mitigate the problem (UP Jal Nigam 2004). The results of screening testing for arsenic in these two districts are given here, along with discussion of the findings, the ongoing work in two districts and details about how the initial phase has informed development of a large scale project to screen for arsenic in Uttar Pradesh and mitigate the problem where found.

25.2 METHODOLOGY

25.2.1 *Sampling strategy*

A simple strategy for selection of a sub-set of handpumps for arsenic testing that would result in a representative coverage of the two districts and give a true picture of the affected areas was required. It was decided by the UP 'Arsenic Task Force', chaired by the Chief Engineer (Rural) of UP Jal Nigam, that one government handpump should be tested in habitations with 1–10 government installed handpumps in total and 2 handpumps should be tested in habitations with >10 government installed handpumps. This generally amounts to around 10–20% of the total government installed handpumps. This sampling strategy was thought to be easy to implement, would give representative results, and was achievable given the supplies and human resources available.

25.2.2 *Testing protocol*

Development of a testing protocol for the state was one of the objectives of the project. Below is given the protocol which has been evolved.
Screening testing stage:

- Screening testing of around 10–20% of government handpumps with field test kit.
- Confirmation of results by Spectrophotometer testing in district or state level laboratory where arsenic found >40 μg/L by field test kit.
- Random checking of 5% of field test kit results (i.e. every 20th test) to validate accuracy of field testing process.
- Location of sources tested using GPS.

Blanket testing stage:

- Blanket testing with field test kits of all habitations where arsenic concentrations confirmed at >50 μg/L during screening testing. Where $>10\%$ of handpumps tested in a block are >50 μg/L then all handpumps of entire block to be blanket tested.
- Confirmation of results by Spectrophotometer testing in district or state level laboratory where arsenic found >40 μg/L by field test kit.
- Random checking of 5% of field test kit results (i.e. every 20th test) to validate accuracy of field testing process.
- Marking of handpumps: blue when field tested <40 μg/L then blue where field tests >40 μg/L are confirmed <50 μg/L by further laboratory testing. The Government of UP has ordered that all handpumps where laboratory tests confirm that groundwater contains arsenic >50 μg/L should be closed by dismantling the handpump.
- Location of sources tested using GPS.
- Communication with local people on dangers of arsenic consumption, health effects of arsenic poisoning, implications of test results and available and potential mitigation strategies.
- Further testing by Atomic Absorption Spectrophotometry or other more accurate methods of all relevant parameters in deeper groundwater sources to confirm acceptability of this source for drinking in a particular area.

25.3 METHODS

25.3.1 *Field testing*

Field tests for arsenic were carried out using arsenic field test kit based on the Gutzeit method developed by Dr. Nambiar formerly of the National Chemical Laboratories (NCL), Pune, India (NCL 2001). This uses two reagents, namely an oxalic acid/sodium hydroxide/ethylene diamine tetraacetic acid (EDTA) mixture and sodium borohydride, to reduce arsenic to arsine gas which is subsequently measured by a test strip coated with mercuric bromide and 4-Dimethylaminoazobenzene-4′-sulfonic acid sodium salt. The kit is manufactured in India by Chem-In Corporation, Pune.

25.3.2 *Spectrophotometer testing*

Arsenic is measured in district or state laboratories by reduction of dissolved arsenic to arsine gas using hydrochloric acid, potassium iodide, stannous chloride and zinc, and subsequent reaction of the arsine gas with silver diethyldithiocarbamate (SDDC) solution. This is based on the method published by the Public Health Engineering Dept. (PHED), Government of West Bengal (2003).

Color change in the SDDC solution is estimated using Merck Spectrophotometer Model UV1 with the following specifications: Photometric system: double Beam, Optics: Quartz coated MIG, Wavelength range: 190–1100 mm, Band Width: 2 nm, Wavelength Accuracy: ±1 nm, Wavelength Reproducibility: ±0.2 nm, Monochromator Drive speed: >6000 nm, Stray Light: +0.005A @ A = 1, Baseline Stability: 0.001A/hr, Baseline Flatness: ±0.002A, Baseline Correction: 2 stage, Noise: <0.0001A, Photometric Range: −0.3A to +4.0A, Mode: Concentration, Abs, % Transmission, Quantitative Analysis: Using up to 20 standards, Rate: up to 8 data points/sec, Photometric accuracy: ±0.005A @ A = 1, Photometric repeatability: ±0.002A @ A = 1, Light sources: Tungsten Halogen & Deuterium Lamps, Detector: Silicon Photodiode.

25.4 RESULTS

The summary results of the screening testing in two districts are shown in Table 25.1.

Table 25.1. Summary data from field screening testing for arsenic in Ballia and Lakhimpur Kheri districts.

Block	Total sources tested	As 0–10 μg/L	As 10–40 μg/L	As 40–50 μg/L	As >50 μg/L	% sources >10 μg/L	% sources >50 μg/L
District–Ballia							
Belhari	191	26	107	6	52	86.4	27.2
Bairiya	204	89	71	13	31	56.4	15.2
Murli Chhapra	198	120	56	5	17	39.4	8.6
Revati	286	74	116	32	64	74.1	22.4
Dubhad	261	66	135	28	32	74.7	12.3
Hanumanganj	290	215	63	7	5	25.9	1.7
Sohaon	285	153	111	14	7	46.3	2.5
Garwar	417	355	61	1	0	14.9	0
Maniyar	261	180	55	7	19	31.0	7.3
Pandah	360	320	40	0	0	11.1	0
Bansdih	290	214	51	14	11	26.2	3.8
Beruarwari	270	253	15	2	0	6.3	0
Chilkahar	390	319	70	0	1	18.2	0.3
Rasra	475	469	5	1	0	1.3	0
Navanagar	312	285	22	2	3	8.7	1.0
Siar	461	354	102	1	4	23.2	0.9
Nagra	700	667	28	5	0	4.7	0
Total	5651	4159	1108	138	246	26.4	4.4
District–Lakhimpur Kheri							
Paliya	316	46	207	23	40	85.4	12.7
Nighasan	419	135	272	6	6	67.8	1.4
Ramiya Behar	357	152	174	12	19	57.4	5.3
Dhaurahara	301	95	171	25	10	68.4	3.3
Issanagar	429	144	231	26	28	66.4	6.5
Mohammdi	351	348	3	0	0	0.9	0
Mitauli	365	354	11	0	0	3.0	0
Pasgawan	299	293	6	0	0	2.0	0
Behjam	296	293	3	0	0	1.0	0
Phoolbehar	332	204	128	0	0	38.6	0
Nakaha	297	215	81	0	1	27.3	0.3
Lakhimpur	336	283	53	0	0	15.8	0
Kumbhi Gola	303	282	21	0	0	6.9	0
Bijuwa	329	176	152	1	0	46.5	0
Bankeyganj	302	237	64	1	0	21.5	0
Total	5032	3257	1577	94	104	35.3	2.1

This data has been mapped block-wise and the maps are given in Figures 25.1 and 25.2. Unfortunately latitude and longitude for each source tested was not available at the time of writing so it was not possible to include a point map of the results.

25.5 DISCUSSION

As can be seen from Figures 25.2 and 25.3 the occurrence of arsenic at concentrations >50 µg/L in the groundwater of Ballia and Lakhimpur Kheri districts has a strong relationship to the course of the major rivers of the area. In Ballia the river Ghaghara lies to the north-east of the state with the river Ganga bounding the south-west. The blocks where arsenic is found in the groundwater of the district are those adjacent to the river, with the exception of Chilkahar (where only 1 out of 390 tests showed an arsenic result >50 µg/L). In Lakhimpur Kheri the river Sarda bisects the district and the river Ghaghara lies to the east of the district. With the exception of Nakaha block (where only 1 out of 297 tests showed an arsenic result >50 µg/L), arsenic is found in groundwater at concentrations >50 µg/L in those blocks which lie between these two rivers.

Figure 25.4 shows the occurrence of arsenic in concentrations at >10 µg/L in groundwater of Lakhimpur Kheri district. It can be seen that occurrence of arsenic decreases further west as distance from the from the rivers Sarda and Ghaghara increases.

When the pattern of arsenic occurrence was compared to geology using the 1: 250,000 'Geological Quadrangle' maps and the 1: 250,000 'District Resource' maps of the Geological Survey of India, the unit referred to as 'Q_2' or 'Terrace Alluvium' respectively on these two series of maps corresponded to the areas already known to be affected. The geological age of this unit is defined as Holocene on these maps. The 'Q_1' unit on the Geological Quadrangle maps or 'Varanasi Alluvium' on the District Resource maps was generally seen to be less unaffected. The geological age of this unit is given as middle to upper (late) Pleistocene. Thus it was concluded that the

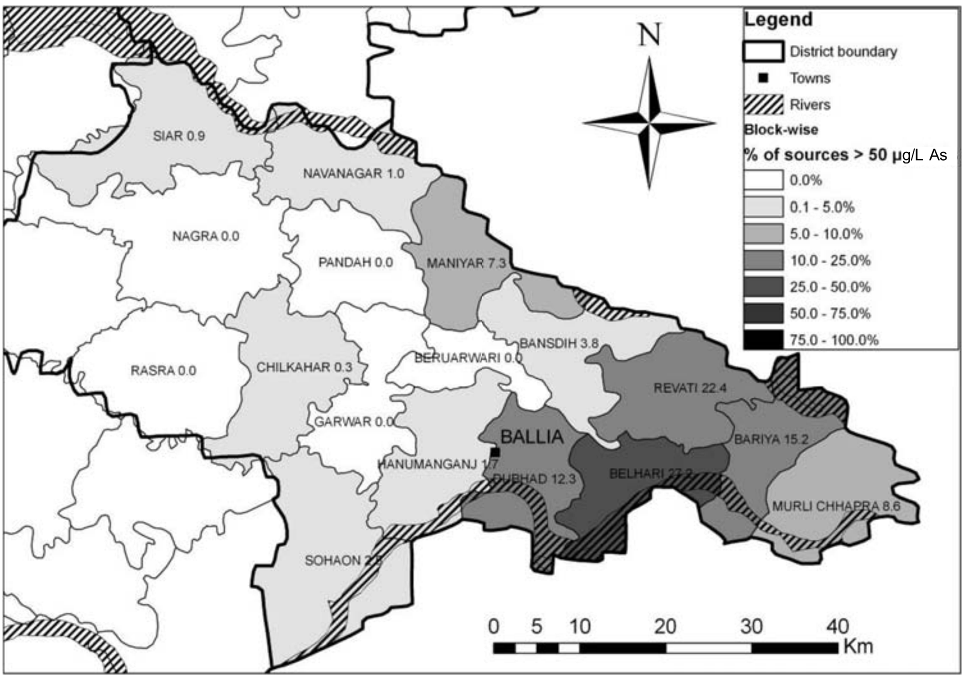

Figure 25.2. Block-wise percentage of handpump sources with concentration of arsenic >50 µg/L in Ballia district of Uttar Pradesh.

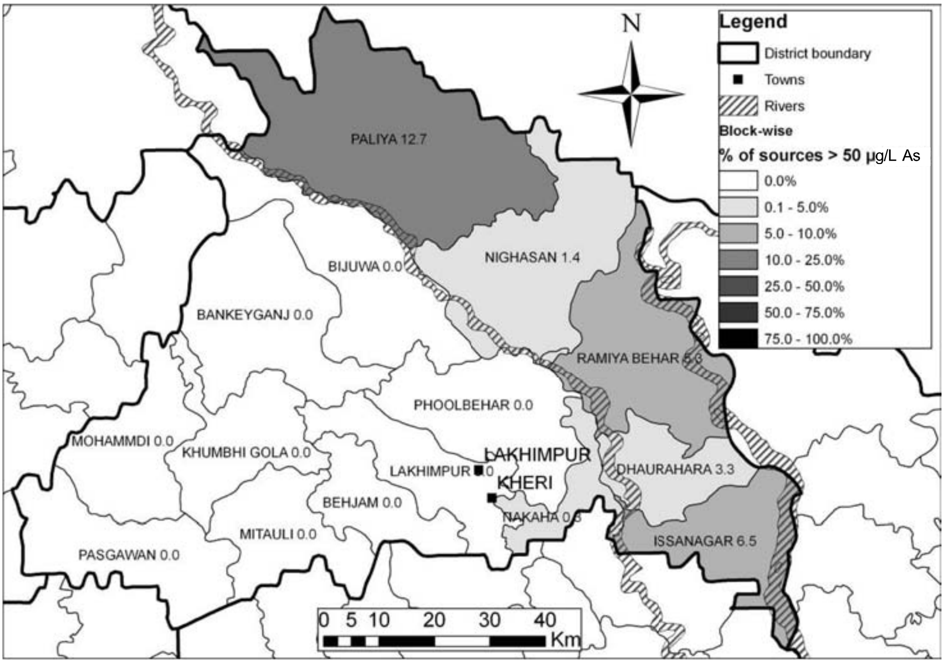

Figure 25.3. Block-wise percentage of handpump sources with concentration of arsenic >50 µg/L in Lakhimpur Kheri district of Uttar Pradesh.

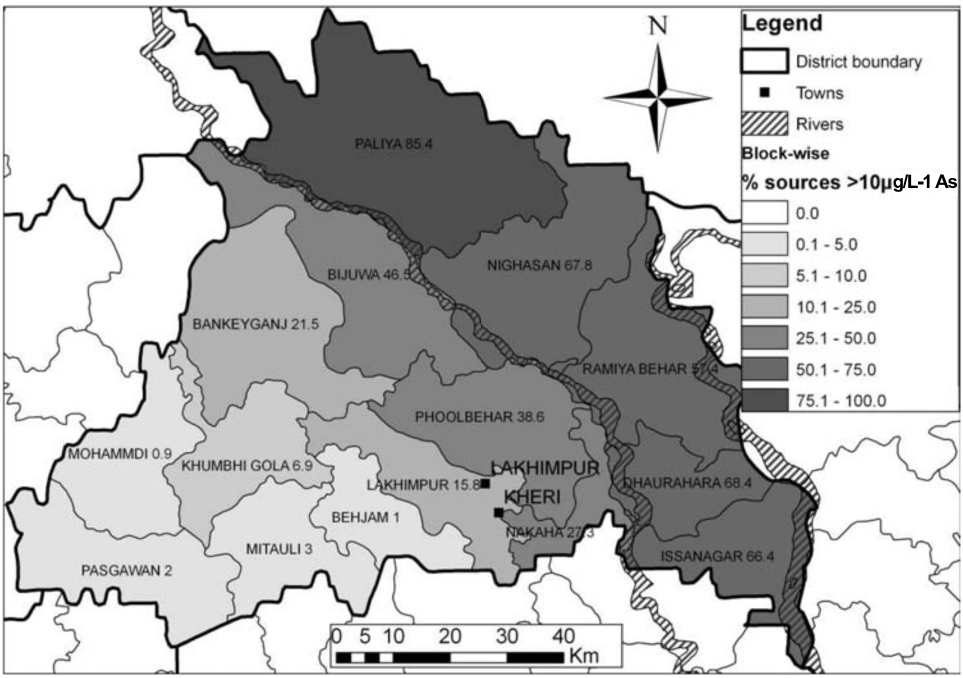

Figure 25.4. Block-wise percentage of handpump sources with concentration of arsenic >10 µg/L^{-1} in Lakhimpur Kheri district of Uttar Pradesh.

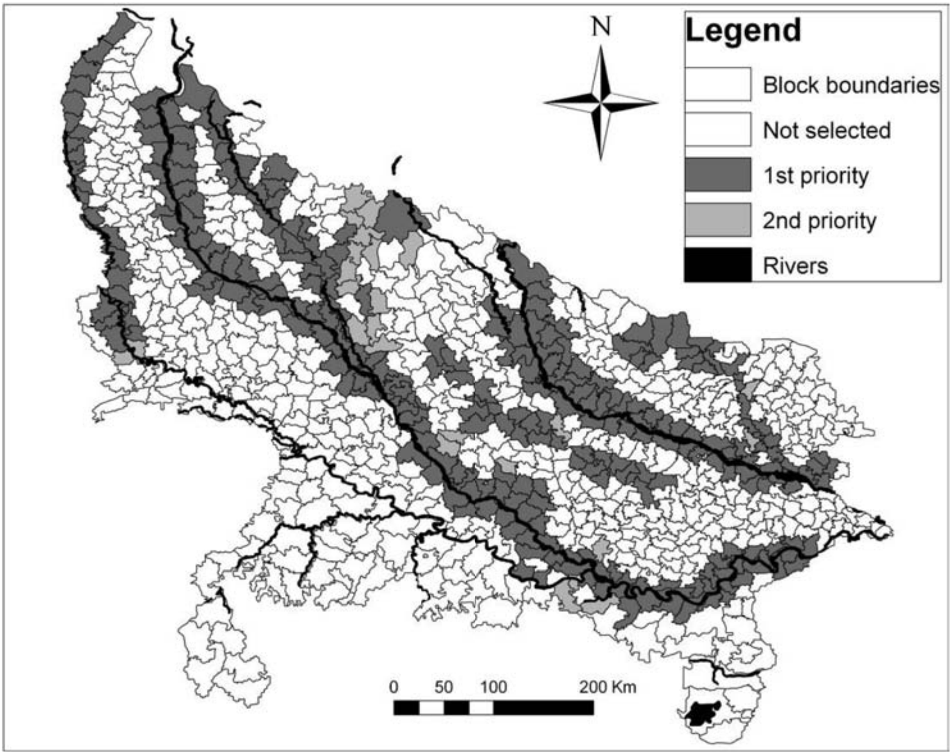

Figure 25.5. 289 blocks identified for state-wide screening testing of arsenic in groundwater in Uttar Pradesh. Blocks in Ballia and Lakhimpur Kheri districts are not included as screening testing is already complete in these districts. 1st priority indicates key blocks for screening testing, 2nd priority indicates blocks to be tested as supplies and time allow.

'Q_2' or 'Terrace Alluvium' geological unit could be considered more 'at risk' of the occurrence of arsenic in groundwater at concentrations >50 µg/L. Unfortunately it was not possible to re-produce the relevant portions of the geology maps here.

This pattern is also seen when the data of SIIR for Uttar Pradesh and Bihar (2004 and 2004 respectively) and PHED/AN College of Patna (2005) data are considered.

This relationship follows the pattern observed elsewhere in India and Bangladesh where mobile arsenic is associated with the geologically younger terminal Pleistocene-Holocene alluvial sediments which contain more organic matter and are thus have a more reducing geochemical environment (e.g. Nickson et al. 2000, McArthur et al. 2004).

25.6 DEVELOPMENT OF A STATE ACTION PLAN FOR SCREENING TESTING

Given the findings of screening testing in Ballia and Lakhimpur Kheri (discussed section 25.5 above) it was decided that priority for screening testing for arsenic should be undertaken in those blocks lying adjacent to major rivers and in particular those where the 'Q_2' or 'Terrace Alluvium' geological unit was present. Following consultation of these maps for the entire state it was found that, in addition to the affected blocks in Ballia and Lakhimpur Kheri, the 'at risk' geological unit was found in the 289 blocks shown in Figure 25.5. It should be noted that no digitized versions of the 'Geological Quadrangle' maps or the 'District Resource' maps were available, so the selection of blocks was done by visual comparison only. There may thus be some inconsistencies in the final selection of blocks for screening i.e. some blocks selected where relevant unit is not present; and some blocks missed where relevant units are found. This will be rectified in further rounds of screening testing.

The screening testing will be carried out using the same methodology previously employed in Ballia and Lakhimpur Kheri districts, namely testing of one government handpump in habitations with 1–10 government installed handpumps in total and 2 handpumps in habitations with >10 government installed handpumps. This will generally amount to around 10–20% of the total government installed handpumps in those particular blocks.

The results of this screening testing should give a broad picture of the areas affected by arsenic contamination in Uttar Pradesh. In future all handpumps in Uttar Pradesh may be tested for arsenic, however, given the financial and human resources available at this time the current strategy is thought to be an effective method of focusing testing on the priority areas. Once the affected blocks/habitations have been identified by screening testing then blanket testing can be undertaken to determine the true magnitude of contamination in the area in question.

25.7 ONGOING BLANKET TESTING, GIS MAPPING, COMMUNICATION, HEALTH AND MITIGATION IN TWO DISTRICTS

25.7.1 *Blanket testing*

In Ballia and Lakhimpur Kheri districts blanket testing is being undertaken following the 'Testing Protocol' outlined in section 25.2 above. This will give the exact magnitude of the arsenic problem in these two districts. The process of testing and marking handpumps also provides information to the community about which sources are arsenic-contaminated and which are arsenic-safe. This provides the first mitigation option to the community for reducing their exposure to arsenic by switching from consumption of water from a source containing high levels of arsenic to water from one with lower or no arsenic where these are available.

25.7.2 *GIS mapping*

All handpumps tested for arsenic in the blanket testing phase will be located using global positioning system units (GPS). This will enable accurate mapping of the arsenic-contaminated and arsenic-safe areas and facilitate design of strategies for arsenic avoidance by communities living in affected areas. Sites for alternative water sources such as deep handpumps may be selected in consultation with the local community using these maps as a guide (e.g. Hassan 2005).

The maps will also serve as an awareness raising and planning tool for the local administration, politicians and planners.

25.7.3 *Communication*

A communication strategy to accompany the blanket testing process has also been devised. Channels of communication which may be explored include UP Jal Nigam testers themselves, NGO's, local elected bodies (Panchayati Raj institutions), health center workers and teachers. A handout has been developed and other materials such as posters, signboards and flip charts are planned.

25.7.4 *Health*

The King George Medical University (KGMU) Lucknow have developed a manual as background and training material for awareness raising of local health service staff on arsenicosis. It incorporates the guidance of the draft 'Field Guide for Detection, Management and Surveillance of Arsenicosis'

(WHO 2006). KGMU will conduct training and awareness raising of district-level senior medical officers and block-level primary health centre (PHC) medical staff and paramedicals in Ballia and Lakhimpur Kheri districts to make them aware of the potential for arsenicosis in their jurisdiction and the symptoms, diagnosis and treatment of the disease.

25.7.5 *Mitigation*

As mentioned above the first option for reducing arsenic exposure available to people is so-called well-switching where a handpump known to contain less arsenic is used in preference to a more heavily contaminated source (e.g. van Geen et al. 2002). This option is only available once all sources are tested and somehow differentiated (e.g. by painting) to indicate high or low arsenic concentration in groundwater.

Table 25.2. Preliminary field test results for arsenic from 35 deep handpumps installed in arsenic-affected areas of Ballia District of Uttar Pradesh.

Block	Village	Habitation	Handpump location	Depth (m)	As in deep H.P. (μg/L)	As in adjacent shallow H.P. (μg/L)
Belahri	Rajpur Ekouna	Khass	Primary School	77	0	90
Belahri	Rajpur Ekouna	Khass	Satrughan Singh	76.6	0	400
Belahri	Rajpur Ekouna	Khass	Anant Kuwar	76.6	0	360
Belahri	Rajpur Ekouna	Khass	Radha K. Yadav	78.6	0	160
Belahri	HaldiHans	Nagar	Dr. D. Rai	64.6	0	70
Belahri	HaldiNem	Chhapara	Primary School	76.6	0	75
Belahri	Bajarha	Khass	Primary School	76.6	0	75
Belahri	Chain ehappara	Khass	Primary School	76.6	0	95
Belahri	Hariharpur	Khass	Primary School	79.6	0	200
Belahri	Repura	Khass	Primary School	79.6	0	75
Belahri	Udawant Chhpara	Khass	Primary School	79.6	0	60
Belahri	Udawant Chhpara	Khass	Shiv Mandir	85.1	0	100
Belahri	Gangapur	Khass	M. L. School	67.5	0	110
Belahri	Gangapur	Khass	Junior P. School	71	0	75
Belahri	Chain Chhpara	Khass	Pratibha Choubey	70.1	10	450
Belahri	Chain Chhpara	Khass	Girija Chaubey	76.6	0	360
Belahri	Udawant Chhpara	Khass	B. Uppadhyaya	79.6	0	115
Belahri	Udawant Chhpara	Khass	Shudarshan Harijan	79.6	0	180
Belahri	Udawant Chhpara	Khass	Narvadeshawer UPS	76.6	0	200
Belahri	Udawant Chhpara	Khass	Harishanker Ram	76.6	0	45
Belahri	Gangapur	Durajanpur	Achhey Lal Yadav	76.6	0	180
Belahri	Gangapur	Tiwaritola	Uma Rawat	76.6	0	140
Belahri	Repura	Hariharpur	Hanuman Mandir	77.6	0	180
Belahri	Nem Chhapra	Khass	Surendra Nath Tiwari	79.5	0	360
Belahri	Nem Chhapra	Khass	Bhikhari Ojha	79.8	0	360
Belahri	Rajpur Ekouna	Khass	Binod Kumar	76	0	130
Belahri	Rajpur Ekouna	Rajpur	Kamaleshwar Singh	75.6	0	180
Belahri	Rajpur Ekouna	Rajpur	Kamaleshwar Singh	80	0	180
Belahri	Gangapur	Tiwaritola	Hari Shanker Yadav	75.4	0	100
Bairiya	Nagar Keharpur	Khass	Barham J. Mathiya	75	0	90
Revati	Gai ghat-I	Khass	Radhey Shyam Singh	76	0	130
Revati	Gai ghat-I	Khass	Yogi Singh	76	0	115
Revati	Gai ghat-II	Khass	Burawa Shivjee	74.3	0	100
Revati	Gai ghat-III	Khass	Laljee Dhobi	75	0	110
Dubhad	Sawrubandh	Khass	P.B. School	79	0	95

Where there are insufficient sources supplying water of low arsenic concentration two options are available: source-substitution to an alternative source, or treatment of the water to remove arsenic. Experience has shown that, although effective under laboratory conditions, very often inadequate operation and maintenance of treatment plants or filters leads to their failure once deployed in the field (e.g. Hossain et al. 2005, Jakariya et al. 2007) for arsenic, or Operations Research Group (ORG) 2005 for fluoride).

In contrast to areas where fluoride occurs in groundwater, the alluvial sedimentary environments of South Asia where arsenic is found in groundwater can be considered as water-rich environments. Water resources are generally plentiful and supply exceeds demand. There are generally four alternative arsenic-safe water sources available in areas affected by arsenic-contamination: deeper groundwater; shallower groundwater; surface water and rainwater.

With these considerations in mind it has been decided that where alternative sources are required in arsenic affected areas in Uttar Pradesh under this project the concept of source-substitution will be followed. The alternative water sources to be trialed include: Deep groundwater through deep handpumps, shallow groundwater through large diameter open wells and rainwater through roof water harvesting systems.

To date 55 deep handpumps have been installed in Ballia district and 3 in Lakhimpur Kheri district. The arsenic concentration in groundwater tapped by these handpumps has been measured only by field test kits to date. The results are encouraging (Table 25.2), however, substantial further work is required before this source can be declared safe for general implementation. Testing of the full range of chemical water quality parameters with implications for human health will be carried out and core sampling and laboratory research into likely rates of downward leakage of arsenic under induced gradients is planned. Essentially a comprehensive programme of supporting research is required in this area.

25.8 CONCLUSION

Arsenic in groundwater used for drinking is an emerging threat to public health in parts of the state of Uttar Pradesh in India. Screening testing carried out using field test kits indicate that 4.4% and 2.1% of India Mark II handpumps in Ballia and Lakhimpur Kheri districts respectively are affected by arsenic in groundwater at concentrations >50 μg/L. The distribution of arsenic in groundwater shows a strong relationship to the course of major rivers in the area, suggesting that arsenic is associated with the geologically younger late-Pleistocene/Holocene alluvial sediments as has been found elsewhere in India and Bangladesh. This knowledge has been used to devise a state action plan for arsenic screening testing which targets blocks where the more recent alluvial sedimentary units are known to occur thereby prioritizing the areas most 'at risk', given the supplies, financial and human resources available. In Ballia and Lakhimpur Kheri districts blanket testing, GIS mapping, communication, training and awareness-raising of health service professionals and provision of alternative water sources where required is underway. This paper is an interim report of work in progress.

ACKNOWLEDGEMENTS

The authors extend their thanks to Peter Ravenscroft of the University of Cambridge for his initial review which helped to improve the manuscript. The authors are thankful to A.B. Mukherjee and Prosun Bhattacharya for their final editorial handling of an earlier draft of this manuscript.

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CHAPTER 26

Arsenic mobilisation in the Holocene flood plains in South-central Bangladesh: Evidences from the hydrogeochemical trends and modeling results

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ABSTRACT: This study presents the results of the investigation of arsenic (As) enrichment in groundwater of three alluvial aquifers at the Bengal Delta Plain (BDP) in Sonargaon in Narayanganj, Chandina in Comilla, and Sirajdikhan in Munshiganj districts in South-central Bangladesh. Water samples were collected from these sites from wells with screens placed at different depths and the hydrogeochemical characteristics and redox status were determined. The highest DOC and HCO_3^- concentrations were found at Sirajdikhan site and lower concentrations at Sonargaon and Chandina sites. In contrast, the highest NH_4^+ concentrations were found at Chandina site and concentrations at other sites were much lower. The correlation between dissolved As and Fe was high at Sirajdikhan and Sonargaon sites, but not at Chandina site. Also, at Chandina site dissolved Mn concentrations were low, suggesting that Mn(IV) redox buffering step was missing. Speciation modeling indicated a possibility of siderite precipitation at all sites, but precipitation of rhodochrosite only at Sonargaon and Sirajdikhan sites. Calculated log P_{CO_2} were very high (reaching -1.37 at Sirajdikhan site), suggesting production of CO_2 in redox processes. The hydrogeochemical trends and modeling results suggest that dissolved As may be de-coupled from dissolved Mn, when Mn(IV) content in solid phase is low (or when released As is re-adsorbed) and from dissolved Fe (when precipitation of Fe(II) minerals controls Fe concentrations). Furthermore, several redox processes may operate simultaneously, depending on kinetic constraints and refractoriness of Fe(III) minerals.

26.1 INTRODUCTION

Natural arsenic (As) present ubiquitously in groundwater of sedimentary aquifers of Bengal Delta Plain (BDP) is an alarming problem in Bangladesh, India and several other countries in the south-east Asian region (Bhattacharya et al. 2002a, 2004, Smedley & Kinniburgh 2002) covering approximately an area of 148,012 km^2. National surveys (BGS & DPHE 2001, Ahmed et al.

2004) indicate that in 60 of the total 64 districts of Bangladesh, As is detected in groundwater above the WHO recommended limit (10 μg/L; WHO 2004) as well as the national drinking water standard (50 μg/L). The source of As occurring in the Holocene sediments deposited by the three major rivers Padma, Brahmaputra (Jamuna) and Meghna is geogenic (Nickson et al. 1998, Bhattacharya et al. 2002a, b, 2006). A vast majority of rural population is exposed to drinking waters containing high levels of As (10–2300 μg/L) in the shallow groundwaters of BDP in Bangladesh that has adverse health consequences to an exposed population of least 35 million (Smith et al. 2000, Frisbie et al. 2002, Yu et al. 2003).

Arsenic was believed to be prevalent in the sedimentary aquifers of the Ganges (Padma) basin (Nickson et al. 1998, 2000), but later studies reveal distinct regional pattern where the Holocene shallow aquifers in the Ganges-Brahmaputra-Meghna (GBM) river system are severely affected (BGS & DPHE 2001, Bhattacharya et al. 2002a, Ahmed et al. 2004, Zheng et al. 2005, Chakraborti et al. 2004, Hasan et al. 2007, Mukherjee et al. in press). However, groundwater in older Plio-Pleistocene aquifers is characteristically low in As (<10 μg/L), which suggest geological and hydrogeological controls on the distribution of As in groundwater. Groundwater As in the BDP shows a close relationship with the depositional environment, texture of sediments and presence of organic matter. Mobilisation of As into the groundwater in the shallow alluvial aquifers in the Bengal Basin involve reductive dissolution of Fe(III)-oxyhydroxide is widely accepted as the principal mechanism (Bhattacharya et al. 1997, 2001, 2002a, b, 2006, Nickson et al. 1998, 2000, BGS & DPHE 2001, Smedley & Kinniburgh 2002, Ahmed et al. 2004, Horneman et al. 2004, McArthur et al. 2004, Hasan et al. 2007). Other solid phases such as Mn/Al-oxyhydroxides and phyllosilicates may also play an important role in As cycling in the aquifers (Breit et al. 2001, Kent & Fox 2004, Saunders et al. 2005, Chakraborty et al. 2007, Stollenwerk et al. 2007). Mobilisation of As due to the oxidation of pyrite (Chowdhury et al. 1999) and widespread application of phosphate fertilizers (Acharyya et al. 2000) are not consistent with the geochemical evidences revealed by several studies (Nickson et al. 2000, BGS & DPHE 2001, Bhattacharya et al. 2002a, 2006, Harvey et al. 2002, Zheng et al. 2004, 2005, Swartz et al. 2004, Horneman et al. 2004, Ahmed et al. 2004).

The present study examines the local scale variations of the chemistry of groundwater with special reference to the distribution and speciation of As and their implications on the drinking water supply in the region. In the present paper we discuss the salient hydrogeochemical characteristics of groundwater from three different areas, and their implications on As mobilization in the region. The current study areas comprise Sonargaon in Narayanganj; Chandina in Comilla; and Sirajdikhan in Munshiganj district of Bangladesh, which are located at the south central part of the BDP. The emphasis is on the comparison of redox zonality with depths and determination of terminal electron acceptors (TEAP) at different areas. The role of Mn(IV) and Fe(III) minerals are also evaluated.

26.2 GEOLOGY AND HYDROGEOLOGY OF THE STUDY AREA

26.2.1 *Geology*

The BDP is one of the largest deltas in the world, formed by huge amount of sediments transported by the Ganges-Brahmaputra (Jamuna)-Meghna (GBM) river systems. The BDP contains extensive plains of relatively uplifted Pleistocene terraces and low lying Holocene fans and floodplains. The Bangladesh part of the BDP includes fan-deltas of the Tista and Brahmaputra; fluvial flood plains of the Ganges, Brahmaputra, Tista and Meghna rivers; delta plain of the lower GBM system south of the Ganges-Meghna valleys, including the moribund Ganges delta and the Chandina plain of Holocene age, Pleistocene Terraces of Barind and Madhupur Tracts and the subsiding basins in the eastern Ganges tidal delta and the Sylhet basin (Morgan & McIntire 1957, Umitsu 1993, Brammer 1996, BGS & DPHE 2001).

The Quaternary sediments in the region were deposited by three major rivers Ganges (Padma), Brahmaputra (Jamuna), and Meghna (Fig. 26.1). The deltaic floodplains consist of channel-fill deposits which are dominated by coarse-grained detritus, whereas the overbank deposits are characterized by predominantly fine-grained sediments. The sedimentary sequences were strongly influenced by the meandering character of these rivers and also by the sea level fluctuations during

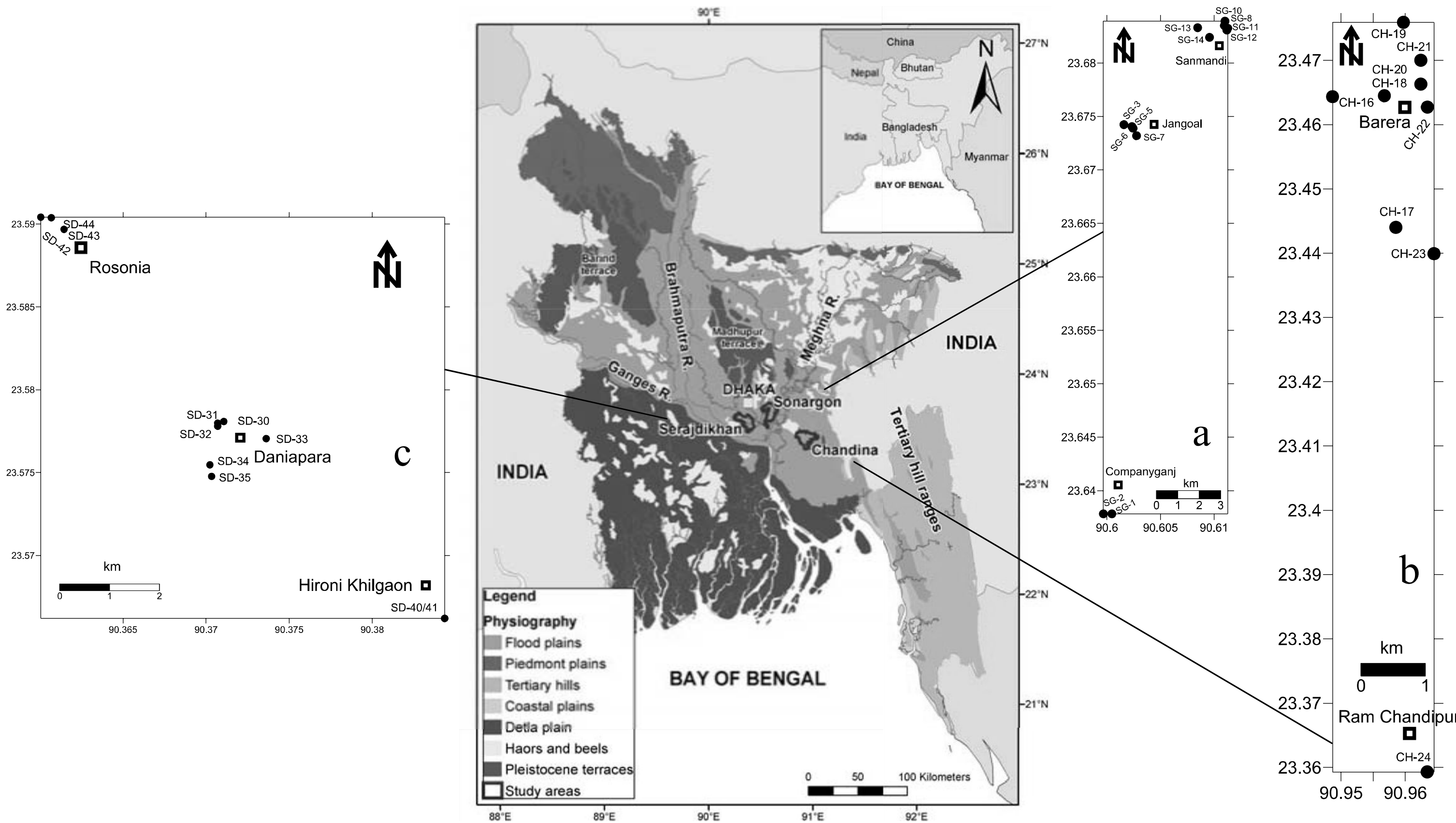

Figure 26.1. Map of Bangladesh showing the location of the study areas. Inset map shows the location of Bangladesh in SE Asia. Groundwater sampling locations are shown in insets, a) Sonargaon (Narayanganj district); b) Chandina (Comilla district); and c) Sirajdikhan (Munshiganj district).

the evolution of the BDP. During the late Holocene period, when the transgression rates decreased, marshy lowlands were developed resulting in the formation of peat in localized parts of the region (Umitsu 1993).

26.2.2 *Hydrogeology and groundwater characteristics*

The extensive occurrences of Quaternary alluvial sediments and abundant sources of recharge from the precipitation and floodwater combine to develop prolific aquifer systems all over the BDP. Groundwater occurs very near to the ground surface in the Holocene sandy sediments and forms extensive unconfined to leaky confined aquifers on the fluvial plain or fan delta. The Pleistocene Dupi Tila sands form the confined aquifers in the elevated terraces. Shallow groundwater from both the aquifers are exploited by the conventional shallow hand tube wells for drinking purposes all over the country from a depth from 10 to 100 m. The same aquifers are also exploited extensively for dry season irrigation of the high yielding varieties of rice by thousands of wells developed in the range of 40 to 100 m. Groundwater abstracted from the Holocene aquifers only contain elevated concentrations of dissolved arsenic in various parts of the country; water withdrawn from the Pleistocene aquifers are safe from arsenic (BGS & DPHE 2001, Ahmed et al. 2004).

Water levels in the Holocene aquifers fluctuate in response to annual recharge discharge conditions (Ahmed 1994, 2005). Water levels start declining at the end of September with the cessation of monsoon rain and reach maximum depth from 3 to 10 m in the month of April–May. This decline is caused partly by natural processes of slow discharge of groundwater to low lying areas and rivers and partly by the impact of large abstraction of groundwater for irrigation during the months from January to March. The larger amounts of groundwater fluctuations are observed in the areas having higher densities of irrigation wells. After the onset of pre-monsoon towards the end of May, water levels start rising and reach the ground surface during August–September. This rise is caused by recharge during the monsoon rainfall and flooding of water. This annual fluctuation pattern is maintained to a large extent all over the basin, indicating a dynamic equilibrium in the shallow aquifers. However, a more accentuated groundwater level fluctuation has been observed in the recent years and in certain parts a declining trend is apparent. Movement of shallow groundwater is mainly driven by surface topography and controlled by the presence of surface water bodies such as rivers and *beels* (Ahmed 1994, Ravenscroft 2003).

The sedimentary architecture of the BDP is characterized by a multiple aquifer system often separated by clay or silty aquitard in between sand formations. However, the lateral continuity of these layers are often not very extensive. The presence of low hydraulic conductivity layers can result into marked variations in observed hydraulic head and water quality. The multiple aquifer system is also present in the study area where most groundwater is abstracted from the Holocene aquifers in the range of 100 m. Some groundwater is abstracted from the Pleistocene Dupi Tila aquifers, particularly in the areas close to the Lalmai Hills. Groundwater abstracted from the Holocene aquifers is generally of very good quality except the presence of higher amount of dissolved iron. Occurrences of pockets of entrapped brackish water and presence of biogenic methane has also been reported (Ahmed et al. 1998, Ravenscroft et al. 2001).

26.3 MATERIAL AND METHODS

Thirty-two groundwater samples were collected during January 2002 from domestic tube wells from the three As-affected regions of Sonargaon in Narayanganj district, Chandina in Comilla district and Sirajdikhan in Munshiganj district of Central Bangladesh (Fig. 26.1; Table 26.1). Twelve wells investigated in Sonargaon (Fig. 26.1a) were placed at depths 6.1–91.4 m, 9 wells in Chandina (Fig. 26.1b) were placed at depths 18.3–65.5 m, while the 11 wells in Sirajdikhan (Fig. 26.1c) at depths 42.7–99.4 m. The pH, Eh, temperature and electrical conductivity (EC) were measured in the field pH was measured using Radiometer Copenhagen PHM 80 instrument equipped with a combination electrode (pH C2401-7). The Eh was measured in a flow-through cell using a combined platinum electrode (MC408Pt) with a calomel reference electrode. Water samples were taken in

Table 26.1. Salient hydrochemical characteristics of groundwater in the investigated areas around Sonargaon, Chandina, and Sirajdikhan in South-central Bangladesh.

Sample ID	Village/Union	Depth (m)	pH	Eh (V)	EC (µS/cm)	HCO_3^-	Cl^-	NO_3^-	SO_4^{2-}	PO_4^{3-}	Na^+ (mg/L)	K^+	Mg^{2+}	Ca^{2+}	IB (%)	Water type	As_{tot} (µg/L)	As(III)	As(III) (%)	Fe_{tot}	Mn (mg/L)	NH_{4+}	DOC
Sonargaon/Narayanganj																							
SG-1	Companyganj, Mograpara	45.7	7.46	−0.42	626	280.7	36.9	bdl	0.99	5.31	10.4	4.86	16.0	75.5	−1.5	Ca-Mg-HCO_3	253.0	240.0	94.9	17.1	0.61	bdl	2.41
SG-2	Companyganj, Mograpara	91.4	6.88	−0.36	1182	195.2	247.0	0.02	8.12	0.35	131	1.80	22.3	59.4	1.0	Na-Ca-Cl-HCO_3	2.0	0.9	45.0	0.62	0.91	0.22	1.75
SG-3	Jangaal, Sanmandi	19.8	7.36	−0.39	525	262.4	23.0	0.05	0.67	5.41	16.4	4.36	12.8	68.4	−8.6	Ca-Mg-HCO_3	857.0	839.0	97.9	2.60	1.11	1.80	1.07
SG-5	Jangaal, Sanmandi	21.3	7.29	−0.35	148	250.2	16.0	bdl	0.55	5.49	12.7	3.37	11.7	59.8	−1.6	Ca-Mg-HCO_3	463.0	438.0	94.6	4.34	0.48	bdl	1.73
SG-6	Jangaal, Sanmandi	19.8	6.65	−0.42	465	250.2	2.8	0.14	0.53	6.16	11.8	3.43	11.9	58.6	1.2	Ca-Mg-HCO_3	436.0	430.0	98.6	4.32	0.44	0.76	1.89
SG-7	Jangaal, Sanmandi	24.4	6.09	−0.36	637	323.4	9.3	0.00	0.77	5.10	17.8	3.82	15.7	75.9	1.7	Ca-Mg-HCO_3	575.0	574.0	99.8	6.22	0.61	bdl	1.42
SG-8	Sanmandi, Sanmandi	24.4	6.62	−0.38	400	183.0	9.9	0.06	0.34	2.47	7.32	2.63	16.5	36.7	3.0	Ca-Mg-HCO_3	322.0	309.0	96.0	3.53	1.59	0.49	1.63
SG-10	Sanmandi, Sanmandi	61.0	7.40	−0.37	668	353.9	19.0	0.12	2.72	0.43	15.7	2.61	24.3	83.7	3.9	Ca-Mg-HCO_3	2.4	2.2	90.8	0.45	1.72	0.36	1.36
SG-11	Sanmandi, Sanmandi	36.6	6.50	−0.39	575	256.3	34.0	0.06	1.77	1.31	10.6	3.70	14.9	76.0	3.1	Ca-Mg-HCO_3	148.0	145.0	98.0	1.94	1.00	0.97	1.89
SG-12	Sanmandi, Sanmandi	36.6	6.49	−0.38	524	268.5	7.7	0.07	0.77	2.60	9.99	3.10	12.4	70.8	3.6	Ca-Mg-HCO_3	151.0	135.0	89.4	2.82	0.72	1.09	1.81
SG-13	Sanmandi, Sanmandi	6.1	6.73	−0.38	639	317.3	2.1	0.22	0.61	4.19	14.9	4.41	16.5	75.5	4.2	Ca-Mg-HCO_3	685.0	656.0	95.8	7.52	2.35	2.70	1.95
SG-14	Sanmandi, Sanmandi	36.6	6.97	−0.37	689	329.5	17.8	bdl	0.66	3.78	14.3	4.84	17.0	75.6	−1.0	Ca-Mg-HCO_3	320.0	312.0	97.5	19.59	0.75	bdl	1.77
Chandina/Comilla																							
CH-16	Barera, Barera	24.4	6.61	−0.39	689	286.8	69.0	0.08	0.55	9.49	54.1	7.55	34.4	24.9	−2.5	Mg-Na-HCO_3	392.0	390.0	99.5	2.89	0.07	9.66	2.59
CH-17	Barera, Barera	22.9	6.24	−0.41	887	292.9	89.0	0.04	0.51	12.58	67.1	8.10	35.3	25.4	−2.9	Na-Mg-HCO_3-Cl	372.0	361.0	97.0	3.43	0.07	10.56	2.36
CH-18	Barera, Barera	27.4	6.81	−0.41	2800	427.1	563.0	0.18	0.67	9.68	337	15.5	70.6	43.5	−0.3	Na-Mg-Cl-HCO_3	460.0	439.0	95.4	4.42	0.08	18.03	2.82
CH-19	Barera, Barera	65.5	6.74	−0.44	2590	439.3	582.0	0.19	0.65	8.66	483	18.6	28.5	12.5	1.1	Na-Cl-HCO_3	390.0	374.0	95.9	1.23	0.06	15.46	1.99
CH-20	Barera, South Barera	32.0	8.13	−0.39	561	268.5	14.0	0.10	0.40	8.75	27.1	7.71	31.4	20.9	−0.8	Mg-Na-Ca-HCO_3	380.0	368.0	96.8	1.89	0.04	6.44	2.42
CH-21	Barera, South Barera	18.3	6.81	−0.41	471	219.6	6.7	0.10	0.52	4.92	12.1	7.01	24.4	24.5	−0.4	Mg-Ca-HCO_3	154.0	149.0	96.8	7.59	0.26	6.05	1.97
CH-22	Barera, South Barera	33.5	6.89	−0.41	480	219.6	8.2	0.07	0.47	4.69	14.5	7.16	24.1	24.0	0.1	Mg-Ca-HCO_3	184.0	177.0	96.2	6.10	0.23	4.64	2.62
CH-23	Ram Chandrapur, Barera	30.5	6.86	−0.42	695	323.4	25.0	0.05	0.63	7.58	45.3	8.01	34.2	25.1	−0.2	Mg-Na-Ca-HCO_3	269.0	254.0	94.4	2.71	0.05	7.08	1.79
CH-24	Ram Chandrapur, Barera	33.5	7.16	−0.40	669	286.8	28.0	0.13	0.36	7.07	70.9	6.04	20.4	17.6	0.6	Na-Mg-HCO_3	225.0	215.0	95.6	3.26	0.05	5.67	2.82
Sirajdikhan/Munshiganj																							
SD-30	Daniapara, Rozonia	45.7	7.12	−0.39	524	311.2	6.7	0.09	0.66	5.76	14.1	6.11	15.0	72.8	1.3	Ca-Mg-HCO_3	538.0	525.0	97.6	6.88	0.48	3.86	5.89
SD-31	Daniapara, Rozonia	45.7	6.80	−0.43	536	317.3	6.8	0.26	0.75	5.69	14.8	8.28	15.1	72.7	1.1	Ca-Mg-HCO_3	619.0	593.0	95.8	7.93	0.36	0.55	4.95
SD-32	Daniapara, Rozonia	61.0	7.37	−0.38	749	463.7	13.0	0.13	0.54	0.43	130	2.23	12.8	41.3	5.0	Na-Ca-HCO_3	3.6	3.0	82.9	0.77	1.41	0.71	6.29
SD-33	Daniapara, Rozonia	48.8	7.75	−0.419	539	317.3	11.0	bdl	0.80	6.66	17.9	3.03	17.4	71.0	0.8	Ca-Mg-HCO_3	431.0	268.0	62.2	4.91	0.10	0.50	4.00
SD-34	Daniapara, Rozonia	76.2	7.02	-	521	305.1	7.5	0.15	0.72	5.20	14.1	4.23	13.7	73.2	1.0	Ca-Mg-HCO_3	564.0	508.0	90.1	5.57	0.41	5.41	5.51
SD-35	Daniapara, Rozonia	79.2	6.86	-	1026	610.1	30.0	0.16	1.93	0.45	143	2.15	26.2	65.9	3.6	Na-Ca-HCO_3	1.0	0.9	90.0	0.32	0.09	0.23	4.91
SD-40	Hironi Khilgaon, Rozonia	42.7	6.84	+0.10	789	538.8	13.0	0.14	0.78	0.95	33.2	3.04	28.6	107	−0.1	Ca-Mg-HCO_3	129.0	69.9	54.2	0.23	2.01	2.96	3.93
SD-41	Hironi Khilgaon, Rozonia	42.7	6.8	+0.13	950	571.4	28.0	0.19	1.00	0.67	46.2	2.74	31.7	110	−0.1	Ca-Mg-HCO_3	49.8	32.0	64.3	0.07	2.07	2.19	4.48
SD-42	Hironi Khilgaon, Rozonia	73.2	6.83	+0.12	844	558.3	7.8	0.47	0.77	0.27	31.2	2.19	39.0	95.5	−0.1	Ca-Mg-HCO_3	4.0	4.0	100.0	0.12	2.06	0.91	5.45
SD-43	Hironi Khilgaon, Rozonia	76.2	6.75	−0.12	816	485.9	19.0	0.18	0.75	2.75	69.3	3.73	22.1	71.9	−0.6	Ca-Na-Mg-HCO_3	236.0	223.0	94.5	6.91	0.18	5.15	5.40
SD-44	Hironi Khilgaon, Rozonia	99.4	6.72	+0.14	942	527.2	39.0	0.21	3.76	0.24	80.7	2.22	29.9	76.0	−0.4	Ca-Na-Mg-HCO_3	1.0	0.9	90.0	0.05	2.13	0.77	5.22

replicates as: i) filtered samples (Sartorius 0.45 μm online filter) for anion analyses; ii) filtered and acidified samples (14 M HNO_3) for major and trace element analyses. Speciation of As(III) was carried out in the field using disposable cartridges, following the procedure discussed in Meng et al. (2001).

Alkalinity was measured on a Radiometer Copenhagen® PHM 82 Standard pH meter equipped with an ABU 80 Autoburette. The alkalinity was determined according to the standard method SS-EN ISO 9963-2 (SIS 1996). Major anions Cl^- and SO_4^{2-} were analyzed by a Dionex 120 ion chromatograph, whereas NO_3^- and PO_4^{3-} were analyzed using Tecator AQUATEC 5400 analyzer at wavelengths 540 nm and 690 nm, respectively. The major cations and trace metals were analyzed on a Perkin Elmer Elan 6000 ICP-MS at the Department of Water and Environment, Linköping University, Sweden. Arsenic(V) was calculated as a difference between total As and As(III) in the samples. Certified standards, SLRS-4 (National Research Council, Canada) and GRUMO 3A (VKI, Denmark) and synthetic chemical standards prepared in the laboratory, and duplicates were analyzed after every 10 samples during the runs. Trace element concentrations in standards were within 90–110% of their true values. Relative percent difference between the original and duplicate samples were within ±10% . Dissolved organic carbon (DOC) in the water samples were determined on a Shimadzu 5000 TOC analyzer (0.5 mg/L detection limit) with a precision of ±10% at the detection limit.

26.4 RESULTS AND DISCUSSION

26.4.1 *Major ion chemistry*

The distribution of major ions in the three study areas showed wide variations. The results of the analytical data from the study areas are presented in Table 26.1. pH values of the groundwater in Sonargaon were from slightly acidic to slightly alkaline (pH 6.1–7.6). Eh varied between −0.42 to −0.36 V suggesting strongly reducing conditions. Electrical conductivity (EC) varied between 148–689 μS/cm, except for one sample with relatively high EC of 1,182 μS/cm. Groundwater in Sonargaon were predominantly Ca-Mg-HCO_3 type (Fig. 26.2).

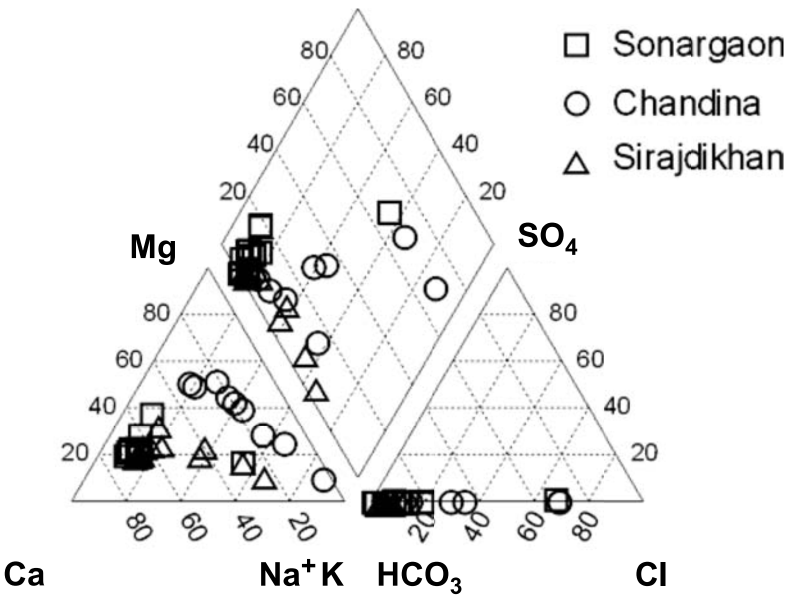

Figure 26.2. Major ion characteristics of groundwaters on a Piper plot.

The major ion chemistry was dominated by HCO_3^-, with concentrations ranging between 183–353.9 mg/L. The concentration of Cl^- varied between 2.1–36.9 mg/L except one high value (274 mg/L). The levels of NO_3^- were very low (below detection limit to up to 0.22 mg/L). Concentrations of SO_4^{2-} and PO_4^{3-} ranged between 0.34–8.12 mg/L and 0.35–6.16 mg/L respectively. The concentrations of Na^+ were generally low (7.3–17.8 mg/L). Ca^{2+} concentrations were comparatively high in all the wells (36.7–83.7 mg/L), while Mg^{2+} and K^+ concentrations were consistent at levels between 12.4–24.3 mg/L and 1.8–4.9 mg/L, respectively (Fig. 26.3a).

Redox chemistry of groundwater in Chandina was similar to those from Sonargaon, with pH and Eh ranging between 6.2–8.1 and −0.44 to −0.39 V respectively. EC ranged mostly between 471–887 µS/cm, except for two wells with very high EC values (2,590–2,800 µS/cm). The water in the Chandina wells showed a mixing trend of Ca-Mg-HCO_3 to Na-Cl type (Fig. 26.2) with HCO_3^- concentrations ranging between 219.6–439.3 mg/L. Concentrations of Cl^- varied between 8.2–89 mg/L, except two samples with very high concentration (582 and 563 mg/L). Both NO_3^- (0.04–0.19 mg/L) and SO_4^{2-} (0.36–0.63 mg/L) were very low but PO_4^{3-} concentrations were fairly high (4.69–12.6 mg/L). The concentrations of Ca^{2+} (12.5–43.5 mg/L) were fairly low, while the Na^+ (12–483 mg/L), K^+ (6.04–18.6 mg/L) and Mg^{2+} (20.4–70.6 mg/L) concentrations showed a considerable variability (Fig. 26.3b).

In general, the groundwater samples at Sirajdikhan exhibited an circum-neutral to slightly alkaline character (pH 6.7–7.8) with field measured redox potential ranging between −0.41 to +0.14 V.

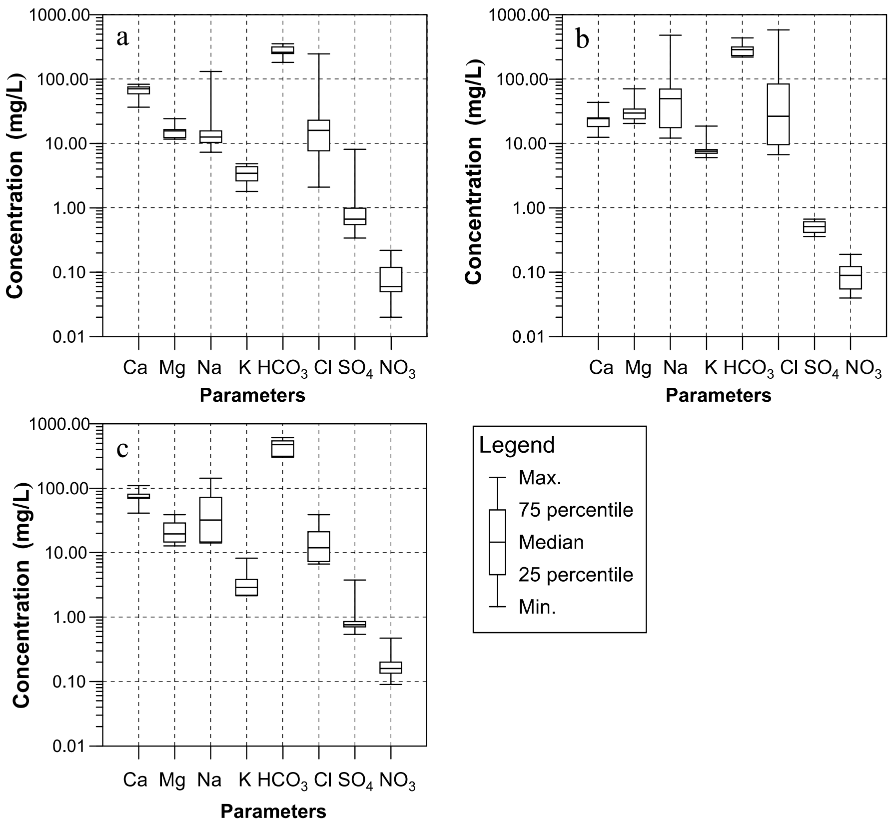

Figure 26.3. Box and Whisker plot for the distribution of major ions in the groundwater from: a) Sonargaon, b) Chandina; and c) Sirajdikhan. Note the log scale in the y-axes.

EC values were moderately high (524–1,026 mg/L) and characterized by elevated concentrations of HCO_3^- (305.1–610.1 mg/L), and depleted in NO_3^-, SO_4^{2-} and PO_4^{3-} (Table 26.1). The water samples were mostly Ca-Mg-HCO_3 type showing a trend to evolve towards Na-HCO_3 type (Fig. 26.2) with concentrations of Na^+ (14.1–143 mg/L), Mg^{2+} (12.8–39 mg/L) and Ca^{2+} (71–110 mg/L). Distribution of K^+ was variable and ranged between 2.15 to 8.28 mg/L (Fig. 26.3c).

26.4.2 *Total As, Fe, Mn, DOC and ammonium*

The analytical data on the redox sensitive parameters, i.e., total arsenic (As_{tot}), total iron (Fe_{tot}), Mn, dissolved organic carbon (DOC) and ammonium (NH_4^+) are presented in Table 26.1, and plotted on Box and Whisker diagrams (Fig. 26.4a–c).

Total arsenic (As_{tot}) in the analyzed wells showed wide range of variability (Fig. 26.4a–c), but were mostly above the Bangladesh drinking water standard (50 µg/L) as well as the WHO guideline value (10 µg/L). Distinct relationship is observed between As_{tot} concentration and the depth of wells, and the peak As_{tot} concentration were noted for wells at depths intervals of 20–60 m and 20–>60 m at the Sonargaon and Chandina sites respectively. However, samples at depths greater than 60 m were significantly low in As_{tot} in Sonargaon. At Sirajdikhan, As_{tot}

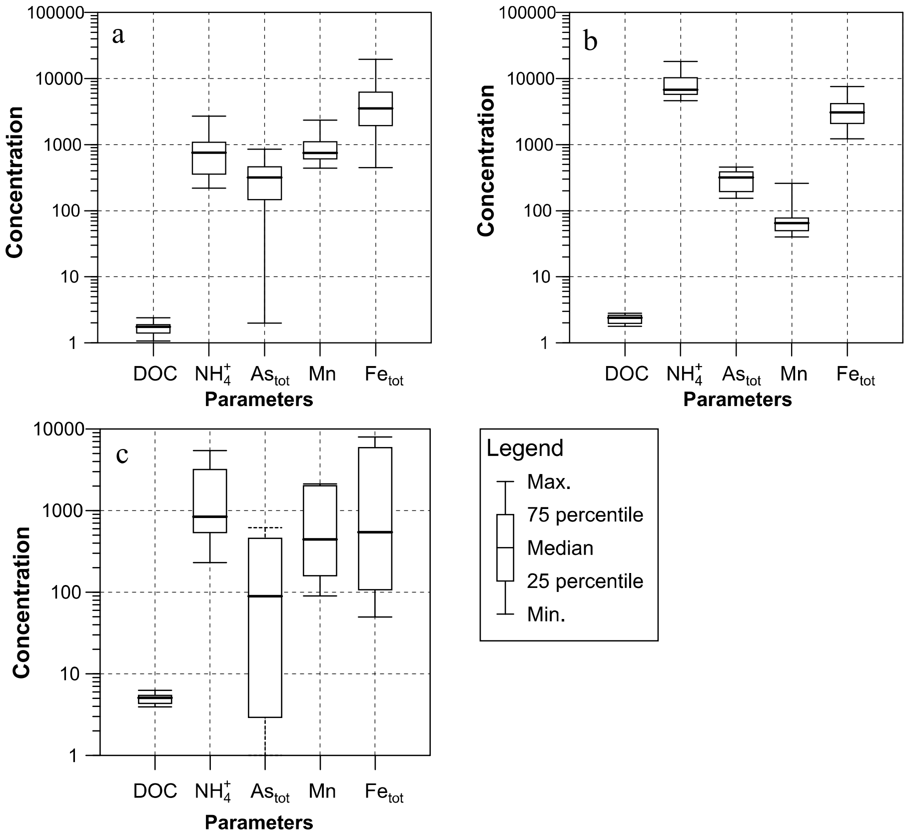

Figure 26.4. Box and Whisker plot for the distribution of DOC, NH_4^+, As_{tot}, Mn and Fe_{tot} in groundwater samples from: a) Sonargaon, b) Chandina, and c) Sirajdikhan. Note the log scale in the y-axes. DOC concentration is expressed as mg/L, while all other concentrations are in µg/L.

peaked at two depth intervals between 40–50 m and 70–80 m. As(III) was the dominant species in the water samples representing about 90–99% of the As_{tot} (Table 26.1).

Concentrations of total iron (Fe_{tot}) varied between 0.45–19.6 mg/L in the water samples in Sonargaon, 1.23–7.59 mg/L in Chandina and 0.05–7.93 mg/L in Sirjdikhan (Table 26.1). Fe_{tot} peaks were consistent with As_{tot} in Sirajdikhan, as well as with the peaks of DOC. However, such relationships were not very apparent at other two study areas. The concentrations of Mn were relatively high in the water samples of Sonargaon (0.61–2.35 mg/L) and Sirajdikhan (0.09–2.13 mg/L) compared to the samples from Chandina, where Mn concentrations were low (0.04–0.26 mg/L). As_{tot} were significantly low in samples with relatively high dissolved Mn, particularly at Sirajdikhan.

In Sonargaon and Chandina areas the DOC levels showed variation between 1.07 to 2.82 mg/L, while in Sirajdikhan, the levels of DOC were higher (3.9–6.3 mg/L, Fig. 26.4a–c). Dissolution of sedimentary carbonates less common in the BDP aquifer systems, and therefore degradation of sedimentary organic matter is most likely to account for most of HCO_3^- and DOC in the BDP groundwaters. Consistent with this, DOC and HCO_3^- levels are high and vary considerably at all the three investigated areas, showing a distinct depth control (Bhattacharya et al. 2006), where degradation of sedimentary organic matter in sediments tend to increase of the levels of HCO_3^- and DOC. High DOC values probably results from microbial degradation of buried organic matter in the sediments as evidenced by previous studies (see Ravenscroft et al. 2001, 2005, Artinger et al. 2000, Buckau et al. 2000, Bhattacharya et al. 2002b).

Ammonium concentrations are typically low in Sonargaon (0.22–2.7 mg/L, Fig. 26.4a), while at Sirajdikhan (0.23–5.4 mg/L) and Chandina the levels are much higher (4.6–18 mg/L, Fig. 26.4b, c). Interestingly, the distributions of NO_3^- and NH_4^+ in the groundwaters reveal a clear antipathic relation in these analysed groundwaters (Hasan et al. 2007, Mukherjee et al. in press).

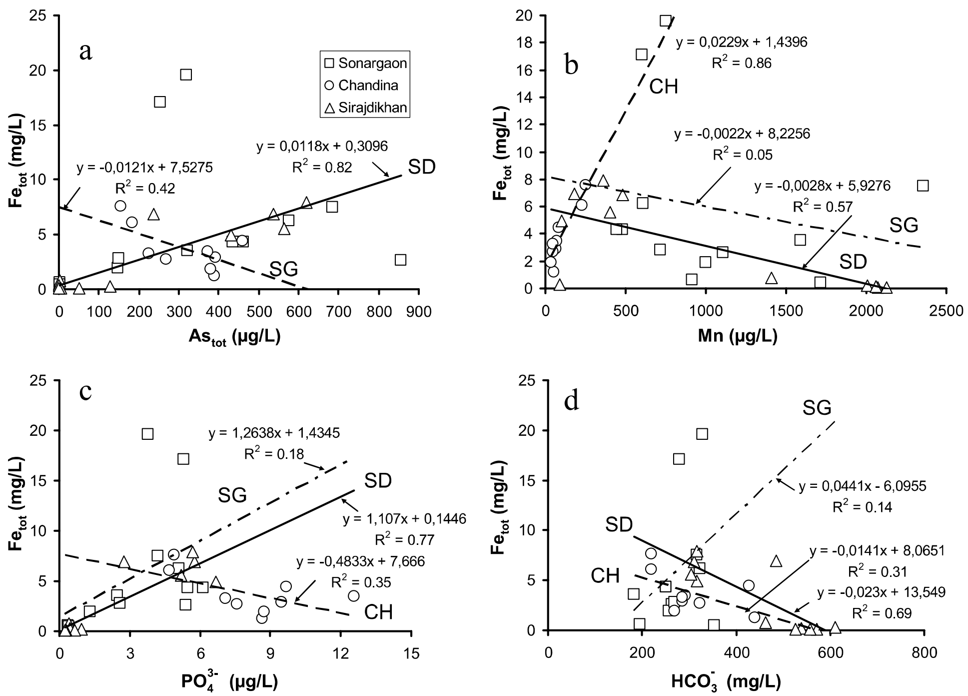

Figure 26.5. Relationship of Fe_{tot} with a) As_{tot}, b) Mn, c) PO_4^{3-}, d) HCO_3^- in the investigated sites. The regression lines ($p \gg 0.01$) are labeled: Sonargaon (SG), Chandina (CH) and Sirajdikhan (SD).

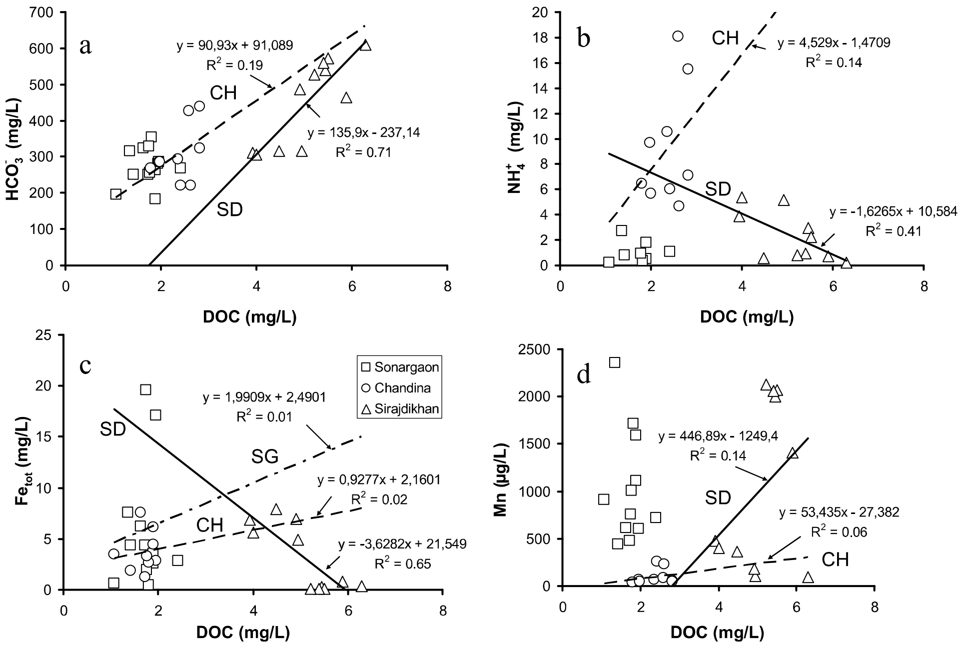

Figure 26.6. Bivariate plots showing relationship of DOC with a) HCO_3^-, b) NH_4^+, c) Fe_{tot}, and d) Mn in groundwaters. The regression lines ($p \gg 0.01$) are labeled: Sonargaon (SG), Chandina (CH) and Sirajdikhan (SD).

26.4.3 *Bivariate analyses*

The bivariate plots reveal considerable variability in the relationships between different hydrogeochemical parameters at the three different sites. At Sonargaon site, Fe_{tot} showed a moderate negative correlation with As_{tot} ($R^2 = 0.42$), no correlation with Mn, low positive correlation with PO_4^{3-} ($R^2 = 0.18$), and low positive correlation with HCO_3^- ($R^2 = 0.14$) (Fig. 26.5a–d). DOC indicated very poor correlation with HCO_3^-, NH_4^+, Fe_{tot}, and Mn (Fig. 26.6a–d) in the wells at Sonargaon. Similarly, the correlation of As_{tot} with both HCO_3^- and DOC was very low, but correlation of As_{tot} with PO_4^{3-} was significant ($R^2 = 0.59$), (Fig. 26.7a–c).

The bivariate plots indicated no correlation between Fe_{tot} with As_{tot}, high positive correlation with Mn ($R^2 = 0.86$), low negative correlation with PO_4^{3-} ($R^2 = 0.35$), and low negative correlation with HCO_3^- ($R^2 = 0.31$) at the Chandina site (Fig. 26.5a–d). Correlation of HCO_3^-, NH_4^+, Fe_{tot}, and Mn with DOC were generally low for these samples from Chandina (R^2 values of 0.19, 0.14, 0.02 and 0.06, respectively) and the Sonargaon sites (Fig. 26.6a–d). Correlation of As_{tot} with HCO_3^- was moderately high ($R^2 = 0.52$), but there was no apparent correlation of As_{tot} with DOC at Chandina and Sonargaon sites (Fig. 26.7b). On the contrary, correlation between As_{tot} and PO_4^{3-} was significant at both sites ($R^2 = 0.68$), (Fig. 26.7c).

At Sirajdikhan site, Fe_{tot} exhibited high positive correlation with As_{tot} ($R^2 = 0.82$), negative correlation with Mn ($R^2 = 0.57$), high positive correlation with PO_4^{3-} ($R^2 = 0.77$), and negative correlation with HCO_3^- ($R^2 = 0.69$), (see Fig. 26.5a–d). Similarly high positive correlation was observed amongst HCO_3^- and DOC ($R^2 = 0.71$), while a low negative correlation was found between DOC and NH_4^+ as well as Fe_{tot}, (Fig. 26.6a–c). The correlation between Mn and DOC was slightly positive ($R^2 = 0.14$) (Fig. 26.6d). Correlation of As_{tot} with both HCO_3^- and DOC was

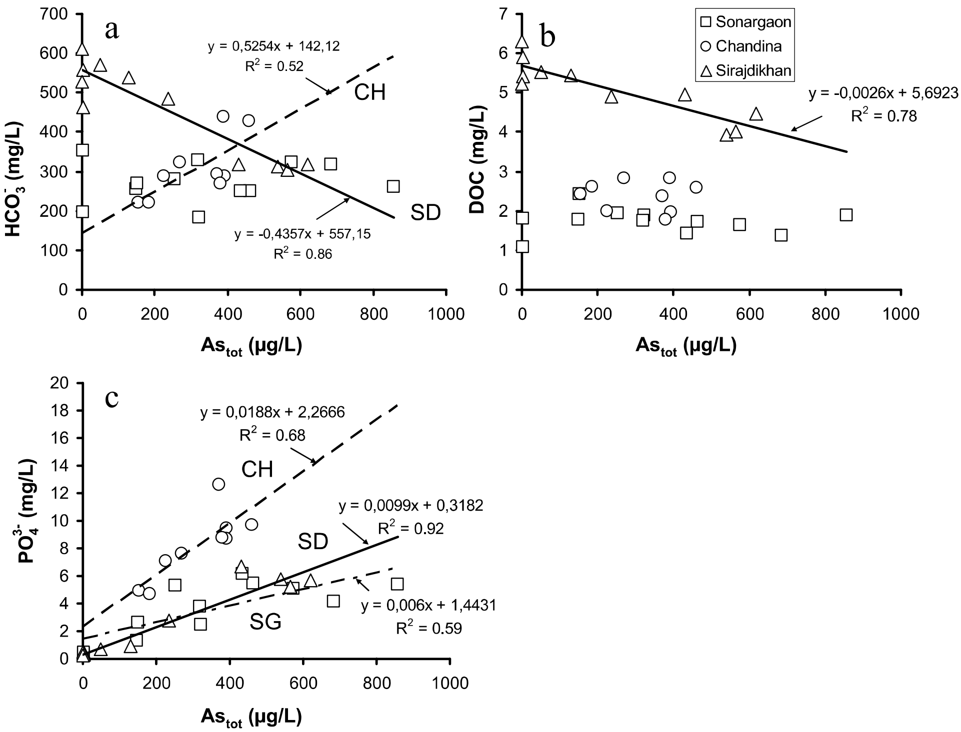

Figure 26.7. Relationship between As_{tot} and a) HCO_3^-, b) DOC and c) PO_4^{3-} in groundwaters. The regression lines ($p \gg 0.01$) are labeled: Sonargaon (SG), Chandina (CH) and Sirajdikhan (SD).

high and negative (R^2 values 0.86 and 0.78 respectively), and correlation of As_{tot} with PO_4^{3-} was fairly high ($R^2 = 0.59$) (Fig. 26.7a–c).

The weak or negative correlation noted between the Fe_{tot} and As_{tot} at Sonargaon and Chandina sites may result from precipitation of siderite ($FeCO_3$), whereas precipitation of vivianite ($Fe_3(PO_4)_2 \cdot 8H_2O$) may affect the correlation between Fe_{tot} and PO_4^{3-}. Both minerals act as a sink for Fe^{2+} ions in anoxic groundwater with high alkalinity and PO_4^{3-} concentrations (see discussion later). While the source for As is geogenic (Mukherjee & Bhattacharya 2001, Bhattacharya et al. 2002a, b, 2006), the source for high PO_4^{3-} concentrations is not well understood. Phosphate can be released during oxidation of organic matter, and may elevate the concentrations of PO_4^{3-} in groundwater.

26.4.4 *Speciation modeling*

Speciation of groundwater samples was calculated using program PHREEQC (Parkhurst & Appelo 1999) and MINTEQ database (minteq.dat) which includes thermodynamic data for As. Thermodynamic data for Ca-arsenate minerals and complexes were taken from Bothe & Brown (1999) and data for formation of Mg complexes with arsenic from Whiting (1992). The modeling results revealed that the groundwater was undersaturated with respect to all As minerals. Saturation indices for samples from Sonargaon site ($n = 12$, Fig. 26.8a). Most of samples are undersaturated with respect to calcite (average SI −0.27, variance 0.11), which suggests that this mineral is not generally a sink for dissolved inorganic C. Calculated values of log P_{CO_2} (average −1.62, variance 0.05) are high, probably as a consequence of CO_2 generation in redox reactions with organic matter. In general, the groundwaters were supersaturated with respect to rhodochrosite, ($MnCO_3$, average SI 0.28, variance 0.13), siderite ($FeCO_3$, average SI 0.63, variance 0.37), and vivianite ($Fe_3(PO_4)_2 \cdot 8H_2O$, average SI 1.22, variance 5.51) which indicate that these minerals are a potential sink for aqueous Mn(II) and Fe(II). The highest degree of supersaturation is reached for vivianite at this site, however with great deal of variability.

Results from Chandina site ($n = 9$, Fig. 26.8b) reveal a situation similar to Sonargaon site for most minerals. Groundwater is again undersaturated with respect to calcite (average SI −0.61, variance 0.07) and supersaturated with respect to siderite (average SI 0.79, variance 0.07) and vivianite (average SI 2.58, variance 0.81). Calculated P_{CO_2} values are again high (average −1.56, variance 0.56), suggesting that CO_2-generating redox reactions are also under operation at this site. Principal difference between both sites is in negative SI values for rhodochrosite (average SI −0.61, variance 0.07) at Chandina site.

Results from Sirajdikhan site ($n = 9$, Fig. 26.8c) reveal that the groundwater is at equilibrium with respect to calcite (average SI 0.03, variance 0.07) and generally is supersaturated with rhodochrosite (average SI 0.36, variance 0.24) and siderite (average SI 0.15, variance 1.11). In contrast to the other two sites, groundwaters are generally undersaturated with respect to vivianite, but scatter of SI values is considerable (average SI −1.37, variance 19.68). Calculated log P_{CO_2} values are the highest of all sites (average −1.37, variance 0.18). At all sites groundwater is strongly undersaturated with respect to amorphous $Fe(OH)_3$(a) and slightly supersaturated with respect to crystalline goethite (not shown). Thus, poorly crystalline As adsorbents as $Fe(OH)_3$(a) with large surface area cannot precipitate from ground water. At both sites As(V) complexes with Ca and Mg may comprise up to 15% of total As(V) concentration. Fe(II) is mostly present as free ion Fe(II), but $FeHCO_3^+$ complex may comprise up to 21% of total Fe(II). The differences in the SI values for rhodochrosite in Chandina site with the values observe for Sonargaon and Sirajdikhan sites is consistent with lower

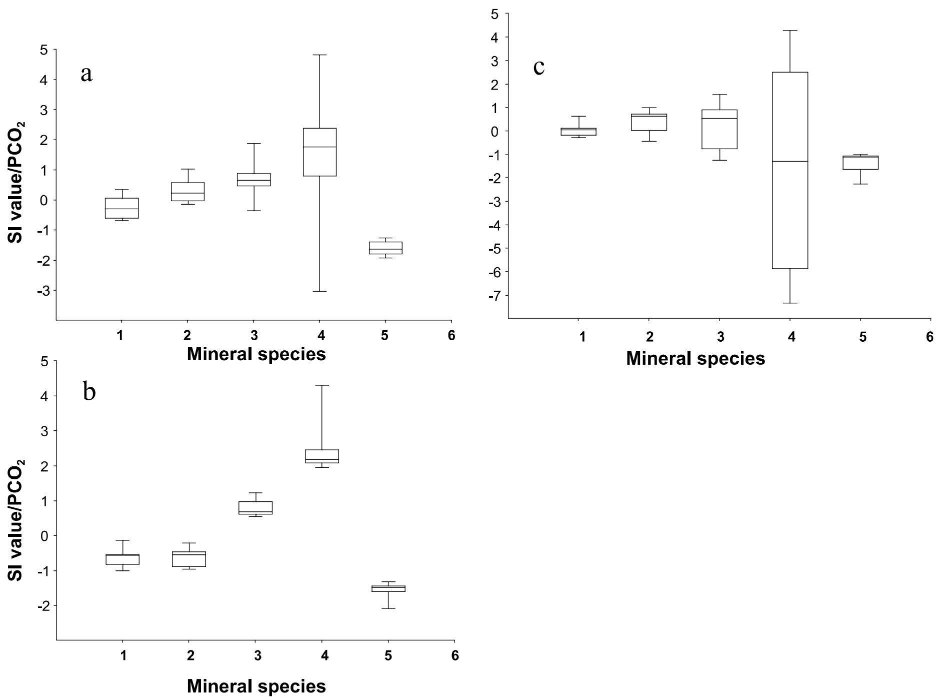

Figure 26.8. Range of SI and log P_{CO_2} values for samples from a) Sonargaon, b) Chandina and c) Sirajdikhan sites. The mineral species (x-axis) are represented as: 1-calcite, 2-rhodochrosite, 3-siderite, 4-vivianite, 5-log P_{CO_2}.

dissolved Mn concentrations at Chandina site and indicate that Mn(IV) redox buffering step was less significant here. At Sirajdikhan site, concentrations of PO_4 are considerably low but variable, that might account for the undersaturation of many samples with respect to vivianite and the wider range of variation of the SI values.

26.5 DISCUSSION AND CONCLUSIONS

The status of natural As occurrences in groundwater is a matter of public health concern in the BDP of Bangladesh. The hydrogeochemical data for groundwaters of the BDP sedimentary aquifers at the three investigated sites suggest their predominantly reducing character, with high HCO_3^-, and low SO_4^{2-} as well as NO_3^- concentrations. Oxidation of organic matter often results in elevated HCO_3^- levels (Mukherjee & Bhattacharya 2001, Bhattacharya et al. 2006, Hasan et al. 2007). Sulfate reduction is indicated by low SO_4^{2-} concentrations and the presence of detectable sulfides in the shallow groundwaters elsewhere in Bangladesh (Broms & Fogelström 2001, Hasan et al. 2007). Further SO_4^{2-} concentrations do not show any definite correlation with As_{tot}, which indicates that the oxidation of pyrite is not the prevalent geochemical process for As mobilization in the aquifers. Elevated NH_4^+ concentrations in these groundwaters reflect dissimilatory nitrate reduction in the shallow aquifers triggered by microbial degradation of organic matter (viz. Hoque et al. 1998, BGS & DPHE 2001). Redox levels for sulfate reduction is fairly low, sufficient for the reduction of Fe(III) and Mn(IV) in the aquifer sediments (Bhattacharya et al. 1997, 2002b) as evidenced by the presence of dissolved Fe and Mn in these groundwaters. However, aqueous concentrations of both Fe and Mn in groundwaters seem to be controlled by the precipitation of Fe(II) minerals such as siderite ($FeCO_3$), vivianite ($Fe_3(PO_4)_2$ $8H_2O$), and the precipitation of Mn(II) mineral rhodochrosite ($MnCO_3$). It is interesting to note that the presence of siderite and rhodochrosite in the sediments was indicated by the selective acetate extraction while acetate extractable P fractions concomitant with Fe indicated the dissolution of vivianite in the reducing aquifer sediments (Dodd et al. 2000, Ahmed et al. 2004, Hasan et al. 2007).

Oxidative degradation of organic matter and DOC seems to be the potential mechanisms for an increase in the levels of HCO_3^- in the groundwaters (Artinger et al. 2000, Appelo et al. 2002). Relatively good correlation between DOC and HCO_3^- at depths <40 m at Sonargaon and Chandina (Table 26.1) sites suggests that HCO_3^- is generated by oxidative degradation of DOC. However, these trends are not evident below 40 m depth and it seems that other processes also contribute to high HCO_3^- concentrations. Similar coupled behavior of DOC and HCO_3^- at shallow depths and their de-coupling in deeper aquifers has been inferred at several sites in Bangladesh and West Bengal (viz. McArthur et al. 2004, Horneman et al. 2004, Mukherjee et al. in press). Several electron-accepting processes seem to operate simultaneously, for example NO_3, Fe(III), and SO_4 reduction. The generic relationships between HCO_3^- and Fe_{tot}, As_{tot}, DOC as well as NH_4^+ in groundwater imply that several terminal electron accepting processes operate simultaneously in the aquifers. One of reasons may be variable reactivity of Fe(III) minerals and their coating by precipitated Fe(II) minerals observed at petroleum hydrocarbons contaminated sites (Vencelides et al. 2007). Reductive dissolution of Fe(III) in the coarse aquifer sediments also results in the release of adsorbed PO_4 along with As into groundwater.

Reductive dissolution processes of Mn(IV) and Fe(III) minerals seem to be de-coupled as indicated by different depth concentrations of dissolved Fe and Mn particularly evident at the Sirajdikhan site, which may be possibly related to the sedimentation pattern. Furthermore, at this site maximum dissolved As concentrations correspond to maximum Fe_{tot} concentrations, but not to maximum dissolved Mn concentrations. This is consistent with assumed As re-adsorption after the dissolution of Mn(IV) minerals as discussed in previous studies (McArthur et al. 2004, Horneman et al. 2004). Similar trends for Fe_{tot} and Mn are observed at the Sonargaon site, but they are less evident at Chandina site, probably due to a lower abundance of Mn(IV) minerals in solid phase. This is consistent with generally negative SI values for rhodochrosite at this site and indicates that Mn(IV) redox-buffering step was circumvented at this site as compared to other sites in the Meghna

Flood Plain and elsewhere (von Brömssen et al. 2007, Hasan et al. 2007, Mukherjee et al. in press). Further detailed studies are thus required to improve our understanding of the mechanism of As mobilization at the Holocene flood plains in Bangladesh.

ACKNOWLEDGEMENTS

This study was supported by the research project funded by Australian Council of International Agricultural Research and the CSIRO's Contaminated Land and Water Environments Program at Adelaide, South Australia. PB thanks the CSIRO for providing the necessary funds for the field work in Bangladesh during January 2002 and the other infrastructural facilities. PB would also acknowledge the financial assistance from the Swedish Research Council (VR) in the form of a travel grant, Swedish International Development Agency (Sida-SAREC) and Swedish Research Council (VR-Sida), Swedish Asia Research Link Program through research grants SWE-1998-193 and 348-2003-4963 respectively on the genesis of high-As groundwaters of Bangladesh. OS was partly supported by the grant MSM0021622412 of the Czech Ministry of Education, Youth, and Sport.

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CHAPTER 27

High fluoride groundwater of Karbi-Anglong district, Assam, Northeastern India: Source characterization

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ABSTRACT: India is one among the 30 countries, where people consume groundwater with more than 1.5 mg/L of fluoride, an upper permissible limit prescribed by WHO. Assam falls in moderately endemic zone with very high fluoride occurring in pockets in certain areas. In the area under investigation fluoride concentration in groundwater varies from 0.51 to 14.7 mg/L. In contrast to other fluoride affected regions of the world, this area receives high rainfall. Granites, granitic gneisses and phyllitic quartzite are the main hard rock aquifers which contain fluorine bearing minerals such as fluorite, biotite and muscovite. Higher concentration of fluoride in groundwater is found to be associated with granites and controlled by regional shear/fault system.

27.1 INTRODUCTION

Nearly 70% of rural world population suffering from fluorosis depends on groundwater. According to WHO (1984) guidelines, in areas with a warm climate, the optimal fluoride concentration in drinking water should be below 1 mg/L, while in colder climatic regions it could be up to 1.2 mg/L. Since the threshold limit of fluoride to cause dental/skeletal fluorosis is 1.5 mg/L, WHO set the upper limit of fluoride in drinking water at 1.5 mg/L. However this guideline is not universal. For example in India this value is set at 1 mg/L (WHO 1994). Those affected by fluorosis are from developing/under developed countries whose main source of drinking water is provided by shallow and deep aquifers. Groundwater occurring in granite contains high levels of fluoride relative to the water occurring in unconsolidated sediments overlying the granites (Liu & Zhu 1989, 1991, Yong & Hua 1991, Saether et al. 1995). Sometimes surface water draining hornblendegneiss terrain in the catchment area provide high fluoride content (>1 mg/kg) to the groundwater (Ren & Jiao 1988, Liu & Zhu 1991). Groundwater with fluoride levels more than 1.5 mg/L, in general, occur in rocks of granitic composition that contain fluorine bearing minerals such as fluorite, hornblende (F: 5–7%), muscovite (F: 2%) and biotite (F: 5.2%) (Yong & Hua 1991). The solubility of fluorite relative to other fluorine bearing minerals is low (Zuane 1990, Appelo & Postma 1993), hence granites with mica and horn-blende appear to be a potential source of fluoride in groundwater (Chaturvedi-Hema & Chandrasekharam 2004). Fluorosis is commonly reported in arid regions with low rainfall and high evapotranspiration where the weathering rates are high, making the fluorine bearing minerals more susceptible to break-down. Low rain fall also promotes long residence time thus enhancing the dissolved ions in groundwater. Due to this reason, incidence of high fluoride in groundwater and fluorosis are common in semiarid/arid regions. Major structures like faults and shear zones, that form loci of mineral particles with large surface area, also play an important role in enhancing fluoride levels in groundwater occurring in granite aquifers (Kim & Jeong 2005). There are cases where high fluoride content in groundwater is reported to be due to leaching re-action between meteoric water and loess deposits, transported from other regions (Kafri et al. 1988a, 1988b, 1989).

Nearly 62 million people, living in the arid and semi-arid regions of India, drink groundwater with fluoride levels >1.5 mg/L and suffer from dental and skeletal fluorosis. Although areas with high rainfall are believed to be free from such problem, Assam, that receives ~2000 mm of rain annually, falls under moderately endemic state with respect to fluorosis (Chaturvedi-Hema & Chandrasekharam 2004). Several cases of dental and skeletal fluorosis have been reported from certain parts of Assam (e.g. Karbi-Anglong). An area of ~200 km^2 within the Karbi-Anglong district in Assam, has been chosen for investigation. Due to high rain fall, groundwater occurs in abundance in shallow aquifers but the water is turbid and hence the local population depends on tube well water drawn from greater depths. The tube well water is crystal clear but contains high fluoride (0.3 to 14.7 mg/L) and iron (around 7 mg/L). Shallow dug wells and surface water have fluoride below permissible limit of 1.5 mg/L. Economically very backward population that lives in remote villages, with stream as their drinking water source and those using dug well water are less affected by fluorosis.

27.2 GEOLOGY AND HYDROGEOLOGY

27.2.1 *The study area*

The study area, covering 200 km^2, falls in Karbi-Anglong district (92°56′–93°20′E; 26°10′–26° 24′N) of central Assam (Fig. 27.1). This area is drained by Dikharu River, a tributary of the Jamuna River. The geomorphological units here comprise of alluvial plain and piedmont zone along denudated hills. The alluvial plains are having very low gradient, well indicated by highly meandering Dikharu and Jamuna rivers and Ox-bow lakes. The climate is humid tropical rain-forest type with temperature varying between 6 and 32°C from winter to summer and the average rainfall is 1,250 mm/ year. The above two rivers are controlled by a regional NE-SW and NW-SE trending fault/lineament reported by Mazumdar and Sharma (1999) and GSI (2000).

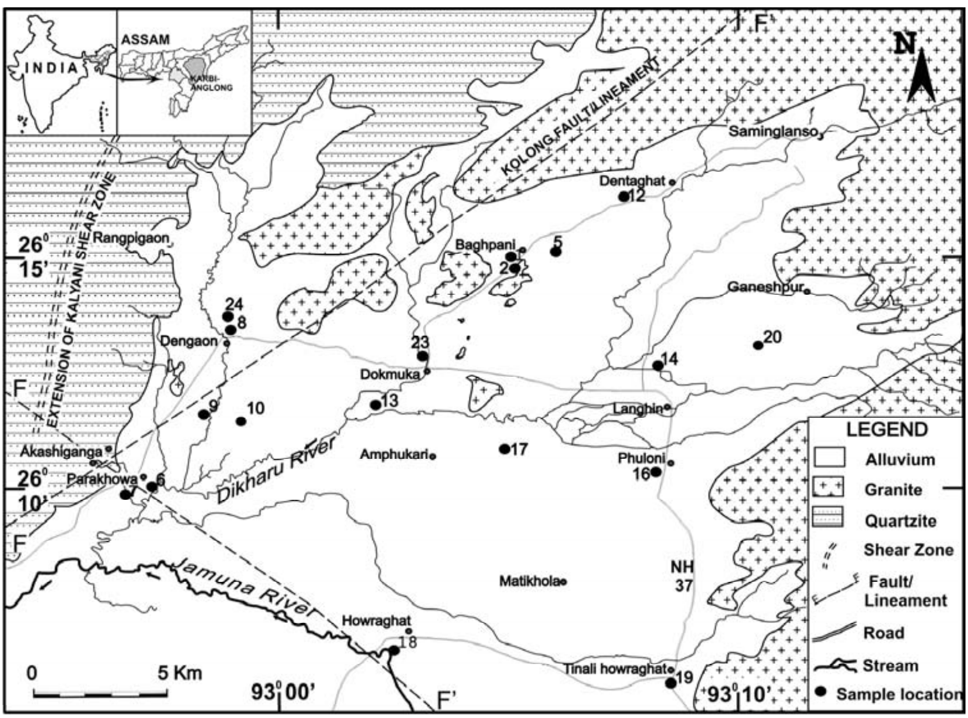

Figure 27.1. Geological and sample location map of the study area (modified after Mazumdar and Sharma (1999), GSI (2000) and CGWB (2002).

27.2.2 *Geological and hydrogeological characteristics*

The lithology exposed in the area includes granites (~500–700 Ma; henceforth known as younger granites), granite-gneisses, micaceous quartzite with intercalated phyllite (phyllitic quartzite) of Shillong Group of rocks of Proterozoic age, that are overlain by recent to sub-recent sediments (Das Gupta & Biswas 2000). The younger granites occur as isolated inselbergs around Bagpani (Fig. 27.1). The thickness of the recent to sub-recent sediments varies depending on the geomorphology. Along the hill slopes it is about 15–30 m while in the other parts it is greater than 80 m. Two prominent structures traverse the area. The Kolong fault with NE-SW trend passes towards west of Bagpani while the Kalyani shear with near N-S trend cuts the Kolong fault near Parakhowa (Fig. 27.1). A typical bore-hole lithology at Bagpani and Dengaon indicates that the top sandy and a deeper weathered granite aquifer is separated by a clay zone (Fig. 27.2). The tube wells appear to be tapping the deeper aquifer. The water table in this upper zone is shallow and occurs at a depth of 1–1.5 m. Many artesian tube-wells are present in the area, indicating the presence of semi-confined or confined aquifer systems. The sediments in the valleys are fine grained while near the foot hills they are coarse (CGWB 2002).

27.2.3 *Petrographic characteristics*

Petrographically, phyllitic quartzite, from the western side of the study area (Fig. 27.1) show mylonitization, indicating the SW extension of the Kalyani shear zone reported by GSI (2000).

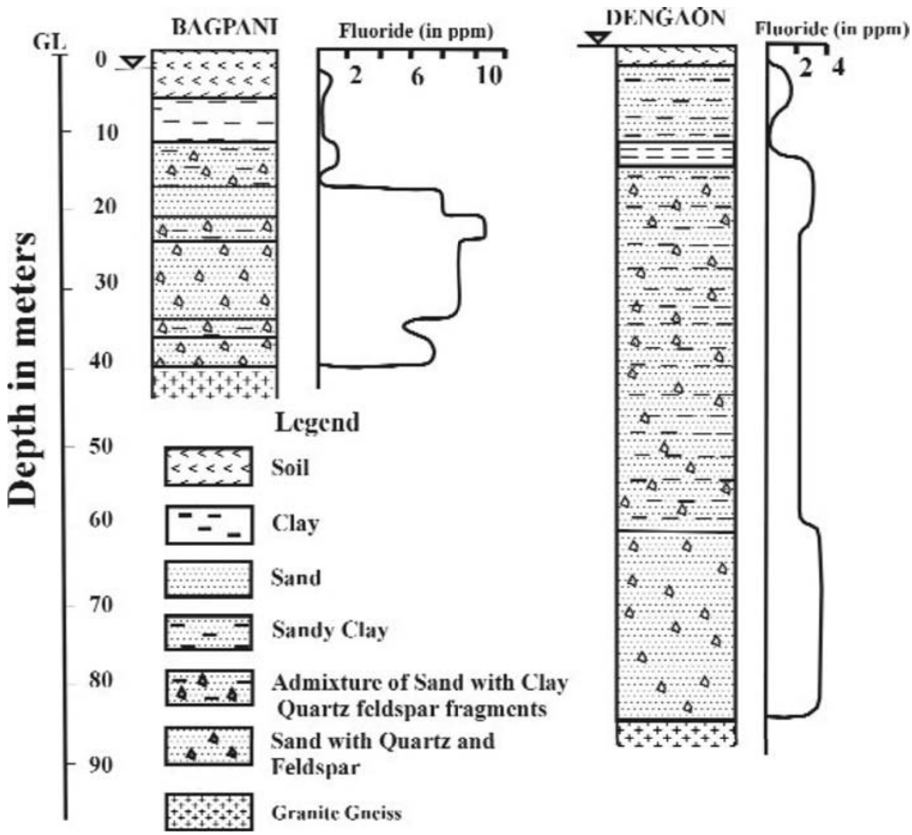

Figure 27.2. Borehole logs of tube wells at Bagpani and Dengaon together with fluoride variation in the water samples with depth (borehole logs modified from CGWB 2002).

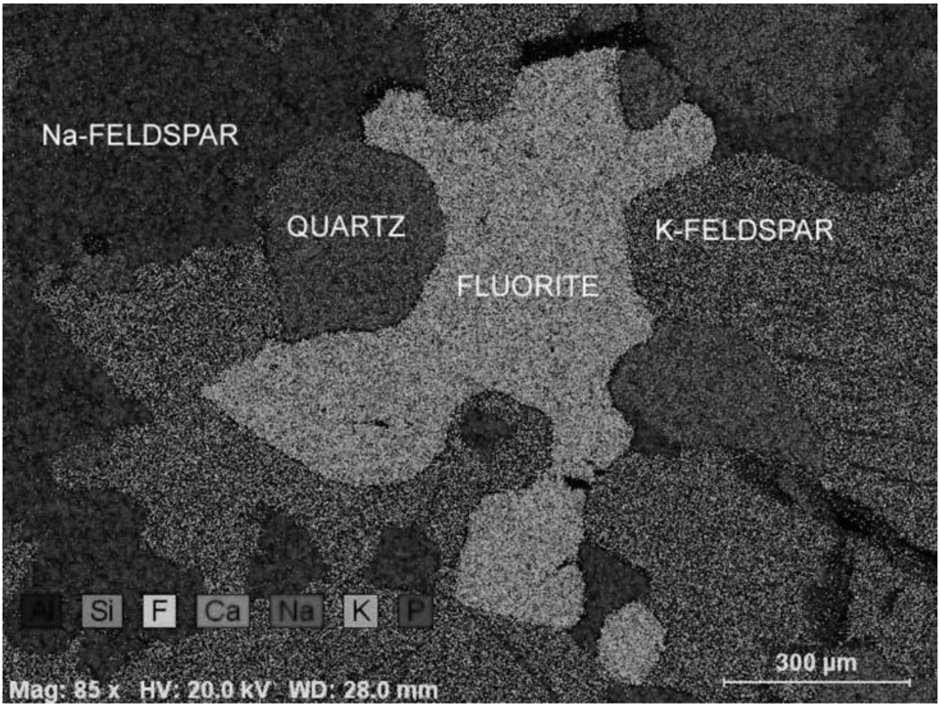

Figure 27.3. SEM-EDAX map showing the anhedral fluorite mineralization in granite near Bagpani.

Presence of elongated quartz grains in the younger granites, with undulose extinction, give evidence of dynamic recrystallization of the grains. Anhedral fluorite, appears to have been formed from secondary fluids (Kerr 1959), is found between quartz and feldspar in granites (Fig. 27.3). This evidence indicates that the region has undergone tectonic disturbance after the emplacement of the younger granites. Biotite and muscovite are the two fluorine bearing minerals present in the granites. The Phyllites, besides muscovite and biotite, contain euhedral grains of pyrite.

27.3 MATERIALS AND METHODS

Twenty four tubewell water samples and representative rock samples (phyllites intercalated with micaceous quartzite and granite) have been collected during the course of the field work. The groundwater samples were collected in 2004 in the months of February (pre-monsoon), June (monsoon) and October (post-monsoon). Water samples were collected in clean polyethylene bottles, which were washed with distilled water and subsequently rinsed with representative water samples. Samples were filtered and two bottles of 250 ml from each sampling location were collected, one acidified with 2% HNO_3 and other non-acidified. Major ions and fluoride content in all the water samples have been determined using Atomic Absorption Spectrophotometer and Orion ion selective electrode respectively. Analytical methods recommended by APHA (1981) were followed in analyzing the samples. Chemical analysis of rocks, major elements and fluorine was carried out using XRF; it was crosschecked using wet chemical analysis, after digesting the samples following the procedure recommended by Shapiro & Brannock (1982) and Bodkin (1977).

Table 27.1. Chemical analysis of tube well of water samples (values in mg/L).

Season	pH	TDS	Na^+	K^+	Ca^{2+}	Mg^{2+}	Cl^-	SO_4^{2-}	HCO_3^-	F^-
Pre	7.8	448.5	100	2.00	20.04	2.42	86.8	100.9	115.0	10.0
Mon	7.7	458.3	110	2.50	15.87	7.21	43.4	106.5	126.0	10.2
Post	7.2	338.3	110	1.80	14.03	4.85	43.7	100.8	106.9	09.8
Pre	7.9	409.5	100	1.60	12.02	2.42	67.5	66.0	145.0	12.7
Mon	7.6	413.3	110	1.70	11.90	4.81	33.8	79.9	152.3	12.0
Post	7.8	427.5	110	0.80	14.03	1.21	34.0	80.1	151.9	11.5
Pre	7.9	456.0	110	1.20	24.05	14.6	96.0	108.5	100.0	14.4
Mon	7.4	477.0	120	0.50	11.90	4.81	67.50	116.9	89.30	14.7
Post	7.2	216.8	37.5	0.80	8.02	7.29	14.86	17.0	174.4	1.9
Pre	7.3	220.5	60.0	2.80	8.02	1.21	24.0	11.6	140.0	7.8
Mon	7.9	225.8	34.5	3.00	15.87	2.40	9.64	10.3	141.8	7.82
Post	7.6	232.5	55.0	2.10	10.02	3.64	9.71	8.1	135.0	6.08
Pre	7.4	209.3	50.0	2.80	8.02	4.86	29.0	14.70	135.0	5.82
Mon	7.9	227.3	31.4	3.50	23.81	16.8	14.5	13.86	147.0	9.48
Post	7.1	189.0	25.3	2.80	8.02	9.72	12.1	5.60	174.4	2.77
Pre	7.5	270.8	67.0	1.00	12.02	4.85	24.0	14.05	126.0	2.37
Mon	8	307.5	65.0	0.50	14.03	12.1	14.6	22.39	173.3	2.2
Post	7.4	281.3	65.0	0.50	14.03	3.64	9.71	11.40	180.0	2.384
Pre	7.5	266.3	80.0	0.80	12.02	2.42	9.60	9.5	185.0	3.74
Mon	7.8	289.5	70.0	0.50	11.90	4.81	19.3	12.7	199.5	4.162
Post	7.4	276.8	70.5	0.60	10.02	3.64	12.1	16.8	174.4	3.68
Pre	7.6	327.8	95.4	0.80	16.03	2.42	9.60	4.66	235.0	2.90
Mon	8.0	352.5	110	0.50	12.02	3.64	12.1	12.8	246.8	3.21
Post	7.8	349.5	110	0.40	12.02	3.64	12.1	11.4	236.3	5.29
Pre	7.1	264.0	62.5	1.60	16.03	2.42	19.0	04.18	180.0	1.57
Mon	7.4	267.8	62.0	1.40	19.84	9.61	4.82	15.46	189.0	1.7
Post	7.5	270.0	62.0	1.40	10.02	7.29	7.28	07.62	174.4	1.6
Pre	7.2	307.5	62.4	1.10	20.04	7.28	9.64	04.40	215.0	0.95
Mon	7.4	321.8	50.5	1.20	20.04	16.83	4.82	17.74	220.5	0.9
Post	7.4	308.3	62.0	0.80	22.04	6.06	7.28	10.12	196.9	1.0
Pre	6.4	301.5	50.0	1.20	20.04	17.0	14.4	04.34	200.0	0.9
Mon	7.3	293.3	42.5	0.50	51.58	19.2	19.3	10.32	220.5	1.0
Post	7.1	292.5	42.0	0.80	20.04	10.9	7.28	12.79	185.6	0.8
Pre	6.4	269.3	31.5	0.80	36.07	9.70	9.64	4.68	185.0	1.39
Mon	7.7	213.8	22.5	1.00	36.07	12.13	19.3	10.38	173.3	0.96
Post	6.5	382.5	27.5	0.80	42.08	20.6	55.9	11.24	174.4	0.62
Pre	7.3	269.3	75.5	1.20	28.06	21.9	9.64	04.88	205.0	1.1
Mon	7.7	269.3	37.5	1.20	27.78	16.8	24.1	09.72	196.9	1.9
Post	7.0	281.3	37.5	0.80	18.04	20.6	7.28	06.50	191.3	0.69
Pre	7.5	328.0	90.0	1.20	28.06	21.8	9.64	04.97	385.0	1.15
Mon	7.6	345.6	110	1.20	28.06	21.86	4.82	09.60	404.3	1.2
Post	7.6	385.0	80.5	1.00	12.02	21.9	4.86	07.95	331.9	1.1
Pre	7.1	338.3	50.0	1.02	36.07	12.1	14.5	04.37	240.0	0.89
Mon	7.8	376.8	110	1.50	20.04	24.2	9.64	16.72	394.2	1.21
Post	7.3	338.5	92.5	0.80	23.81	19.2	12.14	10.51	315.0	1.11
Pre	6.4	292.5	25.0	1.60	48.10	14.6	9.64	04.74	205.0	0.89
Mon	8.0	342.8	70.0	1.00	28.06	21.8	9.64	10.26	252.0	0.92
Post	7.0	339.0	45.0	1.20	20.04	21.8	4.86	07.95	241.9	0.78
Pre	7.2	244.5	34.5	1.40	20.04	20.6	9.64	04.26	190.0	1.19
Mon	7.3	291.75	26.5	1.70	28.06	21.86	9.64	19.13	210.0	0.51
Post	6.6	267	37.5	0.90	20.04	13.7	7.28	09.47	180.0	1.06
Pre	7.4	330.0	70.0	1.20	24.05	7.28	19.3	03.20	250.0	1.33
Mon	7.7	264.0	35.0	1.20	20.04	10.93	4.82	09.12	194.3	1.14
Post	7.2	267.0	37.5	1.20	20.04	13.7	7.28	09.47	180.0	1.06
Pre	7.5	225.8	60.5	2.00	12.02	9.71	9.64	12.80	160.0	2.77
Mon	7.6	321.0	27.5	5.50	11.90	4.81	9.64	17.86	231.0	3.17

Note: Pre: Pre-monsoon; Mon: Monsoon; and Post: Post-monsoon

27.4 GEOCHEMISTRY

The chemical analyses of water samples are given in Table 27.1. The fluoride content in the samples varies from 0.51 to 14.7 mg/L. The highest fluoride was recorded in the tube wells located near Bagpani (samples 1, 2 and 5; F^-: 9.78–14.7 mg/L). The tube well samples 6,7,8,9 and 10 near Parakhowa also recorded high fluoride content (>2 mg/L) but the levels of concentration is less that those wells located near Bagpani. All the groundwater samples fall in the Ca-Na-HCO_3 field (Fig. 27.4) except samples 1, 2 and 5 that fall in the Na-Cl field. The area received very high rainfall, resulting in flooding in the study area, three days prior to the day of sampling. The similarity of water chemistry between monsoon and post-monsoon seasons perhaps is due to this effect. In Post-monsoon, the samples 5 and 7 are chemically different and also show relatively lower fluoride content (fluoride in monsoon: 14.7 and 9.4 mg/L; pre-monsoon: 14.4 and 5.8 mg/L; Post-monsoon: 1.8 and 2.7 mg/L, respectively) compared to other seasons. This change in chemistry is due to the fact that the depth of these two tube wells was reduced in this season to mitigate high fluoride levels in the water. The shallow dug wells in the area contain <1 mg/L of fluoride (except one dug well in Bagpani that has registered 1.7 mg/L of fluoride) that is well below the limit of fluoride in drinking water recommended by WHO (1994).

The Saturation Indices (SI) of both calcite and fluorite in the water samples were calculated using PHREEQC programme (Parkhurst & Appelo 1999). The saturation indices of fluorite and calcite were plotted against fluoride (Fig. 27.5a, b). The SI (F) increase with increase in F^- (Fig. 27.5a), where as SI (C) does not show any significant trend with respect to F^- (Fig. 27.5b). The water is attaining saturation with respect to fluorite at 5 mg/L of F^- (Fig. 27.5a) while calcite saturation is not observed (Fig. 27.5b). This could be because the source rock itself is depleted in calcium (Table 27.2). Also the pH of the water sample through out the seasons is not high enough to facilitate calcite saturation. Both monsoon and post-monsoon seasons show similar trend due to dilution,

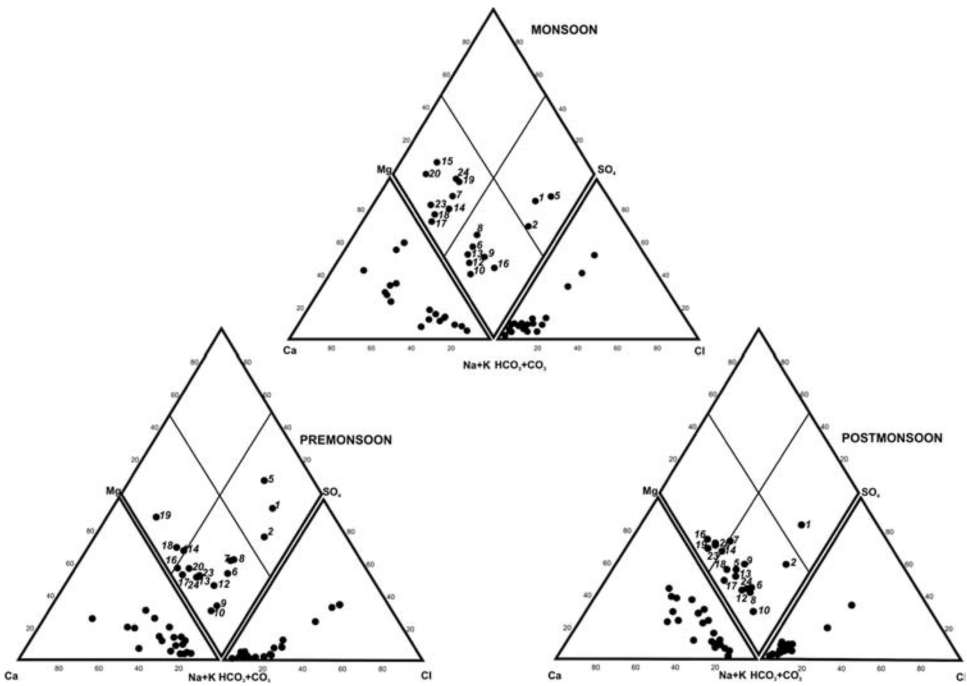

Figure 27.4. Piper trilinear diagram showing chemical characteristics of tube well water from the study area.

as mentioned above, during post-monsoon. The total dissolved iron in the samples vary between 0.9–6.85 and is far above the limit prescribed by WHO (1994). Both granites and the phyllites are enriched in K_2O relative to Na_2O and CaO. Total FeO in the phyllite is higher relative to the other rock types (Table 27.2). The total fluorine in granite is higher (670–1200 mg/kg) compared to micaceous quartzite (270–340 mg/kg) and phyllite (578 mg/kg; Table 27.2).

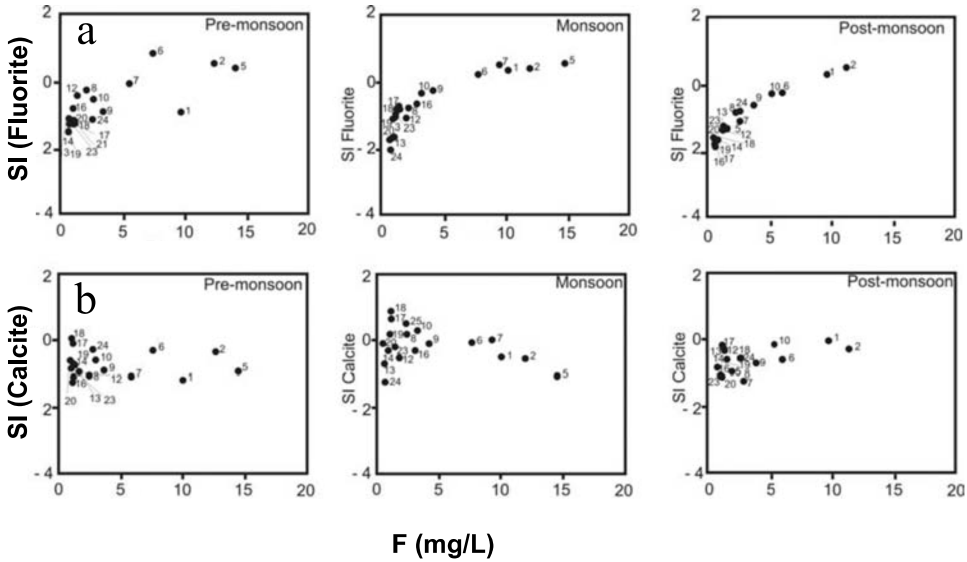

Figure 27.5. Plot of saturation indices of a) fluorite and b) calcite against the concentration of fluoride during the three seasons.

Table 27.2. Chemical analyses of the rock samples.

Oxide %	Gr2	Gr3	PH	Qtz
SiO_2	75.31	73.02	55.41	87.18
TiO_2	0.13	0.28	1.20	0.63
Al_2O_3	13.09	13.89	26.39	6.89
MgO	0.13	0.74	1.03	0.31
FeO	0.10	0.20	0.57	0.18
Fe_2O_3	0.91	1.76	5.16	1.61
MnO	0.05	0.05	0.01	0.01
CaO	0.96	1.49	0.07	0.02
Na_2O	3.39	2.42	0.19	0.22
K_2O	4.78	5.43	6.38	2.07
P_2O_5	0.08	0.11	0.03	0.01
LOI	0.22	0.31	3.15	0.56
TOTAL	99.14	99.69	99.57	99.69
F^- (in mg/kg)	805	1204	578	289

Abbreviations: Gr: Granite; PH: Phyillite lense in quartzite; Qtz: Quartzite.

27.5 DISCUSSION AND CONCLUSIONS

Tube wells in Bagpani (samples 1, 2, 5) and Parakhowa (samples 6, 7, 8, 9 and 10) contain high fluoride content (Table 27.1) relative to the other sampled tube wells. The water from Bagpani (samples 1, 2 and 5) is Na-Cl type with high TDS compared to other locations. The bore-hole data (Fig. 27.2) shows the depth to the granite basement is about 40 m at Bagpani. High Na^{+} content in the water samples (Table 27.1) compared to other cations indicates the source rock to be calcium deficient and sodium rich that is evident from Table 27.2. And also, since the dug wells have lower fluoride content (<1.7 mg/L) compared to the tube-wells (>9 mg/L), it is apparent that the source of fluorine is either the hard rocks and (as evident from Fig. 27.3), their weathered portion lying just below the alluvium. From the subsurface lithology (Fig. 27.2) and Table 27.1, it is clear that the tube wells extending to the depths of the basement are registering high fluoride content (Bagpani and Parakhowa tube wells), though the absolute values are different in the waters from the two areas. This shows the influence of weathered granites in controlling the fluoride content in the tube well waters. Further, it can be observed that all the tube wells with high fluoride content are aligned close to the Kalyani shear/Kolong fault (Fig. 27.1) indicating the influence of the regional structure on the chemical signature of the tube well water. It is quite possible that the Kalyani shear/Kolong fault must be acting as a channel to deeper water in the granites with high fluoride content. It appears that the Kolong fault/Kalyani shear must be the loci for fluorite minerals contributing high fluoride content to the uprising water along these two structures. Such fault system acting as loci for fluorite mineralization has been reported from several regions (Bose & Banerjee 1974, Lahermo et al. 1990). The anhedral fluorite found in the granites (Fig. 27.3) appears to have been crystallized from such uprising, fluoride rich deep waters. Similar situations have also been observed from other high fluoride (in groundwater) terrains in world (Robinson & Kapo 2003, Moore 2004, Kim & Jeong 2005, Ozsvath 2006). Rock fragments with fluorine bearing minerals along the fault/shear zone, such as muscovite and biotite, as described earlier together with the weathered granite, with high surface area may be discharging high fluoride content to the tube wells in Bagpani and Parakhowa. Since Bagpani is close to the Kolong fault (Fig. 27.6), all the tube well near Bagpani has high fluoride content relative to the wells near Parakhowa. The high chloride content in Bagpani tube wells appears to be resulting from mixing of deeper water, with high residence time, from the granite (along Kolong fault). This aspect needs to be confirmed by further investigations through radiocarbon (^{14}C) dating.

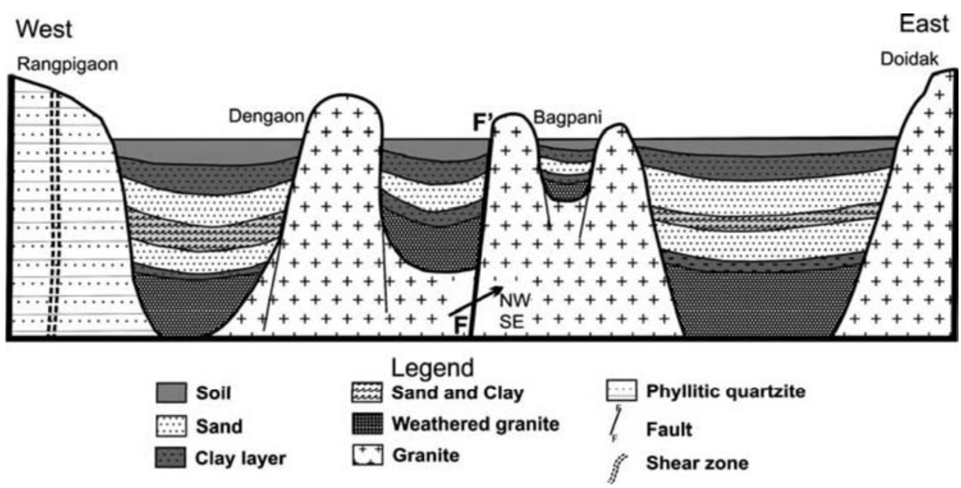

Figure 27.6. Schematic diagram showing the controls of fluoride concentration in the tube wells.

ACKNOWLEDGEMENTS

SS is grateful to Council for Scientific and Industrial Research (CSIR), Government of India for providing a fellowship. The authors thank IIT Bombay for providing the laboratory facilities to carry out the work. Logistic support provided by Mr. Dharam Singh Teron during the field work is gratefully acknowledged. The authors also thank acknowledge Dr. A.B. Mukherjee for his final editorial review.

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CHAPTER 28

Rural Latin America—A forgotten part of the global groundwater arsenic problem?

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ABSTRACT: In Argentina, Chile, Bolivia, Peru, and Mexico at least 4 million of persons depend on drinking water with toxic arsenic concentrations (>50 μg/L), which mostly originate from geogenic sources. In other Latin-American countries the occurrence of the problem and/or the number of exposed persons is yet unknown. This chronic arsenic exposure is associated with neurological and dermatological problems and carcinogenic effects. In contrast to urban areas, practically no action was performed by the authorities to mitigate the arsenic problem for the rural population, which often depends on arsenic-contaminated water as their only drinking water resource. This missing interest hampers the development of low-cost remediation methods for small communities or single houses. So various suitable techniques were developed on laboratory scale, but are only in few cases or even not yet proved and applied on field scale to mitigate the people's arsenic problems. Examples are solar oxidation methods, phytoremediation, or the use natural materials as adsorbents for arsenic removal from drinking water. Hence not a technological problem needs to be solved, but the problem is to convince the responsible authorities to consider the arsenic occurrence as a natural health risk and therefore support the development and the application of remediation methods for rural areas.

28.1 INTRODUCTION

In different countries of Latin America as Argentina, Chile, México, and Peru at least 4 million of persons are permanently drinking water with elevated arsenic concentrations >50 μg/L in a magnitude (>50 μg/L) which converts the issue in some of the countries as in Argentina and Mexico into a public health problem. So e.g. in Argentina (and until 1970 also in Chile) over 1% of the population is exposed to the problem, whereas in Bolivia, Brazil, Ecuador, Costa Rica, El Salvador, and Guatemala, arsenic in drinking water is proved, but the numbers of persons affected are yet unknown. In other Latin American countries, the existence of the groundwater arsenic problem is not yet known. This was for example the case of Nicaragua where the arsenic exposure of population from groundwater and related severe health effects, was detected just two years ago. Additionally it must be taken into account that with advances in the modern analytical methods for arsenic at low levels of concentrations, and with the introduction of new national arsenic limits of (10 μg/L), as already introduced by Nicaragua and being planned to be implemented Mexico, it is expected that in future arsenic will be detected also in several countries with elevated concentrations, where it was presumed until now to be arsenic safe, and that numbers of people exposed will significantly increase.

In most of the these countries, the problem is caused by the occurrence of geogenic arsenic, mostly related to volcanism of the Andes (Argentina, Bolivia, Chile, Peru) and their continuation in Middle America (Nicaragua, Mexico, El Salvador). From these resources, the arsenic is released into the environment (ground- and surface waters, soils, etc.) either by natural dissolution and weathering from the rocks (Argentina, Chile, Bolivia, Peru, Nicaragua, El Salvador, Mexico) or by mining activities (Chile, Bolivia, Peru, Mexico). Other sources of arsenic release, which are of minor and very local importance are artificial, e.g. due electrolytic metal producing processes (Brazil), and agricultural activities (e.g. arsenic containing plaguicides).

This paper comprises three parts. First it gives a country-by-country state of art overview on the occurrence and the respective sources of arsenic in the groundwater and surface water used for drinking purposes in Latin America, and then it addresses the numbers of persons exposed and affected, and the respective already observed and future possible health effects. The third part discusses shortly the experiences from the until now applied remediation methods for both, applications in urban and rural areas, and the future needed measures to mitigate the drinking water arsenic problem of "Rural Latin America", where especially low-cost techniques for small communities and for single households are addressed as possible solutions.

28.2 SOURCES AND PRESENT FORMS OF ARSENIC IN LATIN AMERICA

The arsenic concentrations in the water, mainly in the groundwater, is present by natural process and also through many labour activities as the mining, where the arsenic can be in very dangerous concentrations and the workers are exposed to develop illnesses related with this element. This is a problem of several Latin American countries where the intense mining activity and the natural contamination have generated a high risk for the populations.

Arsenic is typically released to the environment in an inorganic form, and it tends to adsorb strongly to soils. Leaching into subsurface soils is generally not significant, except under reducing conditions. Physical soil characteristics, such as pH, organic carbon content, cation exchange capacity, and iron oxide content, tend to govern the leaching potential. Soluble forms of arsenic in soil may either run off into surface water bodies or leach into shallow groundwater. Arsenic may also be introduced into aqueous systems through natural weathering of soil and rock. The arsenic transport in water depends of the form of arsenic, but also of the interaction with other material. Arsenate is the predominant form in groundwater, although arsenite may be present in significant proportions depending on local geology and water characteristics.

28.3 REGIONAL DISTRIBUTION OF ARSENIC IN LATIN AMERICA

28.3.1 *General*

Since decades, the occurrence arsenic in ground- and surface waters from geogenic sources (including those released through mining activities) is known from Argentina, Chile, Peru and Mexico (Fig. 28.1). Here, common arsenic concentrations in waters are in the range of 10 μg/L to several mg/L. In other countries, as in Bolivia, Brazil, Ecuador, Nicaragua and El Salvador, the problem was detected or investigated within the last few years (Fig. 28.1). So, recently, in Nicaragua—a country where the groundwater arsenic problem was not assumed to exist—elevated arsenic concentrations in groundwater as well as arsenic-related health effects were detected.

28.3.2 *Argentina*

Arsenic contamination in groundwater in Argentina is of primary environmental concern in extended parts of the Chaco-Pampean Plain region consisting of Tertiary aeolian loess-type deposits in the Pampean plain and of predominantly fluvial sediments of Tertiary and Quaternary age in the plains of the Chaco region, all in all covering about 978,634 km^2 (Fig. 28.1) The source of arsenic is geogenic and associated with the Holocene volcanic ash bed (about 6 mg As/kg); the highly soluble rhyolitic volcanic glass component (nearly 20%) of the sand sediments; and sediments originating from metamorphic and acid magmatic rocks (Bundschuh et al. 2004, Bhattacharya et al. 2006).

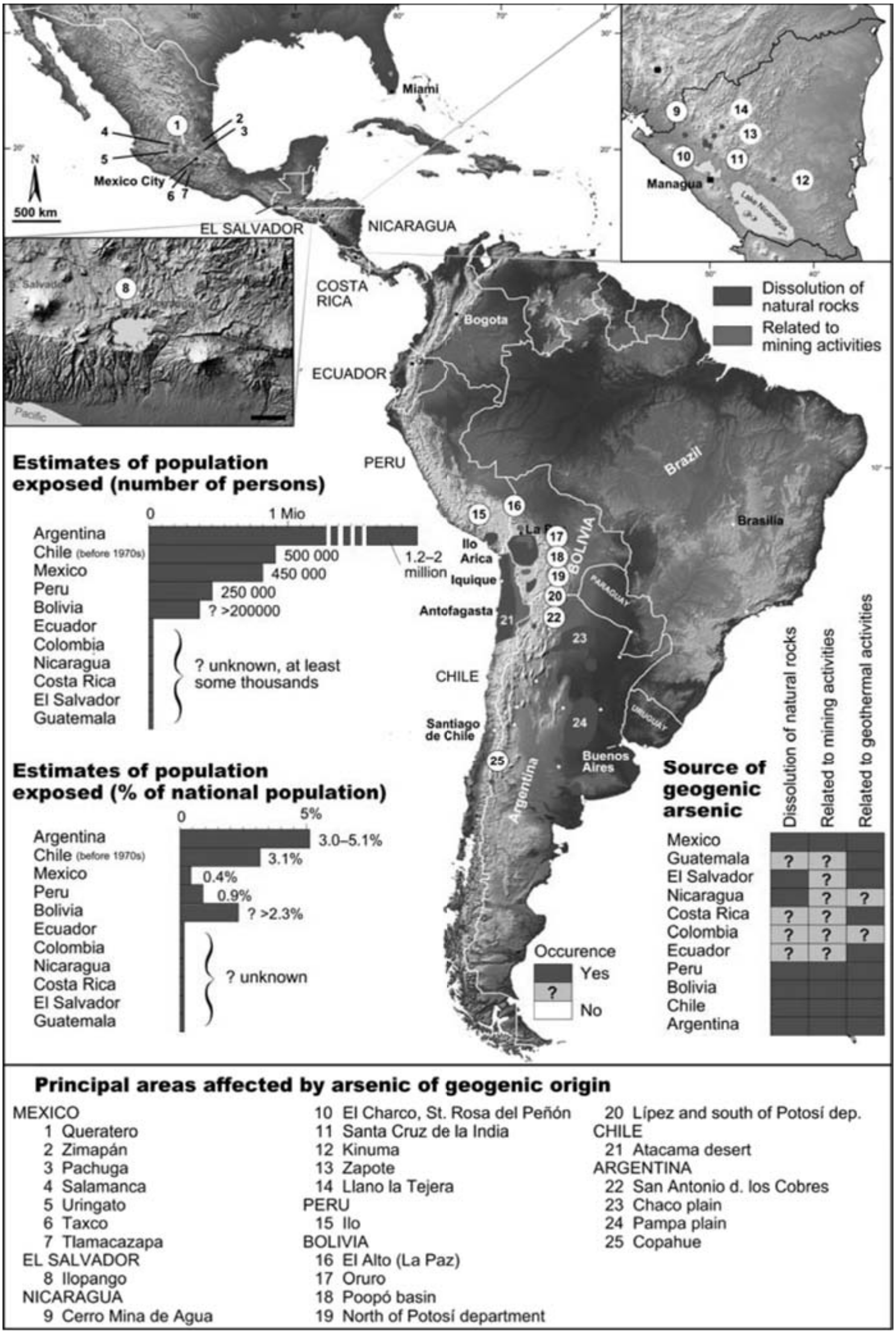

Figure 28.1. Map of Latin America showing location of the principal hotspots of ground- and surface waters with elevated As (>50 μg/L) from geogenic sources, the principal origin and pathway of its release, either by leaching from rocks of favored by mining activities, and the country-by-county data of exposed population. For references see text of respective chapters.

A high variability in the arsenic concentration of groundwater is caused by several hydrogeological and hydrogeochemical factors. This element may be leached from the volcanic glass and is transported into the aquifer. The spatial distribution and variability of arsenic concentrations even in the same (mostly) shallow phreatic aquifer is high. Areas with high groundwater arsenic concentrations were found to correspond to areas with prevailing Na-HCO_3 waters, elevated pH and elevated groundwater residence times; although a more or less homogenous aquifer matrix at the shallow levels characterize the wells exhibiting both low and high concentrations of As in groundwater (Bundschuh et al. 2004).

28.3.3 *Chile*

Several areas of northern Chile count on a long and well-documented history of arsenic exposure by drinking water (Sancha et al. 2004). In this region, the water that supplies most of the towns and villages in the Atacama desert is obtained from rivers, which originate in the Andean mountain range, with its geogenic arsenic sources (Fig. 28.1). Typical arsenic concentrations of the river waters are 0.2 to 0.9 mg/L. The provincial capital Antofagasta (250,000 inhabitants) has had a unique pattern of arsenic exposure. The arsenic concentration in Antofagasta's public drinking water supply rose sharply in 1958 (Hopenhayn-Rich et al. 2000) when a new source for the water supply was used (containing 0.8 mg As/L) and which remained elevated until 1970, when the first arsenic removal plant was inaugurated, reducing As in drinking water to an average value of 0.04 mg/L.

28.3.4 *Bolivia*

The central and the southern part of Bolivia's Andean highland are the areas, with the severest environmental problems related to geogenic arsenic resources. The release into the environment is either by natural leaching/weathering or by mining activities. In Poopó basin (Fig. 28.1), surface water samples from different rivers and Poopó lake have between 90 to 140 μg As/L in areas not affected by mining activities and up to 2000 μg As/L in rivers influenced by mining activities. Arsenic concentrations in groundwater are in the range between 10 to 90 μg/L (García et al. 2006, Bundschuh et al. 2008).

28.3.5 *Peru*

In Peru, geogenic arsenic contaminates present the lagoon of Aricota located in the city of Ilo (Fig. 28.1). This lagoon is feed by the rivers Collazas and Salado River, which pass through the area of Yucamane volcano, whose rocks are the principal arsenic source.

28.3.6 *Nicaragua*

In Central America, for the first time arsenic in groundwater was reported in 1996 from Nicaragua (ECO/OPS 1997). The source of arsenic is purely geogenic and due to dissolution from rocks. Highest concentrations of about 1320 μg/L were found in a tubewell in El Zapote village (Fig. 28.1) (Gomez 2004, 2006, Altamirano Espinoza & Bundschuh 2008).

In 2003, UNICEF studied the arsenic concentrations in groundwater from 77 wells of 5 areas (Barragne-Bigot 2004). This study was delimited in 4 areas (Fig. 28.1) with groundwater arsenic exceeding the national limit of 10 μg/L: (1) Cerro Mina de Agua (municipality Villanueva), (2) communities El Charco and Santa Rosa del Peñón (municipality Santa Rosa del Peñón), (3) community Cruz de la India (municipality Santa Rosa del Peñón), and (4) community Kinuma (municipality La Libertad) (Fig. 28.1). In the about neutral waters (pH 6 to 8) arsenic is predominantly present as As(V) and total arsenic concentrations range from 10 to 107 μg/L. In these areas 1270 persons are exposed to toxic levels of arsenic in drinking water (Barragne-Bigot 2004).

In 2005 UNICEF sampled 54 dug wells in the community of Llano La Tejera (municipality Jinotega; Fig. 28.1), which has 714 inhabitants and where 88% of the houses have their own well. Arsenic concentrations range from the detection limit up to 1200 μg As/L (average 100.4 μg/L) making 87% of the wells unsuitable for drinking purposes (Nicaraguan limit 10 μg As/L) (Bundschuh et al. 2008).

28.3.7 *El Salvador*

In the case of El Salvador, in Ilopango lake (184.9 km^2), whose drainage basin hosts more than 300,000 inhabitants, which use its water even when high concentrations As of 150–770 μg/L make this water unsuitable for human consumption. In Illopango lake, arsenic concentrations range from 0.15 to 0.77 mg/L. Two sources of As in the Ilopango waters can be identified: (1) the internal sediments of the lake that contain As-rich volcanic products of the last eruptions of this caldera, and (2) the material transported to the lake by the Chaguite river, whose As-load originates from leaching and erosion from the volcanic deposits of the Illopango lake basin. (Lopez et al. 2008). The same authors assume that the ash of the last calderic eruption of Ilopango (about 2000 years ago) covers all El Salvador and could be the source of arsenic contamination for other surface and subsurface water bodies and affects other environments.

28.3.8 *Mexico*

In the mining area of Zimapán, the main arsenic-bearing rocks are massive sulfide ores, with pyrite, arsenopyrite, pyrrhotite and others arsenic minerals. Arsenic release into the environment occurs by both, the natural dissolution/weathering of the arsenic rich rocks and by mining activities (e.g. through tailings with up to 22,000 mg As/kg) (Armienta et al. 2004) As consequence, the groundwater in the area of Zimapán has high concentrations of arsenic (190–650 μg As/L; average 380 μg/L) (Armienta et al. 2004).

The Salamanca aquifer system (1900 wells), located in Guanajuato state, is naturally affected by arsenic from geogenic sources, but also from anthropogenic sources, as e.g. the solid wastes of different chemical industries. The highest arsenic concentration observed in groundwater of a well was 280 μg/L (Rodriguez et al. 2004).

In the village of Tlamacazapa (Guerrero state), arsenic release into the environment (groundwater, soil) is natural due to dissolution/weathering of As–bearing rocks. Soil and rock analyses show that arsenic is present at concentrations up to 56 mg/kg and 26 mg/kg, respectively. Arsenic and sewage contaminant concentrations are strongly correlated and the presence of sewage apparently promotes the release of arsenic from aquifer materials. It is likely that arsenic mobilization is the result of a desorption associated with arsenate-phosphate competition for sorption sites. (Cole et al. 2004)

28.3.9 *Other countries*

In other countries, whose geology is in on or the other form related to the Andean or Middle American volcanism, no ore very sparse information on the occurrence of arsenic from geogenic sources in surface and groundwater is existing, calling for further investigations.

28.4 TOXICOLOGICAL PROBLEMS RELATED TO THE ARSENIC ISSUE

28.4.1 *General*

Long-time exposure to As from drinking water is associated with an increased risk for cancer and hyperkeratosis, skin pigmentation, degenerative effects on the circulatory system, neurotoxicity, and hepatotoxicity. However, in Latin America, the problem is not addressed with the requested attention, and the actual number of people at risk for chronic arsenic toxicity is not yet known and that the current status of incomplete knowledge of arsenic occurrence and related health risks deserves serious attention. Another problem is that the national estimates of population to arsenic are using different concentration limits and different analytical techniques, which makes it impossible to compare the data from the distinct countries. So, in Argentina the population exposed to arsenic in drinking water is given as 2 million, given for an arsenic concentration range of 2–2900 μg/L (Sancha & Esparza 2000). In contrast, Bundschuh et al. (2000, 2004) used an arsenic concentration of 500 μg/L, and estimates the exposed population to 1.2 million. In the North of Chile, predominantly in Antofagasta, between 1955 and 1970 about half a million of people were permanently exposed to drinking water with average arsenic concentrations of 600 μg/L. The installation of treatment plants, have reduced these values to an average of 0.040 mg/L (Sancha et al. 1998). In Peru about 250,000 (Esparza 2004), and in Mexico about 450,000 persons are exposed to drinking water exceeding the national limit of 50 μg As/L (Finkelman et al. 1993, Avilés & Pardón 2000). In countries as El Salvador, Ecuador, Nicaragua, no estimates are available, but the numbers are supposed to be less than 50,000 in each of these countries. It must be considered that after the WHO, also the United States EPA has recently (2005) adapted that 10 μg As/L limit. As consequence Nicaragua followed as first country in Latin America, which adapted this value, and Chile is supposed to introduce it in short period, whereas other countries of Latin America discuss whether to introduce it. That means—based on that new limit of 10 μg/L—that the above estimated numbers of exposed population will significantly increase.

28.4.2 *Pathways of human exposure to geogenic arsenic*

Population may be exposed in different way to the arsenic from geogenic resources. The most common and the most studied in Latin America is the uptake through drinking water contaminated by geogenic arsenic. In number of exposed persons, Argentina and Mexico are the most affected countries in Latin America (and located in the third place in the world), followed by Peru and Bolivia. The same regional distribution is valid for the importance of the uptake of arsenic through food, whose arsenic concentration can be increased due to different factors: (1) uptake of As by crops from arsenic rich irrigation water or from arsenic rich soils, (2) use of arsenic rich water for preparing the food. In contrast, the exposure to solids containing geogenic arsenic (solids, dusts, atmosphere) is mostly restricted to areas with mining activities (e.g. Andean highland of Argentina, Chile, Bolivia, Peru, and some areas of Mexico), but can also expected in areas, where winds transport arsenic-rich sediments as in the Pampa-Chaco plain of Argentina. Both, the human uptake of geogenic arsenic through the food chain and through airborne dusts and the atmosphere, are practically not yet studied in Latin America and deserve severe future attention.

28.4.3 *Case studies from Latin America*

Astolfi (1982) found in the population of the Argentine province Cordoba, where the groundwater arsenic problem is known since 1913 (up to 800 μg As/L), lesions of querathosis and epithelioms. From 100 patients, all presented different grades of diverse types of lesions related to arsenic ingestion (hiperhydrosis, melanodermia, hyperqueratosis, infection, ulceration and cancer). Astolfi found a clear association between exposure time and severeness of the lesions.

A study of Concha (2001) in Taco Pozo (Argentina; Chaco province) found an average ingestion of 210 μg As/L in drinking water/person, which together with the staple food soup and maize porridge with 300–440 μg As/L (Concha 2001) result in an daily As-uptake of approximately 1–2 mg As/person., which explains the high occurrence of skin cancer observed in that area (M. Biaggini, pers. com. 2005).

In Antofagasta (Chile) water from the Toconce river was introduced in 1958 as the new main source for drinking water. It was not considered that this river water contains approx. 800 μg As/L. As consequence, in 1962, in Antofagasta the first cases of chronic arsenic health effects were reported. After 12 years of high arsenic exposure of about 500,000 persons, an arsenic-removal plant was installed in 1970 by the public water supply company, which led to a subsequent decrease in arsenic concentrations to an average of 40 μg/L in the drinking water.

Cortes et al. (2004) studied arsenic in urine of pupil (8th grade). In Antofagasta, they found arsenic concentrations of 55.2 μg/L for the year 2000 whereas in 1977 the urine concentrations in scholars were significantly higher (71 to 152 μg/L). In an area not affected by the arsenic problem (Santiago de Chile) corresponding values were much lower (13 to 20 μg/L).

Also in Chile, Hopenhayn-Rich et al. (2000) investigated and compared the mortality of two regions, those of Antofagasta, with its well-documented history of high arsenic exposure, and those of Valparaíso (Fig. 28.1) located to the South which can be considered as a low-exposure city. The results indicate an increase of the late fetal, neonatal, and postneonatal mortality rates for Antofagasta, compared to Valparaíso, for those time periods, where highest arsenic exposure by drinking water is reported from Antofagasta.

The first Central American cases of arsenicosis were reported in 1996 in El Zapote, a rural community of Nicaragua (Gomez 2004, 2006, Altamirano Espinoza & Bundschuh 2008) where the people were contaminated for two years (1994–96) by the water from a public tube-well with 1320 μg/L of inorganic arsenic. Arsenic contamination was also detected in private hand wells used before 1994 and after 1996 (45–66 μg As/L). Between July and October of 2002, 111 inhabitants of El Zapote had undergone a complete medical examination, who lived in this community between 1994–1996. Participants were divided into two groups according to the average As concentration ingested during last 8 years: High As ingestion (80–380 μg As/L) and low arsenic ingestion (<80 μg As/L). Keratosis and hyperpigmentation characteristics of chronic arsenicosis were strongly associated with high arsenic ingestion (Gomez 2006). The typical cutaneous manifestations showed by these patients, confirmed the diagnosis of these first collective case of chronic arsenicosis in Nicaragua and Central America (Gomez 2006). The observed significant statistical relation between the high arsenic ingestion and the respiratory effects adds a strong presumption that chronic ingestion of inorganic As could be associated with important damages in this system (Gomez 2006).

In Zimapán (Mexico), a low-income community in a mining area (activities since 400 years) with nearly 15,000 inhabitants, where groundwater is the only drinking water source, for more than 12 years, the inhabitants consumed arsenic-rich water (190–650 μg As/L; average 380 μg/L), causing different health impacts (Armienta et al. 2004).

In the Salamanca aquifer system (Mexico), which is exploited by 1900 wells, high arsenic concentrations of up to 280 μg/L (from geogenic and antropogenic sources) state a risk hazard for the population (Rodriguez et al. 2004). The people of Salamanca were not only exposed to arsenic through the consumption of arsenic rich drinking water, but are also exposed by other ways since they live in the vicinity of industrial areas, which are further sources of arsenic exposure, e.g. by dusts, and exhalations (Rodriguez et al. 2004). Thereby the arsenic risk level is further increased by the presence of various other contaminants due to by their probable combined action. So, e.g. two carcinogens of different origin, arsenic and organic compounds, acting on the same organ or tissue may increase damage. There is no local epidemiological information that may allow the determination of the degree of correlation between arsenic exposure and cancer, although such is supposed to exist.

The people from Tlamacazapa village (Guerrero state) display toxic health effects related to arsenic and other heavy metal exposures through uptake by drinking water from shallow wells (Cole et al. 2004). However, detailed studies are yet missing. The same lack of health impact studies is missing for other regions of Mexico and many sites of Latin America with high exposure to arsenic, prevailing by arsenic rich drinking water.

28.5 EXPERIENCE AND NEEDS IN ARSENIC REMEDIATION AND IN LATIN AMERICA

The arsenic drinking water problem is already solved in most of Latin America's larger urban areas by installing corresponding treatment plants. However many of them are not working properly or are too expensive. So in northern Chile, the provincial capital Antofagasta (since 1970s), Calama, San Pedro, and other cities and bigger towns treat their water successfully using predominantly flocculation by $FeCl_3$ and subsequent filtration. In Peru, in 1982 a treatment plant (using flocculation by ferric chloride) was constructed in the city of Ilo, but has high operation costs and is not working properly (Esparza 2004). In the cities and some bigger towns of Argentina, e.g. in the provinces of La Pampa, Santa Fe, and Santiago del Estero, predominantly coagulation methods and inverse osmosis is applied to remove arsenic from drinking water. All these treatment methods are expensive and often are not working efficiently. Therefore, the concerned countries, mainly Chile and Argentina, are permanently developing new and improving existing systems and methods of arsenic remediation, looking at the same time on the reduction of the treatment costs and the decreasing maintenance needs by improving autoimmunisation.

In contrast to urban areas, practically no action was performed by the authorities or international and bilateral cooperation agencies to mitigate the arsenic problem for the rural population. This makes especially the dispersed living rural population—which depends often on arsenic-contaminated water as their only drinking water resource, and which often is not aware of its toxicity—to the most disadvantaged group and to an emerging target for further actions to reduce the arsenic exposure.

Above methods are in most cases not usable for small communities, and especially not for the disperse living rural population, which is found all over the continent. This is e.g. the case in the Chaco-Pampean plain (Argentina), where about 12% of total population are living in dispersed settlements with less than 50 inhabitants, which belong mostly to the poorest group of the regional population. That requires low-cost remediation methods for small communities of less than 50 inhabitants down to single houses, which further on require only very simple handling and maintenance.

During the last years some new techniques were developed on laboratory scale, but are only in few cases or even not yet proved and applied on field scale. Examples (1) solar oxidation methods, (2) phytoremediation (e.g. using algae *Lessonia nigrescens* Hansen (2004), or lacustrine algae (Bundschuh et al. 2007), or the use of (3) biomass or (4) natural or activated clay and lime as adsorbents for arsenic removal from drinking water.

One of the low-cost technologies is the so-called Solar Oxidation and Removal of Arsenic (SORAS). Countries like Bangladesh have used this technology to reduce arsenic pollution in drinking water at rural communities level, and some others like Chile and Argentina are trying to test and implement this method. It is based on the oxidation and precipitation of arsenic assisted by light in the presence of citric acid. The practical procedure employed is to fill plastic bottles with the contaminated water, then to add some drops of lemon juice to the bottles, and left them to rest under sunlight for a few hours. The capacity of arsenic removal at the household level by this method is well proven provided the respective adaptation of the technology to the geographic reality is made (García et al. 2004).

The application of the SORAS technology in a rural area of Chile was used (Cornejo 2004) for a simple and low-cost decontamination of water of Camarones river used for human consumption and irrigation. Physicochemical analysis were carried to characterize the water were done, variables such as ion, lemon juice, exposure time, in synthetic as well as real samples.

The photocatalytic method using TiO_2 as catalyst is another recently developed low-cost technologies for arsenic removal, which is suitable also for small community and household-scale applications. Nieto (2004) developed a technique, to cover the inside of borosilicate bottles with a cover of TiO_2 and his laboratory experiments confirmed that a continuous treatment of arsenic-contaminated water is possible by means of a system of tubes in bottles filled with water and exposed to sunlight.

28.6 CONCLUSIONS

The contamination of the environment (ground- and surface water, soils, air) by arsenic geogenic origin, which is released predominantly released many Latin-American countries from the host rocks by dissolution/weathering or by mining activities must be constituted as one of the most important topics by its occurrence, impact and the toxicological effects. Solely the arsenic exposure in drinking water affects at least four million of persons, predominantly in Argentina, Mexico, Bolivia, Chile and properly in Nicaragua, El Salvador and other countries, where the number of exposed persons is not yet known.

In the affected areas of Latin America, the chronic arsenic exposure could be associated with neurological and dermatological problems and carcinogenic effects. As group with the highest risk, the poor population of rural areas, especially those living in dispersed form in communities of less than 50 inhabitants down to single houses could be identified. This group often ignores the risk and consumes the water without any treatment.

Through the last decades various methods for arsenic removal from drinking water were developed and suitably applied in urban areas or larger towns of Latin America. However, missing efficiency and the too high cost, require further improvement.

These methods are in most cased usable for small communities, and especially not for the disperse living rural population. That requires low-cost remediation methods for this target group, which additionally must be very simple in handling and maintenance.

For that purpose during the last years different methods were developed and tested on laboratory scale (e.g. solar oxidation, phytoremediation, use of natural adsorbents for arsenic removal as biomass, clays and limestone), but are practically not implemented in the real world to mitigate the people's arsenic problems in Latin Americas rural areas.

Thereby is must be clear, that—at first—it is not a technological problem to be solved. The problem is to convince the responsible authorities to become interested in the problem. No remediation of the rural water arsenic problem will be possible, if not the, the local, national authorities of the affected countries and the bilateral or international cooperation agencies have recognized that (ground) water arsenic in the rural areas of Latin America is one of the most important natural health risks of the present century, and that they are responsible to solve it. They must recognize that groundwater arsenic is an issue and a problem that would challenge the UN Millennium Development Goals of sustainable development on a global scale, and therefore consider doing its utmost to better equip people for life in those parts, where groundwater arsenic affects population and their sustainable development.

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CHAPTER 29

Global arsenic and antimony flow through coal and their cycling in groundwater environment

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ABSTRACT: The widespread occurrence of potentially toxic elements such as arsenic (As) and antimony (Sb) in coal demands to follow their fate in the environment, due to their dissipation through coal mining, processing, and incineration. Pathways of As, mostly from drinking water and food grown using As-contaminated irrigation water are fairly well-known, while experience with Sb is rather scarce, despite the fact that most humans are exposed to low levels of Sb through food, water and air. For several decades, As-related problems with groundwater, soils, and wastes have been addressed in many countries all over the world, especially in South-east Asia. In the U.S. and the EU, Sb and its compounds are classified as highly toxic. Prior to trading coal, policy and decision makers should have accurate information on their trace metal-related quality. We tried to map flows of As and Sb through coal between seven regions of the world for 2003, and to explain the behavior of these elements in groundwater and the effects on water in global economic development.

29.1 INTRODUCTION

Over million years, plant material in swamps and peat bogs was subjected to elevated temperature and pressure. These processes caused physical and chemical changes in the organic material and transformed it into coal. Over time, and under elevated temperature together with tectonic movements in the Earth's crust, coal was further converted to lignite, then sub-bituminous coals and eventually hard coal (Morris 1997). Coal is one of the most important natural fossil fuel resources in the world, and will remain a principal energy source for at least the first half of the 21st century (Finkelman et al. 2002). Coal reserves are reported in 1996 for 100 countries estimating one thousand billion (1×10^{12}) tonnes. Mankind started to use coal as a source of energy already 20,000 years ago. In the 21st century, the world bituminous coal consumption has increased from 4609.86 Mt (million metric tonnes) in the year 2000 to 4933.47 Mt in 2003 (EIA 2005), and the major hard coal producing countries are China, the USA, India, South Africa, Australia, Russia, Poland, Kazakhstan and Ukraine. In these countries, 12% of the production is exported while the rest is used for domestic purposes e.g., heat and electricity, cement and metal production, and other industries (World Coal Institute 2000).

Coal contains more than 120 elements and trace metal(loids) (hereafter "trace metals"). Using advanced analytical methods it is possible to quantify all these elements in coal (Finkelman 1981) and its products, such as coal fly ash. Many trace elements have toxic properties. Still today, there is sparse information about "hidden" (i.e. unaccounted for) flows of arsenic (As), antimony (Sb) and other elements through coal and coal use. Hence, national and international policy makers need accurate information on coal regarding the quality of products before international or national trade occurs. These policy makers need information to control foreign policy, technology transfer strategies, foreign investment prospects, fly ash used and its disposal problems, and its effects on human health and the environment (Finkelman 2004, Tewalt et al. 2005).

Coal quality and its deposition vary from one region to another. The enrichment of the priority toxic elements, like As, Cd, Co, Cr, Cu, Hg, Mn, Mo, Ni, Pb, Sb, Se, Tl, V and Zn may be hazardous to humans or the environment when they are released and mobilised during coal utilization (Finkelman & Gross 1999, Wagner & Hlatshwayo 2005). For many years, top priority has been given to Hg by the scientific community (e.g., Mukherjee et al. 2004), but today As is of major concerns due to its mobilization from sediments to the groundwater and surface water (Matschullat 2000, Bhattacharya et al. 2002, 2004, Mandal & Suzuki 2002, Naidu et al. 2006). Its presence in drinking water, agricultural soils and its uptake by plants and vegetables in many regions of the world has been reported to cause harmful affects to humans, domestic animals, aquatic species, plants and vegetables. Thousands of people are suffering from arsenicosis in south-east Asia, e.g., Bangladesh, West Bengal (India), Nepal, Inner Mongolia (China), Argentina, and many other regions of the world (Mukherjee & Bhattacharya 2001, Ahmed et al. 2004, Wagner et al. 2005). The groundwater As problem has not only caused health disorders among local populations, but also social problems and reduced the economic development in a country. The pathway of As exposure has been primarily through water, but organisms could also be exposed to As via direct inhalation of dust and/or aerosols and through food stuff (Matschullat et al. 2007). The toxicities of both As and Sb have been well documented by Gebel (1997).

Yudovich and Ketris (2005) reviewed As in coal and their study indicates that due to geologic variation in coal basins there are marked differences of trace element concentrations in coal

Table 29.1. Arsenic and antimony concentration (mg/kg) in coal.

Country, bituminous coal	Arsenic	Antimony	References
Global average	10	2	Reimann & de Caritat (1998)
Australia (26 types)	2.8*	0.7*	Meij (2005)
Spain	25; 83[$]		Alastuey et al. (2001)
Western Turkey	676; 1330[$]	388; 8.8–2347	Karayigit et al. (2000)
Total for Turkey	26**; 1.8–620	1.1**; 0.12–41	Palmer et al. (2004)
Bulgaria	1302[$]; 2–58	0.2–16	Kortenski et al. (1999), Eskenazy (1995)
Germany (Ruhr area)	8.2*	1.9*	Meij (2005)
Poland (8 types)	3.7*	1.4*	Meij (2005)
U.S. coal	24.0*	1.2*	Orem & Finkelman (2004)
U.S. coal, average	6.0		Coleman & Bragg (1990)
Canada, Vancouver Island	1400		van der Flier-Keller & Goodarzi (1991)
South Africa	0.9–8.2		Willis (1983)
South Wales, GB	1254		Gayer et al. (1999)
Former U.S.S.R.	25		Kler (1988)
China	4.7**		Yudovich & Ketris (2005)
Southwest China	8300[$]		Zheng et al. (1999)
India, Raniganj coalfield	3.41–6.36		
India, Sohagpur coalfield	[¤]0.15–40; 2.9*	0.07–1.6; 0.63*	Warwick et al. (2001)
CCRs[#]	5−68		Cited by Asokan et al. (2005)

* Arithmetic mean; ** Median value; [$] Maximum, [¤] Moisture basis; [#] CCRs: Coal Combustion Residues.

including As and Sb (Table 29.1). In the 19th century, Daubrée (1851) mentioned As in bituminous coals in France and lignite samples from the German Saar basin coal and trace elements concentration including As in British Newcastle coal. In England and Belgium, high As-concentrations in vegetation in the vicinity of coal-fired power plants have been reported and cattle have died from poisonous effects of As and Sb (Simmersbach 1917). Recently, information of As in Chinese coal from different provinces and the highest concentration of As in Chinese coal was reported to be 8300 mg/kg from a gold mineralized area (South western Guizohou provice) (Zheng et al. 1999). Hence, there are great variations of trace elements in coal as a result of differences in the factors controlling the dispersal and accumulation of elements in coal forming basins, and different relations between geochemistry and sedimentology of coals (Gupta 1999, Wagner & Hlatshwayo 2005).

The present study focuses on the global flux of As and Sb in coal, their concentration, mode of occurrence in coal, on the fate of these elements from coal mining to burning in a high temperature process, their behaviour in groundwater, and how all this affects global economic development. For this study, we divided the globe in seven regions: North America, Central and South America, Western Europe, Eastern Europe and Russia, Middle East, Africa, and Asia plus Oceania.

29.2 MODE OF OCCURRENCE IN COAL

Both As and Sb are chalcophile, with great affinity to sulphur-containing organic and inorganic components in coal. These elements enter the atmosphere with aerosols from sulphur bearing mineral deposits and their related smelting processes, and from the combustion of coal and oil.

Knowledge of the mode of occurrence of hazardous elements in coal is important to reduce their mobilization into the terrestrial and aquatic environments, transporting from one area to another, for example from activities such as combustion for production of heat and electricity. Thus it is necessary to understand the physical and chemical behaviour of As, Sb and other trace elements in coal and their distribution in coal deposits. If the concentration of an element in coal is below 100 mg/kg, it is difficult to determine how the specific element occurs in coal because of the wide variation in elemental modes of occurrence in coal.

Elements are generally highly scattered in coal and are often covalently bound to the organic matrix of the macerals in distinct forms, meaning that an element is localized in specific minerals (Huggins & Huffman 1996). However, the mode of occurrence of trace elements in coal may be indirectly determined by different methods such as: a) sink-float data; b) statistical correlation with other trace element or coal ash; c) geochemical characteristics; or d) from the behaviour during heating or leaching of coal (Finkelman 1994b). The most versatile techniques to explore the mode of occurrence of trace elements in coal are X-ray mineralogical analysis, X-ray absorption fine structure (XAFS) or microbeam analytical techniques (Finkelman 1994b, Huggins et al. 1994, Huggins & Huffman 1996). The mode of confidence for As and Sb analyses in coal are 8 and 4, respectively (the numerical value 10 indicates highest confidence whereas 1 indicates no confidence; Finkelman 1994b). It is important to note that As and Sb occur in moderately enriched concentrations in bulk coal, being highly concentrated in coal-bound minerals like pyrite where they may reach extremely high concentrations (Diehl et al. 2004).

29.2.1 *Arsenic*

Due to environmental issues and human health effects by various trace elements, currently, there is significant interest in trace elements in coal. Basically, in coal, most trace elements can be associated with the inorganic constituents such as silicates, oxides, carbonates, phosphates, sulphates or with organic constituents (Clarke & Sloss 1992, Finkelman & Gross 1999). Particularly, As and Sb are present in coal as inorganic and organic forms. Concern has been raised due to leaching of As and other trace elements during coal washing, combustion and ash disposal. In bituminous coal, As is generally associated with pyrite, as shown by electron microscopy (Ruppert et al. 1992). The EXAFS (Extended X-Ray Absorption Fine Structure) method indicates that in crushed coal, As in pyrite readily oxidizes into air to form arsenate species especially iron arsenate phase (Huggins et al. 1993). But in low quality coal, As is associated as As^{3+} instead of $(AsS)^{2-}$ or As_2^{2-} as in the pyrite structure. In addition, it is bound to the organic matrix by means of oxygen functional groups (Huggins & Huffman 1996). Yudovich & Ketris (2005) surveyed the mode of occurrence of As in coal and conclude that pyrite is the main carrier of As, and arsenopyrite may be a minor host. It seems that the close relationship between As and Fe is more complicated considering the presence of As in pyrite.

In certain coal fields, As is associated with organic compounds and this organic affinity has been observed in Bulgarian coal deposits (Eskenazy 1995), and occasionally As is found at up to some percent levels in US coal (Finkelman 1994a). In Bulgarian coals, it has been speculated that As has been transported by hydrothermal solution and partly mobilized from the host rocks. Before importing/exporting coal, it may be preferable to reduce the concentrations of certain elements in coal so that there is less severe hazardous effect to the environment during or after combustion.

29.2.2 *Antimony*

There is lack of information on the modes of occurrence of Sb in coal. However, the association varies between coals of different origin. Antimony is frequently associated with Pb-Zn minerals (Clarke & Sloss 1992). In Turkish coals, Sb is associated with ophiolite belts and Sb deposits are mainly of hydrothermal-sedimentary origin (Cina 1990). There are studies on Sb in Bulgarian coals, and the concentration of Sb varies between 0.2 to 16 mg/kg. The mode of occurrence of Sb in Bulgarian coals is of organic affinity based on the following: a) Sb content decreases in the ash with increasing ash content for most deposits; b) the Sb_{inorg}/Sb_{org} ratio for many coal deposits is less than 1; c) in many areas of Bulgaria (e.g., in Goza Delĉev, and Pĉelarovo) Sb in coal deposits correlates closely with Ge (Eskenagy 1995).

However, Sb may be present in coal as solid solution in pyrite and as stibnite, Sb_2S dispersed throughout the organic matrix. Finkelman (1994b) also confirmed the occurrence of Sb in coal as organic form. In the British coal bed, Sb was found as crystals of ullmannite NiSbS (Finkelman 1994b).

Data for Sb in coals from India, South-Africa, the USA and other countries is included in Table 29.1. As the extreme, maximum enrichment of Sb (8.3–2347 mg/kg) in coal was reported from the Gokler coalfield of the western part of Turkey (Karayigit et al. 2000).

29.3 PRODUCTION AND USE OF COAL

Coal is expected to maintain its position as main energy source in both the electric power generation and industrial sectors for some time to come. Table 29.2 indicates production and consumption

Table 29.2. Production and consumption of bituminous coals, Mt (10^6 metric tonnes) (EIA 2005).

	Production		Consumption	
Region	1990	2003	1990	2003
North America	1009.48	1049.91	881.8	1073.86
Central & South America	29.8	56.50	24.1	31.8
Western Europe	791.45	437.46	940.73	646.25
Eastern Europe & Former USSR	1211.20	721.50	1169.2	650.8
Middle East	1.097	0.95	5.15	14.10
Africa	182.42	244.67	137.6	183.8
Asia & Oceania	1624.8	2393.14	1620.8	2332.9
World	4850.25	4903.63	4779.3	4933.5*

* This difference is made up from world stocks.

of coal for 1990 and 2003 in seven regions of the world. It is expected that driven by huge coal reserves and a leading position in world steel production, there will be substantial future demand of coal in China.

29.4 THE FATE OF ARSENIC AND ANTIMONY DURING PRODUCTION AND USE OF COAL

29.4.1 *Coal mining*

The coal mining industry is quite dynamic and the technology of coal mining varies from one region to another. Coal mines are either surface or underground mines (Stocker et al. 2005). Due to increased coal production, its mining has created problems by discharging toxic elements and tailings of high acidity (pH 2–4). Hence mobilization of As, Sb and other trace elements have created hazardous problems in the nearby receiving waters. In an open pit coal mine in New Zealand, As levels in coal bearing environments reached up to 100 mg/kg (Black & Craw 2001).

It is already mentioned that pyrite is one of the carriers of As and Sb in coal. Pyrite is generally abundant in ores mined and the oxidation of pyrite is often catalysed by micro-organisms forming acid mine drainage (AMD). In mining, the geochemical processes at low pH the AMD waters are controlled by the interaction of available oxygen, pyrite oxidation and trace element enrichment. In the open pit mines, waste piles are geochemically heterogeneous and pyrite is generally enriched with elevated levels of As which can be drained through groundwater seepage. In many localities, summer and rainy seasons control the accumulation of trace metals in sediments or surrounding areas. The Punch River (Madhya Pradesh, India) receives large amounts of mine water, but As was found to be below detection limits (BDL). However, in the same area, As in dugwell water (DWW) and hand pump water (HPM) contained 0.36–0.4 mg/L and 0.067–0.18 mg/L As, respectively (Gupta 1999). A recent study revealed also that Sb(III) can easily be oxidised to Sb(V) (Nriagu 2005). During mining activities, trace elements enter the atmosphere in different forms and as aerosol particles they are generally harmful to miners and people living at the vicinity of mines.

29.5 GLOBAL ANTHROPOGENIC EMISSIONS OF ARSENIC AND ANTIMONY

In the mid 1990s, global industrial trace metals (thirteen elements) emissions to the atmosphere have been estimated by Pacyna & Pacyna (2001). Worldwide total As and Sb emissions from industrial sources were 5011 and 1561 tons, respectively, of which 16.0% and 46.8% were shared by stationary fossil fuel power plants (Pacyna & Pacyna 2001). In this study, estimations were made on As and Sb emissions in different regions from coal fired power plants for the year 2003, using reported emission factors 0.2 g As or Sb/t coal (Pacyna & Pacyna 2001), based on data for the year 1995. Since Sb concentrations in coal are typically lower than As concentrations, the identical emission factor implies that Sb is the more volatile of the two. Results are shown in Table 29.3. Using these emission factors, global As and Sb emissions estimates for 2003 are 18% higher than the values presented in that work for the year 1990, presumably due to increased coal combustion since the mid 1990s.

Due to the different composition of coal around the globe, it is quite difficult to estimate world wide concentrations of As and Sb and their hidden movement through coal from one country to another. Global As and Sb concentrations in coal have been cited to vary between the ranges of 0.5–80 mg/kg, and Sb, 0.05–10 mg/kg respectively (Brown 1979, Swaine 1990, Taylor et al. 1998). Yudovich & Ketris (2005) mentioned global As levels in bituminous coal at 9.0 ± 0.8 mg/kg, whereas Zhang et al. (2004) estimated 5 mg As/kg, and 3 mg Sb/kg in coal. However, the distribution of As in coal is strongly affected by the ash and S contents, and less by coal petrography (Yudovich & Ketris 2005). Arsenic and Sb also enter into Coal Combustion Products (CCPs) during combustion.

Table 29.3. Estimated worldwide As and Sb emissions from stationary coal combustion power plants, 2003.

Region	As or Sb (in metric tons)
North America	215
Central & South America	6.36
Western Europe	129
Eastern Europe & former USSR	130
Middle East	2.8
Africa	36.8
Asia & Oceania	466.6
Grand total	986.56

Table 29.4. Arsenic and antimony concentration in world coal and coal fly ash from 30 countries.

		Coal, kt			
Element	Conc. in coal (mg/kg)	1990	2003	Conc. in FA[c] mg/kg	FA, kt 2001
Arsenic, As	9.0[a]	43.0	44.4	32.0	12.8
Antimony, Sb	3.0[b]	14.3	14.8	8.0	3.2

Note: Consumption data for coal has been taken from Table 29.1; [a] Ydovice & Ketris (2005); [b] Zhang et al. (2004); [c] Based on our present survey.

The world CCPs production in 1999 was estimated at 460 Mt (million tons): about 400 Mt of fly ash (FA) per year were produced by 30 countries in 2001–2002 (Smith 2005). The Republic of Korea burns more coal for the production of electricity and heat than any other country in the world and due to high concentration of mineral matter, CCPs production is four times higher per tons of coal than that of the United States (Kalyoncu 2001). Table 29.4 indicates world's total As and Sb concentrations in coal and FA.

The flow of coal from one region to another is presented in Figure 29.1. The amount of hidden As and Sb flows during export and import of coal can be estimated by multiplying with the concentration value of these elements. The black ring indicates the projected follow for these two metals in 2030 and the ring with grey shade indicates the flows of As and Sb during 2002. The maximum export of coal was noted from Australia (Kobayashi 2004), and there is no export of coal from India in 2002, although the country ranked third among the coal producing country in the world.

29.6 HOT SPOTS SITES OF ARSENIC AND ANTIMONY ACCUMULATION

There are more hot spots regarding accumulation of As in the environment from natural sources as well as from mining and burning of coals. China, Bangladesh, and India are a few examples. In the Guizhou province of China, high As concentration coal is burnt for cooking purposes, producing toxic arsenic oxide vapours:

$$2FeAsS + 5O_2(gas) = Fe_2O_3 + 2SO_2(gas) + As_2O_3(gas) \tag{29.1}$$

The As_2O_3 generated by the above reaction will condense into aerosols at below 215°C and subsequently enter the food chain, drinking water and the human body through respiration and skin contact (Zheng et al. 1999). Millions of people are suffering from arsenicosis and cancer due to high concentration of As in groundwaters (>50 µg/L) and foods grown in soils, irrigated by As contaminated water.

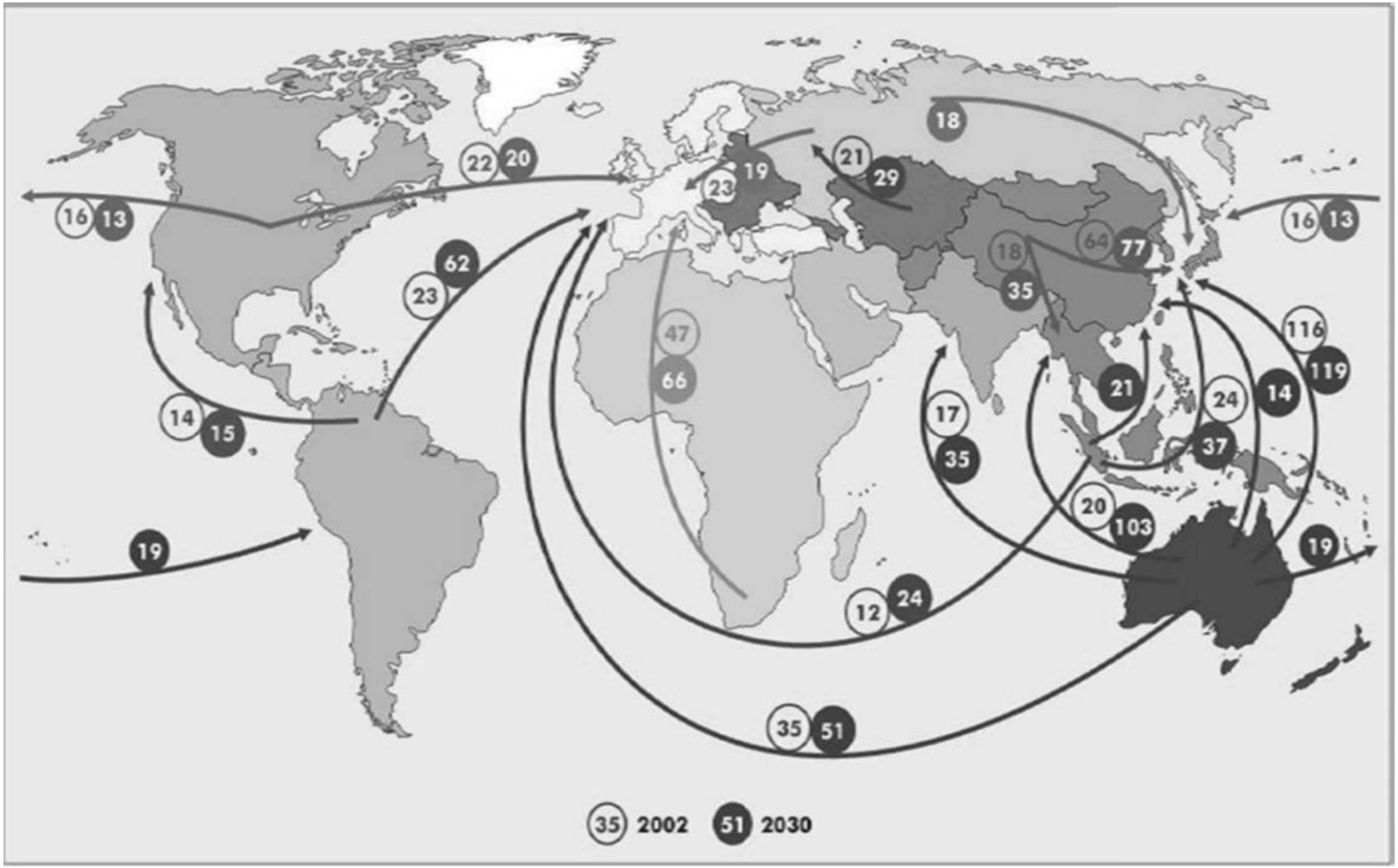

Figure 29.1. Global coal flows. Antimony and arsenic flows can be obtained by multiplying the concentration of these elements in coal cited in Table 29.4 (Kobayashi 2004).

There is not much information regarding hot spots of Sb and its effects to humans, animals and other life. But this element occurs in coal in most coal fields. For the Gokler coalfield in the western part of Turkey, Karayigit et al. (2000) mentioned the elevated concentration of Sb in coal (Table 29.1). In this coalfield, there is no correlation between S and Sb; Finkelman (1994b) believes that Sb is bound organically. The toxicological and physiological behaviour of Sb depend upon its oxidation state and it is believed that Sb(III) compounds are more toxic than Sb(V) species. Antimony trioxide has been stamped as carcinogenic to humans by the International Agency for Research on Cancer (IARC) (Nriagu 2005).

29.7 ARSENIC AND ANTIMONY IN GROUNDWATER

During the last two decades, As in groundwater in different regions of the world has been addressed by many authors (e.g., Tanaka 1988, Bhattacharya et al. 1997, 2002, Karim 2000, Ahmed et al. 2004, Mukherjee et al. 2006). Unfortunately, less attention has been paid on As in coal and coal FA in developing and newly developed countries (such as China, India, and South Korea) and the role in the groundwater in addition to natural releases of As in these countries. All forms of As are partially soluble in water and the oxidation of As_2O_3 to As_2O_5 may cause increased As leachability to soils and groundwater. In groundwater, inorganic As commonly exists as arsenate As(V) and arsenite As(III) the later species is more mobile and toxic for living organisms. In the region of Bengal Basin in West Bengal, India, As-rich groundwater is mostly confined to the alluvial aquifers of the Bengal Delta (Bhattacharya et al. 1997, Nickson et al. 2000) comprising sediments from sulphide-rich mineralized region of Bihar where biggest open pit coal mines are in operation. Weathering of sulphide minerals from the surrounding areas releases As which precipitated under the oxidizing conditions. However, redox potential in the sediments trigger substantial amount of As in aqueous phases through biogeochemical reaction (Bhattacharya et al. 2002, Amaya 2002) causing the high concentration of As in groundwater which is used generally as a drinking water in that region of India.

A study of Sb behaviour in coal fly ash from the United States indicates that Sb, especially as Sb_2O_4 and Sb_2O_5, is fairly water-soluble at pH 5.0 and very soluble at pH 2.0. Hence, Sb will leach from coal, fly ash piles and landfills into groundwater or surface water (Shi & Sengupta 1995). Leaching tests were performed by the US EPA's Toxicity Characterization Leaching Protocol (TCLP) Method 1310 (Seames et al. 2002). Not much is known on Sb reactions with organic matter and our knowledge is rather limited on the role of organo-Sb in the aquatic environment or its movement/release in groundwater. Under oxidising condition, Sb usually exists as Sb(V) and the rate of conversion of Sb(III) to Sb(V) decreases with increased acidity of waters, which may be due to a redox reaction (Filella et al. 2002).

29.8 CONCLUSIONS

In the ecosystem, many important changes are brought about by human-induced material flows. Each material flow affects the environment differently. Arsenic and Sb in coal flow and from coal burning give (eco-) toxic side-effects and put stresses on soil, air and waters. These two toxic elements enter the human food chain through vegetables, cereals and drinking water. In this study, we have addressed world coal consumption, emission from power plants, and presence of these elements in coals and coal FA. The world production of fly ash (FA) has been taken from the works of Smith (2005) which is however incomplete. There is a lack of data for Sb and As concentration in coal FA, hence the data which have been used in this study should be used with caution. Through coal FA, these two elements enter into different products and systems from where leaching and uptake by plants and other life will occur.

Policy makers, coal users, and engineers should understand the occurrence and high concentration in process streams and effluents of these two elements in coal and coal FA during usage and transport. Export and import of these two elements through coal have been visualised in Figure 29.1 for the year 2002. Due to high concentrations of As in groundwater, the people of Bangladesh and many other countries are undergoing severe health problems. Even marketing of products in other countries is sometimes difficult, causing economic problems in a country. Resource management, extraction, uses, disposal of wastes or emissions should not exceed the capacities or tolerable limits of nature or society.

ACKNOWLEDGEMENTS

The authors are grateful to the organizing committee of the 2nd International Groundwater Conference (IGC-2006), which was held in New Delhi on February 1–4, 2006 allowing them to present this paper. ABM acknowledges the Department of Environmental Sciences, University of Helsinki for providing the necessary financial grants. PB would like to thank the Knut Alice and Wallenberg's Foundation, Royal Institute of Technology and the Swedish International Development Agency (Sida), New Delhi for providing the grants for the conference.

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CHAPTER 30

Exploiting precipitation of naturally occurring iron against arsenicosis

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ABSTRACT: Arsenic (As) in shallow wells is a well-documented risk to the population in the large river deltas in Bangladesh and India (West Bengal). The expected health effects can be minimized by a number of mitigating phenomena in spite of the apparent high As intake. Some of these mechanisms are: i) presence of naturally occurring iron in raw water reduce the As concentrations in consumed drinking water. Individual behaviour may also inadvertently limit the As intake; ii) relatively short exposure time not allowing full effects to appear. Most As contaminated wells are no more than 10–15 years old; iii) higher than average dietary intake of phosphates (and other constituents) in food may compete with arsenate and prevent damages. Aeration of the raw water will remove iron and As. Cheap mitigation may be attained adding extra ferrous iron before aeration. This easy alternative to waiting for unaffordable mitigation measures reaching too few, requires building public awareness.

30.1 INTRODUCTION

Millions of people are at risk of getting cancer and arsenicosis due to exposure to high arsenic (As) concentrations in drinking water with the majority of severe cases being reported from South and East Asia like Bangladesh, China, India, Nepal and Vietnam (Smedley & Kinniburgh 2002, Bhattacharya et al. 2002, 2003, Mukherjee et al. 2006). Upon closer investigation further regions in these countries, and new countries from the subcontinent are being added to the list of potential excessive As exposure locations.

The most common initial visible symptoms of As exposure are however dermal changes like hyperpigmentation, and keratosis and are referred together as arsenicosis. The first detailed study of As exposure was from Taiwan, where Tseng and co workers conducted a survey involving 40,421 inhabitants from 31 villages and found respective prevalence rates of hyperpigmentation and keratosis to be 18% & 7% (Tseng et al. 1968). Arsenic concentration was in the range of 10–1820 μg/L with more than 70% exceeding 300 μg/L and people were exposed to As for more than 45 years. Other studies on arsenicosis are also available from China (Inner Mongolia, Xinjiang, Shanxi (Wang et al. 1997, Yang et al. 2002), Chile (Borgono et al. 1977), and Bengal delta (Tondel et al. 1999, Ahsan et al. 2000), showing arsenicosis at all dosage levels. However, studies from the developed world countries did not show any signs of arsenicosis at low to medium exposure (Goldsmith et al. 1972, Kurttio et al. 1998).

Exposure to As in these countries is primarily due to usage of ground water as potable water. Being colorless, odorless and tasteless in nature As is difficult to detect in water. The World Health Organisation (WHO) lowered the guideline value for drinking water from 50 μg/L to 10 μg/L in 1993 based on the then available epidemiological studies on As related skin cancer from As endemic areas of Taiwan. Based on the US population the estimated excess life time risk of getting skin cancer at this present recommended maximum level was $6 * 10^{-4}$ (WHO 1996). In the later years more studies on internal cancers like bladder, kidney and lung related to As exposure were published. According to (Smith et al. 1992) linear extrapolation based on studies from Taiwan and Chile down to 50 μg/L gives a cancer mortality estimate at this concentration of roughly 100 per

10,000 persons due to all internal cancers. However, these effects go unnoticeable for many years due to long latency period of 6–>40 years.

All the studies on skin effects showed lower prevalence rate among women compared to men. A recent study from Northern Chile with good nutrition involving only 11 families showed no arsenicosis signs in women consuming 750–800 μg/L even for more than 40 years, whereas 4 men out of 6 with more than 20 years exposure showed As related skin lesions. Both girls and boys in the range of 10–19 years with exposure showed As skin lesions (Smith et al. 2000).

In the Bengal delta, the biggest As calamity in the world with more than 40 million people exposed to As concentrations above 50 μg/L has been reported. In comparison to the US population, the additional As-related cancer risk estimates for the Bengal delta ought to be higher by a factor of 3–4, due to the lower body weight (50 kg) and higher water intake (4–6 L/day) (Sharma et al. 2006). Such cancer incidences are not reported despite of the literature indication that ground water was introduced as a drinking water source in the 1970s resulting in up to 30 years of excessive exposure to As for the first users. Factors like smoking habits, nutrition level (especially selenium intake), gender, genetic conditions, amount of exposure, and time of exposure have an effect on cancer due to As exposure.

Regarding arsenicosis only few studies are available on prevalence in the Bengal delta, which is not entirely mapped yet. In affected regions both lower (2–10%) (Hadi & Parveen 2004) and higher prevalence rates (30%) (Ahsan et al. 2000) were reported compared to the Taiwanese study (25.4%) (Tseng et al. 1968) when age differentiation was not made.

According to WHO (2003), development of hyper-pigmentation and keratosis would occur within six months to 3 years at a daily ingestion rate of >0.04 mg/kg body weight and 5 to 15 years at a daily ingestion rate of 0.01 mg/kg body weight. These values in terms of drinking water would be 500 μg/L and 125 μg/L, respectively by assuming an average body weight of 50 kg and a drinking water consumption of 4 L/day for people in the Bengal delta. The main factors reported to be responsible for the dissimilarities between various exposure groups and between individuals are differences in nutrition, especially selenium intake, and differences in water intake.

Our hypothesis is that additional factors like choice of preferred drinking water source, differences in nutritional phosphate intakes, and co-occurrence of iron and As in ground water, and elapsed exposure time are relatively unexplored, albeit important, and may thus blur the picture. Therefore the aim of this study is through literature and own field investigations specifically to identify and assess the mitigating effects on health of co-occurring iron in pumped water, time of exposure to As laden water, and possible effects from intake of phosphates.

30.2 MATERIALS AND METHODS

To examine our hypothesis 3 types of investigations were carried out and are described here.

30.2.1 *Water sampling and analyses*

Water samples were collected from 5 villages in the North 24-paraganas to investigate co-occurrence of As and Fe. Experiments were conducted in the field to investigate As removals with naturally co-occurring Fe due to passive sedimentation at initial Fe, As and phosphate (measured as P) concentrations of 5.5, 0.68 and 0.78 mg/L respectively. As(III) and total As analysis were carried out on a Perkin-Elmer 5000 AAS using a continuous hybride generation method. Total iron was measured using Flame-AAS. Soluble P and Si concentrations were determined using the ascorbic acid and hetero-poly blue methods, respectively.

30.2.2 *Duration of exposure and As prevalence rate*

Tubewell owners were interviewed to investigate the age of the tubewell and their drinking water habits. Hair samples were collected to assess the exposure to As. Collection and analysis of hair samples are described in (Sharma et al. 2004). A team with one doctor and some field surveyors visited the study area to investigate prevalence rate of arsenicosis. Twenty one subjects were randomly selected and a detailed interview about both drinking and cooking water habits was conducted. The available literature on As prevalence and their rate in different parts of the world was reviewed to find the differences between various study areas.

30.3 RESULTS AND DISCUSSION

30.3.1 *The prevalence of arsenicosis in the Bengal delta compared to the literature expectations*

The available arsenicosis dose response studies from Bangladesh (Tondel et al. 1999, Ahsan et al. 2000) and India (Guha Mazumder et al. 1998) are shown in Figure 30.1. The figure also illustrates the results of the prevalence of arsenicosis among the population exposed to As in Inner Mongolia (Yang et al. 2002) and Xinjiang (Wang et al. 1997, Yang et al. 2002) provinces of China. The figure further shows arsenicosis prevalence rate from 2 different regions of Chile (Antofagasta (C) (Borgono et al. 1977) and Atacameño (AM, AF) (Smith et al. 2000)), Japan (J) (Tsuda et al. 1995), where the concentration varies in a wide range. The AM and AF represent arsenicosis prevalence rate for males and females respectively among Atacameño people from a village called Chiu Chiu in northern Chile. Results from Finland (F) (Kurttio et al. 1998) are also shown in the figure, where population was exposed up to 500 μg/L, where the duration of exposure is unknown.

The results showed that the observed prevalence rate of arsenicosis in Bengal delta is 3.3 times lower than in Inner Mongolia. Compared to Xinjiang lower prevalence rates were only observed at As concentrations >400 μg/L. The Prevalence rates from Bengal delta were also lower compared to Antofagosta and male population among Atacameño people. Similar prevalence rates were observed compared to Japan at As concentrations <500 μg/L, and lower prevalence rates were observed at As concentrations >500 μg/L. However higher prevalence rates available in

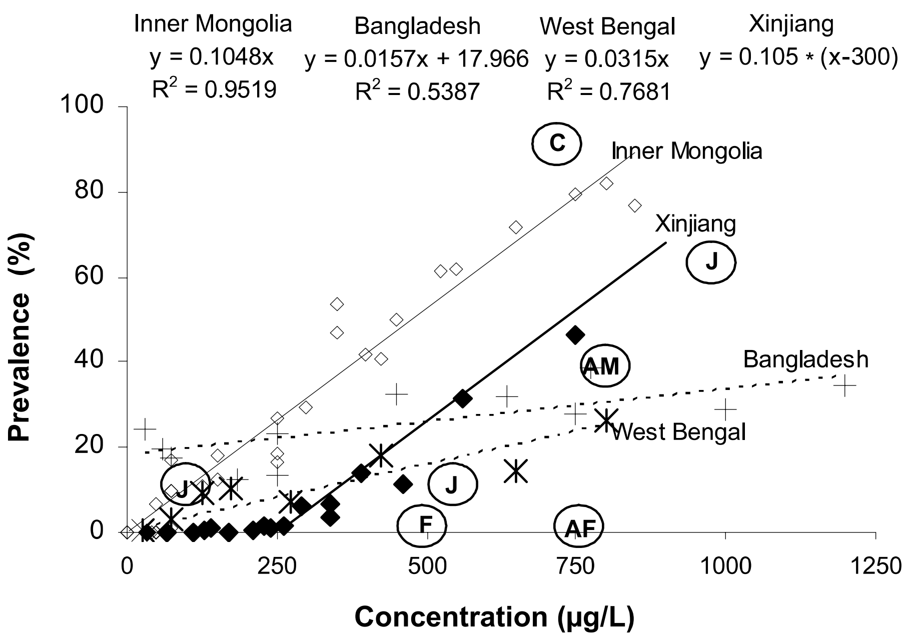

Figure 30.1. Comparison of prevalence rates around the world and Bengal delta. Legend: J = Japan, AM = Atacameño male, AF = Atacameño female, C = Chile, F = Finland.

Bengal delta compared to Finland and female population in Atacameño. There was positive linear correlation between arsenicosis prevalence rate and exposed As concentration in West Bengal. In case of Bangladesh a poor correlation existed between the prevalence rate and As dosage and even at concentrations <10 μg/L a prevalence rate up to 20% was reported.

This may indicate an extra source of As and is also supported by the reported very high urinary concentrations corresponding to lower As exposures (Ahsan et al. 2000). A study from West Bengal showed that the dietary intake of As from the foodstuffs grown in the As contaminated area for adults is as high as 190 μg and according to their findings rice contributes above 90% of the total intake with 44–96% of which being inorganic As (Roychowdhury et al. 2002). Other recent studies from Bangladesh involving all age groups showed lower average prevalence rates in the range of 3–10% (Akhtar Ahmad et al. 1999, Hadi & Parveen 2004) . The study by Akhtar Ahmad et al. (1999) also showed a threshold limit of 82 μg/L where they did not observe any arsenicosis. Based on these arguments if the Bangladeshi dose response data until 250 μg/L is omitted then similar prevalence rates were observed in West Bengal compared to Bangladesh and this would be referred in the future as prevalence rate in the Bengal delta.

Our hypothesis of lower prevalence rate in Bengal delta compared to the expected prevalence rate is proved when compared with Inner Mongolia, Antofagosta, male population in Atacameño and partly proved compared with Xinjiang. However, the hypothesis is untrue when the prevalence rates are compared with Finland, female population in Atacameño and Japan.

30.3.2 *The effect of co-occurrence of Fe and As*

The consumers main concern about high iron concentrations is not effect on health but change of taste, discoloration and turbidity of the water when stored, which is not noticeable at iron concentrations below 0.3 mg/L. WHO reported that 2 mg/L will not present a health hazard although iron concentrations of 1–3 mg/L can be acceptable for people drinking anoxic water (WHO 1996). Our hypothesis is that co-occurrence of iron would either lead to discard of the water for drinking and cooking purpose or storage of water to remove iron, since iron is easily oxidised upon exposure to oxygen at neutral pH followed by formation of $Fe(OH)_3$ which can either be filtered off or settled at the bottom after some time. Therefore both avoidance and storage would result in exposure to lower As concentrations compared to in the absence of iron. Experiments were conducted in the field to investigate As removals due to passive sedimentation at initial iron, As and phosphate (measured as P) concentrations of 5.5, 0.68, 0.78 mg/L respectively and Figure 30.2 shows the obtained % As, iron and phosphate removals in filtered (0.45 μm) and unfiltered samples. The results showed that around 10% of As was present in the particulate form within the first 3 hours of the experiment and increased gradually to 30% after 24 hours. Whereas no As removals were obtained in the unfiltered samples until the first 8 hours of the experiments. The result further showed that though more than 90% of iron is present as particles less than 15% are removed through sedimentation even after 8 hours. Removal of P followed the same trend as Fe for both filtered and unfiltered water.

Our laboratory results showed 35% As in the particulate form at Fe, As & P concentrations of 10, 1 & 2 mg/L respectively. Our previous study of As removals in the field indicated that an iron/As ratio above 80 is necessary to achieve As concentrations below 50 μg/L (Sharma et al. 2005). Previous studies by Roberts et al. (2004) reported a required Fe/As ratio of 120 at initial P and Si concentrations of 3 and 30 mg/L respectively, to achieve As concentrations below 50 μg/L.

The effect of particulate bound As on humans is not mentioned in the available studies. It is however unlikely that people would drink water without prior filtering to remove the particulate matter and thus filtration would result in removal of As and hence reduced exposure to As. The above experiments were conducted at initial As concentrations of 680 μg/L, whereas the extensive data from BGS (British Geological Survey) showed that 98% of the surveyed wells contained As concentrations less than 680 μg/L, while 40% of the tubewells had an Fe concentration above 5 mg/L. 75% of the tubewells exceeded the tolerable value of 2 mg/L for iron and only 3% of the tubewells had iron concentrations ≤0.3 mg/L, where the turbidity and discoloration is not noticed.

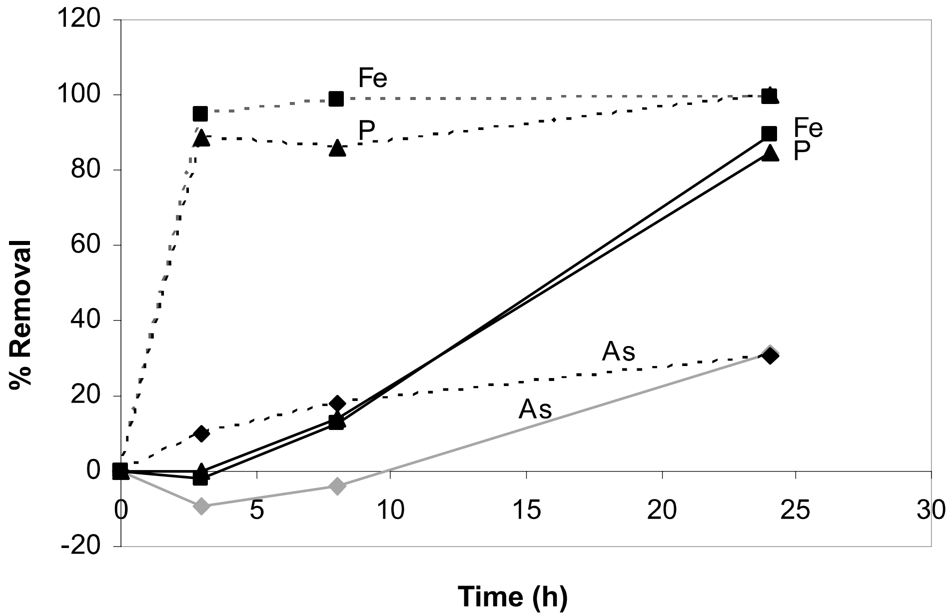

Figure 30.2. Removal of As, iron (Fe) & phosphate (P) in filtered (dotted line) and unfiltered (solid line) water.

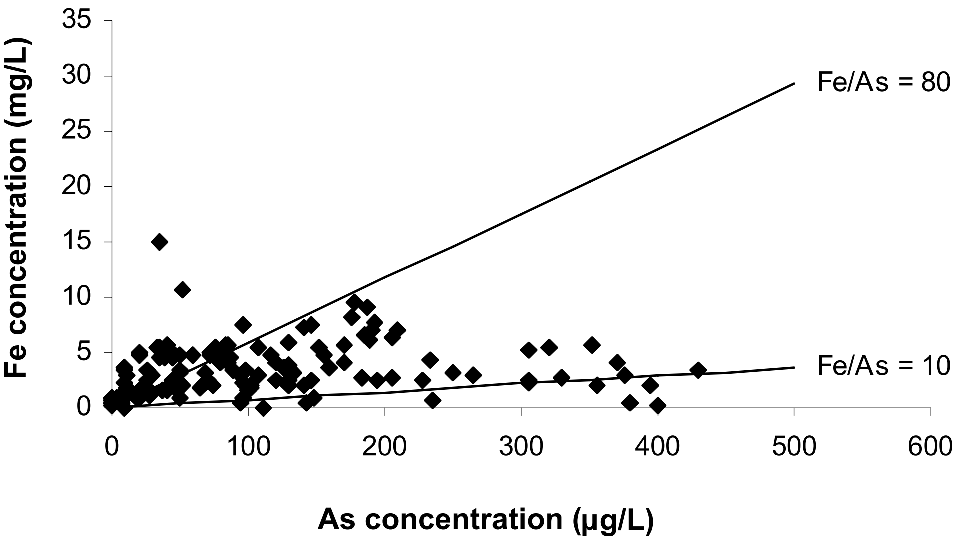

Figure 30.3. Co-occurrence of iron and As in West Bengal. The lines Fe/As = 10 & 80 indicate the concentrations where Fe/As ratios are 10 and 80 respectively.

We investigated 5 villages in the North 24-paraganas and 109 tubewells were investigated for their As and Fe concentrations. These values are shown in Figure 30.3. The analytical results showed that 35% of the tubewells had As concentrations below 50 µg/L. Among the tubewells exceeding 50 µg/L only 10% had Fe/As ratio below 10, and 12% had Fe/As ratios above 80. This indicate that if the passive sedimentation is employed at the surveyed tubewells the As concentrations could be reduced by 20% or more in 90% of the tubewells.

We also examined whether high concentrations of iron were present in studies from other parts of the world. In case of Chile low iron concentrations were observed as the water was taken from a river. In Japan the incidence of arsenicosis was due to industrial pollution and there was no mention about high iron concentrations. In Xinjiang positive correlation existed between As and sulphate, but there was no mention of iron. In Inner Mongolia we could not find any information about elevated iron concentrations. These results indicate that there may not be any indirect effect of presence of iron on exposure to As from China, Chile and Japan.

The interviews in the field showed that habit of storage of water depends on the distance to tubewell, and the longer the distance to a tubewell the more people would store the water. Our survey covering 14 villages in North 24 Paraganas showed that between 14–>90% of the families owned their own tubewell. In the absence of a private tubewell people would fetch the water once or twice a day and store the water and thus benefit from the positive effect of co-occurring iron and As removal. Therefore it is highly probable that the observed low health effects in the Bengal delta are due to the presence of co-occurring iron in pumped water.

30.3.3 *Duration of exposure*

The reported arsenicosis prevalence rates from Inner Mongolia and Xinjiang were after an exposure period of 10–15 years. Further Wang et al. (1997) also reported that the incubation period for the development of arsenicosis in Xinjiang depended on As concentration ranging from 10 years at 120 μg/L to only 6 months at 600 μg/L. Even though exploitation of ground water started during 1970'es the BGS survey shows that extensive installation of tubewells in the Bengal delta began in the 1990'es and hence the duration of exposure to As is only 10–15 years. We investigated As concentrations in all the tubewells of a village situated in North 24 Paraganas in West Bengal and <20% of tubewells were present in the village before 1993 (see Fig. 30.4).

These results indicate similarly to the BGS survey that exposure to As is a recent phenomenon in many parts of the Bengal delta. Our other survey in 2001 covering 14 villages in North 24 paraganas also showed similar installations statistics that the majority of the tube wells were not more than 10 years old. This slightly older survey further showed that very few people in the rural villages were aware of As risks and new installations of tubewells were still going on. Easy access to drinking water though a high number of tubewells would result in change in people's

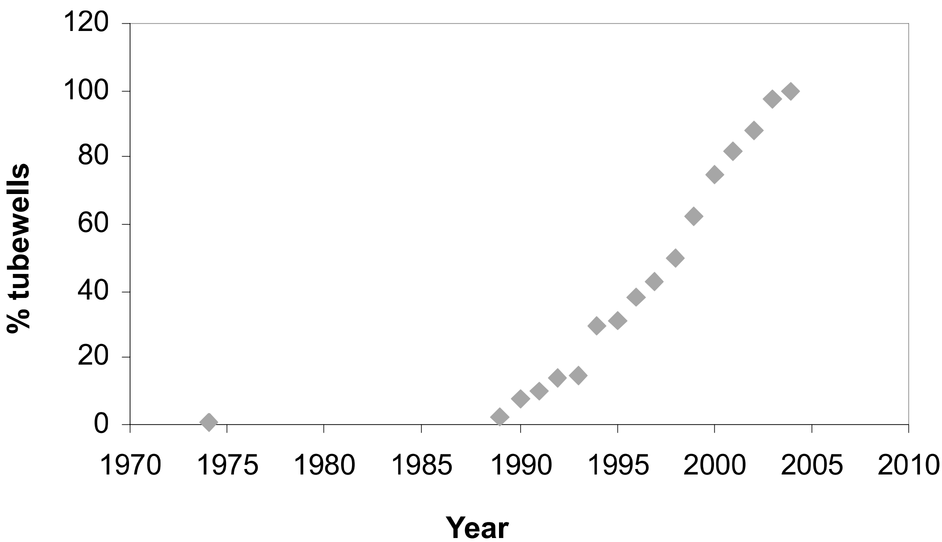

Figure 30.4. Tubewells present (in % of all in 2004) in a given year in a village in West Bengal.

water storage habits and this in turn would result in diminishing of the positive effect of storage with co-occurring iron precipitation, and in future higher arsenicosis prevalence rate may be expected.

30.3.4 *Differences in water intakes*

Figure 30.1 compares the arsenicosis prevalence rate based on the As concentration in drinking water. However, the actual As intake also depends on the amount of water consumed. The amount of water intake will vary with climate, physical activity and culture and hence the As intake would also vary accordingly. Daily water consumption less than 2 L is reported in Canada, Netherlands, United Kingdom and USA and this may also be expected in Finland. However in tropical Bangladesh and India a water consumption up to 4–6 L is reported which is 2 to 3 times more compared to Finland. An equivalent intake of As at 500 μg/L in Finland would be reached in the Bengal delta already at 160–250 μg/L. The low actual As concentrations combined with a possible shorter duration of exposure may explain the absence of health effects in Finland.

30.3.5 *The effect of varying phosphate intakes*

There is evidence of inter conversion of As(III) and arsenate (As(V)) after As has entered into the human body, and this depends on many factors. It has been established that As(V) is taken up in the body via the phosphate transport systems. Since arsenate and phosphate have identical structure and exhibit similar chemistry this allows As(V) to substitute for phosphate in many biochemical processes.

Therefore high phosphate intake may have a mitigating effect on As toxicity, although to our knowledge there is no evidence in literature on the mitigating effect of high phosphate intakes. Some studies reported concentration-dependent inhibitory effect of phosphate on As(V) reduction by human red blood cell lysate and rat liver cyosol. Reduction to As(III) decreased with an increase in phosphate concentration (Nemeti & Gregus 2002, 2005). Since As(III) is more toxic than As(V), the inhibiting effect of phosphate on As(V) reduction may also have a mitigating effect on arsenicosis. A study from West Bengal concluded that low intakes of calcium, folate and fibers may increase susceptibility to arsenicosis, but they did not observe any significant effect of phosphate intake on arsenicosis using a chi-square test at $p = 0.05$, however the results showed an increase in arsenicosis with decreasing phosphate intakes at $p = 0.15$ (Mitra 2004). The average phosphorous intake reported in this study was approximately 1.1 g for both arsenicosis cases and controls with slightly higher intakes among the controls compared to cases. Only about 1% of their studied population's phosphorous intake was <0.4 g (Indian recommended daily intake (RDI)). Previous studies by these authors from the same study area reported a threshold limit of 100 μg/L, below which arsenicosis was not observed. Similarly the study from Xinjiang also reports a threshold limit of 120 μg/L. However, in Inner Mongolia arsenicosis was observed even at concentrations below 100 μg/L. Yang et al. (2002) postulated that the observed lower prevalence rate in Xinjiang compared to Inner Mongolia could be differences of As intake in food, and/or the population in Xinjiang are protected by selenium in their diet. Whereas Lin et al. (2002) tried to explain the difference based on oxidation state of the As. They observed an increase in prevalence rate with an increase in As(III) plus methyl As to the total amount of As. They reported that As in Xinjiang is primarily in the inorganic form with As(V) being the predominant species, where as high contents of methyl As and As(III) were present in Inner Mongolia. None of these studies investigated the effect of nutrition on the observed arsenicosis prevalence. The socio economic conditions of the exposed population in Inner Mongolia are similar. They are poor farmers and live on local farm products. They use the As contaminated water for both cooking and drinking and the adults drink approximately 1.5 L/d. In Xinjiang the exposed population was settlers of farmer and soldiers, and they may have different food habits compared to the population of Inner Mongolia and hence may have different phosphate intakes. But we could not find any literature related to the difference in the nutrient intakes of these 2 regions.

Similar phosphorous intakes as the West Bengal study (1.1 g/d) are reported from China and Japan, and higher phosphorous intakes (1.5 g/d) are reported from UK and USA (Dyer et al. 2003). If elevated concentrations of phosphorous have a mitigating effect on arsenicosis then the dietary intake of phosphate may also be a reason for the observed lower prevalence rates in developed compared to developing countries. The staple diet in developing countries is based on polished rice or potatoes supplemented with low amounts of vegetables, meat and dairy products leading to very low dietary phosphorous intakes.

30.4 PREVALENCE OF ARSENICOSIS AND IMMEDIATE MITIGATION

Large resources are spent on providing clean drinking water to affected regions. Despite this, large segments of the affected population are not benefited due to the high costs of mitigation e.g. drilling for uncontaminated ground water 100–300 m down. This may help initially on drinking water habits but may not affect the cooking water habits. This is also seen in the above studied village in West Bengal where people started using deep tubewell water for drinking purpose, but still use the As contaminated water for cooking purpose. However, in the longer perspective people may start using their own contaminated shallow well both for drinking and cooking due to difficulties in fetching the borehole water if the distance is too far.

Figure 30.5 shows As concentrations in hair and the corresponding As concentrations in drinking water. The results showed that As concentrations in hair increased with an increase in As concentration in drinking water. Only one arsenicosis case was observed which is encircled in the figure giving a prevalence rate around 5%. The arsenicosis was identified as minimal hyperkeratosis. According to the interview this subject was exposed to 97 μg/L of As for the past 3 years, however the entire history of As exposure for this subject is unknown. On the other hand people consuming As concentrations higher than 300 μg/L for more than 10 years in this village did not show any signs of arsenicosis, whereas the As concentration in hair shows that they are exposed to elevated As concentrations. If the WHO's estimation is used for arsenicosis then 16% of the tubewells exceeding 125 μg/L should have existed for more than 5 years, which indicates that if an action is not taken soon many more cases of arsenicosis may appear in the near future.

Our research has shown that high natural iron concentrations reduce the As concentrations in the drinking water to much lower levels if left to precipitate before consumption. Our investigations also showed that this happens often inadvertently due to bad taste or turbidity of iron which is often present in proportion to the As content. Based on the results simple precautionary measures as aerating the raw water and leaving it to precipitate will reduce the risk of arsenicosis. If iron is

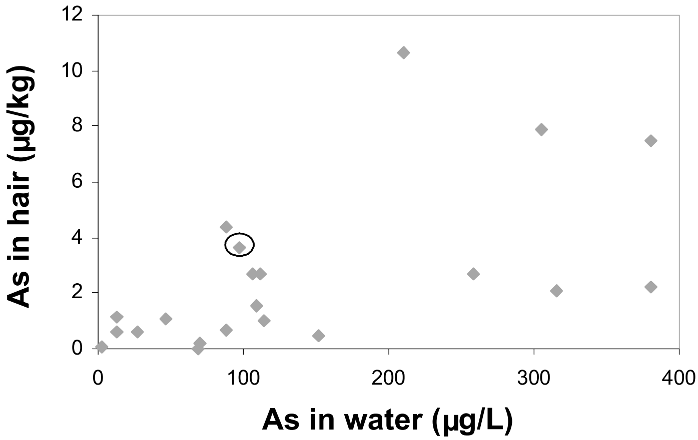

Figure 30.5. As concentration in hair compared with As concentration in water.

absent or low, cheap mitigation could still be attained adding ferrous iron to the raw water before aeration.

Thus, as an alternative to waiting for unaffordable mitigation measures, reaching too few, building public awareness towards exploiting the presence of natural or added iron would help many more against getting arsenicosis. This may be combined with advocating use of As free surface water in cooking.

30.5 CONCLUSIONS

Comparison of the number of patients suffering from arsenicosis in different parts of the world with the Bengal delta indicate lower prevalence rate. The literature review further indicates a threshold limit for development of arsenicosis when exposed to As(V) compared to As(III). This study indicates that lower observed prevalence rate of arsenicosis in Bengal delta could be due to the shorter exposure period. Another possible reason is the presence of iron which upon storage would oxidise and remove As resulting in actual exposures lower than the exposures based on the tubewell concentration. The storage of water depends on distance to tubewell and the longer the distance the more people would store the water. The drastic increase in number of tubewells since 1990s resulted in shorter distance to the tubewell and this in turn changed in peoples habits leading to increase in direct consumption of tubewell water without prior storage. This change in habit would result in higher prevalence rate due to the absence of indirect effect of co-occurring iron on storage. Therefore higher prevalence rate may be expected in the future. This study further indicates that it may not be too late since the exposure to As in most of the cases is not more than 10 years old. An effective solution in the form of building public awareness towards exploiting the presence of natural or added iron would help many more against getting arsenicosis instead of waiting for an alternative unaffordable mitigation measures, reaching too few. This may be combined with advocating use of As-free surface water in cooking. The study further suggested that intake of high dietary phosphate may have a mitigating effect on arsenicosis, but further research is needed to confirm this hypothesis.

ACKNOWLEDGEMENTS

We would like to thank the NGO organisations BSSKS, KTT, SPREAD and JGVK together with Julian Gauthier and Lene Kieler for their support in collecting the data from As affected villages in West Bengal. We would also like to thank M. Sonsoles Quinzaños Lobo and Stefan Heiske for As analysis in hair.

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CHAPTER 31

Arsenic contaminated drinking water and nutrition status in Bagahi village, Terai region, Nepal

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ABSTRACT: Groundwater is the main source of drinking water in the Terai region in Nepal. Analysis of arsenic contaminated groundwater of 287 tubewells and nutrition level of the rural communities of Bagahi village, Rautahat district, Terai region has been performed. Altogether 538 households (average 6 persons) consume the tubewells water. About 6% of the total tested tubewells are considered as risk tubewells based on Nepal Interim Standard (50 μg/L). Of the total risk population about 9% has been identified as arsenicosis patients. About 80% risk populations are taking inadequate nutrition in terms of calorie content. The risk of arsenic among the risk populations is higher because majority of them are under nutrition and using the contaminated water for cooking and drinking. This alarming situation therefore calls for measures to mitigate the problem.

31.1 INTRODUCTION

Arsenic (As) is ubiquitously found in air, water, fuels and soil. Arsenic enters the human body through ingestion, inhalation, or skin absorption. Significant route of As ingestion is drinking water (Smith et al. 2000, Kapaj et al. 2006). Arsenic in natural waters is mostly found in inorganic form, such as arsenite and arsenate; of which the former form is more toxic than the latter (Centeno et al. 2002). The United States Environment Protection Agency (USEPA) and International Agency for Research on Cancer (IARC) As has specified as a known human carcinogen and classed as Group IA carcinogen.

In the As affected areas, As may enter into the food chain from water to soil, and soil to foods of all varieties, viz. roots and tubers, vegetables, fruits, edible flowers, seeds, fleshy foods, eggs, etc. The ultimate recipient is being a man. It is known that most of the As in water is in the form As (III); it is likely that a large proportion of this As remains as such in the plant depending on the rate of bio-methylation capacity of the plant species. In Bangladesh, have found that some varieties of fruits, vegetables, and grains irrigated by As contaminated water have contained high As concentration (Chakrawarty et al. 2003, Huq & Naidu 2003).

In Nepal, As contamination in the groundwater of its Terai region has been detected for the first time in 1999 by DWSS/UNICEF/WHO. Since then, efforts have been made by different organizations to gather information on As contamination in groundwater in the Terai region (Tandukar 2000, Valero 2002, Bhattacharya 2002, Gurung et al. 2005). However, the As threat to human health is relatively new for Nepal. Efforts have been made by the government to harness the groundwater for year round irrigation for increasing agricultural crops in the Terai. Since the groundwater is found to be contaminated with As, continuous use of groundwater for both drinking and irrigation appears to be health catastrophe to the Terai communities. Most As absorbed into the body is converted by liver to less toxic methylated As and then efficiently excreted in the urine. The methylation of process of As is affected by the heath status of the persons (Centeno et al. 2002). This paper examines the As contamination in groundwater and the health status of the population of Bagahi Village in Rautahat district of Terai region of southern Nepal (Fig. 31.1).

The Terai is a plain region and lies in the southern part of Nepal. It represents the northern extension of the Indo-Gangetic Plain bordering India. Lying at altitude ranging from 60 to 310 m above sea level, the Terai region has width of 30–40 km and about 1500 m thick of alluvial sediment (Bhattacharya et al. 2003). This region consists of 20 districts out of total 75 districts of Nepal and occupies approximately 17% of the total area of Nepal. With total population of about 11.5 million, the region represents 49% of the country's total population (CBS 2002).

31.2 MATERIALS AND METHODS

The data and information for this study has been derived from two sources: field survey and existing studies. For the field survey, the following methodological procedures were adopted:

- Firstly, As concentration of the tubewells (287) was tested with 250 Hach-EZ kit. This test kit relies on the reduction of inorganic As to arsine gas (AsH_3) using zinc metal and hydrochloric acid. Thus, produced gas is allowed to pass through the mercury bromide ($HgBr_2$) indicator paper and the intensity of colour indicates the concentration of As. The reliability of this As field test kit was verified on 1% water sample in the field by WAGTECH Arsenator (digital kit) and 2% samples were sent to the laboratory for cross checking by atomic absorption spectrophotometer (AAS);
- Secondly, nutrition status of the As risk households was obtained by administrating questionnaire form, which sought information on quantity and type of foods and frequency of foods consumed, and food and energy consumption per person has been calculated; and
- Thirdly, in addition to the household survey data, the data on health status of children and women was acquired from the village health post.

This study has used the empirical data from the existing studies as secondary sources. From these data, the status of As, health status, and As mitigation measures have been described for the Terai region as a whole.

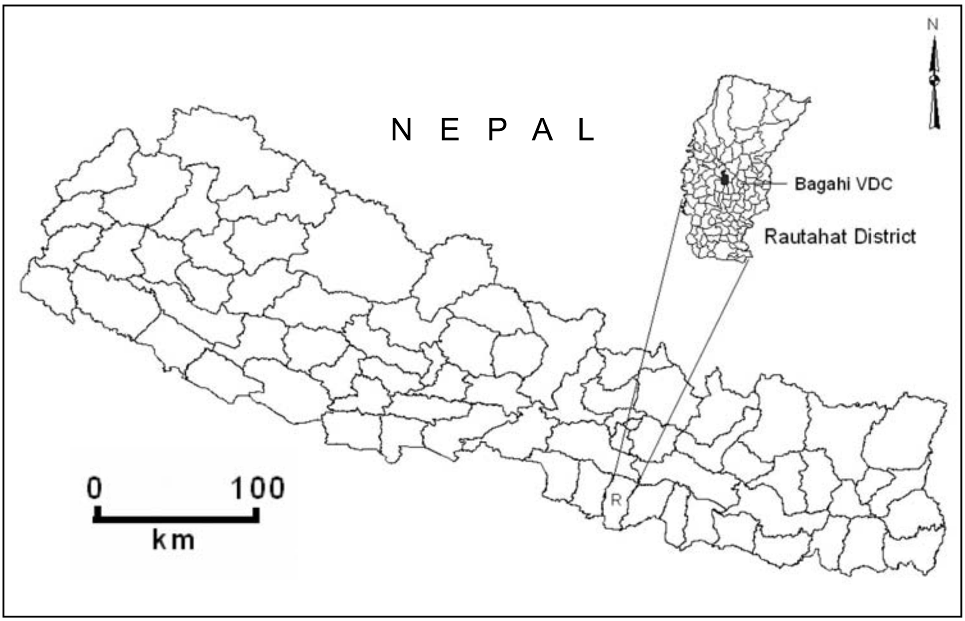

Figure 31.1. Location map of the study area in Bagahi VDC in Rautahat district of Nepal.

31.3 RESULTS AND DISCUSSION

31.3.1 *Classification of arsenic concentration*

Two guideline values such as the WHO guideline (10 μg/L) and Nepal Interim Standard (50 μg/L) are taken as basis for classification of As concentration in water of the tubewells (Table 31.1). Nearly 51 percent of the total tested tubewells in Bagahi village are below 10 μg/L, and the number of tubewells with As concentration above 50 μg/L is nearly 6 percent as compared to 49 percent of the tubewells above 10 μg/L. Compared to the state of As concentration in Bagahi to that of the Terai region, the situation is alarming, as in terms of both guideline values, Bagahi has remarkably larger proportion of tubewells have higher than the Terai region (Tandukar et al. 2001, 2006, DWSS 2005). In terms of severity of As problem, Bagahi lies in the Moderate Extended and Acute (MEA), where 20 to 50 percent of the total tested samples have As concentration >10 μg/L and more than 3 percent of the samples have As concentration >50 μg/L (FAO 2004).

31.3.2 *Year of use of tubewells*

The water of tubewell has been used by the households in the Terai since long time ago. Table 31.2 shows that the class of 5–10 years shares the largest proportion with 53.3 percent, followed by 2–5 years. The tubewells used over 15 years account for 2.1 percent. It is also evident that the largest share of the tested tubewells with >50 μg/L falls in 5–10 years, followed by 2–5 years.

31.3.3 *Distribution of users by arsenic concentration*

Table 31.3 shows that 3.4 percent of the total population has used the tubewells with As concentration >50 μg/L, which is considered as risk population and accordingly, 3.5% of the total households are considered as risk households.

Table 31.1. Distribution of tubewells by levels of arsenic concentration (n = 287).

As concentration (μg/L)	Frequency	Percent	Terai region %
0–10	146	50.9	87.4
>10	141	49.1	12.6
>50	17	5.9	2.5

Source: DWSS 2005.

Table 31.2. Years of use of tubewells by levels of arsenic concentration.

	Tubewells by As concentration (μg/L)							
	0–10		>10		>50		Total	
Year of tubewells	*n*	%	*n*	%	*n*	%	*n*	%
<1	14	4.9	6	2.1	–	0.0	20	7.0
1–2	6	2.1	7	2.4	–	0.0	13	4.5
2–5	32	11.1	41	14.3	4	1.4	73	25.4
5–10	82	28.6	71	24.8	12	4.2	153	53.3
11–15	6	2.1	16	5.5	1	0.3	22	7.7
≥15	6	2.1	0	0	–	0.0	6	2.1
Total	146	50.9	141	49.1	17	5.9	287	100

Based on the Present study.

Table 31.3. Ownership type of tubewells by levels of arsenic concentration.

	Tubewells by As concentration (μg/L)							
	≤10		>10		>50		Total	
Ownership type	*n*	%	*n*	%	*n*	%	*n*	%
Private	117	40.8	133	46.3	17	5.9	250	87.1
Public	29	10.1	8	2.8	0	0.0	37	12.9
Total	146	50.9	141	49.1	17	5.9	287	100

Note: Percentile figures are computed from the total samples (n = 287); the data is divided into two sets, i.e. <10 μg/L and >10 μg/L, and therefore their numerals give total sample units of 287; concentration above 50 μg/L are subset of above 10 μg/L.

Table 31.4. Distribution of tubewells by depth and levels of arsenic concentration.

	≤10μg/L		>10μg/L		>50μg/L		Total	
Depth of tubewells (m)	*n*	%	*n*	%	*n*	%	*n*	%
1–10	28	9.8	13	4.5	1	0.3	41	14.3
10–20	95	33.1	121	42.2	16	5.6	216	75.3
20–30	1	0.3	2	0.7			3	1.0
30–40	12	4.2	3	1.0			15	5.2
40–50	8	2.8	1	0.3			9	3.1
>50	2	0.7	1	0.3			3	1.0
Total	146	50.9	141	49.1	17	5.9	287	100

Based on the present study.

31.3.4 *Type of ownership of tubewells*

Number of tubewells owned by private is larger than by public; the former accounts for 87.1 percent of the total tubewells (Table 31.3). Arsenic concentration above 50 μg/L has not been detected in the public tubewells. It means there is As problem in private tubewells.

31.3.5 *Depth of tubewells*

The number of tubewells lying within the depths of 10–20 in all three levels of As concentration is highest (Table 31.4). The relationship between depth of tubewells and level of As concentration is shown by scattered diagram (Fig. 31.2). All the tubewells with As concentration above 50 μg/L are at the depth of less than 20 m. It indicates that the shallow aquifers are found contaminated with As. Similar result has been found in other Terai districts of Nepal (NASC 2004).

31.3.6 *Public health scenario*

31.3.6.1 *Distribution of arsenicosis patients*

A total of 136 people from 22 households have depended on 17 As contaminated tubewells (>50 μg/L) for drinking water in Bagahi village. They are considered as risk population. Among this total risk population, 13 arsenicosis patients were identified and confirmed by NRCS/ENPHO (2003). This group of population have been provided with As-safe water since 2003. This means that the prevalence of arsenicosis (arsenicosis patients*100/total risk population) is found at 9.5%

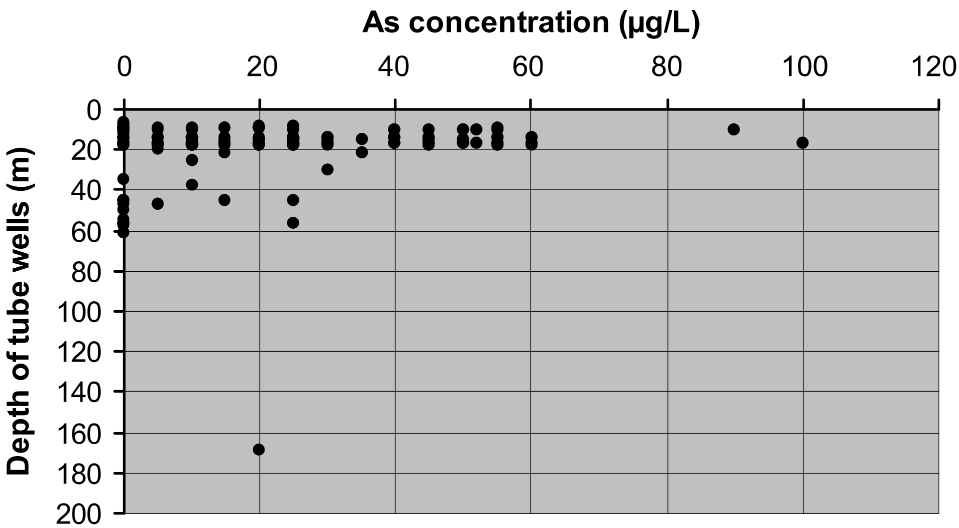

Figure 31.2. Variability of As concentrations in the tubewell with depth.

of the total risk population. This is higher compared to that of the total district, Rautahat (2.7%) which however has been identified as the first level of arsenicosis symptom (WHO 1997). The age group of 50 years and above has the highest prevalence rate (7 out of 13, i.e. 53.8%). It has been observed that the arsenicosis was more prevalent among the male (38.5%) as compared to the female (15.3%) population groups. Melanosis on the trunk and keratosis on the palm are common arsenicosis symptoms in the patients (Pradhan et al. 2004).

31.3.6.2 *Awareness of arsenicosis disease to the health personnel*

A total of 25 health personnel working in Bagahi and its neighboring VDCs were interviewed whether they aware of arsenicosis disease. They were representing from grass root level to officer levels such as MCHW, VHW, AHW, ANM, CMA, Lab. Assistant, HA, Medical Officer and Public Health Officer. Of these, 56% t knew about arsenicosis disease. However, this finding is higher than the finding of the study carried out in other VDCs of the same district, which was 16.1% (Bhagat 2003).

31.3.6.3 *Mitigation options of arsenic problem*

NRCS/ENPHO (2003) has provided the following five types of mitigation measures to the Arsenicosis patients in all VDCs of Rautahat district. They are preventive measures intending to provide As-safe water, viz., i) two-Gagri (water vessel) filter, ii) innovated dug well, iii) arsenic and iron removal plant (AIRP), iv) tubewells from As-safe aquifer, v) modified bio-sand filter, formerly called Arsenic Bio-sand Filter and now called Kanchan Arsenic Filter, and vi) awareness program on nutrition. Among these, the option of two-Gagri filter and awareness program has been provided in Bagahi.

31.4 NUTRITIONAL STATUS

Arsenic is ingested in human body through both water and food stuffs. The food habit, nutritional status and biomethylation activity of the individuals can be related to the manifestation of arsenicosis disease (Huq & Naidu 2003). The analysis of average per person daily food consumption pattern of the risk households in Bagahi shows that four-fifths of the sample respondents have not got

Table 31.5. Daily food consumption pattern and energy obtained of risk households (n = 19).

Nutrition levels	Number of respondents	Percent of respondent	Total food consumption (g)	Average Fat (g)	Average Protein (g)	Energy obtained (kcal)
Adequate*	4	21.1	700–800	65	35	2450–2800
Inadequate	15	78.9	350–600	40	25	1225–2100
Total	19	100				

* Average energy required = 2450 kcal (NHDR 1998).

Table 31.6. Nutritional status of children under five years of age.

	Low		Normal		
Location	No.	%	No.	%	Total
National*	142,830	12.1	1,040,545	87.9	1,183,375
District	3,105	18.4	13,767	81.6	16,872
VDC**	48	23.8	154	76.2	202

Source: * DOHS 2004, ** Health Post Record 2003.

adequate energy supply from their foods (Table 31.5). Similarly, the hospital record as given in Table 31.6 shows that about 23% of children below five years of age have low nutritional level, which is however higher than the national and district level records.

The analysis of food habit of the risk households shows that about 70% were non-vegetarian and about 75% have consumed rice as their staple food and the rest have consumed rice and bread (wheat). The amount of water required to cook food depends on the type of food, and if As contaminated water is used, the retention of As in the food also varies with the type of food (WHO 2003). The information shown in Table 31.7 is derived from the study in Bagahi VDC, and the water requirement for cooking foods and drinking per day per person is determined. Average amount of ingestion of As per day has been calculated based on the factors provided by Ahmed (2003). For examples, the amount of water for cooking 450 g rice, 50 g dal (bean soup), 150 g potato curry, and 200 g wheat flour requires 2,880 ml, 250 ml, 275 ml and 100 ml respectively. The average retention of As (if As contaminated water is used) is 90 percent. The average ingestion per person in the risk households from food and water together is higher based on the toxicological approach, the daily exposure from food and drinking water together must not exceed 2 μg/day/kg body weight or 120 μg/day (body weight = 60 kg) (Ahmed 2003). The average percentage of As intake is higher from food than water (Table 31.8).

In Bagahi, the year of use of tubewells varies from one to 30. The average use of tubewells is 7 years. However, before the use of tubewell, the people had used the water through dug wells. Still there are 4 dug wells using by the inhabitants in Bagahi. However, As concentrations in those dug wells are within the acceptable level.

The risk of As problem may be decreased if As free water could be provided to the people now by more than 8 times. The nutrition status of children of the study area is poor. The children under nutrition is defined in terms of stunting (short for their age which can be a sign of early chronic under nutrition), wasting (thin for their height as an indicator of acute malnutrition), and under-weight (low weight for age) by the Nepal Demographic and Health Survey (NDHS 2001). In Table 31.9, the NDHS report for the national level shows that stunting children shares higher proportion (50.5%) than other two under-nutrition types. Whereas the Health Post report for the Bagahi village shows the higher proportion of under-weight children under five years of age (out of total sample 105 children) than the stunting and wasting children.

Table 31.7. Arsenic concentration and consumption per person per day.

Description of As	Amount
Average As concentration (μg/L)	73
Average Amount of water consumed (liter)	2.7
Average concentration of As ingested per day from water (μg)	167.1
Average Food consumption/day (g)	511.8
Average concentration of As ingested per day from food (μg)	185.3
Average As ingested per person per day (μg)	352.3
Average % of As from water	46.6
Average % of As from food	53.4

Table 31.8. Exposure time and arsenic risk*.

Average years of consumption	7 years
Average life expectancy	60 years
Average concentration of As[1]	73 μg/L
Risk if options are not provided[1]	$4258/10^6$
Risk if options continued[2]	$511/10^6$
Risk reduction (if options continued)	8.3 times

*For carcinogenic substance (WHO 1997), the risk factor is calculated as:
[1] Risk = (As concentration ∗ Contact Rate ∗ Carcinogenic Potential)/Life Expectancy Risk with respect to exposure time (EPA 1997) is based on the relation.

[2] Risk = (As concentration ∗ Unit Risk ∗ Exposure Time)/Life Expectancy.

Table 31.9. Nutrition status of children under five years of age.

			Prevalence of under nutrition		
Sources	Year	Location	Stunting	Wasting	Under-weight
NDHS report	2001	National	50.5	9.6	48.3
Health Post report	2003	VDC	45.2	13.5	48.2

Source: NDHS, 2001.

The nutrition status of women of reproductive age group (15–49 years) is also analyzed based on the mean body mass index (BMI), which is defined in terms of weight of reproductive woman divided by height square. NDHS (2001) has studied about the nutrition status of women of reproductive age group for the Terai as a whole. According to this study, about 36 percent of the reproductive women age group of Terai women was under nutrition, which is higher than the national value of 27% of the same women age group. Though there is no data on women in the Health Post report of Bagahi village, the Terai average value can be considered for this village too.

31.5 CONCLUSIONS

The major source of water for drinking and cooking in Bagahi VDC is tubewell. The VDC has 3,950 total tubewell users, of which the risk population with As concentration above 50 μg/L is 3.4 percent. The private tubewells share 87.1%. Arsenic concentration above 50 μg/L is found within the depth of 10–20 m indicating that the shallow aquifers are contaminated with As. A substantial proportion of the tubewells (53.3%) are used by the rural communities of Bagahi since the past 5–10 years. The average concentration of As in the risk tubewells is 73 μg/L. The average use of tubewells in this VDC is 7 years. In terms of severity of As toxicity, Bagahi lies in the Moderate Extended and Acute (MEA).

Altogether thirteen arsenicosis patients have been identified. Melanosis on the trunk and keratosis on the palm were common arsenicosis symptoms. The highest prevalence rate of arsenicosis was found at the age group of 50 years and above. The prevalence rate was higher in the males than in females. About 78 percent of the total sample respondents (50) have not got adequate energy. The VDC health post data shows that about 50 percent of the children under five have different types of nutrition deficiency, ranging from chronic to acute. The average As ingested per person from food and water is estimated to be 352 mg/day. During the study, about 56 percent of the health personnel working in different levels were aware about the diseases caused by As. The inhabitants in Bagahi village require foods with adequate nutrition and As free water. The village requires the mitigation measures as given above for As free water. If those measures are adopted now more than 8 times risk can be minimized. Further research on hydrogeological characteristics of the aquifers and water quality surveillance of the tubewells should be made for finding effective and sustainable means of As mitigation.

ACKNOWLEDGEMENTS

The author is grateful to the Department of Community, Medicine and Family Health, Institute of Medicine, Tribhuvan University for providing logistic support for this study. The author is thankful to the organizers of the 2nd International Groundwater Conference for their encouragement and support for presenting this work. The author acknowledges Dr. A.B. Mukherjee and P. Bhattacharya for their constructive criticisms and editorial handling of the manuscript.

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CHAPTER 32

Ensuring arsenic-safe water supply in local communities: Emergent concerns in West Bengal, India

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ABSTRACT: In recent years, the arsenic menace has come to threaten the lives of several millions in the state of West Bengal in India. A number of technological options have been developed to supply arsenic-safe water to affected communities. These can be conveniently clubbed under the categories of alternative technologies for supplying arsenic-free water, community level arsenic removal plants, and domestic filters. While each of these alternatives has its own strengths and weaknesses, this paper demonstrates that the common challenges determining their fate constitute a triad of 'appropriateness', 'accessibility' and 'sustainability' and that these are essentially linked to the question of their management at the level of the community. However, an understanding of these issues is yet to be adequately developed and addressed. Based on a detailed primary study in 35 affected villages in the state, this paper outlines these major 'software' issues and argues that there is a need to develop a strategy centred in the 'management approach' in order to ensure that arsenic-safe water is accessible and is used equitably and effectively by all exposed to the problem.

32.1 INTRODUCTION

Arsenic in drinking water sources in West Bengal (WB), India was first detected in 1982 when symptoms of arsenic-related ailments were noticed among residents of three villages in two districts. Today this seemingly minor arsenic incident has grown to a menacing size, with arsenic reported in drinking water sources in 79 blocks in eight districts of the state and the metropolitan area of Kolkata. These drinking water sources are in the form of shallow tubewells (with handpumps attached) that are otherwise believed to supply microbially the safest water by virtue of being dependent upon groundwater. The districts affected are Bardhaman, Haora, Hugli, Malda, Murshidabad, Nadia, North 24 Parganas and South 24 Parganas. According to recent studies about 13.5 million people in the state are at risk due to consumption of arsenic-rich water at levels beyond 50 μg/L (GoWB/UNICEF 2002, Nickson et al. 2007). Around 300,000 patients suffering from arsenic-related diseases have been reportedly identified in the affected areas of the state (ASEM 2006).

In order to fight the menace of arsenic, a number of initiatives have been undertaken over the last two decades. In 1987, considering the gravity of the situation, the Government of West Bengal. (GoWB) implemented a project for comprehensive investigations, one of the objectives of which was to develop measures for mitigating the problem. The recommendations of the project included tapping of the third aquifer for drinking water and the need for research to evolve techniques for eliminating arsenic from handpump on-site. This led to further research on developing technically and commercially viable arsenic elimination plants. As a result, a multi-pronged approach in different districts was proposed in 1994 depending upon the local situation. For instance, in some areas such as North and South 24 Parganas, tapping of deep arsenic-free aquifers for drinking water was recommended while in others like Malda that lack such arsenic-free aquifers, surface water treatment was recommended. In general, use of domestic filters as well as community-level filters was also suggested.

Since 1995 an 'Arsenic Task Force' has been established at state level to coordinate and monitor all activities undertaken by various institutions and organizations to control the arsenic menace in the state. Some of the key actors are the GoWB [especially the Public Health Engineering Department (PHED), Division of Health Services (DHS), Panchayat and Rural Development Department (P & RDD)], UNICEF, All India Institute of Hygiene and Public Health (AIIH & PH), School of Environmental Studies-Jadavpur University (SOES), Central Groundwater Board-Eastern Region (CGWB-ER), Bengal Engineering College (BEC), and a number of non-governmental organisations (NGOs) in the affected districts. Also, a Joint Plan of Action has been developed through alliance between GoWB and UNICEF to undertake coordinated action for mitigating arsenic in the state since 2000 (GoWB/UNICEF 2002).

While at one time, especially during the early phase of the mitigation efforts, search for appropriate technologies for supplying the 'at-risk' populations with arsenic-safe water was the major thrust, today multiple technological alternatives exist for supplying arsenic-safe water in the affected communities. It has been argued that a persistent lack of coordination in the mitigation efforts undertaken by the different actors interested in the problem have acted as a barrier in success of the efforts. Given that each of the technological alternatives has its own strengths and weaknesses, different actors have actually favored one or the other alternative. On the whole, many in the government favor the provision of alternative arsenic-safe water sources, while others like the AIIH & PH, UNICEF and the NGOs etc. favor arsenic-removal units (WSP 2000). Also, the lack of a more holistic approach for resolving the arsenic problem has been expressed as a problem (GoWB/UNICEF 2002). Presently, the Joint Plan of Action envisages a multi-pronged strategy with the following integral components: a community-based water quality surveillance system, a disease surveillance system with special reference to patient identification and management, and a multi-level awareness campaign, besides provision of arsenic-safe water supply because drinking arsenic-safe water is the only way to avoid arsenic poisoning (GoWB/UNICEF 2002).

Notwithstanding the details of the specific components, the overall planning for the last part of the strategy adopted and practiced in the state by different actors has been seen as consisting of two basic alternatives: i) provide an arsenic-free water supply, i.e., an alternative to contaminated tubewells, and ii) provide an arsenic-removal technology, i.e., treat water from contaminated tubewells. In either case, the intervention involves development and implementation of technological options for provision of arsenic-safe water to populations at risk. Much research has been conducted by some of the key actors mentioned, as also by the private sector, to develop arsenic-removal filters of different scales—domestic and community-based (WSP 2000, Basu 2003). The installation of arsenic-removal filters, generally referred as arsenic removal plants (ARPs) when installed at community level, began in late 1998. These are based on adsorption, co-precipitation and ion-exchange techniques. Over 5000 ARPs have been set up all over the state at an average price of USD 1500 per plant (personal communication with Head of the 'Arsenic Task Force'). A number of other options are also being considered and installed including the large surface water projects for Malda district that aims to cover 0.8 million people in 5 blocks of the district, and for North and South Parganas; and construction of protected ringwells and rain water storage ponds, properly treated and disinfected (GoWB/UNICEF 2002).

32.2 THE PROBLEM

Given the overall situation, much thrust has been put on developing technologies for providing arsenic-free water as 'hardware' and on promoting their distribution and installation. But how much has the emphasis on the 'hardware' helped resolve the problem? Preliminary observations made in 28 villages in North and South 24 Parganas and Nadia districts in WB revealed that despite the availability of hardware options for supply of arsenic-safe water, coupled with fairly high level of awareness of the arsenic problem, a large number of people still of the view that they lack access to appropriate technologies that can provide them with sustainable and reliable supply of arsenic-safe drinking water.

What are the emergent concerns in relation to the arsenic-safe water supply options that negate their presence in remote villages of the state? What is missing from the present arsenic mitigation strategy adopted in the state? Answers to these questions were investigated through a first-hand qualitative field study undertaken in WB. These questions were further explored from the gender perspective, especially because first, women are the domestic water managers who shoulder the primary responsibility for procuring domestic water from different sources, including the arsenic-safe sources, and second, women (and also children) tend to show symptoms of arsenic poisoning earlier (GoWB/UNICEF 2002).

32.3 METHODOLOGY

In the present study, the various technological options already made available in 35 affected villages in the districts of Nadia, North and South 24 Parganas in WB were evaluated from the users' perspective in the local communities, with a focus on women as domestic water managers. Techniques for data collection included in-depth key informant interviews, observation and focus group discussions. Both women and men were targeted as informants. Technologies under study were surface water supply through pipeline, deep tubewell, community-level ARPs of different kinds and domestic filters. The data analysis was qualitative in nature.

32.4 ARSENIC-SAFE WATER TECHNOLOGIES AT WORK: REFLECTIONS FROM FIELD

During the study, the impact of each of the three kinds of technological options adopted by different factors for supplying arsenic-safe water in the sample villages was assessed. The findings in this regard are presented in Table 32.1. It has been contended that under the first approach provision of additional/alternative community water supplies, such as deep tubewells, hand-dug wells, pond sand filters, or surface water supply to all affected areas will require enormous funds. Rainwater harvesting is a good alternative but can provide water for only a limited period in the year. In contrast, ARPs and filters are believed to offer a cheap and rapid technological option.

32.4.1 *Alternatives for supplying arsenic-free water*

The study showed that women as the primary water collectors in the villages tend to choose a variety of water sources for different domestic purposes. In case of 10 villages studied that have been supplied with piped treated surface water from river, women users of the public taps provided by the government were of the view that the tap water is not suitable for drinking and cooking. It was observed that these public water points are extensively used on-site for washing clothes, utensils and vegetables by women, and for bathing by children and men. On being asked about the reason for rejection of tap water for drinking and cooking, women specifically complained of the taste being unusual, not fit for drinking. The men identified the smell as being of chlorine that some know as being the agent used for treating river water.

Another complaint about the quality of the water was it's being occasionally turbid or muddy, making it unfit for drinking from health perspective. It was found that the pipelines that are spread over the villages originate from a distant pumping/treatment station, get damaged at times, leading to unclean water supply. It was also observed that many of the taps had been damaged and the water kept flowing when the supply was on. Both women and men as users complained of irregularity and/or discontinuity of water supply for several days in continuity. More importantly, they found themselves helpless in situations of discontinuity or quality problems because there was no local agency that could register the complaint or act to restore the lines. At the time of research, no tap

Table 32.1. Arsenic-safe water technologies at work in West Bengal.

Approach	Technological option	Impact	Reason for adoption/rejection
Alternatives for arsenic-free water supply	Treated surface water though pipelines	Used largely for non-drinking purposes e.g., washing, bathing, etc.	• Taste unsuitable • Turbidity making it unfit from health perspective • Irregular supply making sustainable dependence difficult • Lack of knowledge regarding basic purpose of pipeline supply
	Deep tubewell	Accepted as a good source for drinking and cooking	• Water is sweet in taste, clear and 'safe' from bacteriological as well as arsenic contamination
Arsenic removal from contaminated water	Community-based arsenic removal plants (ARPs)	• Large number of ARPs installed by government defunct • ARPs installed by NGOs through community participation are active, used for drinking/cooking but not equitably used by all targeted beneficiaries	• Broken down due to lack of operation & maintenance • Ethnic differences among targeted users hamper equitable access • Lowering of interest in community management over time • Multiplicity of approaches leading to confusion in community
	Domestic filters	Mixed response—sustainably used by very few families	• Regarded as less user-friendly by women • Problem of efficiency in terms of meeting family's water needs on daily basis • Technical limitations due to lack of knowledge • Question of costs

committees for maintenance and management of the pipelines in the local communities had been constituted, and an external mechanic employed by the PHED was actually responsible for the task who did not owe any responsibility to the community.

An important observation made during the study was that although the pipelines had been installed in the villages under study, there was a general lack of awareness in the community about the water supply as an alternative especially provided for addressing the problem of arsenic contamination. Consequently, the people tended to take the taps as an 'add-on' source that could be used for multiple purposes.

On the contrary, the deep tubewell was generally regarded as a good source of water for drinking and cooking wherever available. In as many as five instances, it was recorded that women prefer to fetch drinking water from deep tubewells that were located in the adjoining villages either by themselves or sending their children even when a pipeline was available in the vicinity. In another four instances, it was recorded that women preferred to fetch water from the tubewell in the next village rather than from the existing community-based ARP in their own village. The deep tubewell water is reportedly sweet in taste, clean and 'safe' from bacteriological contamination. Further, it is seen as free from any chemicals and hence any side effects, thereby being medically more suitable, in comparison particularly to the water from ARPs and domestic filters.

However, with regard to the deep tubewells, it was found that these were installed and maintained by the government—the PHED, with no attempts to involve the community in maintenance and management of the technology. Only in a few cases, the deep tubewells shared in the community were privately owned. This raises the question of sustainability of the deep tubewell as a public alternative in the long term. Given that electricity is largely regular in supply, questions would still arise about regular operation of the pump, regular payments to the operator, and repair of the pump when required. If electricity were irregular, question would also arise on availability of funds for the fuel required to operate the pump.

32.4.2 *Arsenic treatment plants at community level*

The ARPs have been installed in affected communities at a large scale through the years by different agencies, including the PHED and the NGOs. A large number of ARPs were visited during the study. A common situation encountered in the case of the PHED-installed ARPs was reported to be lack of maintenance that made them fall into disuse soon after installation. In none of these cases had the community been involved in operation and maintenance of the plants.

The ARPs present a number of technical challenges that can lead to the question of their sustainability over time especially in community situation. On the one hand, life of the medium used determines the efficiency of the unit in arsenic removal, which must be monitored on a regular basis to ensure effectiveness of the plant. On the other hand is the regular need of backwashing, especially a pressing need in the high-iron zones in a number of the districts. This, in turn, requires the services of a regular operator whose presence on-site is necessary. The question of costs involved in maintaining such an operator, that for regular change of medium and the technical requirements for regular quality surveillance are issues that have been overlooked in the process of ARP installation by the government agencies. Consequently, none of the government installed ARPs installed in the sample villages were found to be in use at the time of the study.

On the contrary, the NGOs have initiated the process with community involvement, constituting user groups with number of families varying between 15 and 25. A project funded by the India-Canada Environment Facility (ICEF), New Delhi was under implementation at the time of the research that aimed to install ARPs in 400 villages through local NGOs based upon the approach of 'community participation'. Twenty-two such villages under two different NGOs were visited during the study. The ICEF project is based on the 'demand-driven' approach where interested user families are to come together in a formalized user committee that agrees to undertake the costs as well as responsibilities for operation and maintenance of the ARP to be installed by the project. Once installed, they contribute a monthly subscription for the upkeep and maintenance as also voluntary labor for the purpose.

It was found that even after organizing community participation, a number of problems persisted. All targeted at-risk families were prevented from accessing the ARP on a sustainable basis: socio-cultural as well as economic reasons prevented these from equitably sharing a common ARP. Religion was a basis for losing access to a common plant for certain families residing in the vicinity while everyone was not in agreement to share the costs of operation and maintenance of the plant. Consequently, it was found that in a number of instances, the ARP was not community-based but instead was 'owned, managed and used' by only 3–4 families, the entire cost being shared by only these users. In some other instances only a segment of the targeted group of users—that was homogenous in terms of religion—was actually making use of the plant. Women were found to be more active in deciding on cases of inter-ethnic sharing of water sources in the village. In such cases of exclusion, the other water users in the community continued to depend upon arsenic-contaminated sources.

A large number of cases, despite community participation, a continued interest in the plant did not exist and people refused to pay up their regular contributions leading to disputes and problems of equitable use. In some cases, multiple choices existed for the community—an ARP installed with support of the NGO where the users regularly pay for the operation and maintenance, and a deep tubewell or another ARP installed by government that is state-managed and hence involves no expenditure from the user's pocket. This led the users to a state of dilemma and lack of certainty about which source is really 'good'. Also, irrespective of the component of community participation, continued interest in backwashing by a single person appeared to be a problem, there being a lack of volunteers. In those instances where women were especially encouraged to receive training as plant caretakers, they did not wish to be involved in the task because they found the task quite demanding. On the whole, the community perception about the ARPs appeared to be one of technological complexity that only a few wanted to support.

32.4.3 *Domestic filters for arsenic removal*

The domestic filters that have been lately promoted by the UNICEF also show a mixed response. Qualitative analysis of the data collected shows that managing the domestic filter is perceived as quite a formidable task by women who are supposed to handle it regularly for treating drinking water at home. Though a number of different designs exist, commonly posed problems from women were the weight and size, and the efficiency of the unit for meeting daily water needs of the family especially when the family is large. Among technical limitations mentioned especially by men were a lack of knowledge about life of the medium and the right time to go for a change, logistic problems about regeneration or replacement of the medium, and the extra costs involved in the process. Questions about safe disposal of the arsenic sludge from the domestic filter were also raised by the users during the FGDs. On the whole, it appears from the study that use of the domestic filter is not regarded as a foolproof method of arsenic mitigation by a number of recipients, especially women who also regard it as less user-friendly especially because adoption of the domestic filter requires a basic behavioral change among the women who are otherwise not generally used to undertaking any preliminary treatment of their drinking water.

32.5 THE EMERGENT ISSUES

From the study it emerges that the various actors involved in resolving the arsenic crisis in WB have taken a 'technocentric' view of the problem. They have perceived the problem—especially that of safe water provision to the affected communities—as one of technology provision, keeping themselves isolated from the context of the community of users. Perhaps the assumption is that once a technology will be provided, it will be automatically accepted by the users. Awareness about the arsenic problem exists among most of the local water users in affected communities through different means—directly by the key actors or indirectly through other sources. But even where the awareness exists and alternatives have been provided to supply arsenic-safe water, the adoption has not been holistic or else sustainable. This poses a formidable challenge to the success of the endeavors of different agencies involved in the issue.

Taking together the results of evaluation of the three different kinds of arsenic-safe water supply technologies from community perspective, it emerges that there is a need to look beyond the technologies as physical structures. In fact, technology has been essentially conceptualized as an artefact with social dimension that constitutes a triad of the following components: (a) development and design that are social processes; (b) use which is social because there are 'social requirements' for the purpose; and (c) effects that arise from the interplay with the surrounding social context (Everts 1998). The social meanings and contexts associated with technology are, in turn, essentially gendered as these correlate to women and men in different ways (Fox et al. 2006). In connection with water technologies for mitigating water quality problems as discussed here, the correlations between society and technology can be understood from the perspective of 'use' especially by women who are seen as the primary beneficiaries.

The observations emerging from evaluation of the three different technology options described in the earlier section may be further classified into three critical areas of: i) appropriateness, ii) accessibility and iii) sustainability. The relevant criteria considered by the users with respect to each of these areas are summarized in Table 32.2. Appropriateness of the so-called 'safe' water technology is judged by the users from the perspective of their cultural parameters, and

Table 32.2. Results from evaluation of different technical options from users' perspective.

Technical quality evaluated	Relevant criteria considered by users
Appropriateness	Determined by cultural parameters defining 'good quality', 'safe' water'—these may be different from the scientifically conceived criteria
Accessibility	Defined by social parameters like membership of particular ethnic group rather than physical closeness
	Accessibility as a physical criterion also overruled by culturally determined criteria defining appropriateness
Sustainability	Influenced by interest expressed by users, partly influenced positively by factors of appropriateness and accessibility & negatively by that of 'cultural alienness' and 'technical complexities'

consequently rejected for drinking/cooking when found unsuitable. Accessibility is not determined by physical closeness to habitation but defined by essentially social parameters such as membership of particular ethnic group. Further, accessibility is also influenced by culturally determined criteria of appropriateness, which leads women to traverse longer distances even when so-called 'safe' water sources exist close by. Finally, sustainability of the technologies is determined by interest expressed by the user community, influenced partly by factors of appropriateness and accessibility (positively) and by that of 'cultural alienness' and 'technical complexities' (negatively). Finally, institutional factors such as the government's approach to the issue (such as state-sponsored operation and maintenance) and introduction of 'community participation' as an alternative approach has created further complications as the messages coming down to the user communities are contradictory.

32.6 CONCLUSIONS

Thus it emerges clearly that there is a need to understand the relationship between community and water technologies for supply of safe water in arsenic-affected areas, exploring the problems as well as prospects of each option before implementing program initiatives. The various problems identified above may be clubbed under the broad rubric of 'management' concern. The limitations experienced by the users—particularly women—as highlighted here point to the need of developing a strategy centered in the 'management approach'—particularly about how the technology options introduced for supplying arsenic-safe water to communities at risk can be (better) managed in terms of the technology-society and society-institution interface so as to ensure mitigation of the problem at the users' end. Only when the different management concerns will be identified, integrated and addressed will it become possible to ensure that those affected or at-risk can enjoy sustainable access to safe drinking water.

ACKNOWLEDGEMENTS

We wish to thank the Swedish International Development Cooperation Agency (Sida-SAREC) for the financial support through the research Project No. SWE-2002-108 and also the Knut and Alice Wallenberg Foundation for providing travel support to the lead author for making the presentation at International Groundwater Conference (IGC-2006).

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CHAPTER 33

Role of social factors as determinants for chronic arsenicosis in India

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ABSTRACT: Millions of people living in eastern state of the West Bengal (India) are exposed to arsenic by consuming arsenic contaminated groundwater. The present study seeks to explore the role of social factors as determinants of arsenic exposure and adverse health outcomes. A cross-sectional survey was conducted in five arsenic affected villages of one of the affected districts in the state. Out of 7,678 arsenic exposed people, 410 had developed dermatological manifestations, were taken as cases for the study. Severity of manifestation was found to be associated with arsenic exposure level, gender, occupation, socio economic status, nutritional status, accessibility and affordability of health services. Poor landless population were more suffered from arsenic due to malnutrition, higher arsenic exposure on account of their occupational pattern, inaccessibility to medical care and inability to shift to alternative arsenic safe water sources due to economic reasons. The research has revealed the multidimensional perspectives of arsenic exposure and adverse health outcomes and their complex relations.

33.1 INTRODUCTION

In 1983, the first case of chronic arsenicosis was diagnosed in the state of West Bengal (India) and since then thousands of people were found with similar kind of manifestations. People were exposed to arsenic from domestic hand pumps and irrigation pumps, which have extracted water from underground contaminated aquifers (Das & Chatterjee 1995). On the basis of guideline value of World Health Organization (10 μg/L), around five million people inhabiting 40,000 square kilometer in nine districts of West Bengal are exposed to arsenic contaminated water and nearly one million people among them had arsenical skin manifestation (WHO 2005, Mitra & Guha Mazumder 2004).

Arsenic, a metalloid known for its toxicity and carcinogenicity, is soluble in water and occurs naturally in many minerals (WHO 1981). Arsenic contamination of groundwater of this region is widely accepted as geological in origin (Mukherjee et al. 2006, Bhattacharya et al. 1997). There is still debate over the exact cause of groundwater arsenic contamination, which began since the first case reported. There is a gradual rise in ground water dependence for irrigation and domestic purpose over the past few decades. People engaged in agricultural activities often drink water from contaminated irrigation pumps while working in their fields. Thus, groundwater extracted for domestic use and irrigation has become the sources of arsenic exposure to the population. Furthermore, in early 1970s, as a part of public health policy groundwater was promoted for drinking in order to prevent water borne infectious diseases. As of now, a majority of the villages of West Bengal (95% of total population) are dependent on groundwater reserves for drinking and other domestic uses (Chowdhury et al. 2000). In agriculture development, groundwater was given a major priority in West Bengal. Between 1970 and 1990, groundwater irrigation in the state has increased by 575% in terms of irrigated land area to meet the growing demands of agriculture (Rawal & Swaminathan 1998).

Despite the fact that most of the ingested arsenic is eliminated through urine in physiological metabolism (detoxification); the residue, which is deposited in various organs begins to show pathological manifestations in the form of various symptoms of chronic arsenicosis. For instance, melanosis (spotted darkening or pigmentation) of the skin is the early common symptom, which is usually seen on the chest, back and limbs. Leucomelanosis (white and black spots side by side) is also seen among many patients. Thickening of palm and sole and corn-like swelling (collectively known as keratosis) are the next features of arsenical skin lesions (Chakraborti & Sengupta 2004, Guha Mazumder & Haque 1998). Neurological and respiratory manifestations and occlusion of blood vessels causing gangrene of limbs are also found among a number of cases in latter phase of the disease. (Guha Mazumder et al. 2000, Rahman et al. 2005a) It crosses placenta leading to premature labor, spontaneous abortion, and congenital malformation of the offspring. (WHO 2005) Arsenic causes cancer of various organs after its prolonged exposure. (Rahman et al. 2005b, Smith & Smith 2004) Symptoms of chronic arsenicosis may develop insidiously between six months to 10 years or more after the initial exposure, depending on the extent of contaminated water intake and its arsenic concentration (Haque & Guha Mazumder 2003). Good nutrition plays a vital role in preventing the onset of symptoms (Rahman & Chowdhury 2001). Several studies suggest a strong association between nutritional status (including dietary intake) and clinical manifestations of chronic arsenicosis. People with low dietary intake of protein and micronutrients (calcium, selenium, vitamins A, C and E) are more prone to chronic arsenicosis (Mitra & Guha Mazumder 2004, Vahter & Marafante 1987). Deficiency of these nutritional factors slows down the physiological metabolism including urinary elimination of arsenic. Eventually, the body burden of arsenic increases, which leads to early appearance and faster progression of symptoms of chronic arsenicosis (Hsueh & Cheng 1995).

Although there is no specific medical treatment for chronic arsenicosis, some experts advocate using chelating agents (dimercaprol, d-penicillamine, dimmer-captopropanesulfonic acid, and dimmer-captosuccinic acid) to eliminate arsenic from body through urine in order to reduce its body burden. Despite improvement of symptoms due to administration of chelating agent, there is further need to evaluate these medications as these have adverse side effects (Guha Mazumder & Ghosal 1998, ROHC 2006). Moreover, these drugs are expensive and not available in rural areas where most of the chronic arsenicosis cases have been diagnosed.

Majority of the scientific literature on chronic arsenicosis cover natural and biomedical sciences including epidemiology of chronic arsenicosis. Current mitigation strategy which is essentially based on these research findings has emphasized the promotion of new technology. But selection of new technology, appropriating it in local context and its equitable access are very much relied upon existing social structure and social determinants of diseases. There are some studies, which have analyzed the implication of social factors, such as gender and poverty on chronic arsenicosis. (APSU 2006, Hassan et al. 2005) But there is little information on comprehensive analysis of social determinants of chronic arsenicosis, their complex relationship and its implication on policy development and implementation.

33.2 METHODS

In this study, the social epidemiological approach has been adopted to explore the social determinants influencing the extent and distribution of chronic arsenicosis. The overall objective was to study the extent, distribution, severity of manifestations of chronic arsenicosis, the factors influencing them and their interrelationship. By definition 'social epidemiology' is the study of the relations between social factors and diseases. In other words, it is an approach that provides a foundation for researching, understanding the contribution of social factors, processes to patterns of health and illness in populations (Kaufman & Cooper 1999, Lowis 1990). The study has utilized primary data collected during field survey; combining them with secondary data from various sources.

To collect primary data a cross sectional study was conducted in Murshidabad, one of the worst arsenic-affected districts of West Bengal. In fact, it was also one of the first arsenic affected districts in West Bengal. Five remote villages were selected from Domkol, which was one of the first arsenic affected administrative blocks in Murshidabad. The survey, analysis of data, and reporting were done from 1997 to 2004 (Fig. 33.1).

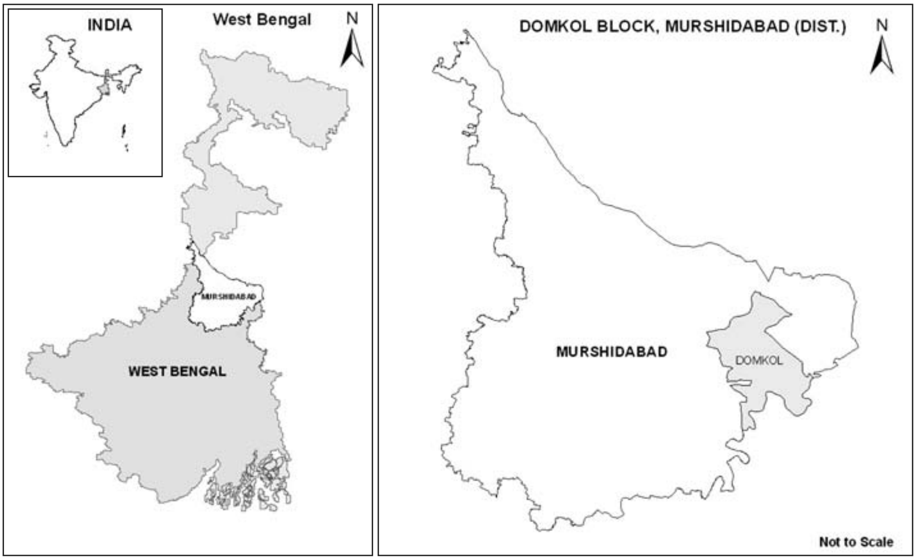

Figure 33.1. Location of West Bengal (in India), Murshidabad district (in West Bengal) and Domkol (in Murshidabad).

The selected villages were severely affected by arsenic and reported several deaths due to severe form of manifestation. Majority of the villagers were aware of arsenic contamination of groundwater and its adverse health impacts as the villages were earlier surveyed by government agencies. This was an important selection criterion, since the people's response to arsenic contamination was one of the study components. Moreover, the village administrative office (*panchayat*) could provide the complete report of household socio-economic survey, which was underway during the field visits. This report helps to get baseline information of socioeconomic status of individual households and their stratification (discussed later). The study villages were representative of rural West Bengal, as they had similar kind of agriculture based economy. Most of the population was landless laborers coming from lowest socioeconomic layer. Land-owned farmers, from higher socio-economic status supervise farm activities done by the landless laborers. Dairy, fishing, artisan, business and service were the next major occupations. Women usually do household activities and participate in secondary economic activities operating from home irrespective of their socioeconomic status. In these villages, hand pumps were the major source of water for drinking and cooking purpose. Despite sufficient rainfall and availability of water bodies, groundwater was usually used in irrigation. The study has aimed to screen whole population living in the five villages. Being early symptoms, clinical diagnosis of arsenical skin manifestations (melanosis, leucomelanosis and keratosis) was used as the basic screening tool. After clinical diagnosis, further detailed investigation of the individual cases was carried out. Complete water intake history and available records of arsenic level of groundwater had been noted to assess the exact number of exposed people. A number of focus group and informal discussions were conducted in each village in order to get more qualitative data.

Report of arsenic level of all hand pumps were collected from Public Health Engineering Department (PHED) of Murshidabad district and School of Environmental Studies (SOES), Jadavpur University, Kolkata. Rest all unrecorded water sources (irrigation pumps, ponds, dug well) were measured by using field test kits (Merck). Water consumption pattern in the villages were very complex, since people drink water from multiple sources on a regular basis. Therefore, measurement of average arsenic exposure (i.e. microgram of arsenic per litre of water per day or μg As/L/day) was done by including arsenic levels of both; domestic hand pumps and workplace of every individual and calculating the average daily intake of water from each source in the past one year. This multiple source exposure assessment is expected to minimize the error of conventional methodology, which takes into account only household level hand pumps. For practical reason, dermatological manifestations (melanosis and leucomelanosis) were taken for analysis as total numbers of cases with other symptoms were much less. Based on standard clinical practice, the symptom was measured into mild, moderate, and severe categories (Guha Mazumder & De 1999, Muralidhar 1996). Nutritional status of individual was assessed by measuring Body Mass Index (BMI). BMI of 20–25 is considered normal, 18.5–20 as low normal and below 18.5 as malnourished (Asthana & Gupta 1998). Socioeconomic categories were constructed on the basis of a participatory assessment to reflect the local context. During survey, people in the study areas, across economic and occupational status groups were asked about their perception of subsistence living and its requirements and was also of relatively comfortable and affluence. Further, they were also asked about nature and amount of household property required to maintain various levels of socioeconomic status. As it was essentially agrarian society, people responded in terms of landholding, livestock, and horticulture. Then the household socioeconomic survey report was referred to categorize the household socioeconomic status. Based on the local context, the households were divided into five categories of socioeconomic status (SES) namely, i) below subsistence, ii) subsistence, iii) just above subsistence, iv) relatively comfortable position, and v) rich—corresponding to the SES categories I, II, III, IV & V respectively.

33.3 RESULTS

Around 90% rural population was dependent on hand pumps for domestic consumption of water. Arsenic levels in different sources of drinking water (hand pumps, irrigation pumps, dug well, and pond) were within the range from 0 to 600 μg/L (95% confidence limit at concentration range in the range 76–120 μg/L). Approximately 62% of the domestic hand pumps and 73% of the irrigation pumps were contaminated with arsenic. The demographic profile of screening, exposure and adverse health outcomes are mentioned in Table 33.1.

Proportion of exposed population in screened population was higher (81%) than proportion of contaminated hand pumps and irrigation pumps, probably due to sharing of same hand pumps by multiple households. Moreover, the farmers who worked in the fields were also exposed to arsenic from the contaminated irrigation pumps, although many of them had arsenic safe source at domestic level.

33.3.1 *Arsenic exposure and severity of manifestation*

The severity of manifestations increased with higher arsenic exposure level and this association was found to be significant (Chi-square test, $p < 0.05$). This finding supports similar kinds of observation by several other researchers, who conducted exposure vis-à-vis manifestation studies in West Bengal and Bangladesh (Smith et al. 2000, Mandal & Roy Chowdhury 1996).

Table 33.1. The demographic profile of total population, screening, exposure, cases in the study villages.

Total population of the study villages	9844
Total population screened*	9427
Total population with history of arsenic exposure	7678
Proportion of exposed population in screened population	81%
Total number of cases detected (melanosis and leucomelanosis)	410
Proportion of cases in exposed population	5%

* Rest 417 villagers could not be contacted mostly due to their migration.

33.3.2 *Arsenic exposure, manifestation and gender*

In a total population of 9844, male to female gender ratio was 1.06:1, but the gender ratio of chronic arsenicosis was 1.97:1. Table 33.2 shows the gender wise exposure rate (proportion of exposed in screened population sub group of same gender), conversion rate (proportion of cases in exposed population sub group of same gender), arsenic exposure level (μg As/L/day), severity of manifestation, age, duration of symptoms and number of death. Males had higher exposure and conversion rates than females. Among the cases of chronic arsenicosis, a greater proportion of males were exposed to higher levels of arsenic than females. There were a number of households where chronic arsenicosis cases were diagnosed only among males but not even a single household was found with only female case.

Women were less exposed to arsenic due to their occupational pattern. Women were essentially involved in household activities and drank water exclusively from domestic water sources. On the other hand, men had multiple water sources for drinking due to their outdoor activities and thus the chances of arsenic contamination were greater. At present, people have become aware of the arsenic problem due to its wide publicity, awareness program and some proactive steps taken by the government and various agencies. There were a number of households in the study villages where women fetched water for domestic use from nearby hand pumps, which were declared arsenic safe by the authority. But their male members still continued to drink water from polluted irrigation pumps in the field during farm activities. Therefore, after shifting of households to arsenic safe water, chances of its exposure to women became further less than their men counterparts. Moreover, average water intake was found to be more among males, particularly those who worked in the fields. In summer and cultivation periods, males used to drink half of their daily water intake from irrigation pumps.

Marriage was found to be an important determinant for the gender differential in chronic arsenicosis. All villages in the area were not arsenic affected and in the affected villages all households were not exposed. In fact, 57% of the villages of the affected blocks are arsenic contaminated. There were a number of instances where married women's parental homes had arsenic safe water till their marriage and they got exposed to arsenic only after their marriage in exposed households. On contrary, men in the same households had been exposed from beginning of contamination of domestic hand pumps, which often occurred before their marriage. The women, whose parental houses were known to be arsenic contaminated, had more severe form of manifestation with longer

Table 33.2. Gender wise distribution of exposure rate, conversion rate, arsenic exposure level, cases, severity of manifestation, mortality, age, and duration of symptoms.

	Male	Female
Exposure rate (%)	89.2	66.1
Cases	272	138
Conversion rate (%)	6.01	4.37
Arsenic exposure level*		
<50	33 (12.1)	20 (14.5)
50–200	152 (55.9)	79 (56.4)
200–400	57 (20.9)	25 (20.0)
400–600	30 (11.1)	14 (10.7)
Severity of manifestation (%)		
Mild	39.7	74.6
Moderate	42.3	22.5
Severe	18.0	2.9
Age of cases in years (95% Confidence Interval)	33.5–37.8	37.1–39.4
Duration of symptoms in years (95% Confidence Interval)	4.1–6.8	2.8–4.3
Number of death	18	4

* Percentage in parentheses.

Table 33.3. Nutritional status and severity of symptoms.

Nutritional status	Severity of symptoms: Mild	Moderate	Severe	Total
BMI < 18.5	19 (14.1%)	76 (56.3%)	40 (29.6%)	135 (100%)
BMI 20–25	95 (87.9%)	11 (10.2%)	2 (1.9%)	108 (100%)

Table 33.4. Occupation wise distribution of cases.

Occupation	Number
Landless laborers	181
Household activities (essentially women)	129
Milkmen, artisans, barbers, other services or business	39
Migrant laborers	24
Students	19
Landed farmers	18
Total	410

duration of symptoms as compared to the women whose parental houses has arsenic safe water. It was found that in all instances, duration of symptoms were more among husbands than wives. This difference was high when parental houses of female cases were arsenic safe.

33.3.3 *Nutrition*

In order to examine the influence of nutritional status on chronic arsenicosis, the severity of manifestation was analyzed in relation to population sub-groups classified by nutritional status and body mass index (BMI). The association of severity of manifestations and BMI was found to be statistically significant (Chi-square test, $p < 0.05$). Cases with poor nutritional status (i.e. lower BMI) had more severe manifestations of chronic arsenicosis. Table 33.3 shows higher proportion of severity of manifestation among malnourished cases (BMI less than 18.5) as compared to normal nutritional status (BMI 20–25).

BMI was found to be significantly associated with the socio-economic status (SES) (Chi-square test, $p < 0.005$). Expectedly, intake of staple food (balanced combination of pulse and cereal) was less among lower SES. As compared cereal (i.e., rice and/or wheat) intake, pulse intake greatly varied by SES. Ratio of per capita mean pulse consumption of two extreme SES categories (i.e., SES V and I) was 4.7, where as ratio of per capita means cereal consumption of same SES category was 1.2. In other words, intake of staple foods was less among poorer section and probably thus BMI was also low. Quantity of fresh vegetables and fruits (sources of micronutrients) intakes were also less among the poor due their high price.

33.3.4 *Arsenic exposure, occupation and manifestation*

Table 33.4 shows occupation wise distribution of chronic arsenicosis cases. Majority of the cases were landless labourers followed by household activities (women) irrespective of their socioeconomic status and others (for instance milkmen, artisans etc).

Nature of arsenic exposure level has been intimately linked with occupation. Landless laborers were exposed to higher levels of arsenic. Table 33.5 shows that as compared to landed farmers, landless laborers had higher arsenic exposure level.

Landed farmers, who had access to alternative source of arsenic safe water at their household level, were exposed to high arsenic only while engaged in agricultural activities like supervision.

Table 33.5. Arsenic exposure level among landless and landed farmers.

	Arsenic exposure level (μg As/L/day)				
Occupation	<50	50–200	200–400	400–600	Total
Landless laborer	13 (7.2%)	104 (57.4%)	36 (19.9%)	28 (15.5%)	181 (100%)
Landed farmers	3 (16.7%)	10 (55.5%)	3 (16.7%)	2 (11.1%)	18 (100%)

Table 33.6. Socio-economic status and age of the cases.

SES	Age of cases (years) (95% Confidence Interval)
I	17.4–25.7
II	19.1–26.2
III	24.7–32.3
IV	28.2–39.8
V	36.3–48.9

Hence, their arsenic intake has been lower than that of the landless agricultural laborers, who spend most of their time in field. There had been several instances of landless laborers being heavily exposed to arsenic from contaminated irrigation pumps but sparing other family members, who were involved only in household activities.

33.3.5 *Arsenic exposure, socio-economic status (SES) and manifestation*

The severity of symptoms is significantly associated with SES (Chi-Square test, $p < 0.05$ and $p < 0.005$ respectively). Poor people suffered from more severe form of manifestations even with similar level of exposure. Table 33.6 shows persons suffering from chronic arsenicosis at younger ages were mostly found in lower SES.

The explanation of lower range of age interval among the poor could be the early exposure to arsenic and/or associated factors such as poor nutrition. People from lower SES quitted education and started working to contribute in household income at younger age. Lower the SES, higher the school drop out rates and participation in household economic activity. Therefore they got early additional exposure to arsenic from polluted water of irrigation pumps. On the other hand, same age group from higher SES could continue their education and hardly got exposed to arsenic from irrigation pumps.

As surface water was arsenic safe, health authority has advised the community to shift to dug well from hand pumps for drinking and cooking. Dug wells provide water from sub-surface layer, which is above the arsenic contaminated aquifers and is believed to be relatively arsenic safe. (Smith et al. 2003) But, households from only higher SES (IV & V) could afford to sink new dug wells and many of them have experienced in improvement of symptoms. Lower SES as a major constraint to promote arsenic safe dug well water has also been witnessed in India's neighboring country of Bangladesh, which is also severely affected by arsenic (van Geen & Ahsan 2002). Based on documents and the history given by health workers, household members and neighbors of the diseased, 22 people died of suspected chronic arsenicosis. Among them 21 belonged to lower SES (I to III).

33.3.6 *Medical care*

Proper medical treatment facilities were only available in Kolkata (the state capital), nearly 250 kilometer from the study villages. Most of the poor villagers could not afford to go to seek treatment

Table 33.7. Seeking medical care by cases.

Source of medical care	Proportion of cases sought treatment
Local private doctor/quack	93%
PHC	87%
District hospital	17%
Kolkata state hospitals	3%

in Kolkata and hence they sought medical care only from local quacks (unqualified doctors or traditional healers), government doctors in rural health care facilities (known as Primary Health Center or PHC) and district hospital. Table 33.7 shows that regardless of SES, majority of the cases (93%) sought treatment first from local private doctors or quacks, followed by government doctors of PHC, district hospital and lastly in Kolkata.

Total 11 cases with chronic arsenicosis, who visited Kolkata to get proper treatment; eight belonged to higher SES (IV & V). Rest three cases were migrant laborers, who despite their low SES (II & III) sought medical treatment in Kolkata, as they were familiar with the city, which encouraged them to stay during their treatment. Apart from affordability, confidence with regard to visiting the city was also an important determining factor in seeking proper treatment.

The source of expenditure on treatment was mainly household savings and also borrowing money or selling property (ranging from meager household belonging to valuable property). There was an apparent gender disparity regarding treatment-seeking, for instance, 97% of males sought medical care, which is higher than females (88%). As far as seeking treatment from local doctors was concerned, there has not been significant gender disparity. Gender disparity was mostly found in seeking treatment at block level or higher (i.e. in district hospital or in Kolkata), as transport cost became an additional economic burden over treatment expenditure. Moreover, in male dominated traditional society, women were not usually allowed to travel alone and this has restrained them to limit with in the health care facilities available in the village. For example, only males went to Kolkata for proper treatment. Nature of spending on treatment has also reflected gender discrimination. Borrowing money or selling property, mostly found for medical care of males, despite the fact that it was less preferred choice of household expenditure on treatment to avoid any possible debt burden. For example, money was borrowed for medical care of 83% for males as compared to 15% for females and property was sold for 15% for males as compared to 3% for females. More attention has been paid towards male due to two reasons, a) traditional gender bias against women b) more severe form of manifestations among males.

33.3.7 *Government response in the study areas*

In the study area, government was mostly involved in water testing and generation of awareness among the population through PHED and health department respectively. Authority analyzed all public water sources (i.e. hand pumps) and advised the villagers to drink water from the sources, which were identified as arsenic safe. Villagers living nearby the handful number of relatively safe water sources could get some benefits. But due to lack of comprehensive intervention program, plight of the people, particularly of poor remained unchanged. Lack of decentralization of medical care has resulted in its poor access to the large section of the community.

33.3.8 *Complex relationship of various factors*

The adverse health outcomes of chronic arsenic exposure were primarily a result of (i) exposure and entry of arsenic in human body, (ii) its metabolism (detoxification and partly retention—body burden) and (iii) elimination through the excretory system (either by physiological mechanism

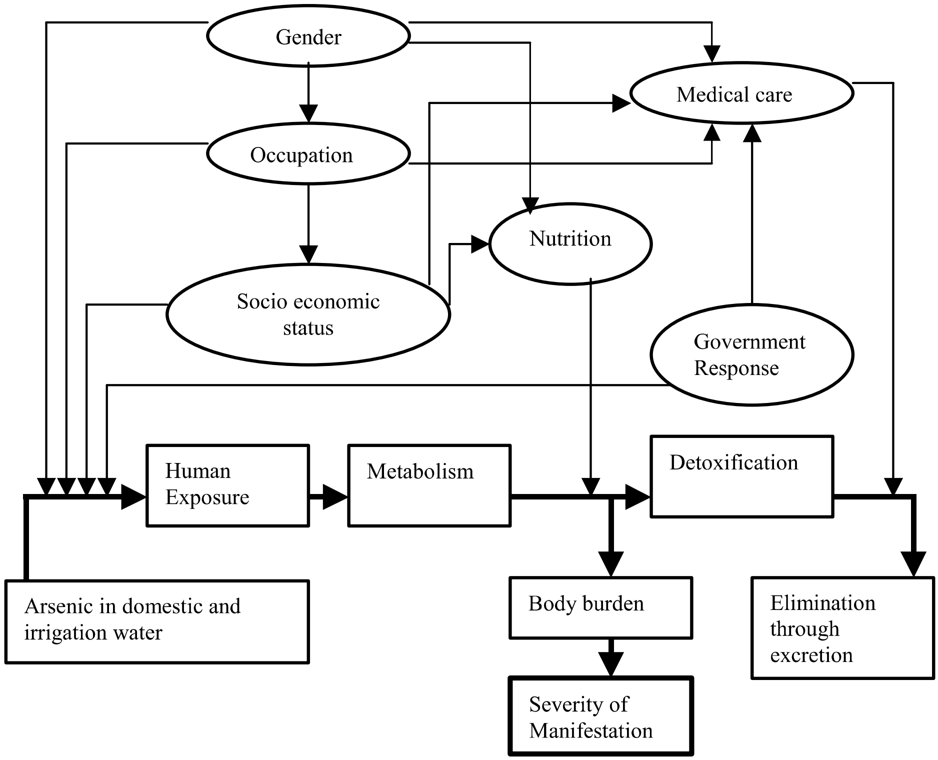

Figure 33.2. Factors and processes influencing arsenic contamination of groundwater, its human exposure, metabolism, body burden and manifestation.

or further enhanced by medical intervention by chelating agents). Multiple factors within the social context and active intervention influenced these three phases either directly or indirectly, in combination or in isolation. The study has revealed the role of individual factor on exposure, biological and social characteristics in relation to distribution and magnitude of manifestation of chronic arsenicosis. But the relations were not linear and independent in nature. Rather these social and biological factors were interrelated in various dimensions. Understanding the dynamic relation by which these factors have influenced the extent and out come of chronic arsenicosis is necessary for effective intervention. This complex relationship is demonstrated schematically in Figure 33.2.

33.4 CONCLUSIONS

Groundwater arsenic contamination has established as India's serious public health issue, which is the outcome of unsustainable agricultural practice and water supply policy. More than four-fifths of the total population in the study villages was exposed to arsenic and more than 5% of the exposed people developed various forms of clinical manifestations. The severity of clinical manifestations was dependent on individual's arsenic exposure level. Arsenic exposure was associated with contamination level of water sources, water consumption patterns and duration of exposure, which in turn was influenced by the dynamics of the social structure—including gender, occupation and socio-economic status. While arsenic exposure was a contributory factor to determine the severity of manifestation, nutritional status along with food intake pattern were likely to be another stronger determinants. There were complex interlinkages among various factors and it was found that poverty, occupation coupled with malnutrition, gender discrimination and the absence of social security (including medical care) aggravated the situation.

The study has unfolded the implications of disregarding social determinants of arsenic poisoning in the current mitigation strategy, which mostly adopted a biomedical and technocentric approach. But this approach has failed to curb the progression of arsenic menace due to existing inequalities in rural West Bengal. The benefits of mitigation programme did not reach the poor because the policy was not appropriated to serve the purpose of poor. Furthermore, if the medical care is decentralized up to district or block level, the existing gender inequalities will be the major obstacle for women to get its access. Chronic arsenicosis among women has serious implication on progeny due to its transmission through placenta and breast milk. Hence, there is an urgent need to include social and cultural dimensions into arsenic mitigation strategies. The present study has found that contamination of irrigation pumps has a big role in causing exposure, especially through direct intake. Other studies also showed that the contaminated irrigation water could affect the agricultural products and food chain. (Heikens 2006, Meharg 2004) Therefore, the ecological impact of arsenic poisoning needs more attention as it has long-term consequences beyond the local region and its population. Appropriate rain water and surface water management will reduce the dependence on groundwater for irrigation and eventually further contamination can be minimized. Lastly, the surveillance of chronic arsenicosis—the key indicator of quality of mitigation program needs to be emphasized. The selected villages were well known for arsenic contamination before the present study was conceived. A number of experts visited the villages to conduct various kinds of studies. Despite this, during the conduct of present study a number of undetected cases were diagnosed. It reveals the importance of surveillance in order to get the right epidemiological picture of chronic arsenicosis.

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CHAPTER 34

Low-energy reverse osmosis (RO) membranes for arsenic removal from groundwater

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ABSTRACT: This work was part of a project on technology partnership for innovative treatment of drinking and industrial water (INNOWA) dealing with innovative treatment of drinking and industrial water in Bangladesh and China. The aim of the project was amongst others to find out an appropriate, reliable, simple and cost-effective technology for removal of arsenic from ground water. Several million people world wide are exposed to high arsenic levels due to contaminated groundwater. In Bangladesh an estimated population of 35 million people—nearly one fourth of the population of the country is at risk. Hence cost-effective and viable arsenic removal techniques for drinking water are badly needed. Recently a new generation of energy efficient so called low pressure reverse osmosis (RO) membranes for brackish and tap water application has been emerged on the market. It is the purpose of this work to investigate the performance of these membranes on arsenic removal. All of the experiments were carried out on laboratory scale using local tap water spiked with As(III) and As(V), respectively. The membrane screening experiments were conducted depending on pH, temperature, pressure and concentration. As(V) rejection exceeds 95%, whereas As(III) rejection is only in the range of 60–80%. The permeate quality of the tested membranes can comply with the recommended Maximum Contaminant Level (MCL) of 10 μg/L up to feed concentration of 2000 μg/L for As(V) and 50 μg/L for As(V), respectively. In the examined pH range (5–9) and temperature range (15–30°C) no considerable change of arsenic rejection has been found. Finally the experimental data have been analysed using the solution-diffusion model. This preliminary screening test provides a basis for subsequent experiments using technical modules.

34.1 INTRODUCTION

Arsenic (As) is a naturally occurring toxic, tasteless and odourless drinking water contaminant. Being ubiquitous in the rocks and soils, As is mobilized in groundwater and enters food chain either through drinking water or consumption of plants and cereals that are grown on contaminated lands. Humans may be exposed to As through drinking water or through the food chain. Arsenic dissolved in water is acutely toxic and can lead to a number of health problems (WHO 1981). Long-term exposure to As in drinking water causes increased risks of cancer in the skin, lungs, bladder and kidney (Smith et al. 1992, IPCS 2001, Kapaj et al. 2006). Arsenic has therefore been recognized as a Class I human carcinogen (IPCS 2001, Centeno et al. 2002). Due to the lowering of the Maximum Contaminant Level (MCL) for As in drinking water from 50 μg/L to 10 μg/L by the U.S. Environmental Protection Agency (USEPA) in recent years, research into appropriate removal technologies has been increased considerably.

Groundwater is the preferred source of drinking water in rural areas, particularly in developing countries, because treatment of the same, including disinfection, is often not required and its extraction system can be placed near consumers. The groundwater of vast areas in the Ganges Delta—in West Bengal and Bangladesh—is highly contaminated by As. The problem of As contamination of groundwater is more serious in Bangladesh, where the groundwater in 59 of the 64 districts is contaminated with As and about two-third of the population is exposed to "the biggest mass poisoning case the world has ever known" (Washington Post 1999). In some places the As concentration in groundwater is higher than 500 μg/L (Bhattacharya et al. 2002). Besides Bangladesh and India, high concentrations of As in drinking water are found in various parts of the world like Argentina, Cambodia, Chile, China, Hungary, Iran, Mexico, Myanmar, Nepal, Pakistan, Thailand, USA, and Vietnam (Nriagu et al. 2007).

The most common valence states of As in geogenic raw water sources are As(III) known as arsenite and As(V) known as arsenate, depending on surrounding oxidation-reduction conditions. Under oxidizing conditions and pH range of 4–9, As(V) is dominant as $H_2AsO_4^-$ and $HAsO_4^{2-}$, whereas under reducing anarobic conditions As(III) prevails as arsenious acid H_3AsO_3 (Ferguson 1972, Schnoor 1996). Hence in this pH range the prevalent As(III) compounds are neutral in charge, while the As(V) compounds are negatively charged.

Arsenic removal from drinking water is part of a project sponsored by the European Commission in the Asia Pro Eco Programme dealing with innovative treatment of drinking and industrial water (INNOWA 2005). The aim of the project was to establish a European—Asian network on innovative water treatment technologies with focus on membrane technologies. This idea represents the steps required to find out an appropriate, reliable, simple and cost-effective technology for removal of As from ground water in particular in rural areas. The following partners are cooperating in the INNOWA project: University of Applied Sciences Karlsruhe (UASK), Germany; Institute for Membrane Technology (ITM-CNR), Rende, Italy; Shah Jalal University of Science and Technology (SUST), Sylhet, Bangladesh and Jiangsu Polytechnic University (JPU), Changzhou, P.R. China.

This paper reports on results of laboratory scale experiments using low pressure reverse osmosis (RO) membranes for the removal of As in valence states +5 and +3. The focus of this work was on DOW FILMTEC™ membranes manufactured by DOW (see DOW Water Solutions 2007a). The purpose of the experiments was to find out the performance of the membranes regarding As removal properties depending on pH, temperature, pressure and concentration. This screening test provides a sound basis for subsequent experiments using technical modules. The long term objective is to develop a simple, low cost pump for rural areas in developing countries.

34.2 AVAILABLE TECHNOLOGIES FOR ARSENIC ELIMINATION

34.2.1 *Overview*

A variety of instrumental techniques are available for elimination of As from drinking water primarily due to the increased awareness about this naturally occurring As in drinking water sources. Several technologies have been developed for the removal of As (see e.g. Sorg 1978, Jekel 1994). The principles of available technologies for As removal can be summarized as below:

- oxidation and sedimentation
- coagulation and filtration
- sorptive filtration
- membrane filtration

Each of these techniques have its benefits and its disadvantages. However, this topic will not discussed in detail here. But the requirements for an acceptable technique for removal of As from drinking water can be summarized as follows:

- high efficiency
- safe technology to ensure the maintaining of the Maximum Contaminant Level (MCL)
- simple operation
- minimal residual mass
- low cost

34.2.2 *Membrane technology*

In recent years, a new generation of high performance membranes has been developed. Membrane technology is relatively well known, efficient and commercially available. It has relatively low energy consumption compared to alternative techniques (like evaporation). In addition, membrane technology is performed without addition of chemicals. Among pressure driven membrane processes reverse osmosis (RO) is the cross flow filtration which produces the highest quality permeates. Due to its dense membrane barrier it ensures physical separation not only of particulates, bacteria and viruses, but of dissolved inorganic and organic contaminants as well. For many years reverse osmosis is well-established in the production of drinking water from seawater and brackish water. For this reason we selected reverse osmosis technology for further investigation for elimination of As from drinking water.

In comparison with other membrane techniques (e.g. nanofiltration) reverse osmosis had been formerly a process consuming relatively high energy. But in recent years, a new generation of so called low energy RO membranes has been emerged on the market.

A comprehensive overview on As removal by pressure-driven membrane processes was published by Shih (2005). Kang et al. (2000) studied removal of As by use of two types of RO membranes (ES-10 and NTR-729HF) manufactured by Nitto Electric Industrial Co., Japan. The removal of arsenite [As(III)] was generally lower than that of arsenate [As(V)] over the pH range of 3–10. Amy et al. (Amy et al. 2003) conducted bench-scale RO experiments using a membrane of type DK2540F manufactured by DESAL (Amy et al. 2005) comprising single element testing and flat sheet on lake water and on spiked deionized water. The results show very high removal efficiency for arsenate up to 96%, but low removal efficiency from 60–85% for arsenite.

Waypa et al. (1997) observed that thin-film composite-type membranes exhibit better removal efficiency of arsenic than the cellulose-acetate type could be the cellulose-acetate type. Thin-film composite membranes also showed higher permeate flow rate and needed much lower applied pressure.

The Environmental Technology Verification (ETV) Programme, created by the U.S. Environmental Protection Agency is seeking the acceptance and use of improved and more cost-effective treatment technologies. Within this programme the performance of two RO membrane modules manufactured by companies Koch and Hydranautics used in package drinking water treatment system application have been evaluated (ETV Report 2001a, b). The field tests have been carried out treating As contaminated groundwater at Park City, Utah, USA. The dominant species in this source is As(V), the total As concentration averaged up to 60 μg/L. Dissolved As showed typically an average removal of 97 to 98% (ETV Report 2001a, b).

34.3 MATERIALS AND METHODS

34.3.1 *Analytical method*

Detection of exact levels of As in drinking water is very important since the calculation of the particular recovery depends greatly on this values. For this case study we followed an analytical method called DIN 38405 D12 (German Standard). With this method dissolved As(V) and As(III) is transformed into arsine AsH_3 by chemical reduction with KI and Sn(II)chloride. Arsine gas is stripped into a silverdiethyldithiocarbamate (DDTC-Ag) pyridine solution forming a coloured complex of arsine, which can be analyzed by using spectrophotometry at wavelength of 525 nm. In this work a Merck Spectroquant Nova 60 spectrophotometer has been used. The standard deviation of this method was observed in our experiments as ±1 μg/L.

A liquid As(V) standard (1000 μg/L) purchased from company Merck served as parent solution to spike tap water and for preparation of the calibration curves. Arsenic trioxide As_2O_3 in powder form, also delivered by Merck has been used for the preparation of the As(III) parent solution (350 mg/L) according DIN 38405 D12. The As trioxide was dried for 24 hours before preparing the standard solution. The 0.462 g of As trioxide, As_2O_3 was dissolved with 12 mL of NaOH (2 mol/L)

Table 34.1. Average annual values 2003 of local tap water in the city of Karlsruhe, Germany (Karlsruhe water quality, 2003).

Parameter	Unit	Value
Aluminium	mg/L	<0.03
Calcium	mg/L	113
Chloride	mg/L	16.3
Electrical conductivity	μS/cm	631
Hydrogencarbonate	mg/L	314
Iron	mg/L	<0.01
Magnesium	mg/L	9.5
Manganese	mg/L	<0.01
Sodium	mg/L	8.9
Sulfate	mg/L	58

and subsequently the solution has been neutralisation with sulfuric acid. Finally the solution has been filled up to 1000 ml with DI water.

34.3.2 *Water quality*

All the experiments were conducted with local tap water. The quality of local tap water used for the experiments is summarized in Table 34.1. This water was spiked with standard solution of As(V) and As(III) to prepare the desired As feed water concentration Besides As rejection, overall salt rejection and permeate flux have been determined as well.

34.3.3 *Experimental rig*

All the experiments were done in a lab scale experimental setup called Lab Module, Type 20 referred hereafter as "Lab-20" produced by DDS RO Division, A/S De Danske Sukkerfabrikker (Denmark), which consists of a plate and frame module where permeate can be collected from a corresponding membrane together with several pressure devices. The module consists of 10 plates. The membrane area per plate is 0.0341 m^2. For the experiments two types of membranes have been installed in alternating order (5 plates each). The flow chart of the experimental set up is depicted in Figure 34.1. A 200 L barrel served as feed tank. Each sample needed only 350 ml of permeate for analysis. After each series of sampling the barrel had been refilled. Thus the recovery rate of membrane filtration can be regarded as almost zero.

34.3.4 *Operational conditions*

Membranes are very sensitive and may be damaged somehow during installation. Therefore, before starting As removal experiments the integrity of *the* experimental rig has been verified by standard test according recommendation of the DOW company. For this purpose the rejection of a sodium chloride solution with electrical conductivity of 3800 μS/cm has been checked. The salt rejection at 5 bar, pH 7 and 25°C exceeded 98%.

Several As removal experiments have been carried out in the Lab-20 unit under different operational conditions:

- pressure dependency
- concentration dependency
- pH dependency
- temperature dependency

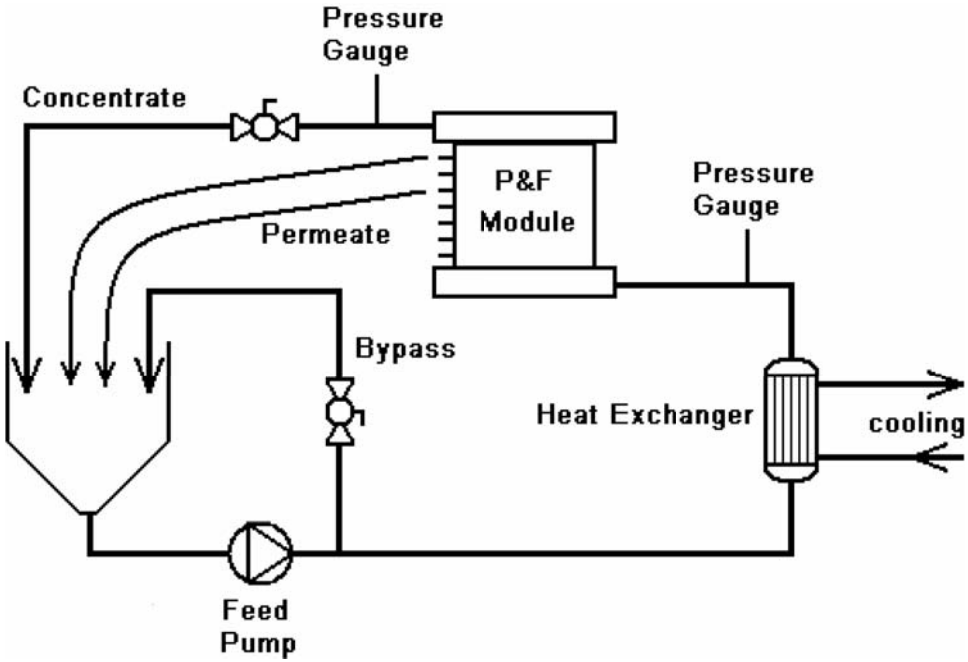

Figure 34.1. Flow chart of the experimental device Lab-20 (DDS Lab-20 Manual 2007).

For As removal experiments pressure range has been choosen between 5 and 20 bar, which is the typical pressure for RO application. The operating conditions of pH (5–9) and temperature (15–30°C) were selected in accordance with the expected range of the ambient conditions in the groundwater in Asian countries like Bangladesh and China. The As feed concentration was measured up to 2000 μg/L, which is comparable to peak concentration in groundwater. Only few results depending on these conditions are presented in this paper, the vast majority of the results are presented in a detailed report in the department of University of Applied Sciences Karlsruhe (Deowan 2005).

34.3.5 *Applied membranes*

We selected two types of FILMTEC^{TM} membranes from company DOW which are both polyamide thin-film composite membranes (see DOW Water Solutions 2007b):

- type XLE (Extra Low Energy)
- type LE (Low Energy)

From other membrane producing companies we also applied other two types of low energy RO membranes. For the secrecy agreement with company DOW we have mentioned these membranes in disguise as Competitor A and Competitor B.

34.3.6 *Pressure dependency*

Figures 34.2 and 34.3 give the As concentration in permeate and the percentage rejection. For As(V) and As(III) at feed concentration of 100 μg/L, pH 7 and 25°C at the applied pressures 5, 10, 15 and 20 bar.

For both DOW membranes XLE and LE the rejection of As(V) is distinctly better than the corresponding As(III) rejection. This can be explained as follows. Both membranes consist of

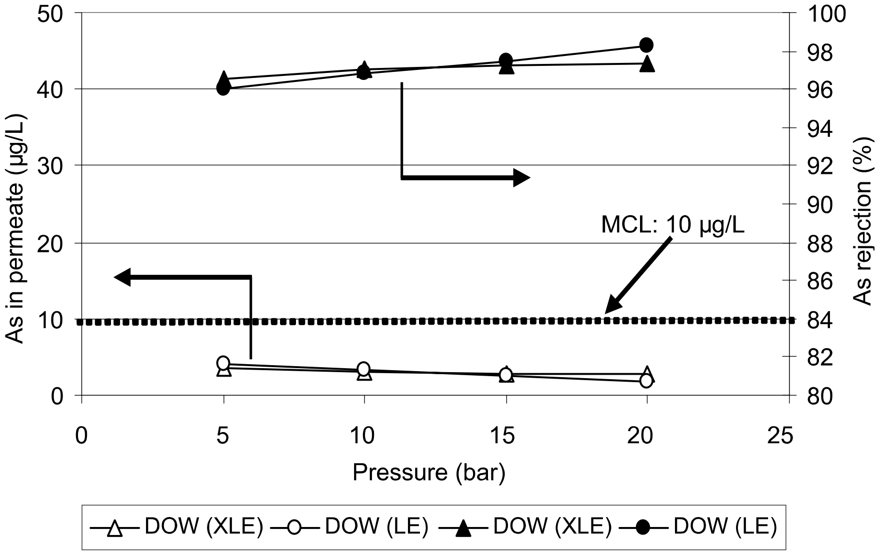

Figure 34.2. Effect of pressure on arsenic in permeate and on arsenic rejection for 100 µg/L As(V), pH 7 and 25°C.

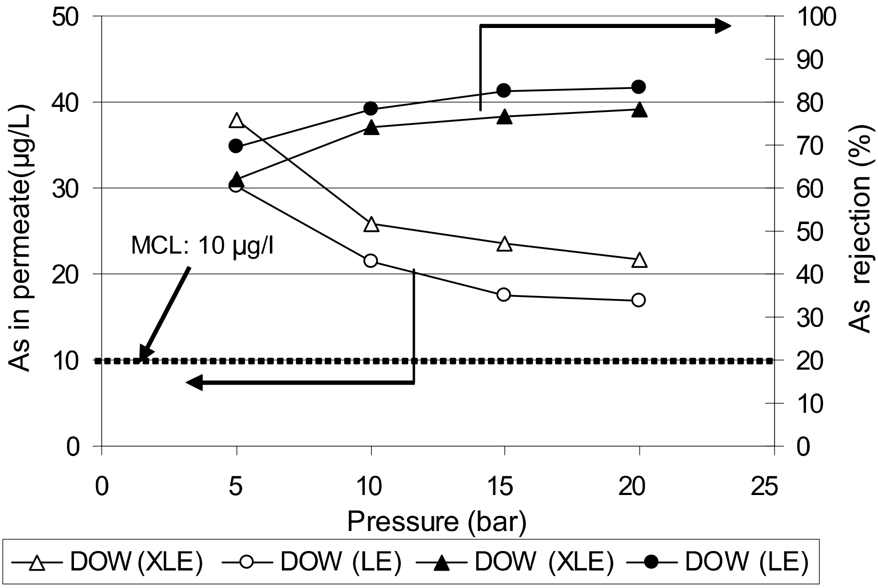

Figure 34.3. Effect of pressure on arsenic in permeate and arsenic rejection for 100 µg/L As(III), pH 7 and 25°C.

polymers with negatively charged groups. Through a charge exclusion effect the rejection of negatively charged ions as for As(V) is enhanced (see section 34.1). In the observed pH range As(III) exist only neutrally charged, hence it is less rejected.

For As(V) the permeate concentration is in the whole pressure range below the MCL of 10 µg/L and the rejection is 96–98%. With higher pressure the rejection rate is slightly better. For As(III) all values in the permeate are higher than the MCL. Unlike As(V) the experiment shows a higher

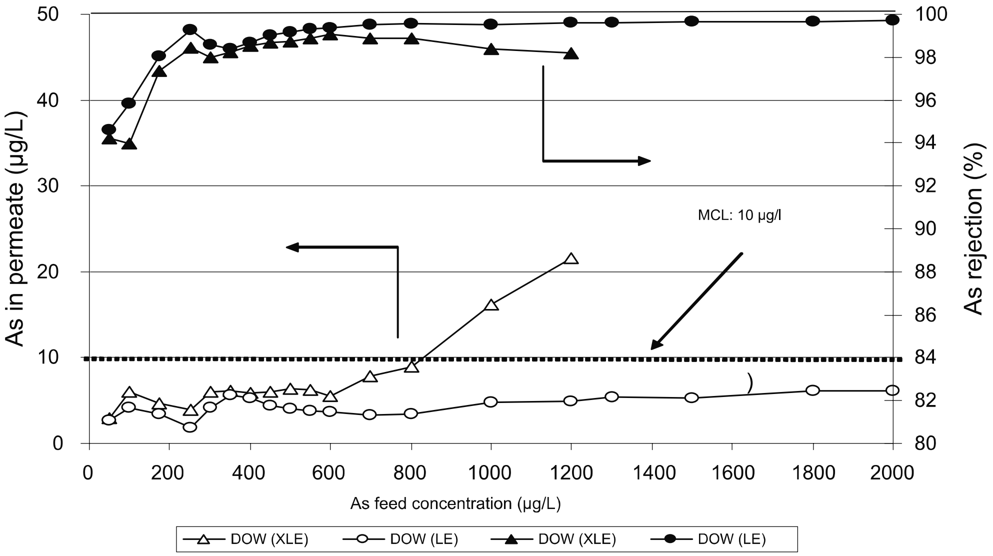

Figure 34.4. Effect of feed concentration on arsenic in permeate and arsenic rejection for As(V).

pressure dependency for As(III). The increase of the rejection rate with higher pressure can be explained by the increase of water flux through the membrane. The increased water flow is showing an increase for the rejection through higher dilution, since the the permeability for As is hardly subject to change.

Other experimental results indicate that for As(III) only at a feed concentration of 50 µg/L provides permeate concentrations below MCL (Deowan 2005).

34.3.7 *Concentration dependency*

Figure 34.4 shows the concentration dependency of As(V) in permeate and the percentage rejection for both DOW membranes at 10 bar, 25°C and pH 7.

The membranes of type LE and of type XLE suggest different flux rates at 10 bar (about 40 L/(m^{2}h) and 60 L/(m^{2}h), respectively). For better comparability all As concentrations have been linearly normalized to 40 L/(m^{2}h) as As concentration in permeate depends on the permeate flux rate. Both membranes show excellent As(V) rejection. It is noteworthy that for type LE up to feed concentration as high as 2000 µg/L As in permeate is still lower than the MCL.

34.3.8 *pH dependency*

Figure 34.5 shows the pH dependency of As(V) in permeate and the percentage rejection for both DOW membranes at feed concentration of 100 µg/L, 10 bar and 25°C. From Figure 34.5 we can observe that DOW (XLE) and DOW (LE) membranes are not very pH sensitive. With the exception of DOW (LE) at pH 5, As rejection increases slightly at higher pH. The results are similar for As(III) (Deowan 2005).

34.3.9 *Temperature dependency*

Figure 34.6 gives the temperature dependency for DOW type XLE and LE between 15 and 30°C. It can be seen that DOW (XLE) and DOW (LE) membranes show almost no temperature dependency in the range between 15–30°C. The slight increase of the rejection rate with higher temperature can be explained by small increase of water flux through the membrane (see also section 37.4).

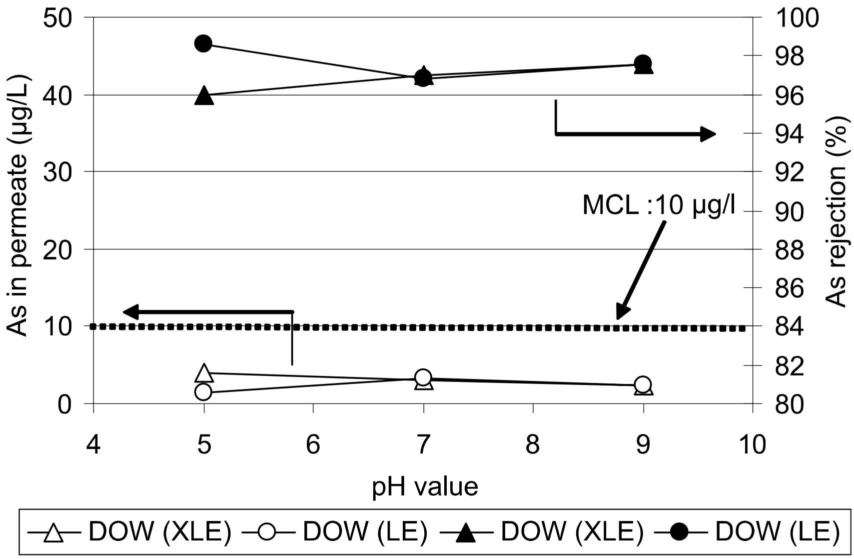

Figure 34.5. Effect of pH dependency on arsenic in permeate and arsenic rejection for 100 µg/L As(V), pH 7 and 25°C.

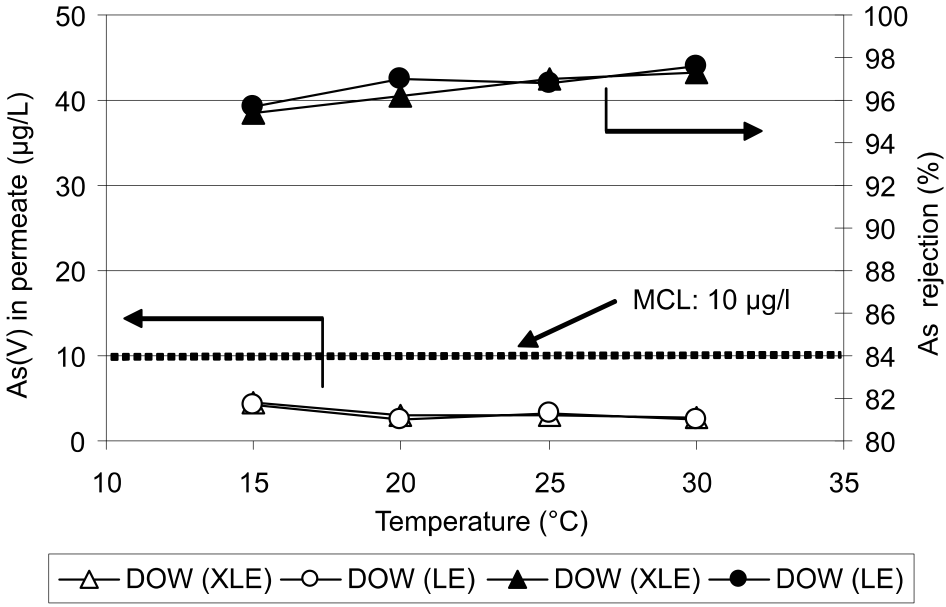

Figure 34.6. Effect of temperature on arsenic in permeate and arsenic rejection for 100 µg/L As(V), 10 bar and pH 7.

34.4 SOLUTION-DIFFUSION MODEL

The solution-diffusion model is a well established model to describe the properties of dense membranes (see e.g. Baker 2004). In this model total mass transport can be considered as sum of permeate and solute transport. The solution-diffusion model has been applied to the experimental data in order to characterize and compare the applied membranes in terms of water permeability and

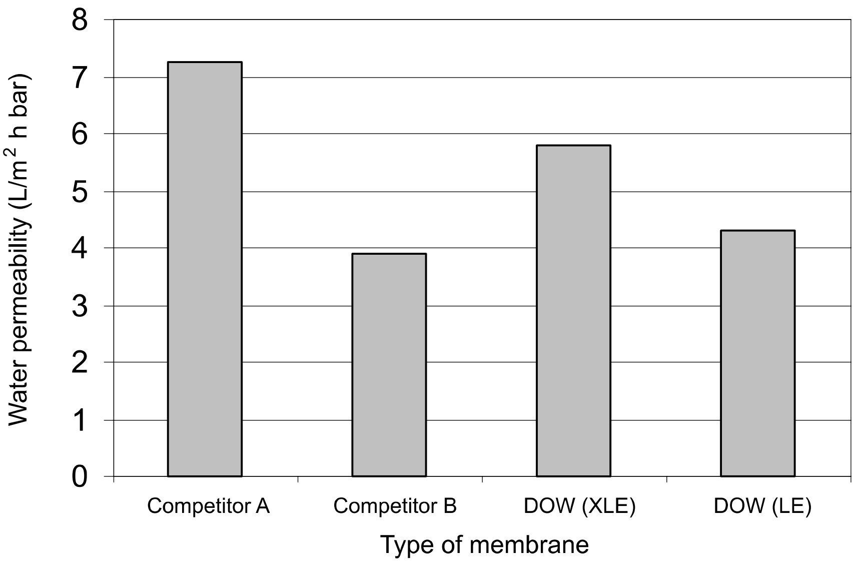

Figure 34.7. Water permeability coefficient A of the tested low energy RO membranes.

solute (As) permeability. Besides DOW membranes our experimental results for two competitors have been analyzed as well. As it is generally known that good membrane performance implies high water and low solute permeability.

34.4.1 *Determination of the water permeability coefficient*

The water permeability coefficient A of the membranes has been calculated from pressure dependency of the permeate flow as shown below:

$J = A(\Delta P - \Delta\pi)$
$J = \textit{Permeate volume flow (flux)}$
$\Delta P = \textit{Applied pressure difference}$
$\Delta\pi = \textit{Osmotic pressure difference}$
$A = \textit{Water permeability coefficient}$

From Figure 34.7 we can observe that Competitor A and DOW (XLE) have higher water permeability—between 6 to 7 L/(m^2 h bar)—than Competitor B and DOW (LE), which show approximately same values of water permeability around 4 L/(m^2h bar).

34.4.2 *Determination of the arsenic permeability coefficient*

Using the solution-diffusion model the solute mass flux can be described by

$$\dot{m}_s = B \cdot \Delta c_s$$

where, B is the solute permeability coefficient and Δc_s is the solute concentration difference across the membrane. From solution-diffusion model we know that the rejection coefficient R can be described as follows (Baker 2004):

$$R = \frac{A(\Delta P - \Delta\pi)}{A(\Delta P - \Delta\pi) + B}$$

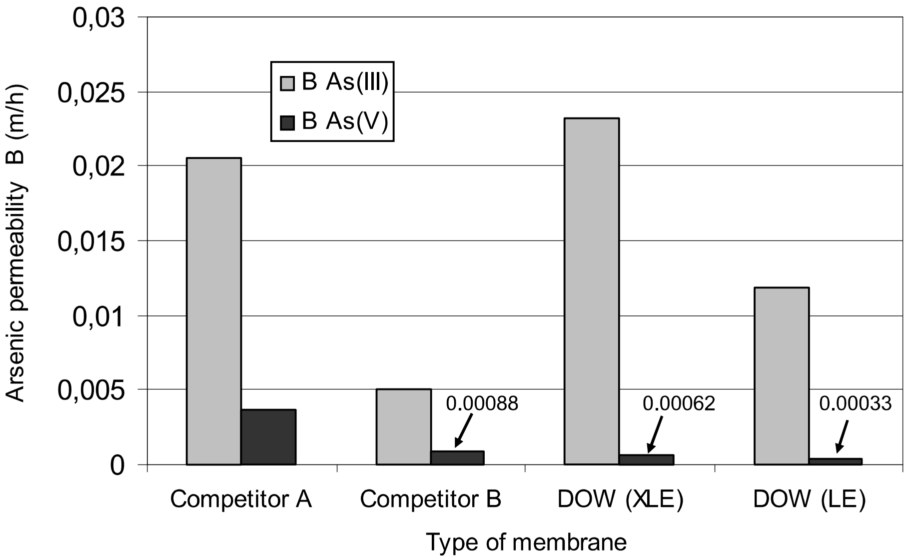

Figure 34.8. Arsenic permeability coefficient B of the tested low energy RO membranes.

which gives the *B* value in a simplified form as mentioned in the following equation:

$$B = A(\Delta P - \Delta\pi)\left(\frac{1}{R} - 1\right)$$

The equations show that *B* and *R* are reciprocally proportional. This implies that the higher the solute permeability through the membrane the lower is the As rejection. From Figure 34.8 we can observe that the permeability of As(III) through the membranes is generally much higher than that of As(V). Competitor B has the lowest permeability for As(III), hence it has the highest As(III) rejection capability. It should be pointed out that for all of the tested membranes As(III) exceeds permissible limit MCL of 10 μg/L already at feed concentration lower than 100 μg/L. The As(V) permeability of DOW (LE) is lowest of all tested membranes, whereas those of Competitor B and DOW XLE are of similar size, whereas permeability of Competitor A is considerably higher.

34.5 SUMMARY AND OUTLOOK

After verification of the laboratory results on the DOW low energy RO membranes from this study, we can summarize the following points:

Overall it was found that the As rejection is significantly higher for As(V), which typically exceeds 95%, than for As(III), which is usually below 80%. Regarding As(V) concentration in permeate DOW (LE) can comply with the MCL of 10 μg/L up to feed concentration of 2000 μg/L. The rejection for As(III) is only at feed concentration around 50 μg/L sufficient to keep As concentration in permeate below the MCL. Overall DOW (LE) shows a slightly higher As rejection rate compared to XLE, whereas XLE has a higher water permeability. Both membranes exhibit only a very slight pH and temperature dependency.

Competitor A has higher water permeability compared to the other membranes, but it has higher As permeability as well. For Competitor B, it is characteristic having low As permeability for As(III), even lower than DOW (LE).

These results indicate that when dealing with underground water having high values of As(III) a pre-oxidation step will be recommended. Simple cost-efficient pre-oxidation processes are further investigated in the INNOWA project.

The results of this laboratory scale investigation serve as preliminary screening test for subsequent experiments using technical modules. Eventually the INNOWA project aims at the development of a simple and cost effective water filter for developing countries like Bangladesh. In this regard As adsorption materials will be investigated as well. In recent years, a new generation of high performance sorption materials are emerging on the market. For instance ADSORBSIA^{TM} GTO^{TM} is a titanium based As adsorption media which is newly offered by company DOW (see DOW Water Solutions 2007b).

ACKNOWLEDGEMENTS

This work was supported by the European Commission under the Asia Pro Eco Programme (BD Asia Pro Eco/07/96638). The authors are also grateful to company DOW (Rheinmünster, Germany) for their financial support. We deeply appreciate the coments of the anonymous reviewers and their valuable comments on an earlier version of this manuscript.

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CHAPTER 35

Bioremoval of arsenic by green alga

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ABSTRACT: Removal of heavy metals from drinking water has been a long term-challenge. During the recent era of environmental protection, the use of microorganisms for the recovery of metals from contaminated water has generated growing attention. The present study focused on isolating algae from the areas in Matlab Upazila in southeastern Bangladesh, with elevated concentrations of arsenic in groundwater and studies on the removal of arsenic by algae from contaminated water. The algae was grown in the synthetic water with a composition that is similar to the average groundwater in the region around Matlab Upazila containing 50 μg/L and 100 μg/L arsenic exposed to 16 hrs light period to get substantial biomass. The major population of algae from the arsenic contaminated area has been identified as green alga, *Chlamydomonas*. The growth rate and arsenic removal was monitored in the presence of high and low phosphorus content at pH 5 and 7 in the media over a period of time up to 45 days. The alga was able to grow at concentrations of arsenic up to 1 mg/L in the media. Results from this study shows that growth rate increased with time. Media containing high phosphorus content and at pH 7, alga could be able to remove 80% of the arsenic with initial concentration of 50 μg/L arsenic; whereas at 100 μg/L arsenic the reduction was up to 50%. *Chlamydomona*s was found to have capabilities of endurance against high concentration of arsenic and accumulation of arsenic that has a potential application for the removal of arsenic from contaminated water.

35.1 INTRODUCTION

Drinking water is a major source for the ingestion of inorganic arsenic (As) in the human system. Because of the high As toxicity, numerous efforts have been dedicated to the study of its chemical behavior in natural systems and in acid drainage (Maeda et al. 1985, Mok & Wai 1994, Francesconi & Edmonds 1994, Carlson et al. 1994, Daus et al. 1998, Williams 2001, Das et al. 2005). Several physico-chemical methods based on coagulation, ion-exchange, reverse osmosis and flocculation have been developed for As removal (Chang et al. 2002, Steven et al. 2002, Sancha 2006). However, these methods are associated with inherent disadvantages such as high material costs and the generation of toxic sludge that are causal agents for diseases and carcinogenetic effects which are often difficult to manage (Okuda et al. 2001, Narasiah et al. 2002).

Arsenic is ubiquitously present in soils, sediments and in the living tissues of plants and animals, as well as in zones of biological activity in the oceans. The dominant chemical speciation of As occurs in the form of arsenate [As(V)] in surficial water. However, other chemical forms of As such as arsenite [As(III)] are also prevalent in water more preferably in groundwater due to the influence of natural biogeochemical processes (Smedley & Kinniburgh 2002, Bhattacharya et al. 2002, 2007, Naidu et al. 2006). Microbes potentially contribute to As cycling in the subsurface *via* several mechanisms. Both As(V) and As(III) are potentially transformed by biologically mediated redox reactions by bacteria, fungi and algae (Craig 1989, McSheehy & Szpunar 2000). They could directly reduce solid phase As(V) into more toxic mobile As(III) (Pontius et al. 1994, Goh & Lim 2004). Reduction of aqueous phase As(V) could similarly increase the overall toxicity and mobility of As in groundwater. Microbes like *Bacillus*, *Citrobacter*, *Clostridium*, *Desulfitobacterium* and *Desulfomicrobium* grow by reducing arsenate methylation and dissimilation (Macy et al. 2000, Niggemyer et al. 2001, Herbal et al. 2002, Yakamura et al. 2003). Some oxidize As to detoxify it and assimilate at higher concentration that results in biomagnification that might hold diverse ramifications for fishing industry. The various As species are accumulated by living organisms and exert different toxicological impacts on higher plants and on phytoplankton or macrophytes and toxicity of As species to lake phytoplankton generally differ (Goessler et al. 1997, Meharg & Hartley-Whitaker 2002). Organometallic compounds are reported to be less toxic than inorganic forms, methylation and external secretion are assumed to be main defense mechanisms in algae (Maeda et al. 1992). Synthesis of sulfur-rich peptides called phytochelatins (PC), which form complex with heavy metals and render detoxification possible in higher plants like *Silen vulgaris, Rauvolfa serpentine, Arabidopsis, Holcus lanatus* and *Cytisus striatus* (Zenk 1996, Sneller et al. 1999, Bleeker et al. 2003).

An increasing awareness has been felt to the conservation of renewable natural resources. For long period, it has been suggested that heavy metals from polluted/contaminated systems may be removed by phytoplankton algae (Meagher 2000). However, few reports have been published concerning how algae metabolize As into their tissues (Maeda et al. 1985). It is of significance to study the responses of algae to inorganic As in culture media. An algal strain which has a high accumulation capability for As could be useful for the elimination of environmental As contamination. The aim of the present study was to isolate and characterize algae from the As contaminated sites; find out the tolerance of isolates in different concentration of As; efficiency of the isolates to uptake As in *in vitro* condition under determined growth conditions; effect of pH and phosphorus in As uptake were discussed.

35.2 MATERIALS AND METHODS

35.2.1 *Sampling site*

Algal samples were collected from seven sites in Matlab Thana located 55 km south-east of Dhaka in Bangladesh (Fig. 35.1). The region was reported to contain elevated amounts of As in the groundwater. The specific sites used for the field studies were Mubarakdi and North Dighaldi villages, located along the western border of the Matlab Upazila (23.30–23.33°N: 90.68–90.69°E). Recent studies carried out in Matlab Upazila reveals that nearly 81% of the tube wells in Dighaldi contained As levels above the national drinking water guideline (Jakariya et al. 2007).

35.2.2 *Isolation and characterization*

Algal samples were taken in a plastic tube and shaken to make a uniform suspension. From the suspension 1 ml was taken and inoculated in microalgae rich nutrient liquid culture media (25 mL) and incubated for a week with 16 hrs light period. After that the cultures were sub-cultured by serial dilution, 100 μL were inoculated on fresh rich nutrient agar media in triplicates. Individual algal colonies were picked up and sub-cultured periodically until obtaining pure cultures were obtained. The morphology of the algal colonies were characterized through microscopic studies.

35.2.3 *Algal growth rate and arsenic analysis*

To determine the growth rate of algae, synthetic groundwater water was prepared following the average groundwater composition in Matlab Upazila (von Brömssen et al. 2007). The typical groundwater composition used in the present study is given in Table 35.1.

Table 35.1. Representative groundwater chemical composition at Matlab Upazila (reproduced from von Brömssen et al. 2007).

Constituents	Concentration (mg/L)
HCO_3	340
Cl^-	28.4
NO_3^-	0.29
SO_4^{2-}	0.028
PO_4^{3-}	6.32
NH_4^+	2.571
Na^+	35.9
K^+	5.1
Mg^{2+}	24.7
Ca^{2+}	70.8
Fe	7.94
Mn	0.31

The synthetic groundwater was spiked with As at concentrations at 50, 100, 500 and 1000 μg/L and absence of As as control. In both control and test flasks, the P concentration was 2 mg/L and 6.3 mg/L. The cultures were kept in Erlenmeyer flask at room temperature with 16 hrs light period. Stock cultures were maintained at room temperature. Growth rate was determined by taking 1 mL of culture from each flask and measuring in spectrophotometer at 750 nm every seventh day (Olympus BX51 with AnalySIS) and images were captured using Sony PC 120 camera.

35.2.4 *Arsenic analysis*

The efficiency of algal isolates to remove As from the synthetic groundwater media was tested by growing algae in the presence and absence of As over a period of time. Algal cultures were inoculated in a synthetic groundwater media in an Erlenmeyer flask (500 ml). The initial culture volume was maintained and measured the density of culture by spectrophotometer with an optical density of 0.1 and incubated at room temp with 16 hr light over a period of time. To determine the As uptake by algae, every seventh day a known volume of culture suspension were withdrawn from the flask and centrifuged at 3000 g for 10 min. Hach As test kit was used as a preliminary indication of the inorganic As concentration in the samples. The supernatant was measured using As test kit and according to the method described by Hach (2000). Briefly, 50 ml of water sample was poured into a reaction chamber followed by the addition of sulfamic acid and powdered zinc. The test strip was placed in the flap region of the cap and the production of arsine (AsH_3) gas was observed by change in color of the test strip. The color difference was compared with the control and color chart which could be able to detect as low as 10 μg/L concentration. Media without As was used as control.

35.2.5 *Atomic absorption spectrometer (AAS)*

A volume of 10 ml aliquots from each sample was taken and filtered through 0.45 μm polycarbonate filters (Poretics Corp.) and measured using GFAAS (Spectra AAS 200, Varian) with a nickel modifier and As specific lamp (Lake 2002). Arsenic was measured using continuous flow of hydride generation method. The hydride was generated using concentrated HCl and 0.35% NaBH4/0.30% NaOH in deionized water. A borosilicate/sodium hydroxide solution was used to volatilize arsenite by measuring the presence of unused As in the culture media.

35.3 RESULTS AND DISCUSSION

An alga was isolated from the As contaminated area from Matlab, Bangladesh and studied the As tolerance of algae and uptake efficiency with different concentration of As, pH and phosphorus in the media. All the experiments were performed by growing algae in the presence of synthetic groundwater media.

35.3.1 *Isolation and characterization*

Samples from seven different locations (Fig. 35.1) within the As contaminated area were grown in rich media and synthetic groundwater media to find out the rate of growth and As tolerance. The colonies were observed under the microscope to see the morphology and purity of the culture. Sample sites 1, 4 and 7 have similar colony and population of algae. The samples sites 2, 3, 5 and 6 having similar algal species (Fig. 35.2). Based on microscopic observation and description of the organisms they were falling into the genus *Chlamydomonas* of green algae under the division Chlorophyta. They are generally having flagella which help in movement. Colonies which have grown in the groundwater media in the presence and absence of As were selected for further study.

35.3.2 *Rate of growth at low and high phosphorus concentration*

Growth rate of algae was measured in media supplemented with 50 µg/L and 100 µg/L As concentration at pH 5 and pH 7 with time. As can be seen in Figure 35.3a, in both As concentrations, the growth rate of algae is increasing with time. The highest growth rate was observed at low phosphorus content on week 7 irrespective of the As concentration and pH. The growth rate increased in week 1, stagnant in week 2 and 3 and after that there is an increased growth even after week 7, whereas the growth rate reached stationary phase after week 2 in media with pH 7. On the other hand, in media with high phosphorus content the growth rate is comparatively lower than media

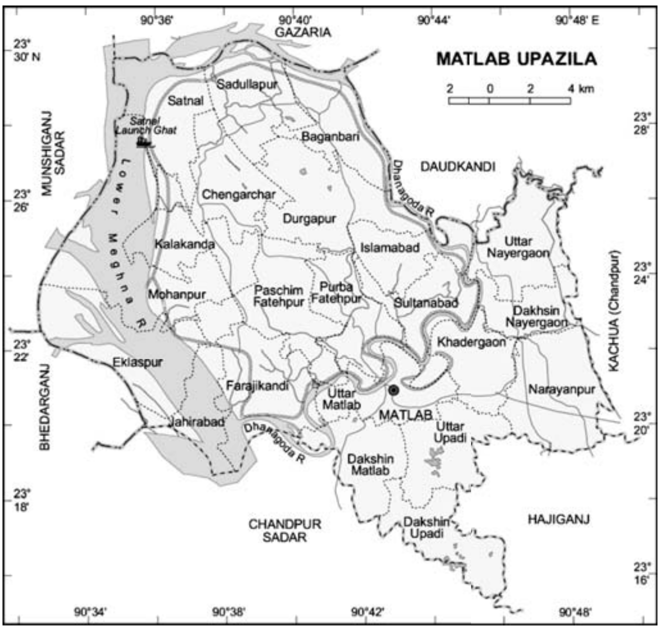

Figure 35.1. Location map of the sampling site within an area with high arsenic concentration at Matlab Upazila at southeastern Bangladesh.

with high phosphorus content. The growth rate reached to stationary phase after week 3 irrespective of the ionic strength of the media (Fig. 35.3b).

35.3.3 *Arsenic uptake at low and high phosphorus concentration*

This experiment was carried out to find out the efficiency of algal isolate on the uptake of As in the synthetic groundwater media and the influence of P and ionic strength of the media. As can be seen in Fig. 35.4a and b, the As concentration is reduced in the media with time. The pH of media influences the uptake of As by algal isolate. At pH 5, 50–55% As was reduced in the media with high phosphorus concentration irrespective of As content. On the other hand, at pH 7, 80% As reduced in the media with 50 μg/L As content irrespective of the phosphorus concentration whereas in media with 100 μg/L As content there was 55% reduction of As. From these results it is clear that As uptake was influenced by the ionic strength of the media and the concentration of As. The isolates from the present sampling sites are taking 5 weeks to reduce approximately 80% As present in the culture media.

Phosphorus and pH are the most important criteria in influencing plant growth and As uptake (Marschner 1995). Maeda et al. (1992a, b) reported that freshwater algae will release As rapidly in the late log phase of the growth. In this study it was observed that the As concentration is reduced gradually in the media with algal cultures in both As concentration (50 μg/L and 100 μg/L). The As reduction was increased by 20% in media with 50 μg/L As over a period of time in both pH and phosphorus concentration.

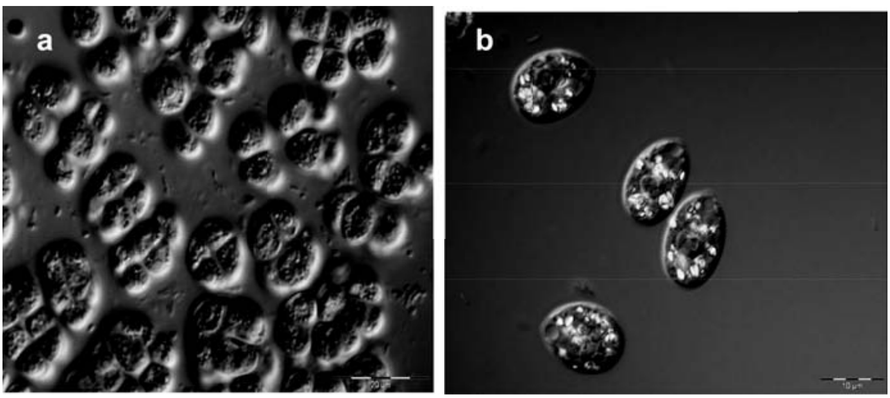

Figure 35.2. Colony morphology of the isolated algae. a) shows the aggregates from sampling site 1, 4 and 7; and b) individual colonies from sampling sites 2, 3, 5 and 7.

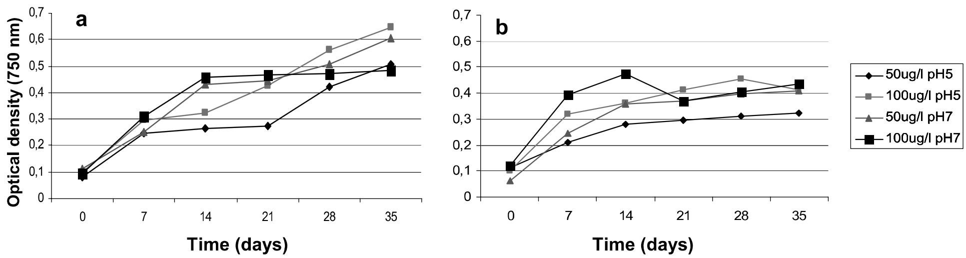

Figure 35.3. The optical density of algal growth in the presence of arsenic at concentrations of 50 μg/L and 100 μg/L respectively. a) growth rate at pH 5 and pH 7 with low P content; and b) growth rate at pH 5 and pH 7 with high P content.

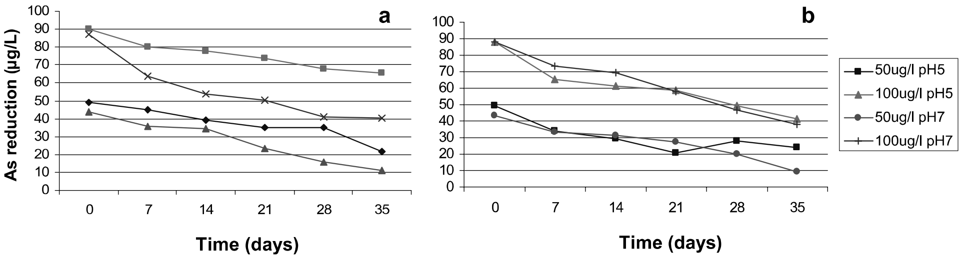

Figure 35.4. The uptake of arsenic by algae at low and high phosphorus concentration. a) arsenic uptake at low phosphorus content, pH 5 and pH 7 in the presence of 50 and 100 μg/L arsenic concentration; b) the arsenic uptake at high phosphorus content, pH 5 and pH 7 in the presence of 50 and 100 μg/L arsenic concentration.

This imply that the As is accumulated in the cells rather than by releasing. It has been stated that arsenate is taken up by two distinct phosphate transport pathways of bacteria (Poole & Hancock 1984; Willsky & Malamy 1980). Theil (1988) suggested that intracellular competition between phosphate and arsenate in cyanobacteria which are believed to be tolerating arsenate by possessing resistance mechanisms that include both efflux systems and intracellular metabolic pathways. The algal isolates from this study shows that the rate of removal was high if the initial concentration of As is lower irrespective of phosphorus concentration and an increase in growth rate at week 5. These could be explained by the species variation, type of media used and the initial concentration of As used in the experiments. In the present study we have used 50 and 100 μg/L concentration this is because in our sampling site the concentration of As contamination is approximately 100 μg/L.

Arsenic is less toxic when the plant is well supplied with phosphorus and some isolates takes up As in a short period of time (Kabata-Pendias & Pendias 1984; Pais & d Jones 1997; Markley 2004). Further isolation of algal strains from the contaminated areas need to be done for fast and reliable As reduction. Detailed studies are required to find out the influence of phosphate in As removal by algal strains, understand the biochemical processes involved with the removal of As by alga.

35.4 CONCLUSIONS

In the present investigation isolated alga from the As contaminated area in Matlab, Bangladesh. The isolated culture was purified as single colony population. Based on the microscopic observation and morphology the isolate was identified as *Chlamydomonas* species. The alga is capable of growing in the presence of upto 1 mg/L As concentration and could able to remove approximately 80% of As from 100 μg/ml supplemented media. The As uptake was influenced by phosphorus and pH of the media. This could be useful information in Matlab area where As remediation is required immediately so as to use the organisms that efficiently decrease the As concentration contaminated in local areas. Further study is needed to identify organisms which could able to remove As in a short period of time.

ACKNOWLEDGEMENTS

The authors are grateful to the Swedish International Development Cooperation Agency (Sida-SAREC) for partial funding through the project "Arsenic in groundwater of Bangladesh: Options of safe drinking water" (dnr: 348-2003-4963). AB is thankful to BRAC and the NGO-Forum of Drinking Water and Sanitation for providing the logistic support to the field investigations. We are thankful to Dr. A.B. Mukherjee and an anonymous reviewer for their constructive criticisms on an earlier draft of this manuscript.

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Section VI
Groundwater management

CHAPTER 36

The autonomy of local drinking water institutions in rural Bangladesh

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ABSTRACT: Most of the current research in relation to the arsenic issue is directed to the technical aspects of the problem. Very few have seriously looked into the development of the organizational and institutional structures that are essential in the successful implementation of any solution. Because the underlying processes in setting up these structures tend to have large time constants there is an inevitable tension between the need to solve the immediate problem and the sustainability of their supporting organizational and institutional institutions on the longer term. Contrary to social changes, pure technical developments and installations can be done relatively quickly. Reasons for the lengthy character of the social process are found at various levels. Within the rural system there are direct social, institutional and economic limitations that make it difficult for the local communities to contribute in solving their own problems. But also in a broader context there are influences from outside the system, which further restrict the autonomy of local communities in their search for endogenous solutions. In face of the problems caused by the arsenic contamination the questions are about the consequences of these constraints and how they influence the setup of new local drinking water supply systems. In this paper we present an implementation scheme based on the different time-constants of its technical and social sub-processes. These processes are linked by using field implementation activities and scientific work as so-called 'triggers.' For example a water-testing program can be used to spark off the participatory dialogue in the villages. In the same way geological data acquisition, the construction of a social map, a risk analysis, or other research input can be used as triggers. At the other hand, a short-term solution for the immediate problem might create room for further research on long-term solutions and the use of endogenous knowledge and information from the local people is an essential 'trigger' for this. The approach presented here incorporates attention to internal as well as external factors, as both will have their impact on any future solution.

36.1 INTRODUCTION

Described as the worst mass poisoning in history, some 40 million people in Bangladesh are drinking from groundwater supplies that are naturally contaminated with arsenic (BGS/DPHE 2001, Bhattacharya et al. 2002, Nriagu et al. 2007). Over 400,000 incidences are likely to occur if nothing changes (Ahmed 2002) but the figure might be considerably worse when taking into account the arsenic intake from agricultural products irrigated with contaminated water (Huq & Naidu 2004). While the problem is generally referred to as one of the biggest environmental calamities of the last decades, little has been achieved to manage the issue in an integrated way. Most research initiatives today are looking into the geological, chemical, health or technical aspects of the problem. While these activities are obviously very important to clarify a number of uncertainties, few programs have seriously looked into the development of essential organizational and institutional structures needed for the successful implementation of any solution. Without going into the details, technologies for arsenic treatment include: pitcher filters, passive sedimentation, underground sedimentation, reverse osmosis, fill and draw, ion exchange, iron removal plant, oxygenated water injection, adsorption-co-precipitation using aluminum salts, or photo-oxidation by adding lime. Options to avoid drinking the arsenic contaminated water include: boiling surface water, buying clean water, pond sand filters, pond/river bank infiltration, rainwater harvesting, clean shallow tube well sharing, deep tube wells, or dug wells.

In our view, the major problem lies not in technology itself, as several options already exist. However, it is their application in the local context that makes the situation problematic. In a nutshell, the current health hazard is caused by contaminated water resources as well as inadequate means to access those resources. In other words, appropriate organizational structures are critical in order to solve the problem. However at the rural level these are far from being adequate or even lacking, and at the national level the central government is struggling with a weak (1) financial, and (2) institutional capacity.

The Bangladesh Planning Commission calculations ended with a huge resource deficit which could only be met, if at all, by massive foreign assistance. This implied some surrender, at least, of the autonomy as a sovereign nation. The country's economic structure also gets locked into a large import-dependence; this along with the debt burden would perpetuate the overall continued dependence on foreign assistance (Rahman 1992).

The statutory responsibility for the Water Supply and Sanitation sector lies with the Ministry of Local Government, Rural Development and Co-operatives. The functional responsibility is delegated to the Department of Public Health Engineering (DPHE), which is responsible for planning, designing and implementing water supply and sanitation services in rural and urban areas (except for Dhaka and Chittagong). In short it is responsible for supplying safe water to the community. Its organizational structure is limited down to the Upazila (sub-district) level so that it has no direct organizational instruments to reach to a local community.

Based on these initial observations we put forward an approach that starts building the needed institutions at the village level, where the problem emerges and where people have to rely on their own resources to tackle it. The analysis strongly suggests that where the highly centralized Bangladeshi administration and national policies fail, local NGO's could be the missing link in the process of implementation of solutions (Rammelt & Boes 2004). However, the complex and urgent character of the arsenic issue also requires an approach linking the necessary scientific knowledge to local knowledge, priorities and institutional building. It is believed that a joint effort between local development organizations to translate the mitigation agenda into grassroots activities, and the experience of research organisations on a scientific and technical level will be successful. This article is limited to an outline of this approach. Practical experiences will be published in future articles.

36.2 BASIC PRINCIPLES

From a scientific point of view there is still much to learn about processes of implementation of technology. Conventional one-sided technology transfer has long dominated the development discourse. Difficulties and delays in the successful practice of participative and bottom-up concepts imply that still a lot has to be learned. With the proper setup, research could be closely linked to a number of pilot projects and result in new insights in the dynamics of the implementation process and the construction of useful institutional frameworks.

36.2.1 *The dilemma*

In the more conventional views on development there is only one stock of knowledge, manifested in modern technology and located in industrialized countries (see 1 in Fig. 36.1). If transferred properly it may benefit other countries as well. This is not self-evident. According to prof. Reddy of the University of Bangalore, technology is a cultural artifact that carries the genetic code of the culture in which it has been developed, resulting in a mechanism where technology sustains and

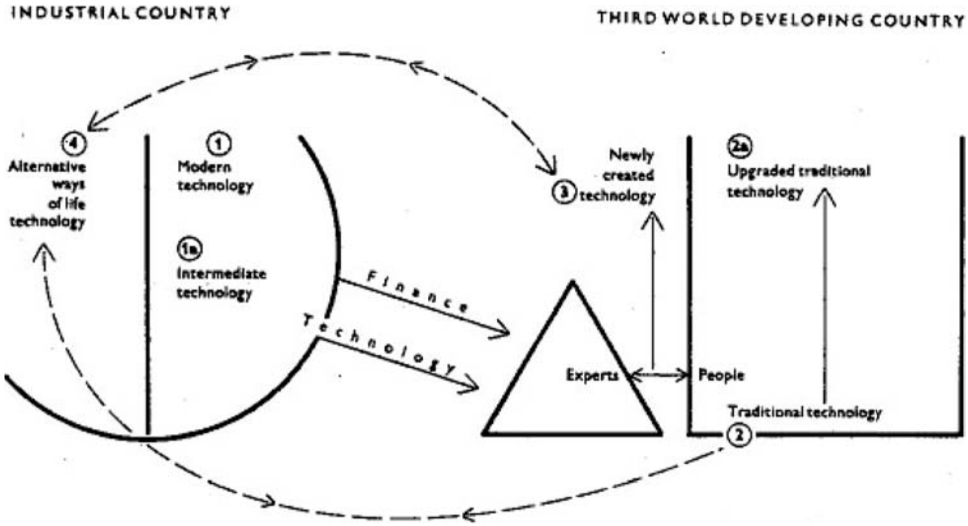

Figure 36.1. Different technological knowledge (Wignaraja 1984).

promotes the interests of the dominant social group of its original society (Dickson 1974). As a result, local endogenous knowledge (having an internal cause or origin) or traditional technology (see 2 in Fig. 36.1) has generally been undervalued. The issue is what is 'appropriate' and who decides what is 'appropriate?' The objective should be to gradually enable local communities to make their own knowledgeable choices regarding the involvement of exogenous technologies (related to, or developed from external sources).

Starting at the local level implies that one has to counter a huge lack of awareness as one of the main obstacles. One view is that people need to raise their level of consciousness and form their own organizations which are central in the development process. Once this participatory self-reliance base has been built, external technical and financial inputs of one kind or another can better be absorbed and will not skew the benefits (see 3 in Fig. 36.1). On the other hand, bringing about the involvement of the community will surely require more than just an awareness programme. Without prospects for solutions to an acute problem on a rather short notice there will still remain much social discontent (Einwachter et al. 2001). However sustainable solutions will take a considerable time to be developed and implemented. Firstly the scientific/technical knowledge needed for this is not ready to use and must be further developed and secondly the implementation of these solutions will be most severely delayed by the lack of proper institutional support.

One of the main dilemmas of the current situation is therefore a matter of priorities. Should the emphasis be laid upon the understanding of the social and technical mechanisms of the problem so that it might be easier to come up with the best solution possible or should priority be given to mitigate the arsenic contamination in order to save lives as much as possible?

Obviously both should be done, but the issue remains on how to streamline these short-term and long-term priorities. The main problem stems from the fact that contrary to social changes technical developments and installations can be done relatively quick. This friction is very fundamental. Because the underlying processes in setting up these structures tend to have large time constants there is an inevitable tension between the need to solve the immediate problem and the sustainability of their supporting institutions on the longer term. In practice it leads often to disregard the endogenous organizational and institutional setup.

For a more sustainable implementation of a technology one has to realise that certain organizational structures are needed. The main question is therefore on how to integrate the different technical and social processes. One of the basic ideas of this approach are the so-called 'triggers'.

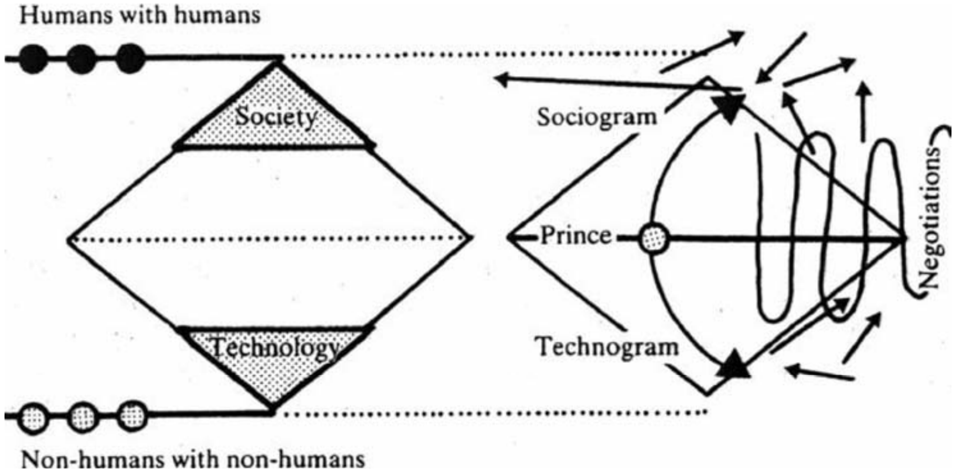

Figure 36.2. The Prince as a mediator and negotiator between the technical and societal sphere (Latour 1988).

This tool could be used to match the apparent mismatch between the short-term/technical and the long-term/social processes.

36.2.2 *Triggering socio-technical processess*

The main technical problems for designing feasible solutions are the uncertainties about the scientific aspects of the contamination itself as well as the cost and complexity of available remedial technologies. The main social obstacles are the lack of adequate local awareness about arsenic and the lack of well-functioning institutional arrangements for drinking water provision, inadequate involvement of local communities in managing the problem and the insufficient consideration of the gender-based roles and responsibilities in water management in general and their potentials with regard to mitigate the effects of arsenic in particular.

An understanding of technology and society as two different poles of the human progression fails (Fig. 36.2). In our approach we link up to those socio-technical approaches that increase the mixture between both spheres at each turn of the 'negotiations' (Latour 1988, see right part of Fig. 36.2).

The hard part in the 'negotiations', directed by a local NGO that fulfils the role of the "Prince" as the mediator, lies in the earlier mentioned different time constants of the technical and the social process. These should be carefully tuned as suggested in the figure above. This can be done in a learning-by-doing process where the different parties are involved in a dialogue. One of the basic ideas of this approach are the so-called 'triggers'. This tool could be used to match the apparent mismatch between the short-term/technical and the long-term/social processes.

36.3 PROPOSED METHOD OF IMPLEMENTATION

The overall approach of the implementation methodology is conceptually decentralised. The justification for this choice has been given in the introduction. An approach starting at the local community level but also seeking a relation with the private sector and the governmental institutions at the regional level could be successful. Matching the local implementation process with the activities of the government at the regional and national levels could lead to a number of synergy effects.

36.3.1 *The local implementation process*

The available technical options that can be used as a base for a safe water supply are easily identified. However, the appropriateness of the short-term solutions depends heavily on the local situation

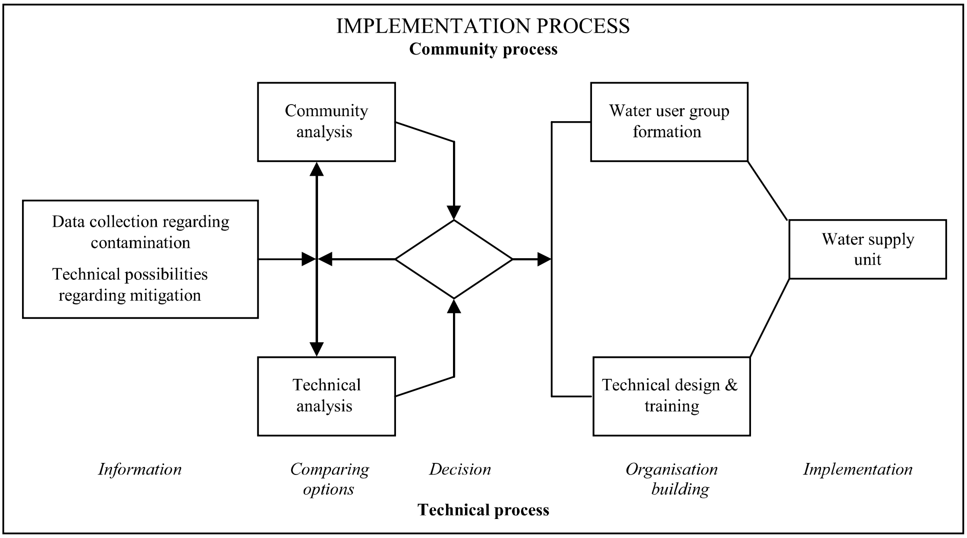

Figure 36.3. Local implementation process.

(Ahmed 2002). On the longer term, solutions based on wells seem to fit the socio-economic structure of Bangladesh. However, a number of problems have to be solved before those can be implemented safely again. Other technical options (dug-wells, pond sand filters, river filtration, etc.) might also play a key role in the short-term mitigation (see for example Jakariya et al. 2005, Jakariya 2007).

The development of institutional and organizational arrangements for local water supplies that meet the needs of the most vulnerable groups in the community will be a very time consuming process. This process of institutional development should therefore start as soon as possible, so the time lost in future implementations of safe water options can be minimized. In other words, it will be easier to shift to better technologies by the time they are available provided the institutional process has been built-up. There will be a better support for local decision-making. Figure 36.3 illustrates the local field management methodology, which essentially represents the community and institution building process as it evolves in time during the planning period of the implementation of a new technology.

Short-term mitigation of the arsenic problem will be done after options are compared from both a technical and a community perspective. In the first phase of institution building user groups or village committees and a technical design will be developed, eventually leading up to the installation of a water supply unit. The earlier described triggers and dialogue will be imperative instruments in this process. This method will provide the opportunity to observe important factors in the building process of community-based institutions. Integrated issues like gender, formal and informal institutional restrictions, physical land conditions, food pattern, health conditions, etc. can be investigated during all phases of the process.

In the beginning especially the implementation process should be of an exploring nature following roughly the outline as visualised in Figure 36.3. It is important to note that this is an ongoing process that will gradually be described with a greater degree of certainty. Because of the inherent uncertainties regarding the problem itself, the intervention strategy should be flexible and adaptive to new findings. Methods to continuously support and monitor this process should be applied and adapted when new insights are generated. The characteristics of a monitoring and evaluation system are elaborated further below.

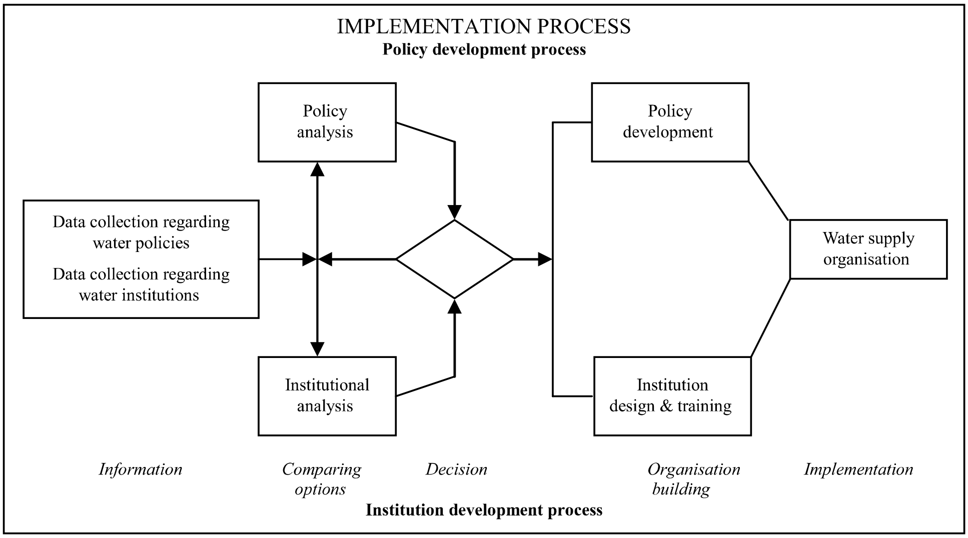

Figure 36.4. Regional management methodology.

36.3.2 *Links with regional management*

To extend the implementation of sustainable water supplies beyond mere experimental decentralised pilot projects the support from the regional level, mostly represented in district towns, might very well become essential in the future. Moreover, to safeguard food security at the town level and their surrounding rural food supply areas, a regional monitoring system with regard to the water quality is also essential. In line with the methodology used at the local level a comparable method could be applied at the district town level to introduce an implementation process of regional institution building for a safe water supply. Figure 36.4 shows the main components.

As mentioned, the DPHE organizational structure is limited down to the Upazila level and the current official water policies do not reach beyond the district town level. The proposed methodologies might fill the gap and might add a valuable instrument in the development of integrated local and regional institutions with appropriate policies and organizations.

36.3.3 *Links with national co-ordination*

The results of such a programme at local and regional (town) level should be communicated to the national level by bringing together various stakeholders. As a general outcome it might be expected that the experience and information gathered from this programme, especially with regard to the set-up of the participatory approach would also be applicable to other complex environmental and resource management issues.

36.4 CONSTRAINTS

For the sake of argumentation, a very broad distinction can be made between two types of technical knowledge. From a 'bottom-up' perspective, one stock of knowledge is primarily endogenous and the other is primarily exogenous. These different technological stocks of knowledge also have quite different social, institutional, cultural and economic characteristics. In practice one will generally find a 'mix' of both.

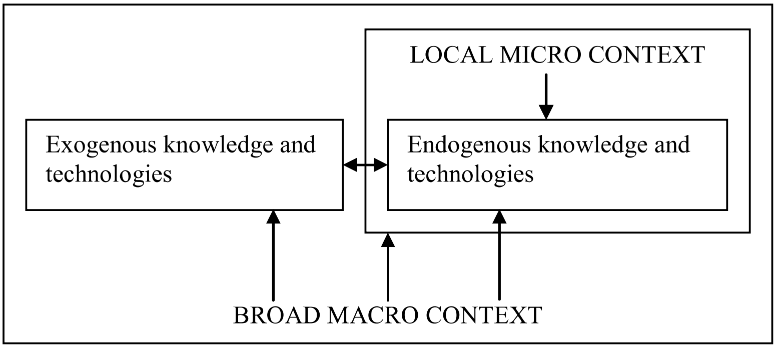

Figure 36.5. Exogenous and endogenous knowledge and context.

The form and success of this 'mix' may vary firstly according to the inherent dynamics of the sub-system on the micro level where the technologies are being implemented and secondly as a consequence of direct external influences that sometimes facilitate and sometimes frustrate the internal development process. Thirdly, in a broader context there are also indirect influences in the macro context that are relevant. The point is that these direct and indirect forces might favor one form of technological knowledge over the other and therefore have varying impacts on different social groups. Figure 36.5 gives a schematic overview of the situation.

More precisely, within the rural context there are direct infrastructural, institutional and economic limitations of influence on local problem-solving capacity. External influences and trends might further restrict the local communities to take matters in their own hands. We will now give a quick impression of that context.

For example a lack of proper health care and a lack of formal support for the elderly often result in high fertility rates as a mechanism to minimise risks. As a result, the pressure increased on a decreasing quantity of arable land, which favoured an intensification of the agriculture during the green revolution of the 1980s. At a first glance this provided a good opportunity for securing food supplies and for increasing income after selling the products at local markets. The income is however for a large part directly spent on the needed agricultural inputs such as seeds, pesticides, fertilisers and irrigation water.

With these dependencies in place a new trend set in. In order to maintain a level of production, more and more chemical fertilisers were needed as certain natural nutrients in the soil were depleted. To worsen the situation, subsidies fell and the prices for inputs increased considerably. So while the agricultural sector has grown and other varieties of vegetables diversify the menu, the reliance of the farmers on local markets for both selling and buying has also increased.

These developments have not benefited the position of the landless poor. The need for manual labour decreased and a growing number of labourers now have to find work outside the village, in the cities, in the jute or the garment industry for example. These sectors are largely influenced by national and international fluctuations. The vulnerability of these people has clearly been increased.

Central in any programme aiming at sustainable development are the priorities of the local communities within their situation. As a consequence it might very well be possible that the long-term objectives in the approach that we are suggesting are over-shadowed by more pressing needs besides arsenic-free drinking water. The success of any programme will therefore depend not only on its capacity to channel the short- and long-term priorities but also on its capacity to understand and include broader development issues.

Many believe that this understanding will be generated through so-called participatory approaches, as popularised particularly by Chambers (1997), Pretty et al. (1995), and others. Participation has now become a 'buzz-word' and many tools have been developed to put it into practice. Point remains that with the above constraints and informal dependencies the needed interaction might (understandably) not always be entirely transparent. As a consequence, the way local people will ultimately participate might often seem incomprehensible when perceived for the outside. We therefore conclude that the form of participation has to be shaped during the course of the programme as a part of the learning process in the form of a participatory monitoring and evaluation system.

36.5 MONITORING AND EVALUATION

Local communities, especially the poor and women, must be involved at all stages in the program, including monitoring and evaluation of the implementation process. There are large differences between a monitoring and evaluation system based on external criteria and one that finds its quality in the internal dynamics of the development process itself. An important tool for shaping participation on the local level will be a joint Participatory Monitoring and Evaluation (PME) System. Based on people's priorities, measurable criteria can be chosen and activities be monitored and evaluated. Learning about the development process and its objectives will lead to a dynamic set of criteria that ideally supports a process that is well understood by its participants, because it reflects primarily their own rationality (Neggers 1998, AMRF 2005). This is illustrated in the following metaphor.

> "*I remembered one morning when I discovered a cocoon in the bark of a tree, just as the butterfly was making a hole in its case and preparing to come out. I waited a while, but it was too long appearing and I was impatient. I bent over it and breathed on it to warm it. I warmed it as quickly as I could and the miracle began to happen before my eyes, faster than life. The case opened, the butterfly started slowly crawling out and I shall never forget my horror when I saw how its wings were folded back and crumpled; the wretched butterfly tried with its whole trembling body to unfold them. Bending over it, I tried to help it with my breath. In vain. It needed to be hatched out patiently and the unfolding of the wings should be a gradual process in the sun. Now it was too late. My breath had forced the butterfly to appear all crumpled, before its time. It struggled desperately and, a few seconds later, died in the palm of my hand.*"
>
> — Nikos Kazantzakis—Zorba the Creek

This simple story tells in a nutshell why it is important that a PME-system should a product of a learning process of the organisation itself. Only in that way it can become a 'professional' tool for the organisation. A more classical way of accounting should also be executed by local NGO partners in terms of the number of checked tube-wells, the status of the 'red/green' (unsafe/safe) codification of the tube-wells, the extent of the arsenic poisoning, status of the patients, etc. These results can be confronted, if necessary, with the PME system.

36.6 CONCLUDING REMARKS

The approach presented in this paper has the objective to streamline the technical and social processes and their inherently different time-constants of its technical and social sub-processes. It incorporates attention to internal as well as external factors, as both will have their impact on any solution in the future. Work on this approach is ongoing as new findings from research activities and project work will come up in the coming years.

ACKNOWLEDGEMENTS

The ideas presented in this paper reflect our collaboration with Dr. Zahed Md. Masud and Mr. Palash Torfder, respectively of AITAM and PRIDE. We would like to thank them for their contributions. The authors also thank P. Bhattacharya and AL. Ramanathan for their editorial comments.

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CHAPTER 37

Management of the Salalah plain aquifer, Oman

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ABSTRACT: Salalah is situated on a fresh water aquifer that is replenished during the annual monsoon season with an average precipitation of 245 mm/year in the highlands of Jabal AlQara to less than 100 mm/year in the Salalah plain, most of which falls during the 3 month long wet season. Salalah plain aquifer is the only source of water in the city for a population of more than the 134,000 inhabitants living in the city. Precipitation in the Jabal AlQara supplies the plain with significant renewable fresh groundwater resources that have allowed agricultural and industrial development to occur. In Salalah city where groundwater is used extensively since the early 1980s for agricultural, industrial and municipal purposes, groundwater is withdrawn from the aquifer more rapidly than it can be replenished by natural recharge. The heavy withdrawal of large quantities of the groundwater from the aquifer leads to the encroachment of seawater. Currently (2005) agricultural activities utilize over 70% of the groundwater. For the last 10–15 years the aquifer is facing acute challenge due to the over-extraction that has taken place since the 1980s and led to salinity intrusion problems. Safe yield has been calculated by the flow model at 51 million cubic meters (MCM)/year, of which 98% of this water is underflow originates from Jabal AlQara, and the remaining 2% is the recharge on plain. The water budget presented in this paper shows that the renewable groundwater resources meet only the 78% of the present demand. The water budget for coming years is predicted and shows that the deficit will increase. The prediction shows that the shallow on-farm wells will virtually lead to the abandonment of farming activities if no solutions are implemented since excess salinity in many wells will occur. The paper suggests a number of recommendations to be taken into account in order to protect the aquifer from further deterioration. A very urgent suggestion of this paper is to take an immediate action to relocate Garziz farm immediately and the fodder production farm of MAF. The two farms located on the freshwater zone and pump currently over 23% of the total discharge for irrigation on the plain. Another measure is to conduct the reuse of the treated wastewater in Salalah plain in order to halt seawater intrusion. These measures would improve the water supply situation for Salalah.

37.1 INTRODUCTION

Salalah, the second largest city in the southern coast of the Sultanate of Oman is located 1,100 km south of the capital Muscat, with a population of 134,000 and a growth rate of about 2.2% a year (Oman Census 2003). Due to an increase of national wealth over the last 37 years coupled with dramatic improvements of public health and quality of life has resulted in significant increase in population. The Salalah plain is underlain by a shallow limestone aquifer, which extends north up to the foot of Jabal Al-Qara, and south to the Hafah coast (Fig. 37.1). Quaternary deposits include Wadi alluvium, a calcarenite layer, underlain by limestones (Table 37.1), constitute the Salalah aquifer. The coastal plain aquifer is historically the only natural source of water supply in the Salalah plain. The residents of Salalah have traditionally used this aquifer as the only source of water for agricultural, drinking water and other domestic needs. In the past, the population of Salalah was fairly small, and demands on the aquifer were in balance with its recharge capacity.

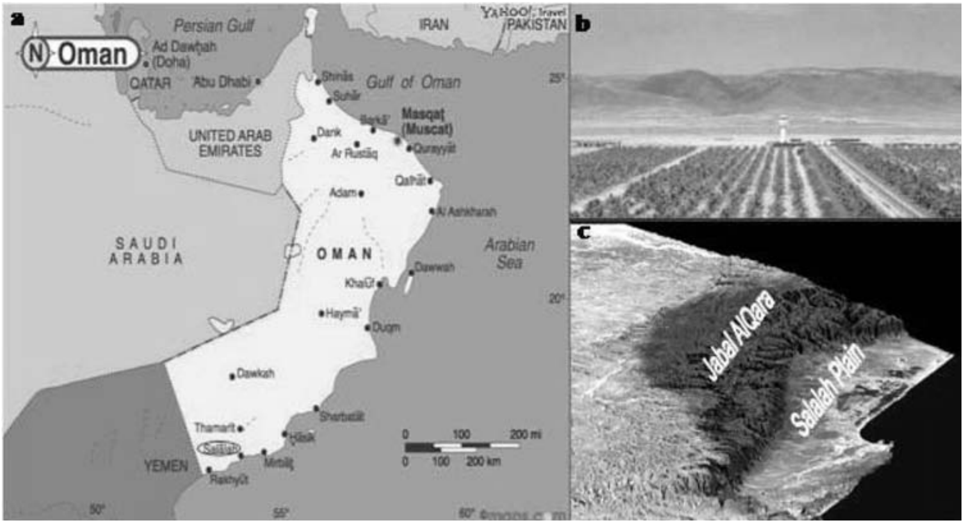

Figure 37.1. a) Map of Sultanate of Oman with the location of Salalah city; b) panoramic view of Salalah plain and Jabal Al-Qara mountain; and c) and satellite image of Salalah plain in front of the Jabal Al-Qara mountain.

Salalah municipality uses 9022 m^3/day of groundwater, equivalent to 3.3 million m^3 (MCM) annually for the irrigation of the public gardens, landscaping and green areas covering 31 hectares (Renardet 2004), which accounts for more than 5% of the current total amount of groundwater that is discharged from the aquifer. Currently the average pumping rate is more than the recharge rate. Seawater intrusion has been reported in the Salalah plain aquifer by Shammas (1998, 2002), and Geo-Resources Consultancy (2004). The recharge into the aquifer takes place through karstic limestone in AlQara Mountains. The average precipitation is 252 mm/year in Jabal Al-Qara Mountain and 114 mm/year in the Salalah plain (Al-Mashaikhi 1997). Annual rainfall averages 260 mm in the Jabal and 110 mm in the plain (FAO 1992).

Abstraction of water for domestic supply was historically from dug wells in the coastal agricultural belt. Development of larger supplies began in the early 1970s with the construction of the Umm Al Ghawarif Wellfield. In 1982, this was replaced by new main Wellfield of Salalah consisting of 15 bores located across the freshwater zone, some 9 km from the coast that was then supplemented in 1986 from the smaller Saada Wellfield of three wells to the north-east of the plain. The demand of potable water in Salalah in 2004 is 14.8 MCM and expected to double to reach 28.36 MCM/yr within the next 20 years. Currently Salalah is totally dependent on groundwater supplies and is mining the groundwater due to increasing agricultural and domestic demand, thus incurring the risk of increasing salinization from the inflow of adjacent brackish waters and seawater intrusion (Shammas 1998, 2002).

A previous study by D&MI (1992) estimated the annual recharge rate at 32.4 MCM/yr using finite digital model and showed that the groundwater discharge from Salalah was more than 40 MCM in the year 1990. The model was calibrated utilizing eight years of available groundwater level and salinity data.

With the overdraft of the aquifer the water flow towards sea was reversed, groundwater levels declined and saline intrusion took place. A major reason for this was irrigation in agriculture taking more than 70% annually. As a result of the annual over-abstraction of the groundwater from the aquifer more rapidly than it can be replenished for agricultural purposes, seawater intrusion occurs. The aquifer deterioration has continued since no solutions are implemented to halt saline intrusion. In 2005, for instance the agricultural sector alone discharges 46 out of 65 MCM whereas the average natural recharge is estimated by (Shammas & Jacks 2007) at 51 MCM. The groundwater quality is deteriorating in the aquifer since 10 to 15 years back (D&MI 1991),

Table 37.1. Stratigraphy of study area.

Age	Group	Formation	Max. thickness (m)	Lithology, comments
Quaternary		Beach deposits	~25	Coastal dunes
		Alluvials and alluvial terraces	5	Limestone conglomerate, partially indurated
		Razat formation	200	Travertine and conglomerate
~~ Unconformity ~~~~~~				
Tertiary	Fars group	Nar formation	60	Reddish conglomerate with sandstone and clay palaeosoils; aquitard
		Adawnib formation	140	Conglomeritic limestone and biocalcarenite; karstic aquifer, fresh to saline
	Dhofar group	Mughsayl formation	~800	Carbonate turbidite with breccia and thin bedded limestone; aquitard, saline
	Hadhramaut group	Damman formation	250	Massive and thin bedded nodular limestone with marl and yellow to orange shale with marl and limestone
		Rus formation	90	Breccia, chalky dolomite, marl and laminated gypsum; aquitard, source of NaCl
		Umm Er Radhumma	700	Massive to nodular bioclastic whitish limestone with shale intercalations and cherts; karstic, fresh to brackish aquifer
~~ Unconformity ~~~~~~				
Late Cretaceous	Wasia group	Dhalqut formation	130	Massive micritic limestone with green marl; aquitard, saline
		Kharfot formation	60	Alternating green marl and micritic limestone, aquitard, saline
	Thamama group	Qishn formation	160	Sandstone grading into well bedded limestone with marl or dolomite; potential aquifer, good quality water at Tawi Attair

and the situation of the aquifer will increase in a negative way in the coming future, so the need to protect this important resource against pollution is an issue of great concern.

Groundwater salinity has increased in some areas on the Salalah plain to levels above the standards of drinking water and the water in irrigation (Shammas 2002). The main causes of these problems are as follows:

- Decline in quantity of fresh water flowing towards the coast line due to over-utilization and continue extraction from wells on the coast, particularly in Dhofar Fodder Company, which used to pump for irrigating the fodder farms in the previous years a percentage that exceeds

20% (average 12 MCM/yr) of the total annual discharge from the aquifer. The current discharge amount is approximately 20% in percentage of the total annual extraction rate, for the irrigation of the farms that belong to a company in Salalah city.
- Excessive pumping from wells existing on the coast line to irrigate the traditional farms in Salalah plain, which in 2002 consumed over 43% (31.2 MCM) of the total annual extraction from the aquifer. This rate exceeds the average rate recorded before 1998 and the same trend applies to Dhofar fodder farms. Domestic consumption during the 1980's and 1990's was far less than the consumption in the year 1998 and 2004, which utilized 12.68 and 14.8 MCM respectively, while the domestic consumption in 1990 was 8.7 MCM (DWR, personal communication). The demand for potable water in Salalah is expected to double from 14.8 to 28.36 MCM per annum within the next 20 years. This increase in demand for domestic water is due to the steady population growth, while the quantity of water extracted for agricultural purposes in Salalah plain, whether for irrigation of Dhofar fodder farms or traditional farms has remained unchanged, particularly in the case of Dhofar fodder (12 MCM) and the water use for irrigation of traditional farms (31.2 MCM). Thus, the immediate and persistent cause of the problem is the over-consumption of water for agricultural purposes. Most of the traditional farms in Salalah plain use flood irrigation, which easily results in poor water use efficiency.
- The annual recharge from rainfall to substitute for the large quantities drawn from the aquifer. The current average natural and artificial recharge is estimated by the author to be 51.68 MCM while the annual drawing rates are on the increase.

When the freshwater is abstracted from the ground the heavier brackish water situated lower down rises up and destroys the freshwater lenses. With this, in the Salalah plain coastal unconfined aquifer, the brackish/saline groundwater underneath will flow upward and intrude into hand dug wells/boreholes causing the development increase in the salinity of pumped groundwater as the current pumping from the coastal belt of the aquifer is more than 50% of the total annual abstractions, where such process under the current situation will result in seawater intrusion. The abstraction from these wells is to be controlled before the entire groundwater becomes saline. The present study deals with groundwater resources assessment of the Salalah coastal plain in Oman.

37.2 GEOLOGY AND HYDROGEOLOGY

Salalah (17.02°N; 54.09°E), is located in the southern region of the Sultanate of Oman. Salalah plain is a flat-lying coastal plain, which is surrounded by the AlQara Mountain and low hills to the north, west and east and Arabian Sea in the south. Salalah plain is approximately 55 km long and 15 km wide (Fig. 37.2). Jabal (mountain) AlQara has two principal scarp directions. West of Sahalnawt valley the crest trends east-northeast whereas to the east of the valley is east-southeast. The south facing slopes are characterized by deep valleys, which cut across the steep faces. These cliff sections contain limestone and travertine faces outcrops close to the springs in the area (Roger & Platel 1987). The land surface of AlQara Mountains, extended 65 km from east to west, is rather flat in some areas, with occasional valleys small hills. The highest area is Qairoon Hairitti, overlooking Salalah plain, located at an elevation of 872 m above mean sea level (a.m.s.l). The hills are covered with grass and dense tree and shrub.

The mean annual temperature near Jabal AlQara is about 21°C and highest mean annual temperature recorded in Salalah city is 26°C. The climate of Salalah plain and the adjacent Jabal AlQara is quite different from the typically arid climate experienced by the rest of Oman and the Arabian Peninsula. This is due to the combined effect of summer monsoon June to September and local topography. The monsoon provides annual precipitation, although its duration and intensity varies yearly. According to COWI Consult (1992), the moist air condenses over the coast and moves inland to the mountains causing foggy humid conditions and low intensity rain. A significant part of the fog is intercepted by the native grass (Savannah type vegetation) and trees. The average annual rainfall in AlQara mountain ranges between 230–450 mm, whereas the average annual rainfall in

Figure 37.2. Satellite image of the study area around Salalah plain and Jabal AlQara front (GIS 2004).

the plain is around 100 mm (Chebaane & Alesh 1995). This provides the basis for Salalah fresh groundwater supply.

A broad gently dipping Tertiary limestone formation dominates the geology of the Dhofar region. The major aquifer horizons occur in the karstic limestone of the Hadramhaut Group (Table 37.1) and the overlying alluvium in the Salalah Graben (Flint & Rippon 1986). The presence of coarse alluvium, particularly pebbles, cobbles and boulders, gives the soils of Salalah plain highly permeable and well-drained texture. The top soil zone of this region is about 1 m thick and composed of sandy loams and loamy sand (Taylor & Sons 1985). The geological structure indicates that the freshwater on which the city depends is mainly a product of recharge from the adjacent mountain areas (D&MI 1992). Thus the natural groundwater recharge to Salalah plain is primarily derived from precipitation falling onto the AlQara Mountain during the monsoon, which infiltrates and moves towards the plain Dames & Moore International (D&MI 1992).

37.3 METHODS

The current estimated groundwater resource by (D&MI, 1992) shows that the available groundwater is able to meet only 53% of the current demand. However, estimates based on 3-dimensional finite difference model by the authors (Shammas & Jacks 2007) indicate that the available groundwater resources should be able to support 78% of the current demand.

The aquifer freshwater stock or the total groundwater reserves under Salalah plain are estimated at 340 MCM D&MI (1991). The question is, how much freshwater was/is available? And how much is being used? The method that being used to answer these questions was by investigating the relationship between the withdrawal from the reserve and the period of expected exhaustion of the aquifer is considered. The equation is:

$$Q = GW + total\ R - total\ D \tag{37.1}$$

Where Q is is the withdrawal from reserve, GW is the aquifer storage, total R is the total annual natural and artificial recharge and total D is the total annual discharge. The storage and recharge rates are assumed to be abstracted totally before it is flushed to the sea. Based on the above, another question is important. What is the cost of new agricultural water source should this resource become saline? The equation is:

$$OC = TS \times AD \tag{37.2}$$

Where OC is the opportunity cost, TS is the cost of the treated wastewater needed to compensate the agricultural demand and AD is the agricultural annual water demand.

37.4 RESULTS AND DISCUSSIONS

Replenishment of the fresh groundwater resources of the Salalah plain is derived from four alternate mechanisms:

- Subsurface flow from the Jabal AlQara.
- Infiltration of rain falling directly on the Salalah plain.
- Recharge dam contribution as a result of cyclonic/storm activities on Dhofar Governorate.
- Infiltration of surface water derived from springs at the foot of the Jabal (Garziz and Sahalnawt springs that used for fodder irrigation).
- Irrigation returns flow.
- Artificial recharge of the reclaimed wastewater by the injection wells directly into the southern part of the aquifer.

The underflow from the Jabal to the plain aquifer was calculated at 51 MCM (Shammas & Jacks 2007). The current (2005) total natural and artificial recharge is calculated 56 MCM (Table 37.2). The underflow was derived from the established numerical groundwater flow modelling and calibration on hydraulic heads of 1992.

Before the beginning of the 1980's, the aquifer was in equilibrium condition with about 1.5 MCM of surplus water that was flowing into the sea every year. After the renaissance in Oman since 1970, accelerated developments in the fields of agriculture, population, construction and industry in Salalah have seriously exhausted the aquifer and threaten to disrupt the water supply in the city. The fresh water budget for Salalah aquifer is as follows: the total annual inflow is 32.4 MCM and the outflow is 40 MCM. The average yearly use for irrigation is 31.7 MCM (D&MI 1991). The data presented in Table 37.3 indicate that over 80% of the groundwater is being used for irrigation and 20% for public and domestic supplies.

The calculations showed roughly that the renewable fresh groundwater resources aquifer meet only the 53% of the present demand based on the previous data of D&MI (1992). However, the underflow calculation by using the flow model developed by Shammas & Jacks (2007) show that the natural groundwater inflows from the Jabal into the aquifer currently meet up to 78% of the present (2005) demand.

The accelerating developments in the city caused over-draft the aquifer, and caused disturbance to the water umbrella in Salalah. It is clear that the aquifer cannot stand the predicted growth rates unless the water management is radically changed. In 2000, for instance, the total discharge from

Table 37.2. The current (2005) calculated aquifer recharge.

Source of recharge	MCM/yr	Percent
Direct recharge from rainfall	1	2
Subsurface flow from the Jabal	50	89
Artificial recharge	5	9
Total	56	100

the aquifer was 62 MCM, and it will increase due to the development in Salalah, which will lead to the increase in demand for water, and also due the weak annual recharge from rainfall to substitute the large quantities drawn from the aquifer annually.

The annual recharge is 51 MCM, of which over 98% is the subsurface inflow from the AlQara Mountain into the plain sub-basin zone and the balance is the recharge on the plain. The deficit in 2000 was 11 MCM. The prediction of this paper show that the discharge from the aquifer is 65 MCM, and the deficit is predicted to decrease to 9 MCM in the year 2005. The reduction in the deficit is as result of the artificial recharge by the reclaimed water and the additional recharge from the cyclone incident that occurred in May 2002. The situation of the aquifer will negatively increase through the coming years. The prediction of the total discharges is 68, 70.5, 70.46 and 74 MCM for the years 2010, 2015, and 2020, respectively. The deficit is predicted during these years to reach 9.4, 11.2, and 13 MCM respectively (Fig. 37.3). This study shows that the deficit in the aquifer will increase annually, even though the artificial recharge by the injection wells is considered on operation and to continue on increase by 3% annually. This paper reveals water quality deterioration, apparently due to over-pumping, particularly for agricultural practices as the salinity in all zones is increasing gradually.

37.4.1 *Water efficiency and sustainable water management*

37.4.1.1 *The effect of continued existence of Garziz farm*

DCFC (Dhofar Cattle Feed Company) has two farms on Salalah plain, Garziz and Sahalnawt farms. Garziz farm (Fig. 37.2) is situated on the freshwater zone, whereas part of Sahalnawt farm is located

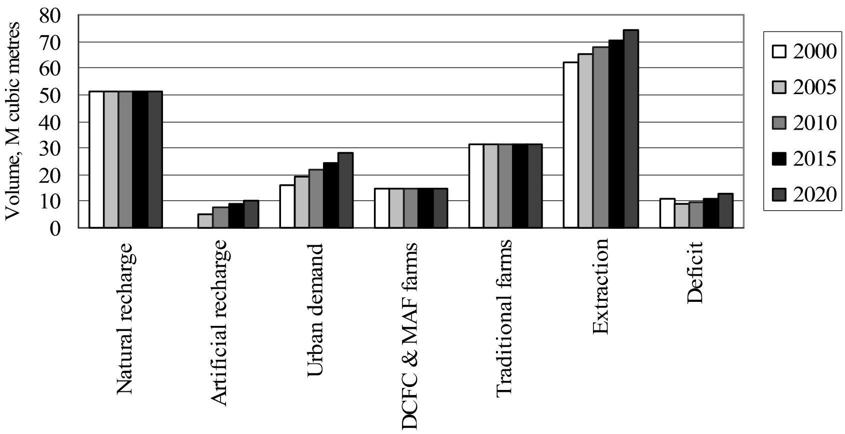

Figure 37.3. The water budget of Salalah aquifer and future projections of its condition under baseline scenario assumes constant underflow recharge.

Table 37.3. Comparison of the estimation of water balance for Salalah aquifer (MCM/yr) based on the data of ENTEC, D&MI and the present study.

	Component	ENTEC	D&MI	This study	Average
Inflows	Jabal front inflow	43.2	30.7	44.2	39.38
	Rain on plain	4.3	0.6	1.1	1.99
Outflows	Agric. demand	44.9	31.7	46.0	40.87
	Potable demand	4.3	7.1	15.7	9.03
Net outflow to sea		−1.7	1.5	−0.01	−0.20
Change in storage		0.0	−7.7	−10.1	−5.91

on the eastern boundary and another part is outside the main groundwater divide of the Salalah plain aquifer. Garziz farm consumes 7.55 MCM annually part of the water (about 1.1 MCM/yr) derived from Garziz spring (Ain). Garziz and MAF farms use Ain Garziz. Both farms use Ain Garziz water and mix this high quality water with water abstracted from their private bores. Sahalnawt farm consumes over 6 MCM annually, part of the water (about 2 MCM/yr) derived from Ain Sahalnawt. Both freshwater springs are located at the Jabal front and distribute the water to the farms through cemented canal. D&MI (1992) study recommended the relocation of Garziz farm away from the Salalah plain aquifer in order to balance the transient aquifer. Subsequently, the Royal Decree No. (23/94) was issued in the year 1994 to relocate the farm. Unfortunately, no action is implemented until date.

By computing the total quantities of water that was extracted from the aquifer since 1994 for the irrigation of (DCFC) and till the year 2006. For instance, the amount of groundwater Garziz farm that is utilizing about 8 MCM/year, so, we multiply 13 years by 8 MCM/yr, we get 104 MCM was extracted from the aquifer for Garziz farm fodder irrigation for the last 13 years, and that since the Royal Decree was issued. In Oman the amount of 1 m^3 of water equals to 0.44 Omani Rials which equal to 1.1 US Dollars (USD) for drinking consumption in municipal purposes and 0.66 Omani Rials (OMR), equivalent to 1.67 USD for irrigation and trading purposes, so when we multiply this number by 104 MCM, we got 68.64 million OMR, equivalent to 174 million USD, which is the water cost that was consumed by (DCFC), with a knowledge that the company did not pay to the government the cost of the used water. The company usually does not pay for utilizing the water in its irrigation purposes, and as the water is a national resource and very important, particularly in a country like Oman which is classified as an arid region that is struggling with a very shortage of water resources in a very bothered situation, as Salalah aquifer is the only source of water for all purposes in Salalah city, thus, we should give it full importance, and the previous mentioned Royal Decree is a very good example on its importance. If the Royal Decree was implemented in 1994, and the government compensate the company for relocating the farm during that time, say by 7.6 million USD as per D&MI (1992) suggestion, the government would save 166.4 million USD, but unfortunately, the decree has not been implemented. By taking no action to shifting Garziz farm, the aquifer looses large water quantities annually, which means loss of money and resource, as the water could be used for domestic purposes. The previous field data of Department of Water Resources (DWR) in Salalah city shows that the water quality in Garziz farm was within the Oman drinking water standards. For instance, nitrate-N concentrations in two bore wells (AD983578AA and AD985718AA) were 4.1 and 6.9 mg/L in 1994 (unpublished data, DWR) and increased to 11.0 and 13.3 mg/L, respectively (Shammas 1998), the EC values were 1084 and 1433 μS/cm in 1994 and increased to 1600 and 1800 μS/cm in 1998 in the two wells, respectively. Omani drinking water standards are 1500 μS/cm for EC and 10 mg/L for nitrate-N. The company is still pumping the groundwater for free. This means the government has lost 174 million USD in the last 13 years. The government is losing over 13 million USD every year by not taking actions to relocate Garziz farm. Therefore, this study strongly recommended implementing the Royal Decree No. (23/1994), so, to immediate relocate of the Dhofar foddering farms, particularly Garziz farm.

D&MI (1992) estimated the economic return of using 1 m^3 (CM) of water in the irrigation of the grass (Rhodes grass) to be 0.05 OMR, which equals currently to 0.13 USD, while estimated more than that for all kind of vegetables if the same amount of water is to be used in cultivating vegetables. The government sells 1 m^3 of water to Salalah inhabitants for domestic purposes at 1.1 USD and 1.67 USD if the same amount of water is to be used in trading purposes. The vegetables products are more water efficiency than animal products in general (Jacks 2003).

37.4.1.2 *Water management efficiency in grass production farms in the brackish zone*

How much would the government pay to the grass production farmers those cultivate in the brackish zone for closing the bores for 4 months? Agricultural returns to water in Oman are generally very low and net benefits contribute marginally to the national economy (<3% GDP) (Al sulaimani 2003). The 6 MCM/yr is assumed as the 4 months period abstraction by the grass production bores that cultivate in the brackish zone, which is to be reduced annually. D&MI (1992) estimated the economic returns of using 1 m^3 of water in the irrigation of the grass production (Rhodes grass) to be 0.05 OMR/m^3. The water quantities that assumed to be abstracted from the brackish zone to produce grass in 4 months is 6 MCM/yr multiplied by 0.05 OMR, the economic returns of each 1 m^3 of water that the farmer would gain. That means the farmers that cultivate grass in the Salalah brackish zone would benefit from the total grass production in 4 months by 300,000 OMR. That means that if those farmers close their bores for a period of 4 months a year, the state should compensate all those farmers annually by 300,000 OMR, which was equivalent to 780,000 USD during 2005. This water management option is more cost effective and sustainable against the high cost Desalination Plant option currently being investigated by the water supply department of Salalah city. The agricultural sector is the dominant water-using sector and currently (2005) accounts for more than 70% of the total consumption in Salalah plain.

37.4.1.3 *Water budget*

A study conducted by D&MI (1991) has shown that the water stock or the total groundwater reserves under the plain are estimated at 340 million cubic metres (MCM) with an annual aquifer recharge of approximately 32.4 MCM/yr, of which 95% is flow from the Jabal and the balance from rainfall and spring flow.

In this study the recharge is calculated by the flow model at 51 MCM/yr, of which about 98% is the inflows from Jabal AlQara and the balance from rainfall recharge on the plain. The current (2005) total freshwater exploit of the Salalah plain is estimated at 65 MCM/yr of which agriculture consumption accounts over 70% and potable use 15.2 MCM/yr (23.4%). The current artificial recharge by the injection into tube-wells of the treated wastewater along the coastal strip is about 5 MCM. Hence, the deficit in the aquifer balance of Salalah plain during 2005 was calculated at 9 MCM. The computations in this paper of the future projections to identify the aquifer's water budget indicate the following:

The stock of the aquifer is calculated, as mentioned earlier, at 340 MCM. The year 1992 is assumed to be in balance, as the calculated subsurface inflow in this paper seems enough to replenish the total abstraction during that period of time that means that the discharge in 1992 was within the recharge budget. The deficit started from the year 1995. Taking the quantity of 11 MCM as an average deficit in the aquifer from 1995 through 2020 and multiply this by 26 years (1995–2020) the result will be 286 MCM some of which has already been extracted. The remaining freshwater will thus be extracted over the next years. This paper computes the accumulating deficit during the time (1995–2020) in 2020 to be 286 MCM, which is lower than the water reserve estimations— i.e. 340 MCM. The average annual deficit is estimated at 11 MCM/yr. The total aquifer reserve will be saline at the end of the year 2012 or 2036 in the case of business as usual under changing underflow by 5% or under constant underflow, respectively, with no-management interference along the predictive period (2006–2020) (Fig. 37.3). The artificial recharge by the injection of the treated wastewater into the plain aquifer was considered from the year 2002 and 2003 through 2020, respectively.

37.4.2 *Sustainable yield of fresh groundwater*

Comparison of the total demand for fresh groundwater of 65 MCM/yr in 2005 with the total recharge which was calculated by the flow model at 56 MCM/yr indicates a fresh groundwater deficit of 9 MCM/yr, which is equivalent to $>16\%$ of the calculated natural and artificial annual recharge. Without introducing changes, the increase demand for fresh groundwater would result in an increased deficit, calculated to be about 9.4, 11.2 and 13 MCM/yr by 2010, 2015 and 2020, which is equivalent to $>16\%$, $>19\%$, and $>21\%$ of the calculated total annual recharge, respectively (Table 37.4). The amount of natural recharge determines the safe groundwater yield for a given hydrological unit, that is, the amount which can be removed without depleting the finite groundwater reserve (Bennett & Doyle 1997). The water consumption during 2005 was 16% higher

Table 37.4. The water budget and future projections under constant recharge during the predictive period.

		Years				
No.	Details	2000	2005	2010	2015	2020
1.	Annual recharge	51	51	51	51	51
	Artificial recharge	–	5	7.6	8.8	10
	Total recharges	51	56	58.6	59.8	61
2.	Total discharges	62	65	68	70.5	74
	Urban demand	16	19	22	24.5	28
	Garziz & MAF	10.8	10.8	10.8	10.8	10.8
	Sahalnawt farm	4	4	4	4	4
	Traditional farms	31.2	31.2	31.2	31.2	31.2
3.	Annual deficit	11	9.2	9.4	11.2	13

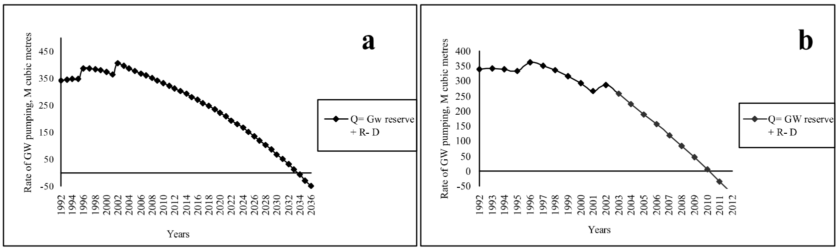

Figure 37.4. The relationship of the rate of groundwater withdrawal and time of the expected exhaustion of freshwater from the aquifers under a) constant recharge, and (b) changing annual recharge at 5%.

than the available resources from renewable and treated wastewater. That will increase to >21% in 15 years if no water demand management actions are implemented.

Figure 37.4 shows the plain aquifer groundwater storage relationship between the rate of the annual abstraction from the reserve and period of groundwater expected exhaustion. The relationship is considered as *Q* (withdrawal) equals the aquifer storage plus the annual recharge minus the annual discharge (Eq. 37.1).

Aquifer storage was calculated by D&MI (1991) at 340 MCM. The discharge starts to increase from the 1995 and the recharge continues the same as calculated in this study, till the year 2003 when the artificial recharge contributes by additional rates that increases the recharge from year to another. The deficit started growing from the 1995, by mining the aquifer reserve annually. In baseline scenario, which assumes constant underflow, the groundwater is expected to be exhausted by the year 2036, whereas by assuming the baseline scenario that considers underflow is changing annually at 5%, the reserve would become saline by the year 2012 (Fig. 37.4). This will of course not be the case as before that pumping will be stopped due to excess salinity in many wells, but is presented here to stress the urgency to change the water management in Salalah.

The other question is what the opportunity cost of new agricultural water source is? The treated sewage effluents are the new source that may replace the current groundwater resource in crop irrigation should the plain aquifer become saline. The treated sewage is cheaper than the Reverse Osmosis effluents. Currently in Salalah, the cost of 1 CM of treated wastewater effluent is equal to 0.17 OMR, equivalent to 0.50 USD, whereas the cost of 1 CM of Reverse Osmosis is proposed at to 0.66 OMR, equivalent to 1.70 USD. The annual agricultural water demand within the main groundwater plain aquifer divide is considered constant throughout the predictive period at about 46 MCM/yr.

The relationship is considered as; the annual opportunity cost (*OC*) equals the cost of the treated wastewater needed (*TS*) multiplied by the agricultural annual water demand (*AD*). Thus, the current

Table 37.5. The average rate of salinity development (µS/cm) in the aquifer versus the average distance from coast (Shammas 2002).

Description	Distance from coast (km)	Year 1985	1991	1998
Coastal strip	1.2	3360	4175	6042
Residential strip	2.4	3073	3210	3307
Garziz farm site	5.5	848	1325	1750
North of Salalah airport	8.5	640	665	840

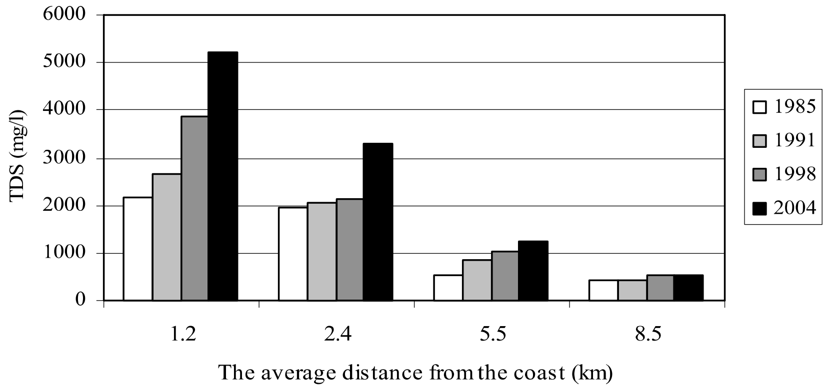

Figure 37.5. The average rate of salinity development (mg/l) in the aquifer versus the average distance from coast (km).

annual opportunity cost of new agricultural water sources if the treated wastewater effluents are considered is about 8 Million OMR (21 Million USD) (Eq. 37.2).

Table 37.4 shows that the consumption for agricultural purposes, whether it for the irrigation of Dhofar fodder farms or for irrigation of the traditional farms in Salalah are the main factor for the over-extraction from the aquifer. The total annual agricultural demand during 2005 was considered at 54 MCM, of which Razat farm consumed over 6 MCM annually. Razat farm and part of Sahalnawt farm are located outside the plain aquifer sub-basin and are excluded. The percentage of the total agricultural consumption during 2005 indicates an average use of 70% of the total annual discharge from the aquifer water.

37.4.3 *Salinity development in the aquifer*

The average of the increase in salinity values as the total dissolved solids (TDS) in the strips are presented in Table 37.5 and Figure 37.5. These data shows the increasing rate of salinity in the wells in all areas of Salalah plain (strips), which means decreasing of the freshwater area in the aquifer. The electrical conductivity and chloride concentrations were considerably high in the agricultural and residential strips and Garziz grass farm. Shammas (2002) concluded that the present status of groundwater quality in most of the agricultural and residential strips are far from drinking water standards.

37.5 CONCLUSIONS AND RECOMMENDATIONS

Water mining has led to the contamination of aquifers due to sea water intrusion. Moreover, saline water will certainly reach the remote areas north of the coast line, if such intrusion continues without measures to solve the problem. The complete loss of the fresh water aquifer, situated north of the Dhofar fodder farms (Garziz farm), will be inevitable.

Due to the increasing extraction of water to meet the demand for various purposes and the imbalance between annual recharge and extraction rates it is expected that the aquifer salinity will rise sharply. The in water salinity will be too severe as to lead to the abandonment of farming activities. This conclusion is based on earlier studies and estimations of this paper. The principle of safe groundwater yield from the plain aquifer must be considered for defining the annual acceptable level of abstraction. The concept of water management efficiency in agricultural sector in terms of the economic returns of crop production must be investigated in Salalah plain.

The results of this study confirm that Salalah aquifer is under serious threat and the area facing environmental disaster. Thus, there is an urgent need to initiate measures to restore the water balance in the aquifer. It is therefore recommended to:

- Immediately implement the Royal Decree No. (23/94) issued on 8th Feb. 1994 concerning the designation of a site in Dhofar governorate for public utility, and relocation of Garziz farm that is located on the freshwater zone, to a site away from Salalah plain aquifer. This will save about 8 MCM of water per annum. The farm abstract annually almost 12% of the total pumping from the aquifer in 2005.
- Immediately relocate the fodder production of Ministry of Agriculture and Fisheries (MAF) Livestock Research Farm that is located on the freshwater zone, the action would improve the water supply situation for Salalah. The farm has 3 centre pivots; all are utilizing 3.12 MCM per annum.
- Suspend the abstraction of the grass production in the brackish zone for 4 months a year.

Because of the existence of the wells for the irrigation of the traditional farms that are located along the coast line extending from Dahariz in the east to Awqad in the west of Salalah plain, the salinity exceeds 5000 mg/L (Table 37.4). When salinity exceeds this level the water will not be suitable for growing most vegetables resulting in poor productivity and quality. Accordingly, we recommend suspension of pumping water for irrigation of traditional farms particularly from wells where salinity exceeds this level.

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CHAPTER 38

The hidden language of rural water supply programmes

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ABSTRACT: Donors have come to recognise the importance of the socio-cultural context within which the implementation of rural water supply programmes is unfolding. Among the technical, institutional and beneficiary components, recent programmes tend to incorporate socio-cultural factors as additional components. There seems to be a gap between the actual benefits from an improved water supply and the targeted benefits specified through programme goals and criteria. This gap resides in the fact that the implicit and unconscious role of culture is often overlooked. If there is no incentive to act upon this cultural phenomenon, this ignorance may lead to cross-cultural miscommunication and deficiencies in the participatory process required to fulfil the programme goals. In this paper, it is shown that water supply programmes in Bangladesh can be improved by making the role of culture explicit by institution building at the local level.

38.1 INTRODUCTION

In development studies, the literature on unsuccessful North-South technology transfer to developing countries is rather extensive (see e.g. Ihde 1993), but the linkage between the key aspects underlying this failure and possible solutions is poor articulated. The reason why technology transfer often fails may reside in the fact that the apparent co-evolution between the technology and the societal context, in which the technology is developed and has to perform its function(s), is often overlooked. It is said that a new field of research emerged in the sixties of the last century, better known as Science and Technology Studies (STS), focussed on interlinking science, technology and society (Ravesteijn et al. 2006). The need for such a line of research came from an erosion of belief in that science, technology and society are apparently involved in an unbalanced development process to be steered for the sake of mankind. Within the context of STS and its subsidiary Technology Dynamics, much research has been done on this co-evolution. Among many others, this line of research is largely indebted to the work of Hughes (1983), who studied the historical development of power networks. In his work, he came to recognise that technology is socially shaped and shaping society. Similarly, philosophers and historian of technology therefore tend to speak of socio-technology (see e.g. Achterhuis 1995, Bijker et al. 1987, Kroes & Meijers 2006, van der Vleuten 2000).

When it comes to developing countries, theoretical and empirical insight from STS and its subsidiary Technology Dynamics might be useful to arrive at a better understanding of North-South technology transfer. The need for such research effort lies in the fact that present-day donor attempts in technology transfer have often an imposed character as a community is usually conceptualised as a relatively passive actor (van Wijk-Sijbesma 1985). This may give the impression that a community is merely perceived as an empty vessel to be endowed with capacities to deal with technology. What remains untouched is the fact that the targeted community provides the socio-cultural environment within which technology transfer unfolds and to which the intended beneficiaries are accounted (Singh et al. 2005).

In line with the socio-technology concept, the technological device has to be unified with the social environment in which it is supposed to perform its function (Ropohl 1999). This suggests that technology transfer within the context of rural water supply programmes does entail change in the layers of cultural codes and values of the receiving society. This change deals with 'new' codes and values introduced by the donor. For that reason, among others, De Jong & Kroesen claim that technology transfer implies value transfer. What is important to realise is that this part of technology transfer is the most difficult one but, nevertheless, decisive in whether it ends up successfully. As the American anthropologist Edward T. Hall already noticed, the difficulty lies within the fact that much what goes on in culture occurs rather implicitly. The implicit character of cultural change often remains hidden in the development of programme goals and criteria. Consequently, this paper aims to gain further insight as to why and how this lead to unfavourable outcomes of rural water supply programmes. For that aim, these questions will be answered within the context of the Drinking water security for the poor and women project. This project is initiated by a Bangladeshi NGO, the Alternative Movement for Resources, Freedom and Society (AMRF) to improve the water supply in a small village. A distinct feature of this project is that it promotes a community-based participatory approach to both its development and implementation.

Finally, taking into account that 'hidden' cultural values are apparently transferred in programme goals and criteria, what does this imply? In what sense does this affect the participatory process that should lead to the fulfilment of programme goals? In order to answer these questions section two will provide a theoretical background on the transfer of participatory approaches but also the involvement of value transfer in that respect. Section three elaborates on the 'hidden' values that are usually transferred in project agenda and implementation. Findings on how 'hidden' values affect codes of behaviour embraced by the community will be presented in section four. The conclusions and recommendations are provided in section five are based on the results from the observations and in-depth research carried out in Bangladesh.

38.2 THE TRANSFER OF PARTICIPATORY APPROACHES

38.2.1 *Institutional transplantation*

The use of participatory approaches in rural Bangladesh can be seen as a form of institutional transplantation: the borrowing of political or cultural codes of behaviour, technological and managerial development practices and policies, the transplant, from a donor country or organisation to a (receiver) host country (De Jong et al. 2002). Institutions can be characterised as mental programmes responsible for structuring a set of repertoires of behaviour e.g., thinking, acting and signalling, in social interactions (Hofstede 1991, North 1990). De Jong et al. recognise two complementary perspectives in policy transfer i.e. the 'goodness of fit' and 'actors pulling in'. The former perspective requires that local actors (i.e. project initiators and implementers) have the final decision on whether the transplant is pulled in and adjusted according to their own requirements. The provision of such manoeuvre space for domestic actors prevents them from restraining and stressing the transplantation process, which is apt to lead to ineffective outcomes. The latter perspective takes into account political, legal and cultural affinities and (dis)similarities between the institution and the country that either voluntarily or involuntarily introduces these institutions. To come to a deeper understanding of cultural affinities and (dis)similarities, the author refers to the work of the Dutch sociologist Hofstede (1991). He conducted a research program on the assessment of national patterns of value orientations from the beginning of the 1970s onwards among IBM employees across the world (Hofstede 1991). He structured his empirical data on countries according to five main criteria:

- Low versus high *power distance*. The power distance relates to the acceptance of hierarchical role differentiation in society. It measures the extent to which less powerful members in society accept and expect that power and wealth are inequitably distributed. High power distance suggests that inequalities in power and wealth have been instilled into society. By contrast, a low power distance is indicative of a society in which differences between people's power and wealth seem to be synchronised. In such societies equality and opportunity for everyone is constrained.

- Low versus high *uncertainty avoidance*. This criterion reflects the urge of society to avoid uncertainty. It emphasises the level of tolerance for uncertainty and ambiguity within the society i.e. unstructured unfamiliar circumstances. High uncertainty avoidance implies less tolerance on the part of society towards uncertainty and ambiguity. In order to expel such uncertainty, society becomes rule oriented through laws, institutions, regulations, etc. Low uncertainty avoidance in society implies that there is less concern about uncertainty and ambiguity and tolerance for variability. Such society is less rule oriented, more likely to accept changes and keen to undertake more and greater risks.
- *Individualism* versus *collectivism*. A collectivistic society is one in which individuals are integrated into groups through strong ties. It reinforces extended families and collectives where everyone takes responsibility for fellow members of their group. An individualistic society is characterised by weak ties between individuals.
- *Masculinity* versus *femininity*. This criterion focuses on the extent to which gender role differentiation is accepted in society. In feminine societies the men share the same caring values of life as the women. Such societies display a low level of differentiation and discrimination between genders i.e. females are treated equally to males in all social aspects. In contrast in a masculine society, these values are rather assertive and competitive. So, a gap between women's values and men's values does exist. It indicates a high degree of gender differentiation and discrimination i.e. males dominate a significant portion of the social and power structure.
- Short versus long-term *orientation*. This reflects to what extent national cultures are long or short-term oriented in decision-making. A long-term oriented society strives to achieve congruence with its variable environment. In order to do so, such society tends to adjust its old traditions and customs to novel circumstances (challenges or threats). By contrast, the short-term oriented societies tend to persist with old traditions and customs regardless of future developments in the unfixed environment.

This resulted in a division of countries into eight clusters. One of these clusters is called "less developed Asian countries", one of which is Bangladesh. These countries are characterised by high power distance, low to medium uncertainty avoidance, low individualism, medium masculinity and a medium to high long-term orientation.

38.2.2 *Community management and the underlying principles*

For technology transfer to be successful, it is obvious that the hidden values unconsciously transferred in project development and implementation has to be exposed. The focus of this paper is on how these values could be integrated in a receiving culture, which is collectivistic, hierarchical, masculine and long-term oriented and has a considerable urge to avoid uncertainty. Equally importantly, this paper aims to provide further insight as to the enabling impact of participatory approaches to project development and implementation on value transfer. To provide the reader a better understanding of the importance of community participation with regard to the implementation of a community-managed drinking water supply system, it is useful to present a brief history of community management.

It is said that the first steps towards community involvement were constituted during the 1977 World Water conference in Mar del Plata, in which the community involvement paradigm was officially adopted by the international society (Schouten & Moriarty 2003). From the 1980s onwards, a series of global initiatives such as the International Drinking Water Supply and Sanitation Decade, the New Delhi Statement, the 1991 Fresh Water Initiative, 1992 Dublin Statement on Water and Sustainable Development and the 1992 Earth Summit in Rio de Janeiro has contributed to community management in its present shape. The focus of these initiatives has primarily been on developing new and enhancing existing guiding principles for the global water and sanitation sector and strengthening the emphasis on community management in water and sanitation programmes. More detailed, these initiatives have identified a set of four core elements to be associated with community-managed water supply systems: collective community *control*, *operation* and *maintenance*, *ownership* of the system and *cost recovery* of the system (see e.g. Schouten & Moriarty 2003, Lockwood 2004).

Although community management has become the leading approach in developing rural water supply programmes, people's tendency to participate in this process still depends on their interests, perceptions and purposes. Arguably, one might then conceive of a community as nothing more than groups of individuals or stakeholders, mainly competing, but sometimes gathered together in interest groups for a common purpose (Schouten & Moriarty 2003). This approach projected on water supply recognises just a user community as a group of individuals who use a drinking water supply system. This, in turn, would imply that anyone who pays the obligatory fee automatically participates in this community that is served by the water supply system. Those who do not are not included in this community. One may then question whether communities truly do exist if interests, perceptions and purposes significantly diverge. Arguably, this question is to be answered in the affirmative. No community can function properly unless most of its members "act" at least to a certain degree as community members since they voluntarily comply with its moral commitments and social responsibilities (Etzioni 1993). A community does exist if the people who belong to it are truly convinced of being a part of it (Schouten & Moriarty 2003).

Having acknowledged the existence of a community, one must then go on to determine its form. The heterogeneity of a community i.e. the fact that people embrace different interests, perceptions and purposes has already been noticed. It is best perceived as a melting pot of continuous negotiations, discussions and conflicts (Etzioni 1993). In addition to its diversity, its unclear boundaries i.e. its physical demarcation, results in its instability. If one considers these two arguments, it seems that a community refers to a dynamic entity persistently changing in its social, economical and organisational composition. Its dynamic character persists since it is linked to and has shared boundaries and interacts continuously with other communities. So, one cannot perceive of community as a distinct isolate and overlook its enabling environment for instance its relation to the administrative units in the country i.e. union, district or country. The community should be seen as a set of people with common but also conflicting interests and ideas and different social and economic and cultural backgrounds (Galvis et al. 1997). Whereas water supply (systems) is of common interest, they can also constitute a considerable source of conflict at the same time. Thus, a community is a set of people with common but also conflicting interests and ideas, subjected to external influences, and different socio-economic and cultural backgrounds. The assumptions often made by project donors, that everyone in a community benefits equally from the arrival of an improved water supply system, that everyone agrees, that everyone participates in managing it and that no one is excluded from its application is a misinterpretation (Singh et al. 2005). No such social coherence does exist in reality.

With this in mind, Hofstede's set of cultural dimensions is suggested to be helpful in exploring the socio-cultural context of Bangladesh in which project implementation seems to unfold and on which the development of project agendas has to reflect. In this paper, the author pays attention to five 'hidden' values generally at work in project agendas and project implementation in development aid (van der Voorn 2006). These are:

- Cooperation across the lines of family loyalties
- Women in public life
- Opposition, criticism and pluralism
- The drive for fundamental change
- Instrumental time and planning

38.3 THE DRINKING WATER SECURITY FOR THE POOR AND WOMEN PROJECT

38.3.1 *Background*

The Alternative Movement for Resources, Freedom and Society (AMRF), a Bangladeshi NGO, initiated the "Drinking water security for poor and women" project in Banagram village, Nanirkhil

Union, Gopalganj District, Bangladesh, supported by its associate organisation the Bangladesh Centre for Advanced Studies (BCAS) and its local partner NGO the Bangladesh Auxiliary Service for Social Advancement (BASSA). The involvement of BCAS and AMRF covers overall coordination, supervision and management, while the implementation of project activities in Banagram village is handled by BASSA. The project goals are the enlargement of the livelihood potential and women's access to and control over natural resources. Moreover, the project endeavours to play a supporting role in the formation and strengthening of a community-based institution (AMRF 2004). These activities are expected to result in the establishment of a water council i.e. *Pani Parishad*. Hence, a community's informed and proactive participation in the development process is to be induced. AMRF and BCAS acknowledge women's vital role in environmental management and development. So, the project tends to develop enabling conditions for women. Generating better collective efforts to address common issues is also envisioned as a project objective.

The proactive participation of the community is indispensable to the establishment of the Pani Parishad in Banagram village. Once the Pani Parishad has been formed, its main function is to undertake the responsibility of taking care of household interests in drinking water security through collective decision-making, consensus building, and resource and technology management. The Pani Parishad is supposed to secure the sustainability and future development of the project. This essentially means is that the Pani Parishad has to develop into a community-based institution that becomes self-reliant—independent of the initiating NGO—in securing the local water supply and demand.

In the first year of the project the Pani Parishad will be set up and its members trained. Pilot demonstrations of safe water options will be reviewed and assessed in Banagram. The second year will continue the activities started in the first year. Lessons learned from first year's exercise will lead to refinement of the project. Safe drinking water options should be replicated on a larger scale. This scaling up should be continued in the last year in workshops, seminars, discussions to disseminate the findings and experiences. In order to claim rights to and ensure drinking water security, dialogue and negotiation with public, private and corporate agencies should be facilitated. For this study, a socio-cultural analysis of the transition towards a Pani Parishad was conducted between April 21 and July 21, 2005 (van der Voorn 2006). The following section illustrates how 'hidden' values affect the participatory process towards the fulfilment of the project goals and how the AMRF has dealt with counterproductive side effects.

38.3.2 *Observations of changes in codes of behaviour*

38.3.2.1 *Cooperation across the lines of family loyalties*

In line with the norms of a collectivistic society, the family is all-important in rural Bangladesh. The in-group consists of the family and from that perspective all the other people are to all intents and purposes outsiders. Creating a job for a family member is not perceived as nepotism, but rather as one's duty and as responsible behaviour. However, if a new technical device like, for instance a deep tube well is to be installed, cooperation between members of different families needs to be created. Bonds of loyalty and trust and responsible behaviour need to be created across family lines to achieve such a degree of integration that even the money of the members of different families can be entrusted to the Pani Parishad.

Throughout several Pani Parishad meetings attended between May 10 and June 26, 2005. It appears that cooperation between families result in attempts to establish coalitions of members from rich and poor families. Particularly at the beginning of this sequence of meetings rich Banagram villagers were enthusiastic to attend the meetings, in order to improve and maintain their situation. Unfortunately, they soon recognised that such thoughts were just illusions, since the community acknowledged that they had to put the necessary effort in it themselves. What kept most poor villagers in fear became the main drive for the rich to participate in the decision-making of the deep tube well still to be installed. The poor were afraid that the rich could claim more rights to ownership because of their higher level of contribution. The rich thought that they could do so. Yet from the side of the rich, there were some doubts as to whether the benefits from the deep tube well should be allocated to those who could not contribute at all. The project team had to convince both poor and rich repeatedly that everybody should equally benefit from and contribute to the deep tube well still to be installed. As a solution to the problem, the project team set obligatory tariffs for future services. Micro-credit loans were offered to those who could not contribute at all. It seems that the most important agreement in establishing poor-rich coalitions is the common desire to operate in such a democratic system.

38.3.2.2 *Women in public life*

As one might imagine, in a masculine society women's role in public life does not extend beyond fetching drinking water in rural Bangladesh. If women take it upon themselves to represent the community, by chairing a meeting, becoming a member of the Pani Parishad etc., that constitutes even one step further in a process of change. This process is gradually occurring in Bangladesh but the situation is still fragile and not commonly accepted. However, the fact that it takes place is of primary importance in view of the centrality of women's role as water provider and managers (Singh et al. 2005).

The Pani Parishad meetings on gender issues held on 12 and 19 June demonstrated that many villagers, particularly the men, did not see and recognise the need and urge to discuss such issues. At the first meeting for instance the majority of people present were women. However, the large absence of men was perhaps a reflection of the preparations that they had to make for the market next day. People needed to prepare for this and, consequently, did not have enough time to join the meeting. Surprisingly, at the second meeting more men showed up but still the women formed the majority. Moreover, gender inequities were exposed in many ways. During meetings women had to sit on the ground, while the men sat on chairs for instance. It seemed that the men allowed the women to give their opinion but somehow they had the final word. A westerner, who was observing the meetings, might draw the conclusion that women needed to raise their voice much higher than the men just in order to be heard. It may imply that attempts on the part of men to avoid women's involvement in decision-making take place implicitly as well. Though women are able to participate in the meetings, the question whether their opinions and views are taken into account in the ultimate decision remains open. Another example in that respect is that the men, particularly the poor, are too busy earning money to support their families. Men are therefore less interested in discussing drinking water issues since those are actually labelled female concerns. Yet those, particularly the older people and the youngsters, who were able to participate in meetings, were keen to contribute to discussions. Again, the question that remains is whether it concerns conscious attempts to constrain women's influence in decision-making.

38.3.2.3 *Opposition, criticism and pluralism*

A hierarchical structure in the community leaves less room for public disagreement. If the people in authority: officials, the educated, the rich, the elders within the community, give their opinion that is all too often the end of the debate. Different interests and conflicting opinions are not easily expressed and explicitly sorted out in an effort to create common understanding and a common will to act. It is difficult for the project to serve the poor and women, if their interests cannot be expressed as interests opposed to the interest of those in authority. But if a plurality of judgments, interests, and values cannot be appreciated, as part of a process that leads to common action, it will be difficult to find the best solutions. Here, too, a learning process is necessary, in which opposition and criticism can be combined with respect and loyalty.

A good example of this very process was the 22 June meeting on the mid-term evaluation of the project. Mr. Khorshed Alam (AMRF), Mr. Johir Bhi (BCAS) and Mr. Bidut Kumar Das (BASSA) discussed the aim of the "*Drinking Water Security for Fringe People*" project, whether to help provide the community with safe drinking water, to make an inventory of the technology is to be implemented according to the community's preference and to make a decision concerning the location for the installing of new drinking water systems with the Pani Parishad and villagers. Yet the water council had already announced that it preferred deep tube wells. Despite several alternatives proposed by project initiators, the Pani Parishad persisted in the choice of arsenic sensitive shallow tube wells since they believed that these prevail over other technologies. Ironically, in their efforts to agree on the technology to be implemented, through consensus building, the persistency of Pani Parishad and community members created dividedness among the project initiators. It seemed that each of them wanted to sustain his individual 'hobby horse' but unfortunately as the discussions continued this proved to be counterproductive. Some members of the project team and project initiators even stepped back during intense discussions. Once the Pani Parishad and the villagers noticed this, they tried to impose chaos through voice-overs just to claim authority. Unexpectedly, that worked most of the time because the villagers like to rally round a 'powerful' and 'influential' speaker or someone who has dissenting views and ideas. Eventually, the project initiators gave in and decided to agree with the Pani Parishad and the villagers. The meeting perceived through the eyes of a westerner, might be characterised as a melting pot of voice-overs and people who continuously persist with their hobbyhorse or rally round a dissenter. The westerner would be inclined to call this organized 'anarchy'.

38.3.2.4 *A drive for fundamental change*

No one refuses improvements, especially if they come as gifts from outside like for instance from NGOs, though a gift comes with a burden. This does not imply that 'anything goes' but it is a warning to project teams and initiators against tactless implementation attempts and to community and Pani Parishad members to undertake responsibilities. The fact that the latter becomes eligible to claim rights to participate without assuming its responsibilities is unethical and illogical (cf. Etzioni 1993, Glendon 1991, Stone 1974). Nevertheless, can people be encouraged and inspired to try radical new ways of life, and, if not, will disaster force that upon them? How much novelty can (or should) be accepted and how much uncertainty?

Again, the 22 June progress meeting proved to be a useful experience from that point of view. It has been mentioned that the community and Pani Parishad members persisted with the implementation of deep tube wells, regardless of the potential hazards and health risks in using these. In compliance with a low to medium urge to avoid uncertainty, they tend to prolong traditions and customs regardless of the possible side effects and individual preferences. It indicates that once a community recognises the benefits of a particular technology, they soon give in, but a lot of efforts has to be made to propose perhaps even 'healthier' technologies. But does such behaviour seem odd if NGOs have been emphasising the use of tube wells as a solution to serious health problems inflicted on polluted surface water already since three decades? It may indicate that uncertainty has been avoided.

38.3.2.5 *Instrumental time and planning*

Since the Middle Ages the clock has been the technical device to bring discipline to labour, organisations, and planning in developing technologies. The coordination of tasks, tight schedules, objectifying time, is a specialism of the West, and it has become much criticised for its feverish rhythm and speed. And certainly there are many things, which should be done as slowly as possible, but there are other things, which cannot be managed, if time schedules are not kept and if labour division and organisation is flawed. Here too an important learning process needs to take place. In order to illustrate the impacts on handling the western way of instrumental time and planning in the community, the 10 May Pani Parishad meeting is used as example. The objective of the meeting was to discuss the composition of the water council and its purpose. Unfortunately, no single Pani Parishad member did show up nor did the majority of the community. The project team waited for almost two hours before deciding to cancel the meeting for that day. Due to the large number of absences no issues could be discussed in this meeting. Those who did show up said that the others were absent because they had to work. Ironically, those who did not have to work were at home and claimed that they did not know that there was a meeting planned for that day. In order to prevent such misunderstanding again, the project team compelled the villagers to start the following meetings at five o'clock in the afternoon. Though villagers and Pani Parishad members obeyed the 'directive' to show up, they never showed up on time. Through the eyes of a westerner, the community handles time according to the saying "when things are ready", which makes appointments meaningless

(Hall 1959). In line with Hofstede, they seem to persist in a limited long-term orientation, while sustaining old traditions. In fact, they are able to do so, since traditions prevent villagers from being confronted with the urge to take decisions or to undertake responsibilities that may imply consequences for the 'future'. So, in that case why does one have to foresee future developments, as 'things that are likely to take place according customs and traditions?

38.4 DISCUSSION AND CONCLUSIONS

The general conclusions that can be drawn from this case study are presented below:

- The above-discussed empirical illustrations have shown that technology transfer involves value transfer. It is observed that value transfer occurs rather implicitly and unconsciously as it takes place and aims to establish changes in the implicit parts of culture.
- The five hidden values create obstacles in the participatory process. If there is no incentive to act upon these implicit tensions, they may affect the fulfilment of targeted programmes goals in a counterproductive way. The difficulty in coping with these tensions resides in the fact that those are highly implicit and thus sometimes not visible for the initiating NGO or donor.
- Although it is rather difficult to act upon implicit tensions, the empirical illustrations have demonstrated that making those explicit in terms of new institutional arrangements constitutes a step forward in improving the management of water supply programmes.
- The two perspectives in institutional transplantation appear to be helpful in guiding the process of value transfer. More specifically, the 'goodness of fit' perspective provides valuable insight into cultural compatibilities or incompatibilities between the transplant and the receiving culture. However, it should be mentioned that the corresponding author, who have developed the set of cultural dimensions, cited in this paper, cannot claim to be all-inclusive in separating cultures at the national or regional level. It looks as though that the set of cultural dimensions does not concern a set of fully independent dimensions. Despite this theoretical and methodological imperfection, this set can be helpful in drawing a first impression of a national or regional culture.
- When it comes to the issue of how to cope with implicit tensions caused by large cultural incompatibilities, the 'actors pulling' perspective provides further insight as to how make them explicit. This perspectives draws on the idea that local actors should be endowed with manoeuvring space, in order to reframe the transplant i.e. new institutions according to their own requirements. Within the context of the DWSFP project, less attention has been paid to two important issues that should have been at the heart of this adoption process. Firstly, the roles of indigenous technology and culture in project development and implementation have been articulated in a negative way. Secondly, the project has thus far given less room for identifying local desirability towards societal changes. As both issues have not been emphasised very properly, the participant observations have revealed that this could not be without any consequence for the participatory process.
- Despite the emerged cultural tensions, the introduction of new institutional arrangements or rules has constituted points of departure for societal changes. Arguably, these changes would have not taken place without the compulsive character of the project. It is then questionable whether the changes that have thus far been achieved during the project or the changes that still have to be achieved will sustain after the project has come to an end. As the proposed changes affect behavioural patterns, both parties should be more aware of the fact that adaptation to new behavioural patterns requires more time than the time available within the project. It then seems doubtful whether these changes can be achieved within the time span of a three year project. Consequently, the author claims that the fact that donors or NGOs being pro-actively involved in project development and implementation does not testify to an act of imposition. In fact, supposing that far-reaching societal changes will be achieved in a three-year project may be more indicative of such an act.
- To elaborate on the previous issue of cultural change, there seems to be a paradox at work in development co-operation. If indigenous development would mean non-interference within a

national culture, this would be a contradiction in terms, in that any form of development would contaminate the indigenous characteristic of the culture in question. The important lesson to be draw from the DWSFP project and many other development aid projects is that changes in the indigenous culture are almost inevitable and, consequently, they should take place in a respectful way.

38.5 RECOMMENDATIONS

Considering it in its first year, it is not surprising that an innovative project such as the DWSFP project has some managerial flaws. Regardless of any attempt to improve the overall management of the project, the donor in charge, ICCO, should come to recognise that technology transfer goes in hand with interference in the old tissue of culture of a receiving country. What this basically means is that it imposes the imperative to do it in a respectful way. In detail, it requires a pro-active donor and a long-term oriented donor to be in charge of the project. By doing so, the donor is forced to take a long hard look at himself as the current level of civilisation of the West is the outcome of a struggle that lasted more than a thousand years. As the past still proves to be a reliable indicator for the future, there is plenty of room for optimism even in the case of Bangladesh. Having said this, it is obvious that societal changes do not emerge overnight. By contrast, it compels both donors and receivers to judge and engage with each other in terms of dialogue on the basis of a common history of equality, loyalty, dignity and respect.

Closely related to the above-mentioned issue of cultural change, donor and NGO effort could be more focussed on increasing the transparency of value transfer. This paper has demonstrated that value transfer unfolds rather implicitly, which may give rise to cross-cultural miscommunication and tensions and, eventually, to deficiencies in the participatory process towards the fulfilment of the project goals. First priority should, therefore, be given to the need for making the apparently implicit tensions between the project goals and the participatory process explicit. The transparency of the DWSFP project can be increased in terms of enhanced overall project management, learning processes, transparency, and community involvement.

The DWSFP project should be organised in a more experimental mode by using flexible time and planning schedules with regard to *overall project management*. The project then becomes less vulnerable to unexpected developments that could result delays. In line with the goal of the project, flexible time and planning schedules will emphasise the importance of the participatory process. During the project, many project activities have been delayed because of insufficient overall project management. In order to catch up with time and planning schedules of the project, Pani Parishad members have been forced to acquire the necessary management capacities and skills in a rather short time period. This time constraint does not only leave any room for drawing lessons from people's performance but it is also in conflict with the guiding principles of the project.

Secondly, as a participatory approach of promoting drinking water security is at the heart of the project, its implementation should be more on the basis of a '*learning by doing*' approach. To give shape to such an approach, project implementation should increase the number of gender and management workshops and exercises, keeping Pani Parishad members well informed about future project activities, intensive monitoring of project activities, exchange experience in and information about past project activities with other partner NGOs involved in the DWSFP project. Moreover, the disadvantage of putting the project in a more experimental setting is that it reduces its *transparency* and responsiveness, which might stress the fulfilment of the targeted project goals. In order to assess the progress of the project, it is recommended to develop a set of clear cut criteria for ex post assessment, which consider the aspects like, for example, the extent of accessibility to safe drinking water, the degree of co-operation with local government bodies and other NGOs (to prevent duplication of drinking water security efforts), the degree of donor or NGO support, the degree of knowledge creation, sharing and exchange, the level of management capacities of Pani Parishad members, and the quality of ownership, control, operation and maintenance, and cost recovery of the drinking water supply systems.

Thirdly, more effort needs to be put in enhancing community involvement, in order to make the Pani Parishad representative for the whole community. Therefore, the project should stimulate co-operation between poor and well-better off families, by widening its narrow-minded pro-poor and pro-women focus, creating physical accommodation for the Pani Parishad so that the idea of institution building becomes meaningful to its members and the villagers. Furthermore, it is important to organise meeting at a reasonable time. The hardcore poor are least able to spare the time, energy or money to be involved in project meetings. To sustain their involvement in meetings and the project, a loss of income should be compensated with a single exception to cost recovery of the drinking water system.

ACKNOWLEDGEMENTS

This study was carried out as a part of graduate thesis programme at the Department of Philosophy, Faculty of Technology, Policy and Management, Delft University of Technology, Delft, The Netherlands. The author would like to thank Wim Ravesteijn, Otto Kroesen and Martin de Jong for providing valuable feedback. Likewise, the wish to thank M. Jakariya and others at BRAC and ARMM Kamal at NGO Forum for their help rendered during the field programme. Financial assistance from the ICCOH Netherlands is gratefully acknowledged.

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CHAPTER 39

"GROWNET" and its relevance to sustainable groundwater development in western India

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ABSTRACT: In April 2005, the UNESCO-IUGS approved the IGCP Project no 523 titled as "GROWNET: Groundwater network for best practices in groundwater management in low-income countries". Under this project, best practices in various aspects of sustainable groundwater management are being assessed through different projects in low-income countries. The outcomes of these projects are posted on the website (www.igcp-grownet.org) for global dissemination and replication wherever possible. Best practices like (a) Watershed management for soil and water conservation leading to increased natural recharge and (b) Recharge augmentation through artificial processes, are already being followed in a few watersheds, with active participation of NGOs and local self-help groups, in schemes promoted by Governments and International Aid Agencies. Sustainability of groundwater development is dependant on successful augmentation of recharge and on demand management through education of stakeholders. The Paper describes an outline of the best practices in recharge augmentation which have eased the situation in some of the overexploited watersheds. These need to be replicated in many other overexploited watersheds in Basaltic, hard rock terrain in western India, which is the largest exposure of Basaltic, lava rocks in the world. GROWNET aims at dissemination of these practices for application in hard rock terrains in India and other low-income countries.

39.1 INTRODUCTION

The concept of sustainability in urban and rural perspective means protection of groundwater quality and quantity. However, in the rural scenario, more importance is often given to protection of the quantity of groundwater available from dug wells and bore wells for irrigation of crops, as this is directly linked to the income of the farming family. In urban areas the residents in apartment complexes are more conscious about the quality of ground water pumped to meet the shortages in municipal tap water supply, because in many cases water from these two sources is mixed in a ground-level tank before being pumped to the tank on the terrace. In rural areas where groundwater is threatened by saline ingression or by arsenic/fluoride contamination, the quality aspects also assume importance.

The efforts for sustainability have the following three types of approaches:

1) Technical aspects such as: Groundwater management, taking a basin or a sub-basin as a unit; Watershed protection for absorbing the forthcoming harsh climatic patterns; Training and capacity building at Government Departments and Institutions in view of supply side of management.
2) Public participation: Educating the public through media and village meetings for demand side of management; (NGOs and Universities play an important role in achieving public participation in Demand Management). Ensuring people's participation in soil and water conservation programs and in artificial recharge schemes.
3) Social, political and legal aspects: Enacting and implementing groundwater protection laws; Controlling pumpage in over-exploited watersheds. Formulation of a policy for allotment of

water resources to various stakeholders. About 70% of the water used for irrigation is consumed by plants/crops for their growth and is lost from the system. City dwellers feel that contribution of Agriculture, which employs 60% of population, is only 21% of GDP in India and that the cities should receive a priority in water allocation (Macdonald et al. 1995).

However, in low-income countries these efforts are only partly effective and success stories are scarce. There is thus a need to create a global database of 'best practices' applied successfully in groundwater development projects. The database, freely accessible over the internet, would foster mutual learning through dissemination of these 'best practices'. The author A proposal was therefore submitted to International Geoscience Program (IGP), formerly known as International Geological Correlation Program (IGCP) for creating such a database. The IGCP is a joint endeavor of UNESCO and IGUS (International Union of Geological Sciences) and operates in about 150 countries, involving several thousands of scientists. It provides a multinational platform for scientists from all disciplines related to earth sciences to exchange knowledge and methodology on a multitude of geological problems of global importance. Recently, in concomitance to the change in name, the IGCP has started encouraging new projects related to practical applications of geology, especially of hydrogeology, so as to bring geosciences 'in service of the society'.

In 2005, out of 15 projects approved by IGCP, three projects were related to groundwater development. Amongst these, the Project no. 523, titled as "GROWNET—Groundwater network for best practices in groundwater management in low-income countries" is convened (Limaye & Reedman 2007). Although many groundwater development projects in the low-income or developing countries are only partially successful, 'best practices' are sometimes observed as isolated and scattered methodologies adopted in different phases in different groundwater development projects. In these projects aimed at promoting irrigated agriculture and/or providing drinking water supply in backward, rural areas, best practices may thus occur in various phases such as exploration & assessment of resource, institutional financing for wells, technology for digging/drilling of wells, pumping technology, utilization of pumped water, marketing of agro-products, recovery of institutional loans, monitoring of water quality and yields from wells, watershed management and recharge augmentation by encouraging participation of beneficiaries, implementing pumping regulations, finding amicable solutions for conflicting interests of stakeholders, using a computer model for guidance and in fostering women's role in groundwater management. Studying such best practices through site—visits and posting them on a common 'website' on the internet, is the purpose of "GROWNET".

It is hoped that GROWNET would be a humble step as a guide towards achieving the Millennium Development Goals (MDGs) in water supply sector. Out of the eight MDGs agreed upon at the UN Millennium Development Summit in 2000, Goal 1 (eradication of extreme poverty and hunger) and Goal 8 (creating global partnership for development) are closely related to groundwater resources and to GROWNET. Groundwater is responsible for over 70% of rural drinking water supply and 50% of irrigational water supply in many countries. Sustainability of groundwater supply of safe quality and adequate quantity leads to better health and reduction of poverty and hunger. Dissemination of best practices by GROWNET through its website "www.igcp-grownet.org" leads to global cooperation, because between 2005 and 2007, GROWNET meetings have taken place from China to Mexico and India to Indonesia.

Out of the several types of best practices in groundwater management mentioned above, only two are considered in this paper with reference to the Deccan Trap (basaltic lava terrain) in western India (Fig. 39.1). These are: (a) watershed management for soil and water conservation leading to more natural recharge and (b) recharge augmentation through artificial processes. The average thickness of each Basaltic lava flow is between 12 and 15 meters. However, flows as thin as 2 meters and thick as 30 meters have also been recorded. (Deshpande & Sen Gupta 1956) The average total thickness of the Deccan traps is estimated to be at approximately 900 m. Ground water occurs in phreatic condition in the soft cover of alluvium and weathered rock overlying hard Basalt and also in semi-confined state in the network of fractures, fissures, joints, cooling cracks and lava flow junctions within the hard, massive rock (Limaye, 1942). However, the occurrence of ground water

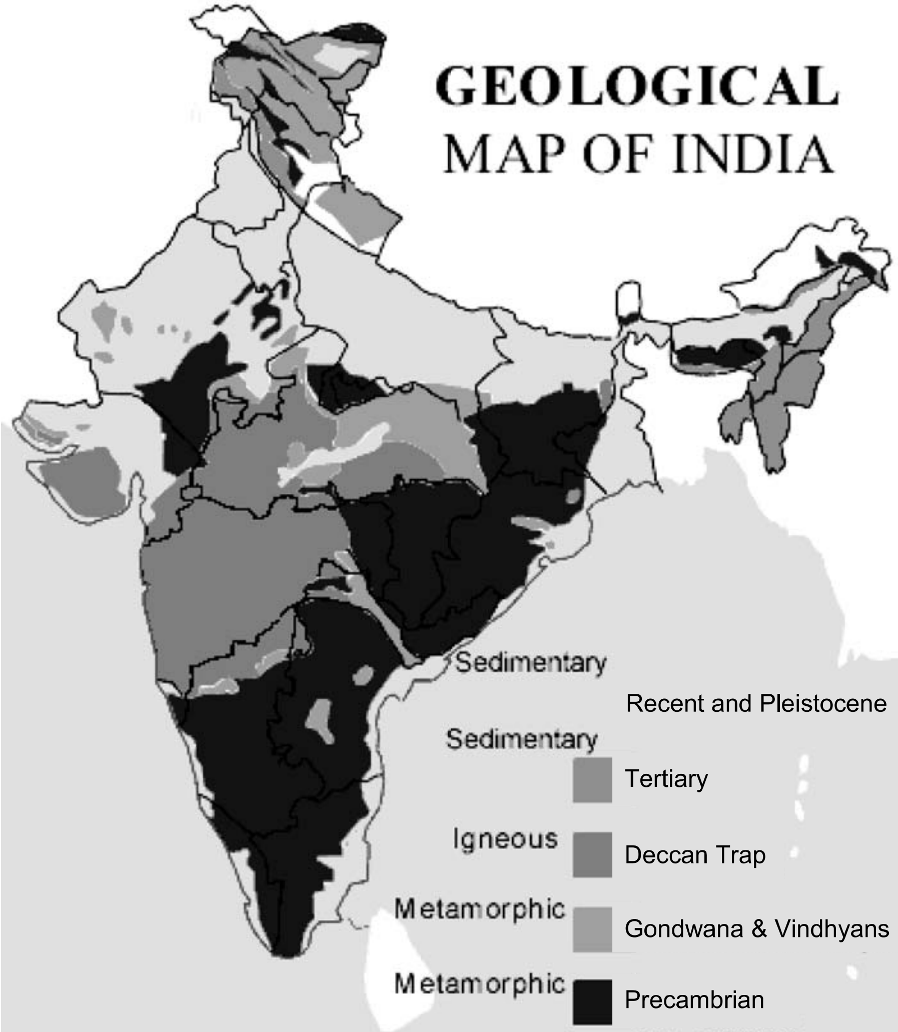

Figure 39.1. Simplified geological map of India showing the extent of the Deccan traps or the basaltic terrain covering the states of Maharashtra, Madhya Pradesh, Gujarat and Karnataka.

is mostly limited to a depth of 60 to 100 m. Dug wells of 5 to 10 m diameter and depths between 8 and 20 m tap the phreatic aquifer. Horizontal and vertical bores are often drilled in the bottom of dug wells for getting additional supply. Bore wells of 150 mm diameter and 60 to 100 m depth are also used for agricultural or domestic supply. Wells are owned by individual farmers and there is a strong sense of ownership of whatever quantity of groundwater available from the well. Pumpage control through legislation is thus very difficult. Out of about 500,000 sq kilometers of Basaltic area in western India about 30% falls in the semi-arid zone. Recharge augmentation has become a crucial issue especially in this semi-arid Basaltic terrain due to over-exploitation of the resource. The change in overexploited and critical areas in India, between 1984–85 and 1992–93 represents a growth of 5.5% per year. If this rate continues, such areas would double after every 12.5 years (World Bank, 1998). Concerted efforts from various players are therefore, necessary for recharge augmentation for balancing the over-exploitation.

39.2 WATERSHED MANAGEMENT FOR INCREASING NATURAL RECHARGE

The Intergovernmental Panel on Climate Change (IPCC) has concluded that warming of the climate system is unequivocal, as is now evident from observations of increases in global average of air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level. (IPCC 2007) Due to natural and anthropogenic causes, the climatic pattern in near future would have more extremes, causing frequent flash floods in some areas and severe droughts in other areas of the same Country or the same State. During the 2004 monsoon season in India, the western coastal area of Maharashtra State experienced ravaging floods while in the eastern portion of the same State, the meager rainfall resulted in near-drought conditions in many Districts, causing several farmers to commit suicide. Thousands of villages had to be supplied with drinking water in Tankers. Ironically, in the countrywide average for India, excess and deficit precipitation got balanced to show that the Monsoons have been normal. Any model on climate will not be able to predict the daily or weekly amount of rainfall over a given area.

Under these circumstances, there is a need to create a buffer, a cushion or a resilient interface to absorb the shocks of the climatic changes and to provide some insurance for the water managers. A watershed, subdivided into mini-watersheds of first order streams which are properly managed for soil and water conservation, provides such a resilient interface. The meeting points of climate on one side and hydrology/hydrogeology on the other side are the soil surface with its cover of grass, bushes and trees; properly tilled farms with contour bunds; farm ponds; check-bunds on small streams or gullies and contour trenches on hill slopes. In low rainfall areas proper management of mini-watersheds promotes recharge to groundwater, thereby increasing the residence time of water in the watersheds. In the form of surface runoff, rainwater flows out of a watershed in just a few days. But when recharged to groundwater reservoir, its residence time increases up to a few years. The frequency of droughts is higher in semi-arid areas and groundwater assumes unique importance in providing drinking water supply in drought years. In high rainfall areas, a degraded watershed gives rise to an evil stream having sharp-peaked, narrow-based hydrograph, while a well-managed watershed generates a beneficial stream with a hydrograph having a gentle peak and a broad time-base. 'Best Practices' in GROWNET project, are selected from land-use planning and watershed management programs promoted by the government departments in which NGOs and local self-help groups have actively participated. Without such participation of local people, the programs often meet only a limited success. Dissemination of these best practices from GROWNET website www.igcp-grownet.org will result in their duplication elsewhere. Planners and policy-makers in low income countries will also be made aware of the importance of NGOs in ensuring people's participation in watershed management programs.

Forestation of degraded watersheds with grasses, bushes and trees is the first step in watershed development. It is however, necessary to exercise caution regarding the choice of species of deep-rooted trees for forestation in semi-arid regions. There have been instances when the plantation of eucalyptus trees in semi-arid watersheds actually decreased the recharge from rainfall due to high rates of transpiration by eucalyptus. In semi-arid regions, eucalyptus may virtually prevent recharge from rainfall. Local species of hardy trees, bushes and grass, with very low water requirement, have been found to be more suitable for forestation.

39.3 ARTIFICIAL RECHARGE AUGMENTATION

Activities for artificial recharge augmentation in Basaltic terrain of western India are mainly undertaken during the monsoon season (June to September). A few of them also continued into the following dry winter season (October to February), using the stream-flow in winter season or the stored runoff water as the source of water for recharge. These activities are more site-specific compared to watershed development activities, which are spread over the whole watershed. In the Basaltic region, percolation tanks are in vogue for the past three decades and are dealt with separately in the next section. The aim of all these activities is to increase the residence time of water in the watershed, from a few days as surface runoff water to several months as ground water. The usual basic considerations given below apply to all these activities:

1) The quality, quantity and timing of water available for recharge. (This quality has to be similar or better than the quality of native water in the aquifer.)
2) The time of residence in the aquifer, chemical reactions (if any) with aquifer material and with native water, and improvement in water quality (if any).
3) The quality, quantity and timing of water pumped out for irrigational or domestic use. Some of the non-conventional means of artificial recharge augmentation being recently introduced are as follows:

   - To blast several shallow bore wells, filled with sand, around a low yielding dug well in summer season and create an artificially fractured aquifer, which eventually gets recharged during the monsoon rains. This practice is followed for water supply to very small villages perched on high hills.
   - To arrest the natural outflow of groundwater by putting underground barriers of clay or corrugated PVC sheets across the bed of a stream. The increased level of water table behind the barrier causes influent seepage.
   - To dig a percolation pit, backfilled with brickbats, gravel and sand, around a dug well or a bore well. In these cases the casing pipe of the bore well has holes in the bottom.
   - To pump runoff water in rainy season from a stream, put it into a farm pond or settling tank and then into a dug well in the farm.
   - To pump silt-free runoff water in winter season from a stream, and put it back into the dug wells away from stream bank.
   - To catch roof-water during the rainy season, pass it through sand filter and put it in a bore well or dug well.
   - To remove clay and silt from the beds of old tanks in villages, so as to increase their capacity and foster better infiltration.
   - To use old stone quarries in water divide region for rainwater storage and then pump the stored water into dug wells for recharge.

39.4 ARTIFICIAL RECHARGE BY PERCOLATION TANKS

In order to achieve artificial recharge on a much larger scale, construction of percolation tanks is a widely practiced technique in Basaltic terrain, especially in semi-arid zone. Due to the high evaporation rates of surface water in the summer months, its storage in groundwater reservoir is a preferred method. In order to augment groundwater storage, runoff water in several seasonal streams in a large watershed is impounded by constructing earthen bunds across the streams. Percolation tanks are formed during the monsoon season, behind such bunds, collecting runoff water from catchments. A typical percolation tank has an earthen bund of 6 to 8 meters height, 50 to 150 m length. Its catchment area is of about 10 to 50 sq. kms. A side waste weir with its top about a meter below the top of bund is provided for discharging excess runoff. Ideally, the water stored in the tank during the Monsoon rainy season (June–September) should percolate within first 3 to 4 months of the dry winter season (October to January) so that the shallow water body is not exposed to excessive evaporation rates in summer months. Ideally a percolation tank should therefore be dry by February–March. When the runoff water collected in the tank percolates to join the phreatic water table, its residence time in the stream valley changes from a few days to a few years, depending upon the thickness of phreatic aquifer. Drought years are frequent in the semi-arid region. It is, therefore, important to collect runoff whenever available, allow it to percolate and recharge the groundwater reservoir, so that in summer of a drought year people, crops, and cattle may depend upon groundwater available in dug wells and bored wells. Sustainability of these percolation Tanks assumes unique importance on this background. At places, where the quality of groundwater is impaired due to high salinity or high fluorides, it is possible to locate the drinking water well of a village on the downstream side of the percolation tank. Here, the quality of native groundwater gets improved due to its dilution by recharge from good quality water from the tank.

An important socio-economic factor, favoring construction of percolation tanks is its capacity to generate rural employment during a drought year. Construction of an earthen bund across a stream gives employment to about 500 to 1,000 men and women, for about 6 months. Government departments therefore take up these constructions as a part of 'drought-relief' programs'. They are useful for providing employment to the villagers and farmers, who have no other work on their farms due to drought conditions. The payment to the construction labors is made partly in cash and partly in food-grains. The cost of a typical work ranges between $ 100,000 up to a million dollars, depending upon the volume of earthwork, number of workers and site conditions.

Sustainable management of such percolation tanks is closely related to the survival of about 15 million farmers and an equal number of cattle, living in the semi-arid Basaltic plateau in western India. However, the silt accumulating in the tank bed, year after year, hampers the efficiency of a percolation tank. The beneficiary farmers have to de-silt the tank-bed when the tank dries in the summer season, in order to maintain the storage volume of the reservoir and also the rate of vertical infiltration. (Limaye & Limaye 1986). NGOs have an important role to play in this field and have come up with some of the best practices in maintenance of tanks and in active participation of the local people. Cooperative societies of fishermen and brick-makers are sometimes interested in maintenance of percolation tanks, although their members are not necessarily farmers. Fishermen wish to breed fish in the tank as long as it contains water. When the tank bed dries, the brick-makers are interested in removing the silt for brick-making.

39.5 CONCLUSIONS

UNESCO-IGCP Project no.523 has a direct bearing on the MDGs and on sustainable management of groundwater because its objective is to verify 'best practices' in groundwater projects through site visits and post them on Website for global dissemination and replication. In semi-arid hard rock regions in India, watershed management for augmentation of natural recharge and construction of percolation tanks for artificial recharge in post-monsoon dry period, are the time tested 'best practices'. Moreover,

- A well-managed watershed not only provides a resilient interface for absorbing variations of climatic pattern but also promotes groundwater recharge.
- In semi-arid Basaltic terrain in India, augmentation of recharge by natural and artificial techniques during and after the monsoon rains has a direct influence on providing irrigational and drinking water supply to millions of farmers, thereby reducing their hardship.
- Construction and maintenance of percolation tanks is of unique importance in augmenting dry-season recharge and in helping the survival of poor farmers.
- In an International Conference such as this, the IGCP Project no 523-"GROWNET" wishes to invite hydrogeologists having experience in best practices in groundwater development in low-income countries, for joining the "GROWNET" team as collaborators and enriching the "GROWNET" website.

ACKNOWLEDGEMENTS

The author would like to thank the reviewers for their thoughtful reviews and suggestions for improvement of the manuscript. Thanks are also due to IGCP Board for continued approval of GROWNET project, which is financed from a grant from UNESCO-IUGS.

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CHAPTER 40

Groundwater management in hard rock areas for sustainable agriculture in Indian subcontinent

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ABSTRACT: Groundwater is of special importance for sustainable development and improvement of agriculture and economic uplift of small and marginal farmers in the rural areas. Wise uses and management of groundwater for irrigation provide the farmer security and sustainability. It is all the more important in hard rock zones since the porosity of the rock is about 3%, the thickness of the weathered zone is about 3 to 20 m and only about 8 to 12% of the precipitation enter into the ground surface in soil pore spaces and form groundwater resources. In most of the hard rock areas especially in the southern peninsula, the groundwater is depleted due to over exploitation/mining and in Tamil Nadu, a few districts the water table has gone down beyond 300 m due to mining of groundwater. Under these circumstances, the groundwater management is very critical in hard rock areas for sustainable agriculture. This paper indicates in detail various groundwater augmentation methods like construction of percolation ponds, check dams, soil and water conservation measures, and water management practices for various crops including use of water saving method of irrigation in surface irrigation and introducing advanced methods like sprinkler and drip irrigation methods and selection of suitable crops based on the availability of water in the wells. If the suggestions are followed, it is possible to maintain groundwater level at reasonable depth and also in a sustained manner.

40.1 INTRODUCTION

India has a predominantly agricultural economy. Water and land are amongst its most valuable natural resources. Hence, groundwater is of special importance and interest in the development and improvement of agriculture and is of great significance in the economic uplift of small and marginal farmers in the rural community. Wise and correct development strategies and proper management of groundwater for irrigation offer the farmer security and confidence and release him from dependence on the unpredictable monsoon. The use of groundwater may extend for full irrigation supply in arid and semi-arid regions and seasonal supply in other areas to supplement irrigation in cases of insufficient or irregular rainfall. In recent years, groundwater has attained great importance particularly in its agricultural use. The planning, development and use of groundwater resource however, are one of the greatest challenges to users, experts and planners.

With the control of water in their hands, the framers have all the incentive to make the investments required for productive agriculture. Thus water from wells would constitute the basic input for providing the basic infrastructure on which the edifice of modern agriculture is being built in developing countries.

Though from time immemorial, groundwater has been used for drinking and irrigation purposes, a more rational and scientific approach to groundwater use was introduced only in the last century. The development of groundwater technology has been tremendous in the last 30–40 years. As a result of the generous policies of the Government of India in lending loans through co-operative

Table 40.1. Groundwater potential in India & Tamil Nadu (in million hectare meter MHM).

Details	India	Tamil Nadu
Potential (Total replenishable quantity)	43.20	2.23
Net draft	11.50	1.36
Percentage of Development	32	61 as on 1992 64 as on 1997
No of wells—both open & bore wells (Million)	16.00	1.80

Source: i) Central Water Commission (2000); ii) Govt. of Tamil Nadu, Chennai (1992); iii) Department of Evaluation and Applied Research, Govt. of Tamil Nadu (2000).

and other financial institutions, large-scale exploitation of groundwater is being done throughout the country. The groundwater potential, utilization and balance in India and Tamil Nadu are given in Table 40.1.

In this study, the data available are for 1997 and in the recent years the available groundwater is drastically reduced. It is estimated that the utilization of groundwater in Tamil Nadu is more than 80%. The groundwater level in some districts of Tamil Nadu, Gujarat has depleted very much and in Tamil Nadu it has gone beyond 223–287 m.

40.2 GROUNDWATER IN HARD ROCKS

The occurrence and distribution of groundwater depends on the characteristics of geological formations. In India, it may be broadly classified into three major hydrogeological units namely: i) hard rock areas, ii) alluvial terrain, iii) consolidated sediments (Posz et al. 1990). Over 70% of the area is covered by hard rocks, which contain less water. These crystalline formations have been formed at great depths in the earth's crest under conditions of high pressure and temperature, they are of Precambrian age and include some of the oldest igneous and metamorphic rocks. The rocks are highly folded and have very steep foliation dips. They have rather poor porosity and negligible permeability. However, small quantities of water accumulate and move along fractures, joints, faults and other channels in rocks. The thickness of weathered mantle on these rocks however, generally ranges between 3 to 20 m. These rocks hold and transmit only about 5% of total groundwater potential (Mayilswami 2006).

Shallow open wells are, therefore, preferred to obtain groundwater from these poor yield formations as they expose a greater surface area of the aquifer for infilteration and store fairly large supplies of water. To improve the water yield from such open wells, bores can be put at the bottom of these open wells both vertically down at the base or radially in the sides, so that the different point system can be penetrated over a larger spread of the formation to tap more water. It is difficult to make any precise quantitative assessment of the yield of water that can be obtained in hard rocks as the water yield of some formation differs from place to place in spite of modern technology like modeling for the assessment is available for hard rock areas.

In general the groundwater potential in hard rock is poor, though relatively high yields may be obtained in restricted location under favorable circumstances. Intensive explorative drilling in hard rock areas in parts of peninsular India have showed that the opening at greater depth becomes less pronounced and they are not favorable for movement of groundwater. Relatively higher yields from hard rocks are obtained within 40 to 50 m depth from surface. Optimum depth of drilling beyond which it is normally not warranted is about 100 m (Mayilswami 2006). But due to the demand and need of water for their livelihood, the farmers in remote areas, drill bore wells even beyond 350 m. In fact in Coimbatore/Dindigul districts of Tamil Nadu, India, farmers who have one to two acres of land, drill 3 or 4 bore wells in the last one decade to get water, but even then they are not able to irrigate more than 1/2 acre to 1 acre of land.

40.3 GROUNDWATER AUGUMENTATION

As the demand of water increasing in all sectors i.e., irrigation, industries and daily drinking purpose, and to maintain the groundwater levels, the government and individuals are going for artificial recharge. Natural recharge in hard rock areas are about 8–12% of the monsoon rainfall. Hence to increase the recharge even up to 20–25% artificial recharge methods are followed by the Government by implementing various watershed development programs. The government of India through its various departments and institutions namely department of land resources, agriculture, environment and forest, rural development, NABARD/CAPART etc. has launched the following watershed development programs under different names/programs.

- Integrated wasteland development programs (IWDP).
- Comprehensive watershed development (CWDP).
- Drought prone area programs (DPAP).
- Desert development program (DDP).
- Western Ghats development programme (WGDP).
- Hill area development program (HADP).
- National watershed development program in rainfed area (NAWDPRA).

Apart from this, the external funding agencies are also financing the development program on watershed basis in agricultural and forest watersheds. The following agencies are extending support in the program:

- Japan Bank for International Cooperation (JBIC) for forest development in Tamil Nadu and Karnataka.
- Department for International Development (DFID), UK Government
- Ford Foundation.
- USAID.
- Sida, Danida and others etc.
- Governments of Germany and Switzerland.

Irrespective of the source of funding for the watershed development programs, the main objectives are to use the land and water resources to get the maximum benefit with involvement of the people living in the watershed. The impacts of watershed development program under various schemes in different regions/states in India have indicated that:

- It helped in reversing the declining water table by raising the same;
- Reduced burden of the state government;
- It helps increased perenniality of wells, recuperation and irrigation areas;
- Soil and water conservation measures have definite impact on replenishment of groundwater;
- The study indicated that there is an increase of groundwater from 16% to 188%.

The impact of watershed development works namely construction of percolation pond, check dam, and soil conservation measures helped in augmenting groundwater in the wells in the watershed. One of the techniques for management of groundwater in hard rock terrain through artificial recharge is to build subsurface dykes and dams (membranes of cement concrete, plastic resins and even clay materials) across small catchments/drainage areas to arrest storm run-off and facilitate increased infiltration into the ground upstream of the membrane. The additional groundwater storage thus created may help utilization of the resource in a timely manner through open wells or bore wells. Indigenous attempts on such structures in selective catchments have been successful. In some advanced countries, fracturing of the rock through proliferated under ground nuclear detonation or through heavy blasting is planned to create favorable conditions for groundwater storage and timely discharge.

40.4 WATER MANAGEMENT IN HARD ROCK REGIONS

From the productivity point of view, water from wells offers several advantages. The flexibility in operation, dependability of the source, timing of water deliveries and low conveyance looses are all in its favor. In addition, the favorable seasonal water availability gives to the schemes served by well water a much wider choice in selection of high profit crops than for most surface water based schemes.

Once water is lifted and brought to the surface, its management becomes a very crucial/important factor. This water is of special importance and interest in the development of agriculture products. Further the cost of lifting groundwater is much more compared to cost of surface water. The magnitude of lowering of water table in Coimbatore District varies from place to place and it has been estimated that over a period of last 40 years, the average total lowering has been of the order of 16.5 m and after 1970, the depletion is about 1 m/year. The above mentioned factors clearly show that the water which is lifted from wells is to be applied very carefully in irrigation in order to get the maximum benefit. In order to maintain the water levels in the wells and to irrigate more acreage from the available water, the following water management techniques may be followed in these areas.

40.4.1 *Horizontal and angular drilling in hard rocks*

Horizontal drilling for revitalization of open wells/dry wells is quiet extensively followed especially to improve the discharge of wells located in crystalline & metamorphic rocks. In recent years, it is extended to tap water from hill slopes at highest elevation also both in soft and hard rock formation. Yet another method of augmenting water supply in the well is pumping many times in a day instead of one/two times as followed by farmers at present since it will increase the flow from the aquifer due to differences of hydraulic elevation.

40.4.2 *Prevention of channel losses*

Presently the lifted water is conveyed through open unlined earthen channels by most of the farmers, big or small. It is estimated that the conveyance loss is about 14–19% in garden land conditions and this water could be saved either by providing pre-fabricated cement concrete or soil cement channels or by conveying through low pressure underground pipe line systems. Necessary control and diversion structures for open lined channels and alfalfa valves for the underground pipe lines may be provided to avoid wastage of water. The cost of making these channels can be reduced by using farm labors during the lean season. Since the area of irrigation and the length of conveyance channels in case of lift irrigation is small, non-pressure pipes with necessary regulation and control structures can be safely used.

40.4.3 *Distribution and control of water*

There is no distribution and control structure in the farmer's fields. The irrigator does the distribution by his own judgment and controls it with his spade. Several types of structures are available to divert, distribute and control irrigation water in the farm, including water measuring devices.

40.4.4 *Land leveling and smoothing*

Farmers use irrigation systems designed to match their soil, water supply, climate, crops to be grown etc. in order to get an application efficiency of 75–80%. In order to achieve this, the important pre-requisite is land leveling and shaping. It has been found that by proper land leveling and shaping alone it is possible to save about 20–30% of water in addition to other advantages like labor saving, increased yield etc. Since more than 99% of water supplied to crop is under surface irrigation, land leveling and smoothing are important. Once the land is leveled and the water is conveyed for irrigation its application becomes important. There are different methods of application of water to garden land crops. Though the surface method is quite common, scientists have found out suitable techniques to save water without sacrificing the yields.

40.4.5 *Alternate and skip furrow/paired row irrigation methods*

Different methods of irrigation namely furrow, border strip, check basin are used for cotton cultivation Furrow method for cotton cultivation is the economical method and it is possible to save 20–30% of water without affecting yield. For row crops, with row spacing of 0.60 to 0.90 m, alternate and paired row irrigation method can be practiced. Experiments conducted on cotton crop have shown that by adopting skip furrow or paired row and alternate furrow, about 40% and 25–30% respectively could be saved. Further in the intervening space in the skip furrow method short duration pulse crops can be raised in the early period. These improved irrigation methods can be adopted for all row crops including sugarcane, vegetables, cotton etc.

40.4.6 *Ring basin method*

Ring basin method is commonly used for the cultivation of most of the fruit crops. The size of the basin used covers the entire area and is equal to spacing of the crops. Experience has shown that size of the basin for crops like banana, grapes and papaya can be reduced very much without affecting the yield thereby large quantity of water can be saved.

40.4.7 *Sprinkler irrigation*

This can be adopted when the land is undulating and the cost of leveling is prohibitive as well as for shallow and sandy soils. About 25 to 40 percent of water can be saved by this method that can be used for almost all crops except rice (Sivanappan 1998). The initial investment cost may be of the order of INR 6000–8000/acre. The area covered is very meager in the country i.e., about 0.60 to 0.7 M Ha.

40.4.8 *Drip irrigation*

This was introduced recently and it works at low pressure with main and lateral tubings. It can be adopted for vegetables, fruit crops and other crops which are planted in rows like cotton, sugarcane, banana or tapioca etc. It was found that only 50% of the water used in surface irrigation is sufficient to grow the crops with an yield increase of 30–100% (Sivanappan 1994). The cost of the system equipment comes to INR 8000–25,000/acre, depending on the spacing of crops, which is rather high for small and marginal farmers, but compared to water consumption efficiency in the field, and due to scarcity of water, this method can be adopted.

40.4.9 *Selection of crops*

The water requirements for different crops depend on the duration of the crop and the critical stages at which water should be applied. When water is a limiting factor, the tapping of the groundwater through pumps is to be made which assures more net return/unit of water/unit time/unit area. As water is costly and scarce commodity, judicious use of it in relation to the stage of the crop besides the selection of the crop to match the availability of the water resources should be practiced.

40.5 WATER MANAGEMENT IN MUNICIPALITIES AND INDUSTRIAL USE

Since more than 80% of available fresh water is used for agriculture (irrigation), management of irrigation water was detailed above. However there is a lot of scope in saving of water for municipal & industrial uses especially in the coming years, the allocation of water is going to be increased by 2 or 3 fold in these sectors.

At present, importance/attention should be paid on water supply for the activities discussed earlier. Once the water is brought to the municipality/industry, nobody cares whether it is used efficiently. The consumption of water in both cases may not exceed more than 20% and the balance is discharged as sewage water in the case of municipality and as effluent water in the case of industries to create pollution problem to the population. Utmost care should be taken to collect the wastewater and feed back it at least for irrigation purposes. This is being followed in different parts of the world, and not systematically in India, whereas water is reclaimed and used for municipalities and industries in advance countries as detailed below:

- In Israel all sewage water are collected at one place and after treatment, water is reused for cotton and other crops through drip method of irrigation.
- In USA (Los Angeles city) all the sewage waste water is collected and reclaimed, and recharged into the groundwater and the same is pumped for reuse in the same city.
- In most of the industries in Europe and developed countries, the effluent water from the industries are treated and reused in the same facility—drawing fresh water only to the extent it is consumed and not the full quota as in the case of Indian industries.

Further in the local municipalities and city corporation, it is estimated that about 25 to 35% of the fresh water brought from rivers or pumped from wells for community use is wasted through leakage in the pipes and taps and other flaws in the distribution net. By proper management of water as stated above and considering water as an economic goods, it is possible that the water scarcity problem in the country can be solved to some extent.

40.6 CONCLUSIONS

Water pollution problem is more complex than air pollution. But it is an important input for agricultural production. Groundwater is available not only in humid areas, but also in arid and semi-arid regions. Open wells and tube wells are constructed to tap water. Well irrigation is very popular in India, especially in the remote areas where canal or tank irrigation is not feasible. Millions of wells belong to small and marginal farmers. Due to the increase in construction and operating costs, it is difficult to build new wells or deepen the existing structures. Sixty percent of farm holdings belong to small and marginal farmers. With only about 50% of the area being under lift irrigation, the government could help in providing irrigating facilities in the form of large sized community wells wherever possible and help in economizing the water application through advanced methods of irrigation. The level of groundwater also should be maintained by suitable recharge practices so that small and marginal farmers can get maximum benefits from their smallholdings. There is great scope for developing wells wherever groundwater is available to bring new areas under irrigation which in turn will increase food production of the country and improve the living condition of rural community.

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Groundwater has become the most important source of water for domestic needs (e.g. drinking water), industrial, and agricultural water supply in the world and its use has increased manifolds. A substantial amount of groundwater is used indiscriminately to fulfill the demand in the agricultural sector, especially in regions with rather dry and/or semi-arid climate to enhance crop production for sustainable food supply. Most rural areas in both the developed and the developing world depend on groundwater sources for drinking purposes. Its high dependence will increase even further during the next decades due to severe limitations in the availability of reliable surface water volume and its continuous degradation in terms of quality. Thus, this important natural resource demands suitable management strategies for sustainable development.

The book is published as '*Special Publication 1*' of the International Society of Groundwater for Sustainable Development (ISGSD), forming a base for discussion and exchange of scientific ideas to identify future targets for research needed to improve the knowledge of groundwater resources development, management and protection. Written in straightforward English, the book would create interest among groundwater professionals, students, academicians, administrators, policy makers and executives, on the diverse problems associated with groundwater resources and to stimulate international cooperation on:

- developing methodologies for the assessment of groundwater resources and aquifer recharge
- understanding the environmental factors that may affect the groundwater quality as well as treatment;
- aspects of groundwater flow modeling and application in aquatic systems
- improved understanding of the coastal groundwater systems and their vulnerability due to natural disasters;
- dynamics of natural groundwater contaminants such as arsenic and fluoride from the aquifers through groundwater-to-food chain and their impact on the society
- groundwater management needed to prevent environmental health disasters.

ISGSD
International Society of Groundwater for Sustainable Development

ISBN 0415407761
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