Urban Groundwater Management and Sustainability

Edited by

John H. Tellam, Michael O. Rivett
and Rauf G. Israfilov

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PREFACE

With increasing awareness of environmental issues and the rapid expansion of urbanized areas, interest in urban groundwater has developed greatly over the last two decades. A number of meetings have been held on urban groundwater issues, for example the International Association of Hydrogeologists (IAH) Congress in 1997 (Chilton et al., 1997), and the urban groundwater session of the Geological Congress of 2004. The IAH has a commission on Urban Hydrogeology, and UNESCO has an Urban Water Management focal area under its IHP VI programme. Concomitantly, there has been a great interest in the wider development of the concepts of sustainability (e.g. Hiscock et al., 2002), including as applied to urban systems. It was in this context that a NATO Advanced Studies Institute (ASI) was held in Baku, Azerbaijan, in August 2004, following a very successful NATO Advanced Research Workshop at the same venue in 2001 (Howard and Israfilov, 2001). The ASI brought together, over a period of 10 days, a very diverse group of researchers from a wide range of countries interested in the whole gamut of urban groundwater issues. The aim was to share experiences and ideas, and to explore the commonality of urban groundwater problems, working towards methods for managing urban hydrological systems in ways that would minimize adverse impacts. This book is one product of the meeting. It covers all the main aspects of urban groundwater issues, from flow systems, through chemical and biological contamination, to engineering and socio-economic impacts, with evidence being drawn from many different countries and hydrogeological/socio-economic environments. It also includes, importantly, overviews which demonstrate how the different processes and impacts coexist and interact within any one city system.

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SECTION I:

INTRODUCTION

TOWARDS MANAGEMENT AND SUSTAINABLE DEVELOPMENT OF URBAN GROUNDWATER SYSTEMS

An Introduction

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Abstract: Urbanization modifies underlying groundwater systems. This may lead to adverse hydrological, water quality, geotechnical, or socio-economic effects, jeopardizing sustainability. To avoid these effects, management is required irrespective of whether the groundwater is to be used or not. This management must be based on a sound technical understanding of the interacting processes involved. The papers in the present volume explore the state of this understanding in the context of a wide range of countries, climates, and geologies.

Key words: importance of urban groundwater; flow; chemical water quality; biological water quality; remediation; engineering; socio-economic issues; water balance.

1. THE IMPORTANCE OF URBAN GROUNDWATER

Urbanization modifies local hydrology, often extensively. Water and chemical fluxes are changed in both surface and groundwater systems, often to the detriment of the environment and water usability.

Changes in land cover will often reduce recharge amounts and change recharge distributions. Recharge will be supplemented by leakage from water pipelines and sewers, discharges onto ground surface, and increased

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infiltration from surface water bodies. Groundwater abstraction will lower piezometric surfaces and reduce flows from the system. These changes in water balance will cause changes in solute fluxes: chemical fluxes will also be affected by dissolution from modified ('contaminated') land.

Depending on which factors dominate, possible implications of these changes include: reduced well and river yields; increased flood hazard; deterioration in quality of groundwater; salinization; poor quality baseflow; migration of polluted urban groundwater into surrounding rural areas; increased ground instability; and increased social conflict.

For a city to be hydrologically sustainable, an equilibrium needs to be established which is acceptable for drainage, flooding, waste disposal, and water supply. Such an equilibrium is unlikely to be established without active management intervention. Even if groundwater is not required for use, an increasingly unusual case, there are still potentially serious consequences of not managing the system actively: inaction has consequences. This is evident from the continuing impacts past un-managed development has had on both water levels and water quality in many cities, as exemplified by the papers in Section II of this volume.

Management of the interacting surface/groundwater system, considering both quantity and quality whilst satisfying other constraints of the built environment, is not a trivial undertaking. It is made more difficult by the transient nature and marked heterogeneity of urban development. Some forecasts suggest that urban populations will double by 2050 (Foster et al., 1999), and climate change will add further pressures: the development of management techniques is therefore urgently needed.

2. THE MAIN TECHNICAL HURDLES

Urban aquifers are hydrogeologically distinctive in at least the following respects:

- the density and heterogeneity of wells and pollution sources in space and time;
- the presence of structures such as sewers, fill material, foundations, pipelines, and various types of ground cover;
- the presence of chemicals such as synthetic industrial organic compounds (and any arising cocktails);
- the concentration of human-associated biological contaminants; and
- the socio-economic structures impacted by water-related issues.

Much remains to be learnt about how to quantify each of these components. However, the most difficult problem is how to integrate all the interacting hydraulic, chemical, biological, and socio-economic systems.

3. THIS VOLUME

The ultimate aim of urban groundwater studies is to develop the sound theoretical understanding necessary to develop management techniques which will deliver sustainability. This volume outlines the current state of understanding across the main areas of the subject, using both review and focused, specific-issue papers. The studies represented come from many different countries, including some for which the English language literature is sparse. This international diversity and the comprehensive range of urban groundwater issues covered are key features of the book. Although each city is unique, it is clear that many of the urban groundwater problems faced are common to many countries. As such, lessons learnt in, e.g., Azerbaijan may well be relevant in, e.g., Portugal, and *vice versa*.

The remainder of the book is divided into eight sections. Section II sets the scene with a series of case studies dealing with regional overviews. Sections III to VIII cover the following issues: flow; chemical water quality; biological water quality; remediation; engineering; and socio-economics. These areas are briefly outlined below.

4. IMPACTS ON GROUNDWATER FLOW: SECTION III

Flow issues include: recharge estimation; flow pattern assessment; integrated flow modelling; and characterizing the hydraulic behaviour of urban-specific structures. The latter includes the effects of foundations, tunnels, and any other buried structure, and the effects of fill material.

Urban recharge assessment is difficult. Low permeability cover promotes runoff. However, the cover will almost certainly be incomplete, encouraging focused recharge (Thomas and Tellam, 2006). In addition, cover may be modified to be permeable (SUDS: sustainable drainage systems). Evapotranspiration will be modified by the urban micro-climate and changes in the radiative properties of the cover material: if focused recharge occurs, evapotranspiration may be limited, and the frequently observed reduction in recharge rate during the hotter parts of the year may not be seen unless urban trees are present. Interflow in filled ground is extremely difficult to quantify. Leakage from water mains has often compensated for by the increase in runoff (e.g. Lerner, 1997), and leakage from canals can be so important that drainage wells are necessary (see Section VII). It is clear that new methods for determining recharge *in situ* would be particularly advantageous, and one such method is described in

Section III. The issue of uncertainty in modelling is also raised in the context of recharge estimation, but its salutary message should be considered in all aspects of assessment in urban hydrogeology.

If pollution risk is to be quantified, flow patterns need to be determined. This is particularly difficult in urban systems where, aside from the difficulties of estimating recharge, the history of pumping is typically complex in space and time. As shown in Section III, the well catchments need to be mapped all three space dimensions. Catchment shapes can become complex even in steady-state and certainly in unsteady-state. In most cases, detailed pumping histories are not available, and prediction becomes very uncertain.

Surface water and groundwater are part of the same system, yet it is much more usual to treat them separately. One reason is that integrated models are often difficult to develop, not least because of the differences in groundwater and surface water response times. However, progress is being made in this area, and an example of an integrated model run in real time to aid operational management is described in Section III.

5. IMPACTS ON CHEMICAL WATER QUALITY: SECTION IV

Many pollutant sources in urban areas are point sources and give rise to plumes which then move through the aquifer in paths often complicated by the time-variant nature of local abstractions. When sampled, often at pumping wells, the water is usually mixed, and may well have concentrations much lower than the maximum within the aquifer (e.g. Tellam and Thomas, 2002), thus making the interpretation of any *in situ* natural attenuation difficult.

In residential parts of a city, chemical pollutants may be limited to Cl, SO_4, and N species, with possibly some chlorination products (trihalomethanes) and occasionally other chemicals associated with minor spills. Cl and especially NO_3 may be associated in part with sewage pollution: presence will depend on the state/design of the sewerage system over time, and the residence time of the groundwater. In extreme cases, wastes could potentially transform the aquifer redox systems. However, in many cases industry will be a major cause of pollution, especially by synthetic organic compounds and inorganic pollutants (e.g. heavy metals, NH_4). The chemicals released will have varied over time as the industrial processes and land use changed (e.g. Lerner, 2003). Sources will vary from short-term, solute sources, to non aqueous phase liquid discharges which may act as sources for prolonged periods of time (Rivett et al., 2005).

Section IV includes examples of mainly residential and mainly industrial pollution from Portugal and the UK. But in addition, other papers provide reminders of a number of issues, including: the importance of recording the quality of the groundwaters before urbanization; pollution by less studied chemicals, including organic airport de-icers, pharmaceuticals, and actinides; the importance of urban landfills and city-adjacent agricultural areas; the presence of thermal waters in some cities; and the difficulty of predicting bypass pathways in systems as different as karstic limestones and clays.

6. IMPACTS ON BIOLOGICAL WATER QUALITY: SECTION V

Possible sources of microbiological pollution include human waste systems, animal wastes, food processing wastes, and waste water injection. Often it is assumed that groundwater contamination by bacteria and viruses in at least matrix-dominated systems can be dealt with in practice by use of 'setback' distances between potential sources and receptors. It is assumed that over this distance the microbes are inactivated by a variety of mechanisms, including predation, attachment to the rock, and breakdown. Recent evidence, summarized in Section V, in both developing and developed countries suggests that this may be a dangerous assumption, and interesting insights into apparently rapid virus movement and possibly extended times of viability are starting to emerge. Fractured systems have long been known to be vulnerable to microbiological pollution but, despite this, sometimes incidents involving groundwater transmission occur, even where management systems are in place: an important recent incident in Canada is described in Section V. Other papers compare microbe occurrence in countries with a range of waste-disposal practices.

Waste water injection is likely to become more popular as water resources become scarcer. Injected water will contain both microbial particles and potential nutrients, and care must be taken that the groundwater system is not degraded. A set of field experiments in a shallow sandy system, reported in Section V, shows that unsaturated zone thickness is critical if stimulation of groundwater bacterial populations is to be avoided. It also shows that such stimulation in turn affects the aquifer's indigenous invertebrate fauna.

7. REMEDIATION: SECTION VI

Section VI deals with remediation in the broadest sense of the term: aquifer protection; *in situ* remediation; and pumped water treatment. Prevention of pollutants entering an aquifer is generally the best approach, and, as indicated in one paper, in some cases waste treatment can even be potentially profitable. However, prevention is not always possible: there will always be cases where pollution occurs despite aquifer protection; in many aquifers there will be pollution present from times prior to the protection procedures being put in place; and in many cities it is simply not feasible to protect all the aquifer. In these cases, remediation can be considered, though often it will be an expensive option. There are many possibilities, from passive to vigorously interventionist, including monitored natural attenuation, permeable reactive barrier methods, pump-and-treat systems, and *in situ* chemical injection approaches. These are reviewed in Section VI from historical-, socio-economic-, and process-based perspectives. A final option is treatment at the point of use. This may involve simple procedures such as aeration or dilution, or rather more complex chemical treatments, an example of which is also described in detail in Section VI.

8. ENGINEERING IMPACTS: SECTION VII

There are numerous engineering impacts of urbanization, and many of these are covered in Section VII. Changes in water balance for an aquifer following urbanization can result in falling or rising water level depending on the relative changes in runoff, leakage, and abstraction. Falling water levels give rise to ground subsidence, especially in aquifer systems of alternating sands and clays. They can also result in the drying out and shrinking of soils or timber foundations, leading to subsidence of buildings. Rising water levels will result in greater pore pressures and hence lower effective stresses. In unfavourable cases, this will result in foundation collapse and landslide initiation. In extreme cases, seismic hazard may be increased. High water levels may result in salinization (through evaporation from the water table) and flooding, and require drainage systems to be installed. Another effect may be dissolution resulting in reduction of strength and building subsidence.

9. SOCIO-ECONOMIC IMPACTS: SECTION VIII

Even if all the technical problems were solved, success in sustainable development would still be contingent on finding satisfactory ways to implement any management strategies. In many countries, for example those of the European Union and North America, environmental legislation continues to be tightened. However, the experience summarized in the papers of Section VIII suggests that in many countries the situation is more complex, with governments unable and/or unwilling to enforce water control methods. This, and the sometimes inefficient performance of public water supply systems, has resulted in households taking more responsibility for developing their own supplies, often with an associated loss of central control over aquifer development. In other cases, this is not technically or economically feasible, and the population has to deal with low water availability. Often the key issue is to gain the agreement of the local populace, and non-governmental organizations are often heavily involved with raising awareness of water (and other) issues.

10. TOWARDS MANAGEMENT AND SUSTAINABLE DEVELOPMENT

The overall water and solute balances for an urban aquifer for a given time period can be expressed in the forms:

$$\begin{aligned} &P + L_s + L_f + D + SWI + GWI + AR \\ &- ET - RO - IS - Q - SWO - GWO - L_o = \Delta S \end{aligned} \tag{1}$$

$$\begin{aligned} &F_{ET} P C_P + L_s C_s + L_f C_f + DC_D + SWIC_{SWI} + GWIC_{GWI} + ARC_{AR} + mdr \\ &- QC_{GW} - SWOC_{SWO} - GWOC_{GW} - L_o C_{GW} = V \Delta C_{GW} \end{aligned} \tag{2}$$

where the symbols are as indicated in Table 1. All variables are functions of space and time, and many are not independent, being connected either directly (e.g. Q/GWI) or via socio-economic responses (e.g. if C_{GW} increases, Q may decrease). The problem faced in urban development is how to manipulate the balances represented by these equations so that abstraction, water levels, and water concentrations are optimized, i.e. do not give rise to unacceptable consequences (e.g. insufficient supply, flooding, subsidence, contaminated water). The variables in Equations (1) and (2) are generally not amenable to direct control: some of the management 'instruments' available to effect the manipulations needed are listed in Table 1. These instruments are subject to socio-economic constraints (e.g. Section VIII), and clearly cannot be retrospective - the history of urban development is a critical factor.

Table 1. Components of urban aquifer water balance, and potential management 'instruments' (regs = regulations; CL = contaminated land; HZ = hyporheic zone; Q = abstraction; SW = surface water; GW = groundwater).

Symbol	Variables	Potential management 'instruments'
Inflows		
P	Precipitation	Air quality; SUDS
RO	Runoff	SUDS; rainwater harvesting regs
IS	Interception storage	Rainwater harvesting regs
D	Industrial discharge	Restrict; landuse regs; education
L_s	Supply leakage	Repair (or allow?)
L_f	Foul leakage	Repair (or allow?)
SWI	Surface water inflow	Q regs; rely on HZ; SW discharge regs
GWI	Groundwater inflow	Q regs;
AR	Artificial recharge	Operational regs
mdr	Mass dissolution rate for CL	Fill regs; CL remediation; landuse regs; animal waste regs; Q regs
Outflows		
ET	Evapotranspiration	Q regs; vegetation regs; leakage repair
Q	Abstraction	Q regs; enforce GW use by industry
SWO	Surface water outflow	Rely on HZ; Q regs; well design regs;
GWO	Groundwater outflow	Rely on HZ; Q regs; well design regs;
L_o	Exfiltration to pipelines	Repair; Q regs
Other Symbols		
C_x	Concentration associated with x	
F_{ET}	ET concentration factor	
V	Aquifer volume	
ΔS	Change in water stored	
ΔC_{GW}	Change in concentration of groundwater	

There will be very different optimizations in different systems at different times, and research needs to concentrate on developing flexible procedures. To do this, two approaches are necessary: detailed analytical investigations of the subsystems where they can be isolated; and integrating, quantitative syntheses of several or all subsystems. As this volume suggests, much is being achieved by the former approach, and progress is being made in the latter. A significant challenge is the integration of the physical /biological /chemical systems with the socio-economic systems. Given the possible feedback relationships, there may well be interesting and important phenomena here which have not yet been fully explored.

In the long run, urban water systems by definition will be sustainable, but this could be at a high human cost. The purpose of urban research such as that summarized in this volume is to seek ways to minimize this cost.

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SECTION II:

REGIONAL OVERVIEWS

ANTHROPOGENIC CHANGES TO HYDROGEOLOGICAL CONDITIONS IN URBAN AREAS

New Perspectives from Azerbaijan

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Abstract: Urban water supply problems are not limited to depletion of reserves, but include issues such as surface and groundwater contamination, deterioration of distribution systems, water table rise or decline, landslides, land subsidence, and increased seismic hazard. These problems often have social implications that can lead to serious economic damage. Analysis of the situation associated with the use and protection of urban groundwater in Azerbaijan, and of the Absheron peninsula in particular (where in our view there are major issues to be resolved), suggests that special approaches are necessary within urban hydrogeology. Evaluation of both static and time variant factors has allowed the setting up of a hydrogeological zoning system which has been trialled in the Absheron Peninsula, with encouraging results. It is hoped that this approach will form a sound basis for prognostic evaluation in urban areas elsewhere.

Key words: protection of catchments; management; environmental assessment; Absheron Peninsula, Azerbaijan; classification; zoning; regional overview.

1. INTRODUCTION

The rapid and large-scale urbanization occurring in various regions of the world disturbs considerably the natural balance and often changes, the environment detrimentally. In these situations, a dynamic component of the environment, the groundwater, is open to damage if it is not protected from anthropogenic input. Results of analysis demonstrate that the main elements affecting the hydrodynamic and hydrochemical characteristics of

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groundwater beneath urban areas are seepage from underground pipes, infiltration of irrigation water, waste discharge, leakage from artificial reservoirs and tanks, and installations which block groundwater flow, such as building foundations. However, there also may be significant outflows such as groundwater production, underground withdrawal from drainage systems and subway discharge wells which partially offset the groundwater inflows.

At the same time due to the combined influence of several factors it is very hard (sometimes impossible) to predict changes to the hydrogeological system. In addition, it is impossible to foresee the damage to structures such as water, oil, and chemical pipelines, or to surface or underground construction works. As a consequence, forecasts are based on analysis of quantitative and qualitative trends. During the first stage of investigation, background data have to be collated. This can be very difficult sometimes due to absence of current and historical observation wells or data. Sometimes use has to be made of data from surrounding areas also, though such data are less reliable. Experience of investigations in Azerbaijan suggests that the results obtained from such collations usually enable successful assessment of the main regional characteristics of the groundwater regime (e.g. Israfilov, 1997; 2002). If observation wells are absent, any available information about the state and changes of hydrogeological conditions must be used. These may be published information from various reports, theses, monographs, or papers, archive data, data from fieldwork, and data from interrogation of well owners, information from remote sensing or aerial photography surveys, and geological / hydrogeological maps.

2. STRATEGY FOR STUDY OF URBAN GROUNDWATER SYSTEMS

2.1 Principles and Approach

Irrespective of the ways of obtaining the information, it is necessary to follow the requirements of a number of principles for developing a correct analysis:

1. In all cases, basic information on all aspects of geological / hydrogeological conditions should be collated;
2. These data should be studied together to give a full interpretation of the whole interdependent hydrogeological (and ideally ecological) system;

3. Seasonal and long term variation of the groundwater regime, including extreme events, should be determined and analysed.

Observance of these principles allows the basic regional characteristics of urban groundwater systems to be determined. These can then be used in forecasting future behaviour. The same approach should be applied at the next stage of investigations where anthropogenic loading is determined. In this case, consideration must be given both to the whole complex of the regime forming factors and to the development of the anthropogenic processes and phenomena.

Three types of groundwater regime formation are found within urban areas (Israfilov, 2002; Kovalevski, 1976):

1. natural and relatively undisturbed areas where annual seasonal fluctuations of groundwater level remain within historical ranges and water quality can be considered stable (the impacts on groundwater are episodic and low or insignificant;
2. abnormal areas where impacts on groundwater are apparent, but they do not cause long term water level changes, and hydrochemical indices range up or down;
3. severely or intensively abnormal areas where unnatural aquifer behaviour occurs, completely determined by anthropogenic factors; here, both short and long term trends in hydrogeological variables are likely to be observed.

At first sight these categories do not seem very important: however, they define the methods of investigation and prediction in urban hydrogeology studies. Depending on the characteristics of the natural setting, anthropogenic impacts, etc., each of these categories may be divided into further sub-categories.

The next basic result is that, independent of the multifarious nature of the impacts of the anthropogenic loading, urban water supply strategies are the main cause of regional changes of hydrogeological conditions. Depending on circumstances, there are three possible water supply strategies:

1. surface water only;
2. both surface and groundwater;
3. groundwater only.

It is clear that the third case will have the most effect on urban groundwater systems. The role of groundwater sources will constantly increase as groundwater is the source most protected from contamination and the most economical.

Location of wells mainly determines the character and variability of the degree of impact. Usually, within urban areas, private wells are located unsystematically, their location usually depending on the development of the city and its life support systems. Frequently, they are located for self-sufficient water supply of private apartment and business premises, other residential areas, and industrial complexes. Usually the public supply wells are located on the basis of special hydrogeological investigations, and after assessment of likely/required yields (hundreds of thousands of cubic metres per day usually). These wells become one of the main water sources (sometimes the only one) of the city water supply. But even in the cases where for any reason (ecological, economic, etc.) surface water is the only source, redistribution of water becomes the main reason for any changes in the groundwater balance.

2.2 Main Aims

The analysis and comparison of the current problems of hydrogeology in urban areas, urban agglomerates, and industrial centres (Chilton et al., 1997; Hiscock et al., 2002; Howard and Israfilov, 2002) allow the defining of the following main issues which should be investigated if a scientific basis of management and sustainable development of urban groundwater systems is to be developed:

1. Definition of spatio-temporal distributions of the groundwater regime parameters, identification of natural and anthropogenic sources of water, and definition of water-balances;
2. Identification and quantitative evaluation of factors which cause of long and sort term changes in hydrodynamic and hydrochemical parameters;
3. Creation of a conceptual model of the groundwater flow system, schematization of aquifer boundaries, and estimation of basic hydrogeological parameters;
4. Assessment of the limits of anthropogenic impacts on the groundwater and the development of predict methods for the definition of possible detrimental impacts on the groundwater and other parts of the environment;
5. Hydrogeological zoning of urban areas on the basis of groundwater conditions and the factors which may affect groundwater protection;
6. Development of principles for the optimal design of a network of observation wells that would provide more complete information on natural and anthropogenic impacts on groundwater at the regional and local scales;
7. Creation of interactive real time mathematical models of groundwater movement for use in management and sustainable development.

To achieve these goals, the following four main tasks should be undertaken.

Firstly, development of databases and GIS needs to be undertaken. The structure of the study database will be determined by the most important features of the local water system, and needs to be consistent with existing software. Periodic updating of the database is required after field and analytical work. The database should be populated, following critical assessment, with: (a) historical data from archives and publications; (b) recent geological, hydrogeological, and ecological data; (c) any special field survey data. A GIS should be created to describe the present condition of urban water resources for every city in the study area.

Secondly, the water supply system, groundwater sources, and any known problems should be examined. Analysis of the water supply strategy within different urban areas should be undertaken, including: (a) assessment of quality and quantity parameters for the public water supply systems; and (b) identification of alternative sources for private water supply systems and their role in the water consumption structure. Assessment of the water distribution system and sewer system conditions should also be undertaken. This will involve: (a) analysis of calculated and actual leakages from water distribution and sewer systems; and (b) determination of the performance of the water supply sources, using qualitative and quantitative indicators,, including their water quality before joining the public water distribution systems, and their associated water quality at points of consumption. An assessment should be made of the natural and anthropogenic impacts on groundwater using an analysis of seasonal and long-term variations of hydraulic and hydrochemical parameters. This will include assessment of: (a) variation in the natural groundwater regime; (b) variation of impacted groundwater regimes; and (c) the characteristics of groundwater regime changes. Finally, comparative studies of the characteristics of the water systems and problems in case urban areas need to be undertaken. This will involve: (a) definition of the general problems and characteristics of groundwater flow and quality changes for the case areas; (b) characterizing the various impacting processes (e.g. contamination of groundwater, flooding, landslides, land subsidence); and (c) classification of the changes in hydrogeology and the dominant processes in each case urban area.

Thirdly, development of a conceptual urban water model needs to be undertaken. This process will usually include: (a) definition of the groundwater balance structure; (b) justification of the estimated hydrogeological parameters; and (c) schematization of the aquifer boundary conditions.

Fourthly, and finally, a model should be developed and assessed: (a) chose a justifiable model type; (b) calibrate the model using the available data; (c) assess the model adequacy; and (d) evaluate modelling results and reliability.

Although the structure of the research presented above can be changed and enlarged depending on specific natural and anthropogenic conditions obtaining in the urban areas being investigated, the main components may form a basis for development of a common approach and methodology for urban water investigations. The rest of this paper concerns application of these principles as carried out in Azerbaijan: urban water problems are present here in all forms.

3. URBAN WATER PROBLEMS IN AZERBAIJAN

As in most countries of the world the majority of the population of the Azerbaijan Republic (roughly 70%) lives in cities. As a rule, the process of urbanization is followed by intensive human impact on the environment, the latter including surface water and groundwater. It usually leads to land subsidence, landslide processes, flooding, and other environmental and social phenomena, which cause huge economic dislocations. At the same time, in most cities of the Republic one can observe a serious worsening in portable water quality. This can cause serious damage to public health and is the main reason for the spread of preventable diseases. Moreover, together with contamination of the main sources of surface water and groundwater supplies, one can observe frequent cases of water contamination in the water distribution systems. This situation is complicated by the fact that after the disintegration of USSR and occupation by Armenia of about 20% of the territory of Azerbaijan, the population of the Absheron peninsula has increased dramatically because of refugee inflows and forced migration, to over 4 million (more than doubled). The cities of Baku and Sumgayit are located within the Absheron peninsula. In this peninsula there are almost no fresh water resources. The water demands are met by surface water resources of the Samur river and groundwater transfer from the Guba-Gusar region of the Republic (more than 200 km from the demand areas). These sources are augmented with water from the Kura river in the Ali-Bayramli region (more than 180 km from the demand area), at rates of up to 30 m^3/s. The available water resources were not intended to meet the needs of 4 million people, this being the reason for the substantial deficit in water supply. At the same time, the existing state of water supply infrastructure and inter-urban water supply transmission has worsened the potable water quality. Often the population is supplied by potable water which does not meet the generally accepted quality standards. In many regions of Baku and Sumgayit, city water is supplied for only a few hours a day by a rotating schedule that is inadequate for sanitary-epidemiological requirements. The paradox is that the existing freshwater deficit is largely due to the poor state of the distribution network with leakages of over 50%. Due to this loss of water, many people in the suburban regions of Baku city and the surrounding Absheron peninsula, in the areas where there is no centralized sewer system, use untreated groundwater for their domestic needs. This uncontrolled use of potentially polluted groundwater is fraught with serious consequences.

Gyanja city, the second most populous urban area with a population of over 500,000 and considerable commercial potential, is in the western part of the Republic. This city supplies water only to 50% of its people from a centralized water supply network with a total production of 2.65 m^3/s. The sources of the water supply are baseflow from the Gyanjachai river and surface waters of the Kura, Agsu, and Kyapazu rivers. The private, decentralized water supply is abstracted using over 150 wells located in a disorderly distribution in different parts of the city. It should be pointed out that none of the wells have any protection zones, and that the region is not completely served by the sewer system. Thus, sources of water supply are under constant threat of contamination and in some cases do not meet sanitary norms. Taking into account the low level of water treatment and water transmission to populations through existing urban water supply networks, in Gyanja city water quality also does not meet potable standards.

The populations of other cities in the Republic are less than 100,000. However, this does not mean that the state of the water supply is better. These cities are only partially served by centralized water supply and sewage systems. To satisfy their water needs most people use sources of surface water and groundwater that are of low quality. On the whole, portable water supply in these cities does not meet water quality standards.

Therefore, one of the main problems in the Azerbaijan Republic is management and sustainable development of urban water systems. Addressing these issues would reduce the risk and extent of diseases significantly for 70% of the population. As a reminder, about 50% of the population of Azerbaijan Republic lives in the Absheron peninsula. In addition, many industries including oil and gas, chemical, mining, and electronics are developed here. In less than a hundred years, the Absheron peninsula has become one of the most urbanized and industrial areas of the world. Therefore, using the example of this region, the results of urban loading upon groundwater will be described.

4. AZERBAIJAN BACKGROUND

4.1 Introduction

Azerbaijan is situated within the Alpine fold belt and includes mountain regions of the Greater and the Lesser Caucasus, the Kura inter-mountain depression and part of the Caspian Sea. Topographic relief varies from -26 to 4459 m above mean sea level (MSL). The altitude of more than 45% of the country (86,600 km^2) is above 500m MSL. The geology ranges from Precambrian rocks to recent materials. Sediments are marine, volcanogenic and continental in origin. Azerbaijan has common borders with Armenia, Georgia, Iran, Russia, and Turkey, and maritime boundaries with Iran, Kazakhstan, Russia, and Turkmenistan.

4.2 Natural Setting of the Case study Area

The Absheron peninsula is located in the east of the Azerbaijan Republic on the west coast of the Caspian Sea (Figure 1). It has an area of 1984 km^2 and is underlain by Cretaceous , Palaeogene, Neogene, Pliocene, and Quaternary deposits. The lithological composition of the sequence up to the base of the Neogene consists of sands, clays, sandy clays, and limestones.

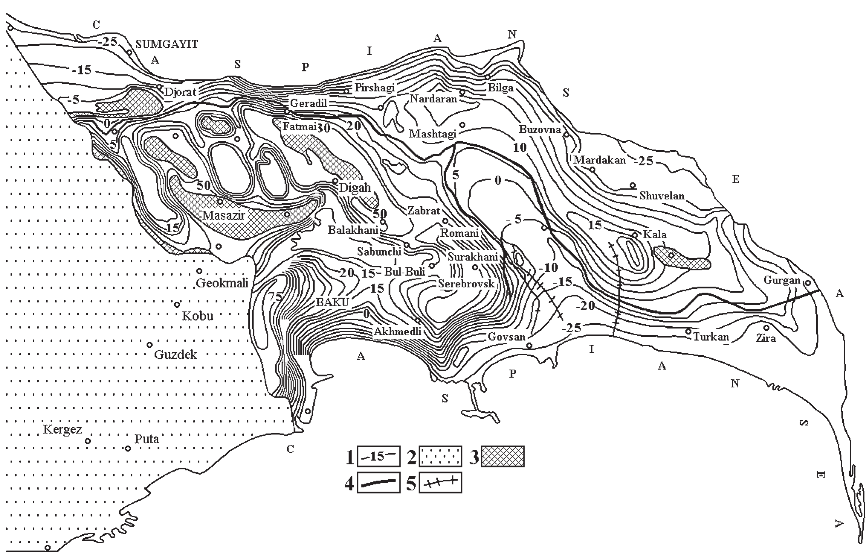

Figure 1. Map of water table contours for the Absheron peninsula for 1999; 1-water table contours; 2 - region where there is only sporadic development of the aquifer; 3 - impermeable deposits; 4- irrigation Canal; 5- channels for waste oilfield waters.

The middle Pliocene sediments are represented by sandy clays, thick where the highly mineralized oil groundwaters occur. The overlying deposits contain lower salinity though rarely fresh groundwater which is cut off from the underlying sequence by thick low permeability clays (200 - 400 m) of the Akchagil and the Lower-Absheron stages (layers). In the west of the Absheron peninsula, lower permeability clays of Cretaceous, Palaeogene, Neogene, and Lower-Pliocene age outcrop. Towards the east coast of the peninsula, these clays become deeper and are covered by Quaternary sediments where the main deposits of shallow, confined and unconfined groundwater occurs. Thus the geological structure determines the isolation of the Absheron peninsula Quaternary deposits from the Great Caucasus Mountains, in spite of the fact that tectonically the peninsula is its south-east continuation. Only the eastern part of the peninsula is interesting from the point of view of groundwater resources.

Tectonically, the Absheron peninsula is situated in the East-Absheron depression where seven sub-structures are identified: 1. Djorat-Sabunchi-Zikh anticline; 2. Kyurdakhani-Kala- Zirya anticline; 3.Bilgyah-Mardakyani-Shuvalan syncline; 4. Zirya syncline; 5. Bina-Hovsan syncline; 6. Baku syncline; 7. Sumgayit syncline (see Figure 4 below).

The climate of the Absheron peninsula is characterized as arid with a dry summer. Winter is warm but precipitation is more than in the other seasons (rarely as snow). However, the most intensive precipitation occurs in the autumn. Typical seasonal variations of precipitation, temperature, and air humidity are shown in Figure 2.

Hydrogeological conditions within the eastern part of the Absheron peninsula are quite diverse. The aquifer is subdivided into a shallow unconfined part and two confined parts. Shallow groundwater occurs throughout the area, predominantly in the Quaternary sediments. Confined groundwater occurs within the Quaternary sediments within the Zirya syncline, the Bina-Hovsan syncline, and the Baku syncline. Confined groundwater also occurs within the Neogene sediments (Absheron stage) within the Bilgyah-Mardakyani-Shuvalan syncline. Because of the low quality of the confined groundwater, which not used for drinking, the main importance of the urbanization influences its effect on the shallow groundwater, and this is the main focus of the study.

Hydrogeological maps of scale 1:25000 provide complete information about the origin, occurrence, and movements of the groundwater. Some of these data are shown in Figures 1 and 3.

The depth to the water table varies from several centimetres up to 50 and more metres. Heads vary from +110 to -25 masl (metres above sea level). The vast majority of groundwater discharge takes place into the Caspian Sea. Within the tectonic structures mentioned above, different groundwater flow systems are developed. Groundwater drainage has been estimated based on values of hydraulic gradient (Katz, 1976). Four categories were identified: 1) intensively drained with hydraulic gradient > 0.01; 2) moderately drained (0.005 to 0.01); 3) weakly drained (0.001 to 0.005) and 4) poorly drained (<0.001).

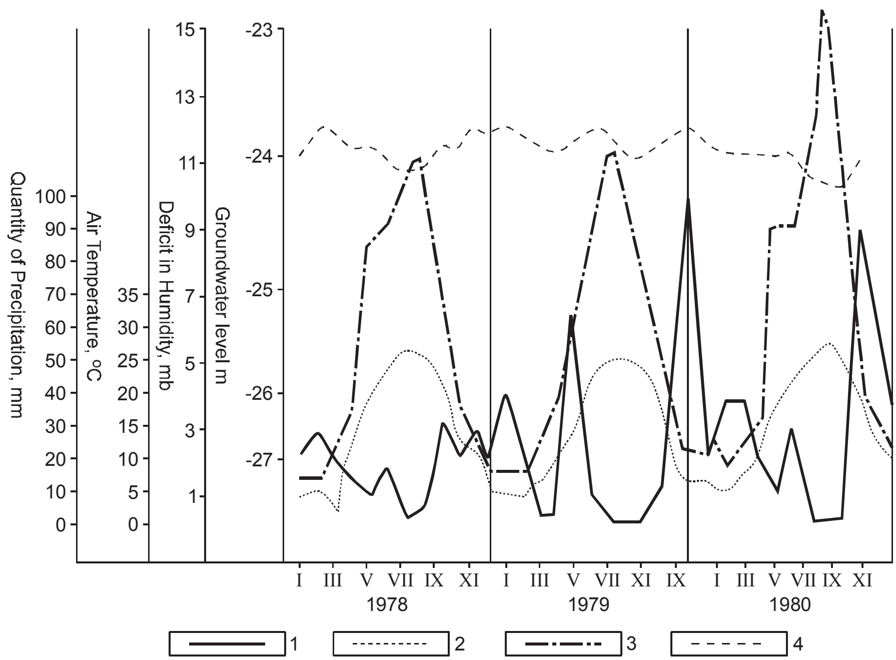

Figure 2. Relationship between groundwater level at well 161 and meteorological conditions in Baku; 1. precipitation; 2. air temperature; 3. humidity deficit ; 4. groundwater level.

Due to variations in the hydrogeological settings, groundwater mineralization varies widely, with salinities ranging from less than 1.0 g/l to more than 60 g/l. As the groundwater mineralization increases, the chemical composition changes from HCO_3-Ca/Na and Ca/Mg through HCO_3-Cl and Cl-HCO_3 to Cl-Na.

5. IMPACTS OF URBANIZATION

5.1 Water Levels, Water Balances, and Time Trends

Hydrogeological data from 137 observation wells collected over the period 1955-1999, and other data, have been compiled in annual reports and summary technical reports, monographs, and papers of the Ecology Ministry of Azerbaijan, the Geology Institute, the Research Institute of Water Problems, Baku State University, etc. (Shishkin, 1939; Israfilov, 1947; Vaidov, 1956; Israfilov and Listengarten, 1978) The interpretation was approached through a complex analysis of geological and hydrogeological data using hydrodynamic, probabilistic, statistical, and water balance techniques.

As in the majority urbanized areas, the groundwater formation within the peninsula depends on the interactions of two complexes of factors - natural and anthropogenic. Therefore without studying the natural groundwater regime it is impossible to assess the impacts of urbanization. Differences in the groundwater regime across the area indicate the relative importance of the various geological-geomorphological, climatic, and hydrological factors.

As mentioned above, the geological distribution is the main reason for the isolated fresh groundwater systems in the Absheron peninsula, as well as for the origin, occurrence, and movement of the groundwater within the deeper tectonic structures.

Natural long-term groundwater regimes in the Absheron peninsula were found to be influenced by both hydrological (Caspian Sea) and meteorological factors. Seasonal variations result in a gradual rise of groundwater level after a minimum in September/October, reaching a maximum in March/April (Figure 2).

More detailed descriptions of the characteristics of the groundwater regime within the Absheron peninsula are available elsewhere [e.g. Israfilov, 2002; Israfilov and Listengarten 1978; Samedov, 1976) This includes research which has been undertaken to determine the possible natural long term trends in groundwater hydrodynamical parameters by the special technique of the World Meteorological Organization. The results have confirmed once again that all trends are related to the impacts of anthropogenic factors only.

The Absheron peninsula is highly populated (4.12 million inhabitants in 2 cities and 83 settlements), and at the same time is one of the agricultural regions, and the main oil-gas-producing region of Azerbaijan. The oil-gas field exploration began here about 150 years ago. In those times, only about 15,000 inhabitants lived here, half of them in Baku. Due to the extreme lack of fresh water resources (both the surface water and groundwater), it had not previously been possible to develop any kind of agricultural or industrial production. But from the beginning of the oil-gas exploitation and the associated influx of local and foreign investments, the Absheron peninsula for about 100 years became one of the most urbanized, industrial, and overpopulated areas in the world. The lack of water resources in the

Absheron peninsula is compensated by water imported from other regions of Azerbaijan.

In 1893, the population of the Absheron peninsula used about 1000 wells for water supply, of which only about 150 yielded fresh water. In 1893, the first desalination plant to use Caspian Sea water was constructed, and provided about 370 m^3/day: in 1908, a second plant with a capacity of about 1100 m^3/day was constructed. From 1908, about 2500 m^3/day of water from the Kura river was consumed by steamships. Between 1917 and 1988, five water pipelines, the Samur-Absheron irrigation canal (about 9.0 m^3/s) and the Jeiranbatan water reservoir (in the northwest of peninsula - useful capacity 95 million m^3) were constructed to convey water from north-east Azerbaijan. The water pipelines supplied the cities Baku, Sumgayit, and nearby settlements with 12.5 m^3/s, 4.0 m^3/s and 3.5 m^3/s of potable water respectively. Industry draws water from the Caspian Sea at a rate of 8.7 m^3/s. About 1.3 m^3/s of groundwater is produced from wells to meet industrial demand, to irrigate, etc. The large amount of imported water is the cause of the significant changes in the hydrogeological conditions of the Absheron peninsula. The anthropogenic impacts have caused both groundwater level changes and pollution. The groundwater balance for the Absheron peninsula for 1999 is shown in Table 1.

Table 1. Groundwater balance.

Inflow	Thousand m^3/y	mm/y	%	Outflow	Thousand m^3/y	mm/y	%
Precipitation	38900	39	11	Evaporation	22806	23	8
Condensation	10984	11	3	Abstraction by wells	29191	29	10
Irrigation	38135	38	11	Flow to lakes	1603	2	1
Underground pipe leakage	228005	227	66	Flow to the sea	227959	227	81
Oil field waste water infiltration	23441	23	7				
Flow from confined aquifers	7096	7	2				
Total	346559	345	100	Total	281560	280	100

The net accumulation of groundwater in the Absheron peninsula is 65 x 10^6 m^3/year and this is the cause of the changes to the hydrogeological systems.

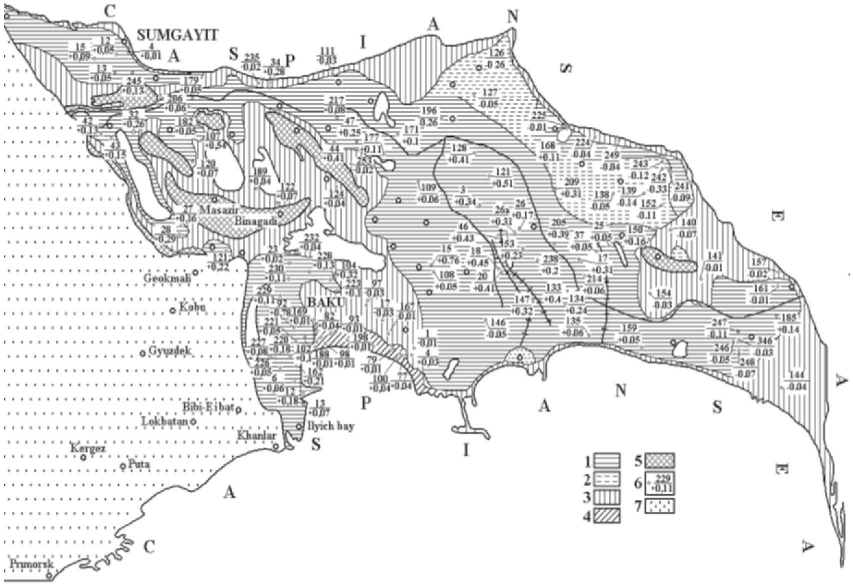

Figure 3. Map of water table trends. 1 Increasing levels; 2 decreasing levels; 3 ~ stable; 4 areas with artificially supported stable long-term level; 5 impermeable deposits; 6 observation wells, with number (numerator) and trend value, m/year (denominator); 7 region where there is only sporadic development of the aquifer.

The dominating factors which affect the groundwater beneath different areas of the Absheron peninsula are infiltration from underground pipes, irrigation, the Absheron Canal, oil field waste waters, and meteorological and hydrologic influences (Figures 4, 5). Across nearly the whole area, these factors influence the groundwater systems in various combinations, eight main combinations having been identified. So, on the maps of Figures 4 and 5 they are presented according to the prevailing factor. The scale and magnitude of the effect of each factor is not the same, and it is hard to define the degree of influence of each factor separately. Thus, for quantitative assessment of the groundwater response to the current loading, trends were examined, linear and non-linear regression calculations being undertaken on all 137 long-term groundwater level data sets. The results have been compiled into a map of groundwater level trend distributions shown in Figure 3. The values of the trends vary considerably, from -0.33 m/year to +0.78 m/year. This indicates the complicated nature of the current processes.

5.2 Hydrogeological Zoning

The hydrogeological zoning of the Absheron peninsula has been attempted by several researchers for various purposes and using different principles. In 1947 a zoning scheme for the peninsula for the purpose of water supply was set up by H. Yu. Israfilov. This scheme was based on the genetic type of groundwater regime, the stratigraphic characteristics of aquifers, and their resources (Israfilov, 1947). With the same purpose, Listengarten made a new zoning on the basis of the lithological characteristics of the aquifers, the depth of occurrence of the groundwater, and its chemical composition (Israfilov and Listengarten, 1978). On the basis of regime types, B.M. Samedov erected a zoning scheme for land-reclamation purposes (Samedov, 1976). The recent studies of the hydrogeological setting of the Absheron urbanized area have allowed a new scheme of hydrogeological zoning to be devised, with the purpose of helping assess groundwater protection, management, and sustainable development (Figures 4 and 5).

The defined taxonomic units characterize the following features:

- - the first order of the classification is characterized by various geological structures, difference in relief, groundwater recharge processes, transit and discharging areas.
- - the second order is characterized by various physical-chemical properties of aquifer;
- - the third order is characterized by various values of hydraulic gradient;
- - the fourth order is characterized by various types of groundwater occurrence;
- - the fifth order is characterized by various areas of normal and abnormal conditions of groundwater formation;
- - the sixth order is characterized by types of groundwater regime formation.

In addition to the abovementioned purposes and depending on the hydrogeological setting, the map developed from applying the zoning scheme can be used while assessing likely hydrogeological changes, for land-reclamation purposes, for groundwater resource assessment, for hydrogeological process modelling, for planning monitoring schemes, etc.

Figure 4. Map of hydrogeological zoning of the western end of the Absheron peninsula. For legend, see Table 2.

Figure 5. Map of hydrogeological zoning of the eastern end of the Absheron peninsula. For legend, see Table 2.

Table 2. Classification system and legend for Figures 4 and 5.

Order	Div	
1st Tectonic Provinces	**1**	I: Djorat-Sabunchi-Zikh anticline; II: Kurdakhani-Kala-Zirya anticline; III: Bilgyah- Mardakyan-Shuvalan syncline. IV: Zirya synclineV: Bina-Hovsan syncline. VI: Baku syncline. VII: Sumgayit syncline. VIII: Coastal region
2nd Lithology	**2**	Clays
	3	Loam
	4	Sandy loam
	5	Sand
	6	Sand with shingle
	7	Shelly sand
	8	Shelly sand with shingle
	9	Shingle
	10	Alternation of clay
	11	Limestone
	12	Limestone with shelly sand
	13	Limestone-shelly sand with shingle
	14	Borders zone of tectonic structure development
	15	Borders zone of different lithological structure
3rd Drainage	**16**	Intensively drained with hydraulic gradient > 0.01
	17	Moderately drained with hydraulic gradient 0.005-0.01
	18	Weakly drained with hydraulic gradient 0.001-0.005
	19	Poorly drained zone with hydraulic gradient < 0-0.001
4th Water Table Depth	**20**	a) < 2 m
	21	b) 2 – 5 m
	22	c) > 5 m
	23	Borders zone of various drainage
	24	Groundwater flow directions
5th Natural/ Anthropogen	**25**	A - natural factors dominant
	26	B – anthropogenic factors dominant
6th Dominant Impact	**27**	Natural – meteorological
	28	Natural-hydrological
	29	Infiltration of irrigation
	30	Leakage from underground pipelines
	31	Artificial underground withdraw of flooding areas
	32	Infiltration from canal
	33	Production
	34	Oil field waste waters
	35	Border between zones of natural & anthropogenic factor dominance
	36	Gutters of waste oil field water
	37	Irrigation canal

6. CONCLUSIONS

In urbanized areas there are many types of influence on groundwater systems. The specific features of urban loadings (natural and anthropogenic) can be described by the following characteristics: multifactor; varying effects on either or both recharge and discharge: irregularity in time and space; concentration within a limited area.

The character, rates, and degree of change in hydrogeological conditions are as much directly dependent on the joint action of the loads imposed by anthropogenic influences as they are on the specific characteristics of the geological, hydrogeological, and hydrological conditions. Irrespective of features of the natural system, the presence of anthropogenic influences on groundwater results in the development of deleterious effects within the majority of urban areas. For the study of urban hydrogeology, it is necessary to develop and use special strategies and methods if the aims of management and sustainable development of urban water systems are to be achieved.

For developing and performing scientifically justified management and sustainable development of urban water systems, it is necessary to obtain as complete a set as possible of accurate information and data on the initial parameters listed above. This information can be presented in a more universal, rational, and informative way by using hydrogeological zoning techniques. These should be based on the characteristics of the tectonic development, lithological make up of the aquifer and vadose zone, the depth of the groundwater table, drainage, the characteristics of the changes in groundwater properties, etc. Such zoning allows assessment of the current groundwater condition, and should be applied to solve prognostic tasks, including the determination of what anthropogenic activities may be allowed. Results of these studies together with worldwide knowledge of urban groundwater problems allow important areas of further research to be identified. They include:

- development of principles for the optimal design of observation networks in urban areas that would provide complete information on natural and anthropogenic impacts on groundwater at regional and local levels;
- development and improvement of methods for assessing impacts of natural and anthropogenic influences on groundwater conditions;
- construction of universal (multi-purpose) interactive real-time mathematical models of groundwater movement in the urban areas to predict changes in hydrogeological conditions and limit anthropogenic impacts on the environment;
- development of improved methods for predicting changes of hydrogeological conditions; and
- development of improved understanding of the relationship between hydrogeology and environmental change, economics, and potential ecological disasters.

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DEMANDS ON, CONDITION, AND ENVIRONMENTAL PROBLEMS OF THE BAKU MUNICIPAL WATER SUPPLY

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Abstract: The historical development of water supply sources for the populations of Baku and the Absheron peninsula are described and their water-quality status indicated. The area is an intensely populated part of Azerbaijan subject to the pressures of the urbanization and increasing population densities. As such there are significant pressures on existing water resources and a need to identify supplementary supplies from groundwater sources. It is proposed that future supplies should come from the good quality groundwaters within the Quaternary deposits of the Greater Caucasus foothills of the Samur-Devechi plain. A further possible supply is from the Ganyh-Agrichai Plain.

Key words: Baku, Azerbaijan; public water supply; pipeline; canal; spring; quality; potable water; contamination; imported water.

1. INTRODUCTION

Baku city is the capital of the Azerbaijan Republic. It is a very urbanized area with an urban population of some 1.9 million (excluding refuges). It is a significant industrial, scientific, and cultural centre, a major west coast port on the Caspian Sea, and the Silk Way gate to Europe. The city is located in the southern central part of the Absheron peninsula, in a natural terraced amphitheatre of 250 km^2. From a water resource and supply point of view, it is reasonably located with freshwater resources potentially sufficient to serve not only the population of Baku but also the surrounding Absheron peninsula agricultural sector. There are, however, significant urban and expanding population pressures upon the water resources present

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and there is an ever present need to protect the currently used resources as well as identify new resources (Israfilov, 1998).

The aim of this paper is to describe the water resources and supplies currently available to the city and its surroundings, and to identify supplementary supplies that are currently under-utilized but have capacity to serve as a future, sustainable supply.

2. HYDROGEOLOGICAL SETTING

Natural conditions such as the absence of rivers, marine sedimentary deposits in which clayey facies predominate, low annual rainfall (180-280 mm/year), high potential evaporation (1200 mm/year), and complex geological structure militate against the formation of a sustainable fresh groundwater resource. This is true for both the Baku trough and the wider Absheron peninsula. Generally, potable groundwaters with mineralization (total dissolved solids, TDS) less than 1 g/l are absent. Moderately mineralized groundwaters (1-3 g/l) suitable for some industrial/agricultural needs are distributed sporadically in the east of the peninsula (Figure 1). In the Quaternary sedimentary rocks of Baku, particularly in the western part of the peninsula, groundwater supplies are sporadically distributed. Only in a small area of the Guzdek trough are the groundwaters fresh; their potential does not exceed 240,000 m^3/day. Confined groundwaters are highly mineralized, i.e. saline, throughout the area.

3. HISTORICAL WATER SUPPLY DEVELOPMENT

The primary source of water supply to the Absheron peninsula and Baku city [Icheri Sheher (“Inner City”)] population before the oil boom was the fresh groundwaters. Their exploitation was largely “manual” with small local-supply wells and “ovdans”. They originally numbered 800-900 in the Baku area compared with 25-30,000 on the peninsula at present.

In 1883, about 12,200 people lived in Baku. The growing population led to a need to locate and develop further sources of water supply. The existing local water quality deteriorated due to general mineralization increases and bacteriological contamination due to poor sanitation and effluent disposal. Water shortages were increasingly apparent. Initially, some of the population used sea water desalinated at Bailov by “Arthur Coppel Co.”. The daily volume of this supply was less than 10,000 litres. Further, their quality failed to meet the requirements of the “Potable Water” standard with ferric oxide typically precipitating out. Other private water-desalination companies also existed: however, because of their high price and dubious quality their water was not made available to all city residents.

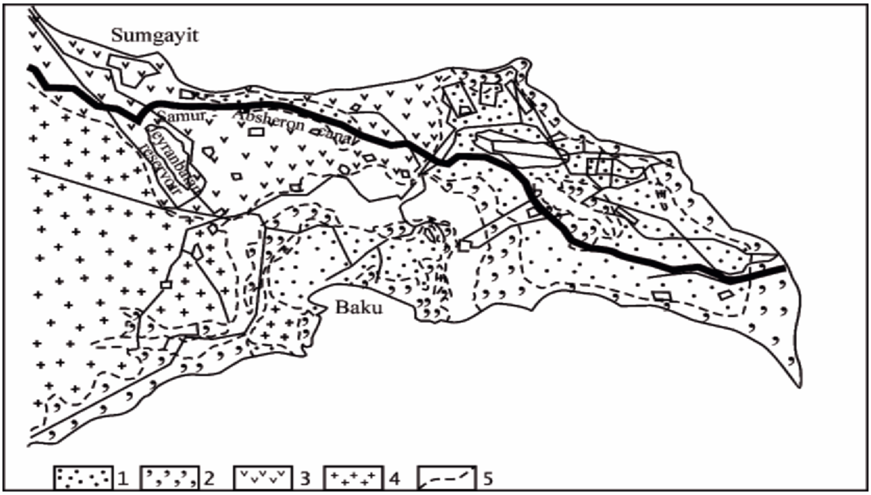

Figure 1. Schematic map of the Absheron peninsula groundwater use prospects. 1. Areas of low TDS groundwater suitable for industrial purposes. 2. Areas of practically unusable mineralized groundwater. 3. Areas of sporadic distribution of groundwater of variable chemical composition. 4. Areas where groundwater is practically absent. 5. Boundaries.

Around this era, the city's population was predominantly composed of labourers for the oil industry drawn from rural Azerbaijan and neighbouring countries including Russia and Iran. The oilfield settlement populations had to use industrial water condensates for their supply which was far from satisfactory. Infectious diseases were observed among the city population and in part ascribed to poor water supply. Since 1897 the town council had supplied additional water from the mouth of the River Kura by barges via the Caspian Sea. Even during the early years when Kura filtered water was less contaminated by wastes derived from urban cities such as Tbilisi, Gafan, Kajaran, and Ali-Bayramli, it still generally failed to meet the potable water requirements. Thus historically water quality issues have been significant.

The neglect of sanitary codes (disposal guidelines), frequent violations of filtration rules, the hot climate, poor general sanitation lead to cholera outbreaks. After the very significant 1892 epidemic, the town council resolved to develop a centralized municipal piped water supply. But development was slow, with little action before 1908 largely because unacceptable sources of centralized water were proposed. These sources did not meet the needs of the existing population, either in terms of quantity or quality, even without considering the burgeoning needs of industry and the rapidly increasing population. According to the results of surveys undertaken by V.G. Lindley in 1909 (IX Water Pipeline Congress), the Shollar pipeline proposal was accepted as the most satisfactory supply scheme. This extracted groundwater from the Quaternary deposits of the Samur-Devichi plain around Shollar (Figure 2). It is known as the I Baku pipeline.

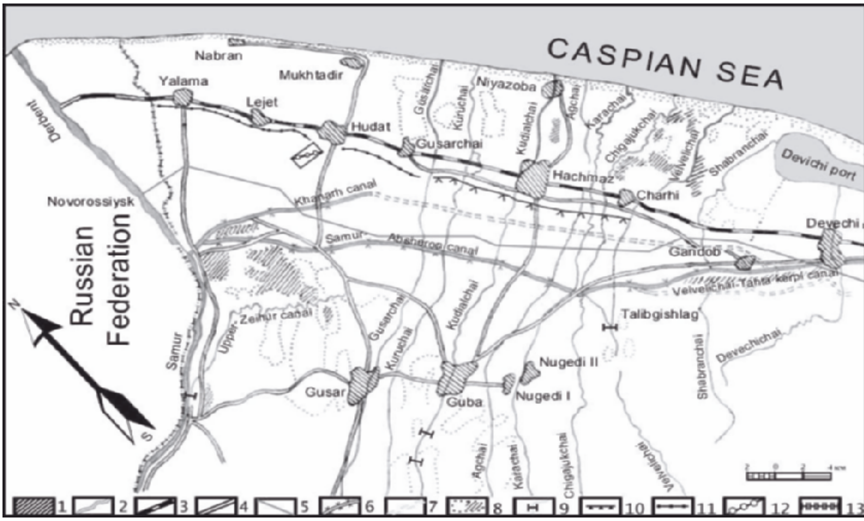

Figure 2. Map of anthropogenic features. Settlements; 2. Rivers; 3. Railways; 4. Roads; 5. North Export Oil Pipeline; 6. Canals to be reconstructed; 7. New canals; 8. Land irrigated / to be irrigated; 9. Water intakes in river beds; 10. Hachmaz water intake (II Baku water pipeline); 11. Water intake of projected III Baku pipeline; 12. Shollar water intake (I Baku pipeline); 13. State boundaries.

3.1 I Baku Water Pipeline

The central water supply project ultimately commenced in 1911 with the first phase completed in 1917 and the second phase in 1926. The pipeline output was 1.27 m^3/s. The Shollar (I Baku pipeline) water intake is located 186 km from Baku, at the village of Shollar near the town of Hudat. The scheme incorporated all the springs in the Farzali-oba and Shollar areas in addition to 24 wells. Well depths varied, with 11 deep wells extending to depths up to 216 m, and 13 shallow wells with depths of 40-70 m. Baku oil magnates were part-funders of the scheme, with H.Z. Tagiyev, a well-known millionaire and patron, being the main contributor. The output of water of the so-called I Baku water pipeline remains stable to this day and the water quality also remains high.

3.2 II Baku Water Pipeline

Urban expansion and the development of non-oil industries led to increased water demand. Since 1926, hydrogeological surveys had been undertaken by N.K. Ignatovich, V.P. Baturin, I.V. Pustovalov, N.D. Kranopevtsov, and others, with scientific supervision by N.G. Pogrebov, F.P. Savaranski, and others. Good supplies were identified in the nearby Shollar-Khachmaz zone and the II Baku pipeline constructed. The Khachmaz water intake had a capacity of 2.65 m^3/s and consisted of: 55 wells of various depths located in two rows of 25 km length; a group of springs; and a horizontal drainage system in the bed of the River Velvelichai. The distance between well rows is 1 km; the distance between wells in the first row is 200-300 m; in the second row distances increase to 500 m. 11 wells were 150 m deep, 44 wells were 300 to 400 m deep. The 9 wells drilled for withdrawal from the River Velvelichai reaches were 170 m deep. The water intake was commissioned in 1942 and had attained full capacity (2.65 m^3/s) by 1956. The water quality is good.

3.3 III Baku Water Pipeline and Jeyranbatan Reservoir

Due to growing demand for economical drinking and industrial water, especially for the food industry, hydrogeological prospecting (A.P.Popov, A.H.Babayev) commenced in the Samur-Gusarchai interfluve. In 1969, approval was obtained to develop reserves in the Quaternary deposits amounting to 1.7 Mm^3/d. At the time the water needs of Baku were 15 m^3/s.

The location of wells for the intake of III Baku pipeline were chosen following mathematical modelling (Askerbeyli et al., 1968). However, the water intake construction had been refused by an expert group of the Council of Ministers of the former USSR (Moscow). The reason for the refusal was a lack of understanding of the impact on the forestland. Additionally, little was known about the potential sources and impact of groundwater contamination from agricultural, domestic, and, in particular, waste disposal sources.

Since 1961, one of the additional sources of water supply for Baku and Sumgayit has been the Jeyranbatan reservoir. It is supplied by the Samur-Absheron canal with water from the Samur river. The water intake capacity is 12.65 m^3/s, with 4-5 m^3/s at present supplying Baku. This open reservoir is constructed in a natural depression with salty bottom sediments. In principle the reservoir was destined for industrial water supply, but at present it is used for public drinking supply needs. Recently, due to the construction of unsewered housing nearby, groundwater flow from Hyrdalan settlement, and the influence of perhaps other sources including atmospheric pollution from Sumgayit, increased contamination has occurred in the reservoir.

4. RECENT DEVELOPMENTS AND CONDITIONS

Baku progressively developed in recent times (Akhundov, 1981). It occupies nearly the whole area of the Baku syncline. This, with the planned construction of big factories such as “AZON”, “Baku Air Conditioners”, “Jewellery”, etc., water shortages forced the Azerbaijan Government to build I and II Kura intakes near M. Talysh village, below the junction of the Kura and Araz rivers.

Kura water contamination has increased since 1970-71. TDS in low-flow periods reaches 1.2-1.3 g/l. The SO_4 and Cl contents are nearly 1.5 times their maximum allowable concentrations (MAC), and phenol, fatty acids, oil products, heavy metals, etc. exceed MACs on occasion by 2-17 times. Waters of the Kura pipelines hence do not meet Potable Water requirements.

Factors such as the occupation of 20% of Azerbaijan by Armenian troops, the discharge of sewage from the cities of Gafan and Kajaran, and the copper-molybdenum plants of Armenia are believed to be responsible for significant increases in the degree of contamination of the River Araz and the Kura pipeline supplies. There is (strangely) some support to use Kura water to cover the deficit of potable water in Baku by withdrawal from the Mingachevir reservoir. It is, however, known that on the border with Georgia in the area of Shykhli village, the water is not potable. Moreover, here and further downstream the contents of heavy metals, oil products, etc. exceed MAC values.

4.1 Proposed Development of the Quaternary Deposits of the Foothills of the Greater Caucasus

Taking into account the degree of Kura water contamination the Government of Azerbaijan resolved to convey to Baku a good quality supply derived from groundwater of the Quaternary deposits located in the Greater Caucasus foothills of the Samur-Devechi and Ganyh-Agrichai plains. The decision was made by the President of Azerbaijan, H.A. Aliyev, at the end of 2002, and the Ministry was charged with its execution.

The operational reserves of the eastern margin of the Ganyh-Agrichai plain (Oguz and Gabala administrative provinces) had been investigated, but a detailed survey had not been completed. Modelling performed assuming regional operational reserves of 2 Mm^3/d supported the possibility of constructing a water intake of 80 km length that would yield a water production of 15 m^3/s. But further analysis showed that in the route of the intake passed through a potential suffusion zone of the alluvial fans of the Rivers Kishchai and Shinchai and that there were fluoride problems. Additionally, the length of the intake would potentially cause river interaction and ecological problems not currently taken into account. Due to the lack of detailed technical, economic, and ecological investigations of Oguz-Gabala province, the construction of the new intake has not yet begun.

Results of further hydrogeological, ecological, and botanical studies indicate that the interfluve of the Samur-Gusarchai is suitable for construction of a new water intake (III Baku pipeline) with varying capacities, potentially of 3 m^3/s, 6 m^3/s, and 9 m^3/s (Aliyev and Mammadova, 2001). Additionally, if there was capture of the spring runoff that drains into the Caspian Sea, a volume of 3 m^3/s, then the overall volume of available water may reach 12 m^3/s. Modelling and hydrogeological calculations (E.K. Askerbeyli, F.Sh. Aliyev, H.A. Huseynov) confirm that the proposed water intake capacity may be met. Large drawdowns are only anticipated in the line between the proposed wells and, for a water intake discharge of 9 m^3/s, stabilization of drawdown will occur within the first five years of its operation. Drawdown is not anticipated to be significant in sensitive areas such as the zone around the River Samur and the border zone with Dagestan.

Botanical and hydrogeological studies performed by the Institute of Botany of the Azerbaijan National Academy of Sciences and the Hydrogeological Expedition of the Ministry of Ecology and Natural Resources (1985-1997) indicate that the water intakes of Shollar, Hachmaz, and III Baku pipeline would not significantly impact the forestry of the piedmont plain. Although there are inevitably some negative impacts on the environment, the functioning of the I and II Baku pipelines over 80 and 50 years respectively suggests that large schemes can be relatively impact-free and sustainable. There has even been relatively little general groundwater level decrease. In fact over the last few decades some trends have exhibited water level (Figure 3) and spring discharge increases (Figure 4).

4.2 Baku: Present Conditions

Significant urban water quality problems exist in Baku, the main population centre. These are related to inadequacies in the water supply, waste water, and industrial sewage systems. This is in spite of a municipal water pipeline system being developed from ~1917 onwards (see above).

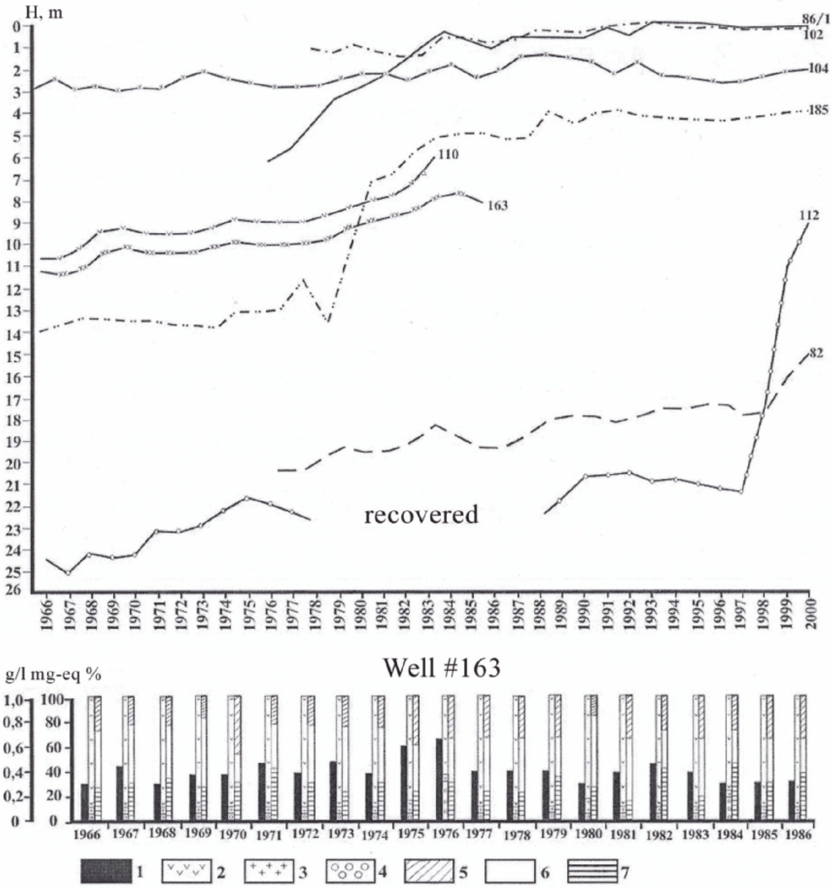

Figure 3. Groundwater level and quality in zone influenced by anthropogenic factors: 1. TDS (g/l); 2-4. Anions (HCO_3, SO_4, Cl); 5-7. Cations (Ca, Mg, Na).

The sewerage system is in a worse state. Its capacity does not match the volume of sewage. Sewers were built without taking into consideration increased use or the salinization and aggressiveness of ground and groundwater. Restoration of leaking sewers is slow, if it occurs at all. These problems were becoming apparent after the development of the Kura pipelines. Groundwater levels had increased sharply due to leakages from the water-sewerage systems. At present in the central part of Baku, groundwater occurs at depths of just 1-3 m (Figure 5) with much flooding of basements. Alternation of clays, loams, and fine-grained sands cause the development of suffusion, subsidence, and the destruction of some basements; e.g. the State Philharmonic and 2nd Clinic Hospital #1 buildings were unusable for more than 10 years due to such problems. Presently over

300 basements in Baku are in poor condition and landsliding has occurred, e.g. at the Bail slope, Akhmedli and Guneshli settlements, and in the inner part of the western flank of the Baku ‘amphitheatre’.

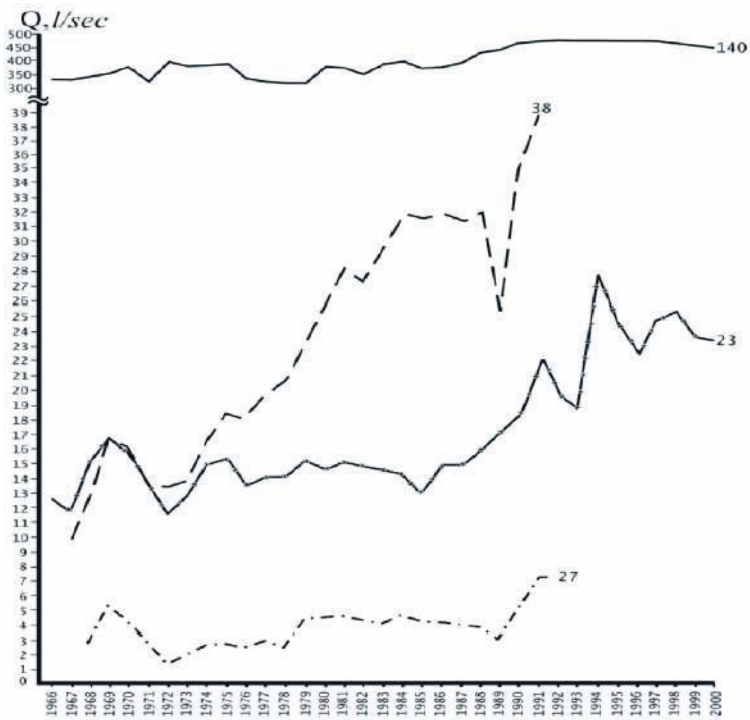

Figure 4. Discharge of springs in irrigation zone.

5. CONCLUSIONS

Baku and surrounding areas represents a challenging urban management scenario, particularly in the sustainable water supply and sewage disposal. The city has a complex relief and geological structure that may contribute to some of its problems, e.g. sewage system leakage and landslides. The area has developed its water supply system over the last century or so, but is still significantly stressed due to both water volume and water quality concerns.

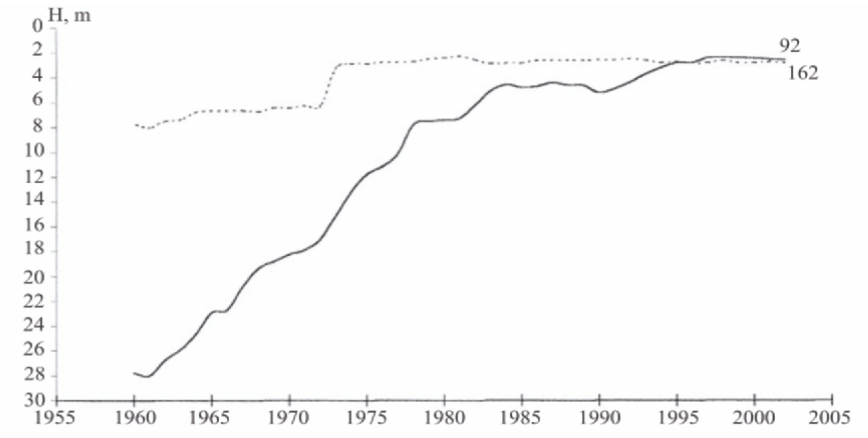

Figure 5. Groundwater levels in the central part of the Baku syncline.

It is clear that the groundwater-based schemes that supply the city have played a pivotal role in alleviating the water supply problems and will continue to do so with the proposed development of the Quaternary deposits outside the city. Clearly with the increasing demands that arise from population shifts to urban areas there remain many challenges in achieving a reliable and sustainable supply for the city. Foremost amongst these is the protection and safe utililization of the groundwater reserves already in use and those proposed for exploitation in the future.

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GEOENVIRONMENTAL PROBLEMS IN AZERBAIJAN

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Abstract: Azerbaijan's environment is under pressure. Flooding from irrigation canal leakage, irrigation return, blocked drainage systems, and leaking water pipes / sewers are the most significant problems. Associated phenomena include groundwater level rise, salinization, and fertilizer contamination. Since the start of the Samur-Absheron irrigation canal project, groundwater level has risen by up to 20 m. Rise in Caspian Sea level has also caused flooding. Landslides are triggered by slope erosion, irrigation, engineering works, and increased loading. Soil contamination is a major problem in the oilfields of the Absheron Peninsula. Mud volcanoes, deflation, internal erosion, and abrasion are widely developed. This evaluation marks the start of a geoenvironmental monitoring project within the Republic.

Key words: Contamination; landslides; erosion; flooding; swamping; salinization; mudflows; mud volcanoes; Absheron Peninsula, Azerbaijan; irrigation; flooding; canals; drainage; Caspian Sea; groundwater level

1. INTRODUCTION

Situated in the Eastern Caucasus, the Republic of Azerbaijan occupies an area of 86,600 km^2. It borders Russia to the north, Georgia to the north-west, Armenia to the west, and Turkey and Iran to the south. The Caspian Sea forms its eastern border. In the 1920s, the Zangezur province of Azerbaijan, situated between the regions of Nakhchivan and Zangelan-Kalbajar, was separated from Azerbaijan and incorporated into Armenia. As a result, Azerbaijan is divided into two separate territories that do not

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share a border. The capital of Azerbaijan is Baku, which has a population of over 8 million.

The territory of Azerbaijan is distinguished by its natural characteristics and distribution of resources. Climatically, the country can be divided into three areas: a permafrost belt, dry semi-arid regions, and humid sub-tropical lowlands. Its reserves of oil and gas, ore deposits, non-metallic and mineral waters are of world importance. Oil exploration and refining, chemical production, machine-building and other industries have been developed, whilst the primary agricultural industries are cotton-growing, grain-farming, wine-growing, and cattle-breeding. At present, practically all of Azerbaijan is being developed under different fields of agriculture, industry, health resorts and sanatoria, and town-planning. It represents a complicated geoenvironmental system, affected by both natural and anthropogenic factors.

2. NATURAL GEOENVIRONMENTAL CONDITIONS

2.1 Geological Structure and Landscape Zones

The mountainous zones of the Greater Caucasus and Lesser Caucasus, separated by the Kur-Araz lowlands, are the major geological structures present in Azerbaijan. Though the mountainous Talish region to the south-east is separated from the Lesser Caucasus by the Araz valley, it is considered part of the Lesser Caucasus structural zone. Thus 3 regions are distinguished as geostructural regions (i.e. having tectonic structures of the first order): A – the mountainous fold zone of the Greater Caucasus; B – the mountainous fold zone of the Lesser Caucasus; C – the Kur-Araz lowlands. Within these regions, 18 hydrogeological basins with matrix + fracture flow, matrix-flow, and fracture-flow aquifers have been distinguished (Table 1).

Altitudinal belts for the mountainous fold territories are distinguished in the landscape-geoenvironmental zonal scheme used here. The territories of hydrogeological basins of pore-fracture waters, and the latitude-altitude zonality of the remaining territories correspond to these belts. The latitude-altitude zones are divided into the zone of leaching, the transition zone (with both leaching and salinization processes present) and the salinization zone.

2.2 Geoenvironmental / Hydrogeological Conditions

2.2.1 Introduction

The geoenvironmental assessment of hydrogeological conditions includes the examination of the availability of drinkable groundwater, the existence of any negative influence of the groundwater on its surroundings, and the contamination state of the groundwater.

Table 1. Hydrogeological basins of Azerbaijan.

Geostructural regions	Hydrogeological basins
A. Mountainous fold zone of Greater Caucasus	I. Basin of Greater Caucasus (matrix+fracture flow aquifers) II. Shamakhi-Gobustan basin (matrix-flow and fracture-flow aquifers) III. Basin of Absheron peninsula (matrix-flow aquifers) IV. Samur-Gusarchay basin (matrix-flow aquifers)
B. Mountainous fold zone of Lesser Caucasus	I. Basin of Lesser Caucasus (matrix+fracture-flow aquifers) II. Nakhchivan basin (matrix+fracture-flow aquifers) III. Talish basin (matrix+fracture-flow aquifers)
C. Kur-Araz lowlands	I. Sheki-Zakatali basin (matrix-flow aquifers) II. Ganja-Kazakh basin (matrix-flow aquifers) III. Karabakh basin (matrix-flow aquifers) IV. Mil basin (matrix-flow aquifers) V. Jabrail basin (matrix-flow aquifers) VI. Nakhchivan basin (matrix-flow aquifers) VII. Shirvan basin (matrix-flow aquifers) VIII. Mugan-Salyan basin (matrix-flow aquifers) IX. Lenkoran basin (matrix-flow aquifers) X. Basin of Jeyranchol Neogene foothills (matrix-flow aquifers) XI. Basin of Hajinohur (matrix-flow aquifers)

2.2.2 Fresh and low-salinity groundwater supply

Fresh and low-salinity groundwater resources are limited within the territory of Azerbaijan and are distributed irregularly, thus causing problems in water supply.

The mountainous regions of Azerbaijan are formed of Mesozoic-Cenozoic rocks and characterized by significant relief, thick weathering zones, fracturing, abundance of mantles of deluvial/eluvial loam, and river valleys and small troughs of alluvial and fluvioglacial sediments. Groundwater is mainly associated with the weathering zone and tectonic dislocations. Shallow circulating groundwater, discharging as springs usually at a rate of 5-10 l/s, is observed in valleys and ravines of the foothills. Springs with much greater discharges (to 60-100 l/s) are associated with karstic limestone areas. Waters are fresh or occasionally slightly saline. In mountainous zones, groundwater contained within alluvial sediments in river flood plains has major economic importance. The quantity of water yielded by these resources totals 40-60,000 m^3/day.

Fore- and intermontane plains are the regions richest in fresh and low salinity water. The aquifers here are formed by the convergence of alluvial cones with Upper Pliocene-Quaternary and Quaternary alluvial and deluvial/eluvial sediments of significant total thickness (300-500 m, rarely 1000-2000 m). These aquifers comprise boulders, gravel terraces, grits, and sands. In the upper parts of the alluvial cones, groundwater is largely unconfined: in the central and peripheral parts of the cones there are both confined and unconfined systems. Depending on the characteristics of the geological fabric, recharge rate, and drainage conditions of the water-bearing beds, fresh and low salinity waters are widely used. Recharge of intensive precipitation and surface drainage takes place in the upper part of the alluvial cones. The central parts of the alluvial cones are zones of groundwater transition and extensive redistribution between separate water-bearing horizons. Continental deposits often pass into marine deposits where thick zones of high-pressure groundwaters are found in lithologies of favourable composition. Depending on the unloading conditions, these zones contain fresh water (e.g. Samur-Gusarchay basin – A-IV) or low salinity and salty waters (e.g. Mil basin – C-IV). In some cases (e.g. Jabrail basin – C-V), underground water is associated mainly with less permeable alluvium in the central and peripheral parts of the alluvial cones.

The total operational reserves of fresh and low salinity groundwater stored within the piedmont plains of Azerbaijan has been assessed as about 24 million m^3/day, 12 million m^3/day of which is maintained and approbated by the State Committee on Resources of the former USSR and the Azerbaijan Republic. The Samur-Gusarchay (A-IV), Sheki-Zakatali (C-I) and Ganja-Kazakh (C-II) basins of matrix-flow aquifers are distinguished by their high productivity: 60% of all maintained resources in Azerbaijan come from them. The water supply for Baku is provided by groundwater from the Samur-Gusarchay basin.

The Jeyranchol (C-X) and Hajinohur (C-XI) basins of matrix-flow systems are formed of Palaeogene/Neogene clays and are characterized within the fore-mountain plains by unfavourable hydrogeological conditions. However, permeable gravels, sands, sandstones, and limestones are occasionally present. Fresh and low salinity water occurs only in small quantities; most of the water is of medium to high salinity. The Shamakhi-Gobustan basin of matrix-flow systems (A-II) has more favourable conditions.

In the Absheron Peninsula, anthropogenic factors affecting underground water reserves, such as leakage from irrigation channels, water pipes, sewage and heating systems, and infiltration of water used in irrigation, have caused the naturally unfavourable hydrogeological conditions of the matrix-flow sequences to change considerably. The watering of this arid area has raised groundwater level and increased groundwater resources, and has also both decreased mineralization in some places and enriched the chemical composition of groundwaters in other places.

In the lowlands of the Republic, particularly in the Mugan-Salyan basin (C-VIII) and parts of the Shirvan (C-VII), Mil (C-IV) and Lenkoran (C-IX) matrix-flow basins, unfavourable hydrogeological conditions occur. Geologically, these areas are formed of continental-marine and marine deposits of Upper Pliocene-Quaternary age, with the continental sediments having a subordinate position. Underground waters are salty, with dry residues of 100-200 g/l.

2.2.3 Natural protection of groundwater

The water-bearing stratum closest to the surface is never protected naturally. All basins containing matrix+fracture-flow sequences are assessed as poorly protected. In the Sheki-Zakatali (C-I), Samur-Gusarchay (A-II), Jabrail (C-V), and Nakhchivan matrix-flow basins, groundwater in the water-bearing stratum closest to the surface is not protected. Other areas where fresh and low salinity groundwater are present are also assessed as poorly protected, as is the groundwater of the Absheron Peninsula. Over the rest of the region and in some fore-mountain plains, groundwater in the topmost water-bearing stratum is relatively protected.

Over most of the Ganja-Kazakh (C-II), Sheki-Zakatali (C-I), Jabrail (C-V) and Lenkoran (C-IX) basins, the water-bearing horizon is relatively protected. In other regions with basins of matrix-flow sequences, the confined aquifers are protected.

2.2.4 Geochemical conditions

Azerbaijan is rich in oil, gas, precious metals, iron, mineralized geothermal waters, and different kinds of non-metallic and building materials. As such, local geochemical and gas anomalies are observed in various areas. Mercury, iron, and lead anomalies are observed in the Lesser Caucasus, whilst in the ore-bearing areas of the Greater and Lesser Caucasus, copper, cobalt, zinc, and manganese anomalies are observed. In eluvial deposits and fluvial sediments, copper, lead, tin, cobalt, and nickel anomalies are found. In areas with oil and gas condensate, methane anomalies are also observed, and there is a high strontium content in the soil of oil fields. High radon concentrations are locally present in some areas of the Lesser Caucasus.

Filizchay, Katsdag, Katekh, Mazimchay, and Sagator are some of the biggest ore and polymetallic deposits in Europe: undeveloped deposits are a potential danger to the environment.

2.2.5 Geodynamic conditions

Azerbaijan is a region of active geodynamic processes, with many complex tectonic structures. Hundreds of seismic events of varying strength occur annually: the most powerful earthquake of the 20th century was the Shemakha earthquake in 1902. Registering 9 on the Richter Scale, the quake destroyed the ancient capital of Azerbaijan. During the 20th century, 20 earthquakes over magnitude 5 were observed, 13 of them registering 7 on the Richter Scale, and four being magnitude 8 (1924 – Imishli, 1931 – Zangezur, 1933 – Gubatli, 1968 – Ordubad).

Mud volcanoes are common in Azerbaijan, particularly at the south-western margins of the Greater Caucasus, including that region of the Caspian Sea. Over 220 mud volcanoes are concentrated in a comparatively small area, many of them periodically active at present. From the crater of the mud volcanoes, gas and water, sometimes with an oil film or containing rock clasts, are educed and volcanic mud spreads down the slope, building up a mud cone. At certain times, eruptions and explosions take place, followed by large gas emissions and discharges of rock clasts to a significant height. These volcanoes are not analogous to magmatic volcanoes and are of great interest as a natural phenomenon: it has been determined that they are associated with oil-bearing strata.

Mud volcanoes are one of the most dangerous geoenvironmental phenomena in Azerbaijan. For example, an eruption from a mud volcano on the Absheron peninsula caused landslides over 3 km. As a result, important economic resources are put at risk.

Exogenous geoenvironmental processes are widely developed in Azerbaijan. Significant environmental deterioration caused by exogenous processes is present over a large part of the deformed, mountainous region of the Greater Caucasus (area A), an area of intensive erosion, avalanches, debris flows, and collapses. Flows, often thick and muddy, take place within the Belokan-Sheki zone. Catastrophic mud flows repeatedly destroyed the ancient city of Sheki and many other villages located in river valleys. Within the Greater Caucasus, to the east of the Geokchay River, increased regional erosion by landslides is seen within areas of Cretaceous and Paleogene-Neogene clay, limestone, and terrigenous sediments. They are observed on the mountain slopes, plateau, and river valleys. The areas affected by landslides cover many hectares. Landslides are especially dangerous and cause great damage in the Shemakha, Ismailli, and Agsu regions. Scores of populated areas, economic resources, and roads are in danger of damage by landslides. Ravine erosion is also widely developed in the region.

In the Jeyranchol and Hajinohur regions, badlands, clay pseudokarst, landslides, ravine formation, subsidence, and river erosion are all prevalent.

Environmental deterioration by exogenous processes is especially notable in the Absheron Peninsula. Landslides, ravine erosion, marine abrasion, rockfalls, deluvium flows, rockfalls, mud volcanism, flooding, soil salinization, subsidence, and deflation processes are widely developed in the region. The largest and most active landslides occur in the western part of the peninsula. Some of them are linked to the Baku trough, located on the inner edges of its western and eastern slopes, outer, steeply sloping areas at the back of the trough, and also in the central part of the peninsula. Historically, the Bailov, Park, Akhmedli, and Zikh landslides have caused great damage.

The Bailov landslide is the oldest and most powerful of these and occurred between the present day Neftchilar Avenue and T. Bagirov Street in Baku, from the old hotel 'Intourist' to the fire station. The Bailov slope is a complex of ancient landslide deposits, formed over many thousands of years with initial movements dating back to prehistoric times. Activation of landslide processes began during a period of high sea level in the Caspian Sea. Landslide processes reduced when the Caspian Sea level receded and also as a consequence of climate change, but were replaced by erosive and solifluxion processes. Evidence of old landslides is preserved today in the form of damaged historical structures. An increase in landslides has been observed over the last century, since the beginning of major construction and oil extraction in the area. The biggest landslide, at a scale of ~ 250 m, was in 1929. Tram lines and a number of buildings were destroyed. In 1936, the construction of a power station began in the area, and visible slope movement became apparent. The rapid construction of a hospital, residential housing and a zoo then increased landslide activity: between 1936 and 1939, slope movement occurred in the vicinity of Bail bridge and the adjacent shoe factory, and the landslides which happened over the period 1939-1943 caused the shoe factory to be completely destroyed. Landslides last occurred in the area in 1952, 1953, 1957 and the 1960s. The results of the investigation into the 1960s landslide showed the main causes to be: large-scale leakage from water and/or sewage pipes into the ground; over-watering of vegetation; leakage from the water supply pool in the zoo; lack of systematic slope drainage; slope cutting during the construction of roads, communication infrastructure, and other buildings; and general slope erosion. In February 1974, new slope movement took place that caused great damage to underground communication lines, and deformation of buildings and roads.

Starting in 1970, remediation works have been carried out on the slope. A cloudburst runoff canal was built along the Bail slope, watering was limited, the city zoo was moved, and water and sewage lines were repaired. These actions helped to stabilize the slope until 1996. In early August of that year, a broken water pipe on T. Bagirov Street on the upper part of the slope produced a long period of water leakage and this, combined with leakage from an underground reservoir, caused large-scale landslide development. Heavy rainfall in July was also a contributory factor. The decollement horizon of the new landslide coincided with the decollement horizon of the landslide that took place in 1974. The landslide was in the direction of the 'Lukoil' petrol station, and landslide tongues formed in Neftchilar Avenue, causing large-scale deformation of the 'Parkcommuna' building. The area of the landslide covered 0.8 ha, with a depth of 12-14 m. The cloudburst runoff canals were reconstructed, slope stabilization was carried out, pipes were repaired, reservoirs were emptied and fractures were repaired. Despite this, the landslide was in a strained condition and remained hazardous to many vital activities of the city.

The investigation into the 1996 event made it possible to predict the timing of the next landslide. This took place on 6-7 March 2000 and, as a result, 17 residential buildings in the Bailov ravine were destroyed, a further 26 became uninhabitable, and 100 more were damaged. Industrial and administrative buildings in Neftchilar Avenue and the 'Lukoil' petrol station were also destroyed. As a result of the advance of the landslide tongue, 200 metres of public road were put out of operation, fractures were formed in neighbouring buildings and in both the foot and the top of the slope. As a result of side dislocation, many cliffs were formed and massive blocks of various sizes and volume left behind. After a large block collapse at the brow of the slope bordering the districts of Memorial and Shekhids, a cliff of 35-40 m height was formed. The size of the region affected by the landslide is 550-600 m from east-to-west and 350-400 m from north-to–south, with the volume of material within the landslide calculated as 10 million m^3. With the prognosis of specialists and the efforts of the first author, the local population was evacuated eight hours before the landslide and human casualties were avoided. Damage caused by the landslide was great and, although some restoration work has been carried out in the area, it is difficult to determine whether the slope has now been stabilized.

Intensive coastal erosion processes take place along the north and east coasts of the Absheron Peninsula. Due to strong, persistent northerly winds peculiar to the peninsula, waves develop of great size and strength and the sea bed along the coast is often exposed to subaqueous erosion. Flooding and swamping of coastal areas are major hazards within the Absheron Peninsula and the Baku region.

Variations in the Caspian Sea level and in particular its observed rise since 1976, have caused a marine incursion over large parts of the coastal regions and thousands of hectares of land have become useless, whilst settlements, important industrial and civil buildings, main roads and railways have been damaged.

Within the Lesser Caucasus (area B), different areas are subjected to plane and ravine erosion, deluvium flow, landslides, and mud flow. The highest mountains are subject to freeze-thaw and rock breakdown, and deluvium flows and debris flows form as a consequence. The erosion of ultrabasites results in easily transported deluvium and flows and landslides are formed. In turn, these form landslide slopes of 3-5 km^2 in area and 20-30 m or more in depth. Hazardous landslides are widespread in the Kedabek region. In many cases deluvium flows and rockfalls are the main sources of mud flows. Active mud flows of mixed clastic debris and water are distinctive of rivers in the Nakhchivan region. A particularly powerful mud flow in the Vanand valley of the Ordubad region of Nakhchivan in August 1998 destroyed everything in its path, causing great damage.

In the headwater regions of rivers in the Nakhchivan Autonomous Republic and the mountainous Talish region, exogenic geoenvironmental processes can cause major damage. Landslide movements on steep slopes in this area periodically destroy whole villages. An earthquake in Talish in July 1998 reactivated many dormant landslide areas, some of which caused extensive damage to the area.

Within the Kur-Araz lowlands (area C), the central Absheron plain is characterized by ravine formation, edge erosion, landslides, subsidence and solifluxion. Actively developing ravines are present on both banks of the River Kur, up to the point where it flows into the Mingachevir reservoir. Bank erosion occurs where the banks are cut into by the erosive part of the river channel, and also on the ledges of high terraces. Conditions favourable to landslide development are observed mainly on the right bank of the Kur. Landslides are also observed within the Ganja-Kazakh plain, and areas of subsidence are seen to develop within loess sediments. The confluence of the rivers flowing from the southern slopes of the Greater Caucasus takes place within the Sheki-Zakatala zone. Transportation and reworking of clasts from older mud flows in the lower reaches and alluvial cones of these rivers often results in the formation of new mud flows. Additionally, there are areas of subsidence, river and ravine erosion, and flooding in this region.

Within the Low Kur area, subsidence in loess, loams and badlands, mud volcanism, aeolian sands in South-Eastern Shirvan, quicksand development within the Kur strip, flooding processes and the formation of solonchak soils are all observed. At the edges of the Low Kur depression, especially around the Mingachevir reservoir, clayey pseudokarst formed of easily eroded Palaeogene-Neogene sandy-clayey sediments is widespread. Serious landslide processes periodically occur at the edges of the Mingachevir reservoir.

In the eastern and central parts of the region, ground salinization processes, which are mainly caused by climatic conditions and a high level of mineralized groundwaters in areas with low drainage levels, are developed. In such natural landscapes, favourable conditions for solonchak accumulations arise. Salinization is mainly by chlorides, with sulphates occurring in some areas. Salinization processes and flooding events restrict soil usage and result in the deterioration of the area.

Flooding of the soil takes place within the lower part of Nakhchivan Autonomous Republic, and the Ganja-Kazakh, Karabakh-Mil and Lenkoran areas. It is caused by infiltration of irrigation water, and leakage from irrigation systems and the main channels.

3. THE CHARACTERISTICS OF ANTHROPOGENIC SYSTEMS

3.1 Background

The natural environment of Azerbaijan is under constant pressure from anthropogenic exploitation and industries belonging to a variety of different fields. Within the agrarian sector, irrigated agriculture and cattle-breeding are well-established in the Republic, with cotton, grain, grapes, tobacco, and subtropical vegetables being grown.

Azerbaijan is characterized by an arid climate and, in many areas, there is insufficient water for the cultivation of land. In locations where water resources are unevenly distributed it became necessary to remedy this situation via the construction of irrigation channels. Within the Republic, there are 205 irrigation systems, 40 reservoirs, and a thousand wells extracting groundwater to serve 1400 thousand hectares of irrigated land. The total length of these irrigation systems is about 450,000 km. The largest channels are the Upper Karabakh channel, with a length of 172 km and a supply of 114 m^3/s, and the Upper Shirvan channel, 123 km long and supplying 78 m^3/s, both taking water from the River Kur, and the Absheron channel, 178 km in length and supplying 55 m^3/s from the River Samur.

The largest reservoirs in the Republic are Mingachevir (16 x 10^6 m^3) and Shamkir (2.6 x 10^6 m^3) on the River Kur, a waterworks facility on the Araz (1.3 x 10^6 m^3), Sarsan (0.5 x 10^6 m^3) on the River Terter, Khachinchay (23 million m^3) on the River Khachinchay, and Ali-Bayramli (22 million m^3) on the River Agrichay. The total volume of these reservoirs is about 22 x 10^6 m^3. About 11% of reservoir water infiltrates the surface and recharges the groundwater system. Annual water use in Azerbaijan is 11-12 x 10^6. m^3; with 18-20% of the water used for irrigation being lost.

In order to monitor changes in groundwater level, decrease and prevent further salinization, and improve ground conditions, drainage collectors have been constructed on 315,000 hectares of the Kur-Araz lowland: closed collectors occupy 248,000 hectares of this area, whilst vertical collectors cover 12,000 hectares. A trap-drainage grid covers 28,000 hectares. Over 35% of the collectors were built in the Mugan-Salyan (C-VIII) area, 29% in Shirvan (C-VII), and 13% on the Mil (C-VI) plains. The average length of collector drainage systems is about 20 m/ha over the whole irrigated area. Annually, the system discharges over 5 Mm^3 water into the sea, with water from the Kur-Araz lowland having an average mineralization of 10.5 g/l (Alimov, 2001).

Groundwater is collected via pits, wells, and kahrizes (subterranean canals), and there are many capped springs in mountainous areas and their foothills. The depths of exploited wells are mainly 120-200 m.

During the most intensive periods of agricultural activity, the annual groundwater output was 2.5-2.9 Mm^3, but it has been 1.3-1.5 Mm^3 in more recent years. The most intensive groundwater output occurs in the Karabakh (C-III), Mil (C-IV), Ganja-Kazakh (C-II), and Lenkoran (C-IX) groundwater basins. Only 8-10% of the total groundwater output is used for drinking water supply and 3-4% for industrial uses. The vast majority, 86 - 89%, is used for irrigation. However, because of irregular water distribution and a lack of access, a considerable proportion of the population uses river and stream water for drinking and economic purposes. Water supply is a particularly big problem in Baku, the Azerbaijani capital. Unfavourable natural conditions (no major river, low precipitation, high evaporation, a local geology formed predominantly of muddy deposits, and high rock salinity) are present in the Absheron Peninsula where Baku is situated. As a consequence, the large population of Baku satisfies its water needs by exploiting groundwater from the Samur-Gusarchay basin and water from the Rivers Kur and Samur.

3.2 Contamination of Surface Waters

A number of Azerbaijani rivers contain chemical and organic compounds in concentrations which exceed the maximum permissible concentration (MPC). The basins of the two main rivers of Azerbaijan, the Kur and the Araz , also occupy large parts of Georgia, Armenia, and Turkey. Contamination from the large Georgian cities of Tbilisi and Rustavi means that when the Kur crosses Azerbaijan's borders it is already contaminated (including a 5 day biological oxygen demand (BOD-5) of ~ 3.7 mg/l, 0.15 mg/l of oil products, and 0.03 mg/l of phenol). Within Azerbaijan, the Kur becomes enriched with agricultural contaminants, from industrial, cattle-breeding, and poultry farm runoff. In some places, the BOD-5 increases to 4.1 mg/l, oil products to 0.24-0.30 mg/l, and phenol to 0.04-0.08 mg/l.

The second main river of Azerbaijan, the Araz, contains hazardous contamination due to input from two of its tributaries, the Razdan and the Okhchu. Copper content in the Araz is 25-50 times over the MPC and phenol 6-15 times greater because the Okhchu River brings in industrial runoff from the Gadjaron copper-molybdenum and Kafan copper-ore mines of Armenia. High aluminium, zinc, manganese, bismuth, and titanium content is also observed in the river runoff, which is red-brown in colour.

Poorly developed sewage systems in Azerbaijan, Georgia, and Armenia mean that polluted runoff is discharged by tributaries into the Kur and Araz systems, making them constant sources of contamination of the river water. Within the Absheron Peninsula, a lack of river drainage results in runoff being discharged into the peninsula's lakes. In terms of contamination protection, the rivers of the Samur-Gusarchay basin are relatively well guarded.

3.3 Contamination of the Shallow Subsurface

Over large lowland areas in Azerbaijan, the shallow subsurface is subjected to natural contamination in the form of salinization. The following main salinization types are distinguished: chloride-sodium; sulphate/chloride-sodium; and sulphate/chloride-magnesium/sodium. The degree of ground contamination increases sharply below the zero horizon (the absolute level of the Caspian Sea is determined to be -26 to -27 m from the level of the Baltic Sea) in areas with poor or no drainage. Its value in irrigated areas changes from 0.25% to 1-2%.

Local contamination of the shallow subsurface has been observed: by organic and mineral fertilizers in irrigated areas and in land adjacent to mineral fertilizer storehouses; by oil products within oilfields and oil refineries; and by various chemical elements and compounds within some industrial plants. In some years on local strips of the irrigated Karabakh slope areas (C-III), nitrate and nitrite contents in the unsaturated zone were more than 10 times those present in the groundwaters below the water table. Near the storehouses of mineral fertilizers, nitrite content in the unsaturated zone was between 1.7 and 97.7 mg/kg, and hexaclorane (Lindane) from 0.1 to 0.4 mg/kg. Nitrate, phosphate, sulphate, chlorine, iron, and aluminium contents are high in the unsaturated zone beneath a sludge storage area of a Ganja alumina industry site. Contamination by oil products is very common within the Absheron peninsula: over 30,000 hectares have been contaminated here. The ground and unsaturated zone are contaminated to a depth of 1.0 to 1.5 m.

3.4 Contamination of Groundwaters

Groundwaters are not subjected to regional diffuse contamination in Azerbaijan, but local, point contamination of communal, industrial, and agricultural character is observed. As has already been mentioned, the main source of communal contamination is the lack or poor development of sewerage and purification systems in many settlements. Waste water is discharged into rivers, the sea, natural hollows, or specially dug holes, and contamination of groundwater takes place by infiltration of already contaminated river water or by migration of contaminants through the unsaturated zone. It should be mentioned that it is difficult to achieve full purification even with the available purification installations. For example, even after purification, phenol (0.007-0.13 mg/l), BOD-5 (0.46-23.9 mg/l), sulphate (960-1280 mg/l), and iron (0.5-5.0 mg/l) have all been found in the groundwater of the region where industrial runoff from Ganja is discharged. Within Azerbaijan, industry is mainly concentrated in the Absheron Peninsula, within the cities of Baku and Sumgayit, and is less developed in Ganja, Ali-Bayramli, Mingachevir, and Nakhchivan. Despite this, groundwater in the Absheron Peninsula has a rather mixed chemical composition and its mineralization varies from 0.4-0.5 g/l to 100 g/l or more. The groundwater is also enriched by a large quantity of chemical and organic compounds. Oil products contaminating the shallow system also infiltrate the groundwater. In different areas of the peninsula, the oil content of the groundwaters varies from 1.4-3.6 mg/l to 40.0-50.0 mg/l or more. In some cases, radioactive contamination is observed in groundwaters beneath the oilfields. Phenols, sulphites, anilines, amines, and aromatic hydrocarbons are also present in the peninsula groundwater. Groundwater beneath Sumgayit is also contaminated with heavy metals.

A particularly negative incident has been discovered recently in the area of the Sumgayit chloro-alkali plant. Over many years production in the area, errors in processing had enabled liquid mercury to penetrate into the ground and the groundwater. Our investigations showed that, just within the vicinity of an old, non-functioning facility, about 80 tonnes of mercury was accumulated at a depth of 6 m and about 8 tonnes of mercury was present in a liquid waste mass of about 130,000 tonnes in the plant waste storehouse. Besides the contaminated waste, a further 18,000 m^3 of material is likely to have been utilized. It is thus highly probable that both the volume of mercury and the area of utilized land within the plant are considerably larger than these initial estimates indicate. Similar situations are also present in other Sumgayit plants.

Periodically, 0.08-3.5 mg/l of aluminium, 3.5-50.0 mg/l of iron, and 0.003-0.004 mg/l of phenol have been observed in groundwater beneath the liquid waste storehouse of the Ganja alumina enterprise. There is also an increase in nitrite, nitrate, ammonia, and sulphate content.

Groundwater contamination by agricultural contaminants takes place mostly in areas closest to mineral fertilizer storehouses. In these areas, there have been found hexachloran levels of 0.01-0.6 mg/l, 0.02-0.15 mg/l of the insecticide sevin, and up to 0.5 mg/l ragar present in the groundwater.

Increased nitrite and nitrate content is observed in the groundwater of irrigated areas, but the quantity does not exceed the maximum permissible concentration (MPC). Nitrate and nitrite contamination of groundwater takes place particularly near cattle-breeding farms: in some cases, the nitrite content is 10-19 mg/l, and that of nitrate 12-145 mg/l.

There are levels of 0.01-1.2 mg/l nitrite and 2-75 mg/l nitrate in kahriz (well/gallery complexes) waters of the Nakhchivan Autonomous Republic. Bacteriological contamination of groundwater is observed in irrigated land within cities, cattle breeding farms, and purification installations. Contamination of confined groundwaters has not been observed.

3.5 Influence of Oilfield Activity

Azerbaijan is famous for its hydrocarbon reserves, with oil and gas production carried out both on- and off-shore. Extracted oil is partially refined in Baku and part of it is transported by the Baku-Novorossiysk and Baku-Supsa pipelines, which distribute oil beyond the borders of the country. The Baku-Jeykhan pipeline is being built, and it is anticipated to be used also in the pumping of Kazakhstani oil. Much of the Absheron peninsula is crossed by oil and gas pipelines. Vertical movements caused both by tectonic and anthropogenic factors take place across all of the peninsula: complex processes of uplift and subsidence of the Earth's crust occur at a rate of 6 to 28 mm/year with the southern part of the peninsula subsiding at a rate of 0.21 to 5.68 mm/year. Local areas with the highest and most unfavourable subsidence rates are linked to anticline arches, which are practically coincident with the Surakhani, Balakhani-Sabunchu-Ramani, and Bibiheybat oil regions where oil production has been carried out -for over one hundred years. Over most of the oil-producing regions, subsidence does not exceed 8 mm/year. Taking into account the importance of endogenous processes in surface subsidence, the long-term development of oil- and gas-bearing deposits, and the decrease in rock pressure we consider that the extraction of large quantities of groundwater during the development of oil- and gas-bearing deposits has been significant too.

Oil-producing areas occupy 15,000 hectares of the Absheron Peninsula, with about 100,000 m^3/day of groundwaters being extracted. Around 35,000 m^3/day of this water is pumped back into the strata to support rock pressure during extraction. The intensity of groundwater extraction varies over different areas from 1.5 to 10 m^3/day/ha. In the older regions of oil production mentioned above, where maximum subsidence is observed, the greatest quantity of groundwater is extracted, with an intensity of 5.4 to 10 m^3/day/ha. In areas where the extraction intensity does not exceed 1.5-2.5 m^3/day/ha, the subsidence rate is 8 mm/year and it is evident that the subsidence rate is dependent on the quantity of water extracted (Alekperov, 2000). In the regions of the oil refineries, contamination by oil products is observed close to some oil pipelines.

3.6 Development of Ore Deposits and Power

With the exception of the Dashkesan group of iron-ore deposits, the ore deposits in Azerbaijan (e.g. the Zaglic alunite deposits, the Paragachay copper-molybdenum deposits, and the Gumushlu polymetal deposits) are relatively small in scale. However, the extraction of materials used in construction is rather intensive. The materials extracted are: limestone, sand, clay, gravel, pebbles, facing materials, and some types of metallic materials. Ten strip mines, extracting limestone, sand for use in construction, and material for the manufacture of cement, operate in the south-west of the Absheron Peninsula.

Using the Mingachevir, Shamkir and Yenikend reservoirs, there are four hydroelectric stations on the River Kur, with a further hydroelectric station at the waterworks facility on the River Araz, which not only provides Azerbaijan with electric power but neighbouring states too. Besides these there are a number of local hydro-electric and fuel-driven power stations.

4. GENERAL GEOENVIRONMENTAL PROBLEMS

The diverse range of anthropogenic influences on already complex natural conditions has promoted the development of many complicated geoenvironmental situations across the large area that makes up the Republic of Azerbaijan.

By scale and intensity, the raising of the water level to the Earth's surface has an important role to play in many of these geoenvironmental situations. Water leakage takes place along all major and minor irrigation channels, the large part of which infiltrates the ground, raising the water level towards the surface. Flooding, swamping, and irrigation of arid areas promote secondary salinization. Over large regions, trap drainage systems have been obstructed and are not able entirely to drain infiltrating irrigation waters. Migration of contaminants, mineral compounds and organic fertilizers into the groundwater takes place. Flooding and swamping adversely affect the Kur-Araz lowlands in particular, areas with poor natural drainage. Flooding and swamping processes also raise anxiety in some areas of the Absheron Peninsula, including Baku and Sumgayit. Lands flooded particularly regularly are located on the outer part of the Baku trough, forming a chain of deep, wide, flat-bottomed valleys with shallow slopes, the valley bottoms being underlain by clays or loams. By their origin, flooding of the Yasamal, Chakhnaglar-Boyukshor, Bulbul-Karachukhur, and Zikh valleys can be attributed both to natural and artificial factors; oil-field runoff is the main artificial factor. This runoff, flooding a large area to the west and northwest of Lake Boyukshor, forms a temporary lake which remains separate to Lake Boyukshor. A similar picture is observed to the South of Lake Khoja-Hasan, around Lake Girmizi, and also at the Putin lakes to the north of Karadag station. Many areas in the vicinity of the Balakhani, Ramana, Sabunchu, and Surakhani oil fields have been flooded because of this runoff.

Much of the land in the centre of the Absheron peninsula, stretching from the Kurdakhan lakes to the Hovsan suburbs, has either already been flooded or is under threat of flooding. The size of this area is 21-22 km long by 1.5-5.0 km wide. Linked to the Bina-Hovsan trough, the cause of the flooding was initially oil field runoff and then leakage from the Absheron main channel. Over many years, runoff from the Buzovna-Mashtaga and Gala oil fields has flowed in the direction of the lower elevation areas of the Bina-Hovsan trough. Under the oilfields' runoff regime, wells drilled here in the 1950s demonstrate that a raising of the groundwater level occurred before the Absheron main channel was put into operation. Seasonal fluctuations are small, and water level variations are more strongly influenced by variations in climate. Following the commencement of the operation of the Absheron main channel and the beginning of irrigated agriculture, the water level began to rise quickly in the north-west from 1959-1960 onwards, and in the southeast from 1964-1966 onwards. The rise in the water level was 10-15 m between 1962 and 1998, and the rise is still continuing. This region is of particular importance due to its high population and economic significance. Large settlements such as Mashtaga and Bina and a number of villages are threatened by flooding.

The second largest flooded area is situated to the north of the Jeyranbatan reservoir. Having formed due to leakages from the Samur-Absheron channel and runoff from the industrial activities of Sumgayit, the flooded region stretches from the north of the reservoir to Sumgayit. Large areas within the city have been flooded because of leakage from communication networks and the basements of many buildings are under water. Chemical analyses show the origin of water taken from the basements and water pipelines or the sewerage system. According to its character, this flooded area differs from Bina-Hovsan. Whereas the lithologies of the Bina-Hovsan trough are represented by permeable limestones, sands, and sandy loams, in the Jeyranbatan-Sumgayit trough they are represented by sandy loams, loams, and, rarely, sands and sandstone. Besides this, the abundance of permeable deposits in the Bina-Hovsan trough is incomparably greater than in Jeyranbatan-Sumgayit. Relief also gives a different character to each area. The Bina-Hovsan trough is bowl-shaped whilst the Jeyranbatan-Sumgayit area is relatively flat and is flooded partly due to a lack of drainage across its borders.

Flooded basements occur in Baku. This can be very dangerous, resulting in a deleterious change in the geomechanical properties of the ground, weakening the foundations of buildings and other constructions. Basement flooding has led to incidents of buildings being rendered unusable and also of building collapse.

The rise in the level of the Caspian Sea from 1976-1977 brought flooding of the seashore, of beach areas, and the main coastal roads. Since 1998, some stabilization of sea level has been observed and this has reduced these flooding events.

Raising of the groundwater level and flooding have resulted in changes in microclimate, a rise in humidity in residential lodgings, a deterioration of sanitary conditions, the appearance of mosquitoes and other insects, and the infection of the population by different diseases. Flooding brings a halt to production in the largest industrial enterprises, can render land unusable for agriculture, and can affect gardens and municipal parks.

Lakes occupy a significant place in the environment of the Absheron region. Due to unfavourable natural conditions, all of them are salty and at present salt is extracted from many of them. Groundwater plays a considerable role in their replenishment. Lakes are formed because of industrial and communal runoff. One main source is the discharge of waste water from oil production. The Samur-Absheron and Absheron channels, irrigation water, and leakage from water pipes and sewerage systems also play a significant role in supplying the lakes. Most of the lakes in the peninsula have become a serious problem, complicating the geoenvironmental situation both hydrodynamically and hydrochemically.

In characterizing the environmental condition of the lakes the following points are relevant:

- unregulated effluent and water input from different sources promotes a rise in lake levels, and an associated expansion of surface area;
- in corresponding conditions, a rise in lake level promotes the recharge of groundwaters and increases in groundwater flow rates; in this regard Lake Ganli Gol presents a great danger;
- in natural conditions, all lake waters are salty and unusable;
- because of industrial and communal runoff flow, industrial water input and the infiltration of water used in irrigation, the lake compositions are enriched in various contaminants, including bacteria, to such an extent that the possibility of using purified lake waters is excluded;
- hazardous concentrations of chemicals and organic compounds and the bacteriological composition of lake waters exclude the possibility discharging to the sea without any corresponding purification (Alekperov, 2000).

Being an integral natural feature of Azerbaijan, the Caspian Sea presents an important natural geosystem influenced by geological, climatic, hydrological, and meteorological factors. Whilst not examining in detail the environmental problems of the Caspian Sea, which are not the subject of this investigation, it must be mentioned that fluctuations of sea level and anthropogenic factors play an important role in the environmental condition of the sea. It has been determined that a rise in sea level brings additional complexities to the complex hydrodynamic and hydrochemical conditions of coastal regions, as indicated in several ways:

- The rise in sea level has caused a rise in groundwater level in coastal areas;
- In order to prevent flooding a barrier over 2 m in height was constructed in some coastal areas, and this obstructed groundwater movement, decreasing the quantity of groundwater discharge into the sea, thus further promoting arise in groundwater level;
- Inundation by rising seawater levels resulted in changes in the chemical composition of the coastal groundwaters.

The other process complicating the geoenvironmental conditions is landslides, activated in the main by anthropogenic influences, such as slope cutting, flooding, construction work, and increased loading. Landslides are particularly hazardous within Baku, around the borders of the Mingachevir reservoir, along the Shamakhi-Ismailli highway, and in many settlements within the Great and Lesser Caucasus, the Nakhchivan Autonomous Republic, and the Talish Mountains.

Mud flows have a negative influence on geoenvironmental conditions, particularly in the Greater Caucasus and Nakhchivan regions: Sheki is under the constant threat of mud flows.

Three large reservoirs – the Mingachevir, Shamkir and Yenikend reservoirs – have been constructed along the 110-120 km length of the River Kur. Locally, primarily within the Karabakh, Mil, and Ganja-Kazakh groundwater basins, a considerable drop in groundwater level has been observed as a consequence of their intensive exploitation during periods of crop growth.

5. SUMMARY

Assessment of the natural geoenvironmental conditions of Azerbaijan, suggests that the Absheron Peninsula, where ground and surface waters suitable for drinking are practically absent, is distinctive in its geoenvironmental conditions. Over much of the Peninsula, landslides, subsidence, aeolian processes, abrasion, ravine formation, and solonchak development have occurred. Large areas have been flooded and swamped, there are many volcanoes, and over 200 salty and contaminated lakes.

Much of the Kur-Araz lowlands within the Mugan-Salyan and Shirvan plains, where there is also a lack of suitable groundwater, is also characterized by unfavourable geoenvironmental conditions. Soils and rocks of the unsaturated zone have become saline whilst vast areas have been swamped and flooded. In the east of the region, subsidence, salinization, solifluxion, ravine erosion, and mud volcanoes are widely developed.

Unfavourable geoenvironmental conditions in the Greater Caucasus are limited mainly to mud and landslide processes and also are prone to large earthquakes. There are useful groundwater reserves in the east.

Based on the large resources of usable, good quality groundwater with limited anthropogenic impacts, the geoenvironmental conditions of the large region of Sheki-Zakatali and the Samur-Gusarchay plains can be assessed as relatively favourable. According to the criteria used to assess anthropogenic influence on the environment, very intensive environmental degradation is limited to the Absheron peninsula. Here flooding and swamping of soils by irrigation systems, the Samur-Absheron channel, city housing constructions, and activation of exogenous processes such as landslides, all lead to the contamination of the ground and groundwater. Ground contamination within the oilfields is a serious problem for the

Absheron area, with incidents of oil products leaking from pipelines and then percolating into the ground.

Environmental degradation in the Kur-Araz lowlands is assessed as intensive due to flooding and secondary salinization from irrigation systems and the main channels. The intensity of environmental change caused by anthropogenic factors can be assessed as moderate over the rest of the region. Recently, large-scale works have been carried out in an attempt to remediate these ecological and environmental problems. Construction of the Main Mil-Mugan collector, which drains salty and contaminated groundwater within the Kur-Araz lowlands, has been completed. The collector provides a means of draining flooded areas, and preventing secondary salinization of the ground. Intensive reconstruction work on the Samur-Absheron channel will help minimize leakage from the irrigation system and improve water supply conditions in the northeastern part of Azerbaijan. Large scale works are also being carried out in the Absheron peninsula, particularly around Baku and Sumgayit, helping to reduce the risk of flooding and swamping, and also to remediate those areas contaminated by oil products.

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SOURCES OF GROUNDWATER SUPPLY TO URBANIZED AREAS IN AZERBAIJAN

National Development of Groundwater Resources

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Abstract: This paper focuses on the reasons underlying the continuing derogation of surface-water quality in Azerbaijan. Surface (river) water has been one of the main sources of fresh water. The major river systems used have their sources in neighbouring countries. There is, however, proven advantage of utilizing groundwater rather than surface water as a supply source for urbanized territories in Azerbaijan. The overarching reason being the greater natural protection of groundwaters from pollution relative to surface waters. An overview of water resources present and contamination problems observed is presented. A case for increased groundwater supply in Azerbaijan is presented as a solution to these challenging problems.

Key words: Azerbaijan; water resources; surface water supply; pollution; monitoring; international boundaries; transboundary issues

1. INTRODUCTION

With rapid urbanization, water users have increasingly noticed a reduction in supplies of naturally clean water. More than one in four people now consume water that does not meet sanitary standards. The environmental impacts of anthropogenic activity upon water resources in particular have been significant. Mortality caused by infectious diseases transmitted directly by poor quality drinking water has increased among urban populations in recent decades, even in European countries, not to mention countries with arid climatic conditions. During the 20th century, International Hydrological Decades (IHD) were frequently declared by UNESCO or UN for the purpose of water resources preservation. A general

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programme for establishing monitoring systems at representative river basins, including groundwater basins, is now being proposed. Data collated from monitoring results obtained both globally and within individual countries have allowed regional patterns of water-salinity balance in both surface and underground waters to be determined. The scientifically soundest measures have been proposed for response to declines in water quality, especially those caused by factors associated with urbanization. Anthropogenic impact on the natural environment is frequently underestimated that may ultimately have catastrophic consequences.

In the Republic of Azerbaijan, complex studies were also carried out on the basin of the River Gyandjachay (representative of basin type II within the IHD classification), which is located in the central eastern part of the Lesser Caucasus. Though Azerbaijan is several times larger than its neighbours Armenia and Georgia, its total surface-water resources are four times less than those of Armenia and seven times less than those of Georgia. Azerbaijan is situated at the downstream end of the River Kura and, consequently, over 72% of its water resources are formed outside its borders. Thus, in terms of the quantity and quality of waters reaching its territory, Azerbaijan is dependent on its neighbours located within the river basin.

The surface-water resources of the republic (29.2 km^3/year) are not sufficient to meet its general water requirements, particularly sanitary and potable water demands. Water resources within the republic are not unsurprisingly irregularly distributed with groundwater in some locations forming the main, or indeed only, source of water supply. This paper outlines the water resources present in Azerbaijan and contamination problems present are overviewed. A case for groundwater supply is presented as a solution to these problems.

2. WATER DISTRIBUTION IN AZERBAIJAN

Both surface water and groundwater are distributed irregularly within Azerbaijan (Aliyev, 2000; Listengarten, 1983). Over large areas, they can be highly mineralized or practically absent. This pattern is a consequence of complicated geomorphological conditions, geological structures, lithological variations in the upper levels of aquifers, the origin of water-bearing rocks, and climatic aridity. Approximately 60% of the country lies within mountainous areas, while 40% is in lowland-piedmont areas. The altitude of the mountainous areas reaches 4480 m (Mountain Bazar-Duzi) in the Great Caucasus, 2993 m (Mountain Murguz) in the Small Caucasus, 2500 m in the Talysh Mountains, and 3902 m (Mountain Kapydjyg) in the

Nakhchyvan part of the Small Caucasus. The lowest parts of the Kura-Araz Lowlands are 3-10 metres below sea level, with parts of the Caspian zone being 26-29 m below sea level. The climate is sharply continental, from mountain tundra to arid and semiarid climate, and altitudinal zoning is observed in relation to precipitation that ranges from 150-200 mm to 900-1600 mm annually. Air temperature varies from sub-zero to 35-45ºC, with an annual average value of 10-14.5ºC. River drainage density varies from 202.2 km/km^2 in mountainous areas to 0.1 km/km^2 in the lowlands (Fig. 1).

The geology of Azerbaijan is formed of Mesozoic, Cainozoic and modern deposits, in which underground waters, from fresh to saline, are formed with some waters having thermal or mineral water designations. Resources of the latter have facilitated development of sanatorium-resort zones and hydropathical and mineral water bottling plants, their use as alternative energy sources, and industrial production of iodine, bromine and other micro-elements and salts for export.

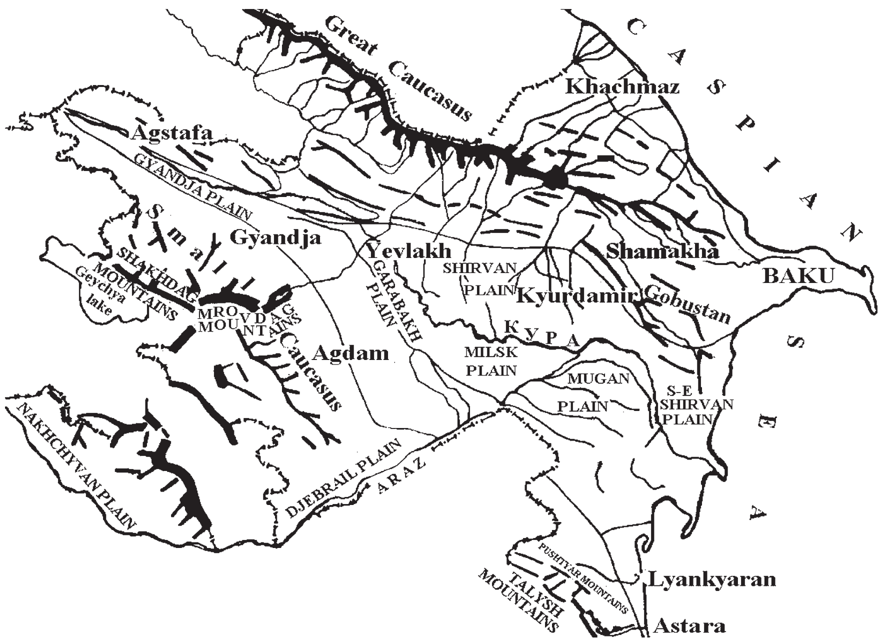

Figure 1. Sketch map showing main geographical features of Azerbaijan.

The main resources of pure and low-mineralized groundwater are concentrated in sedimentary deposits of piedmont plains, fractured and vein-fissured rocks of mountainous regions, and alluvial deposits of river valleys. Further details of these resources are provided later in Tables 5-8.

3. WATER RESOURCES

In Azerbaijan, as in other countries, the original choices of settlement locations were influenced by the availability of permanent water sources, including the proximity of rivers, springs, and groundwater outlets. However, local water resources have gradually become insufficient to meet population requirements. Factors include the urbanization of territories, increasing populations, establishment of industrial areas. Hence, additional sources of water supply have been, and continue to be, sought. Water supply to the population of urbanized territories in Azerbaijan is currently undertaken via the following:

- Local surface water resources;
- Local groundwater resources;
- Local groundwater and influent surface waters;
- Local subsurface waters, influent surface waters, and groundwater.

Along with an improvement in living standards and the increase in water requirements in urbanized areas, the ecological and geoenvironmental situation has become more complex. Areas affected include Greater Baku, Sumgayit, the Absheron Peninsula as a whole, and other large towns of the republic. The River Kura and River Araz play a major role in the water supply to many towns.

The River Kura has a total length of 1515 km, of which 906 km is in Azerbaijan. It has a total catchment of 188,000 km^2, and is the arterial water course of the republic. Its waters are used for the supply of sanitary and potable water to populated areas in the Central and Lower-Kura regions, as well as for irrigation of lands on the Gyandja Plain and the Kura-Araz Lowlands. Water is taken from the River Kura by canals, such as the Main-Shirvan Canal ($L = 210$ km; $Q = 78$ m^3/s), Upper Karabakh Canal ($L = 178$ km; $Q = 113$ m^3/s), Main Mugan System ($L = 191$ km; $Q = 134$ m^3/s) and other canals of minor importance, as well as by pump installations. The area of lands irrigated by the River Kura and its tributaries is some 2.2 million ha. Water taken from the River Kura is used for centralized water supply to the towns of Mingechevir, Yevlakh (also partial groundwater supply), Zardab, Gadjigabul, Alibayramly, Salyan, and Neftchala and to Baku City (via the First and Second Kura Water-Intake Systems). Water taken from its main tributary, the River Araz, is supplied to the towns of Imishli (also partial groundwater supply) and Sabirabad.

4. CONTAMINATION PROBLEMS

The urbanization of towns within the Kura basin and the development of irrigated agriculture using fertilizers and pesticides have contributed to an increase in demand for potable water and water for industrial and agricultural purposes, which has caused a lowering of water quality within the basin. Since infiltrated surface waters and infra-bed waters in the head regions of fans of the Kura's tributaries are the main contributors to water resource formation in the piedmont plains of the Kura Depression, the importance of protecting these sources from contamination is paramount. The River Kura is a regional drain for the unconfined aquifers and, in some cases, confined aquifers along its length. Its total catchment area of 183,000 km^2 is truly international and distributed as follows: 24,000 km^2 in Turkey, 24,500 km^2 in Iran, 36,400 km^2 in Georgia, 29,800 km^2 in Armenia, and 68,900 km^2 (downstream) in Azerbaijan. The main source of river contamination is water discharged by towns situated within the transit zone of the rivers Kura and Araz and of their tributaries. Of these towns, Borjomi, Khashuri, Tbilisi, and Rustavi (in Georgia) and Dilijan, Gafan, and Kadjaran (in Armenia), which discharge untreated water into the Kura and Araz, and Mingechevir, Yevlakh, Zardab, Sabirabad, Imishli, Salyan, and Neftchala, which discharge partially treated water, are of particular importance. The latter five towns in the list are provided with canalisation to the level of max. 30-40%.

The greatest contamination of river waters was observed over 1970-90 that coincided with a period of increased urbanization and development of irrigated agriculture. After the collapse of the former USSR, many plants and factories closed, and agricultural areas used for producing cotton, grapes and other crops have reduced. Despite this, waters of the River Kura presently fail to meet the requirements of the appropriate potable water criteria for some contaminants, even at the border with Georgia as shown by the Shikhly water sampling data (Tables 1 and 2).

According to data from water-protection institutions in Azerbaijan, Georgia, and Armenia, the total volume of water discharged into the River Kura during 1992 was estimated at ~575 million m^3 with the following allocation: Armenia - 300 million m^3 (52%); Georgia - 250 million m^3 (43%); and Azerbaijan - 25 million m^3 (5%). These values reduced during 1992 due to economical difficulties associated with the maintenance of existing plants and factories, and the total volume of water discharged in the Kura basin was 453 million m^3 (~120 million m^3 less). The respective discharged volumes were reduced to 212 million m^3 in Armenia, 229 million m^3 in Georgia, and 12 million m^3 in the Azerbaijan Republic.

Table 1. Concentrations (mg/l) of N and P substances along the River Kura, Azerbaijan in 1999.

Location	NH_4	NO_2	NO_3	N_{total}	Mineral P	$P_{total.}$
Shikhly	0.1	0.31	7.72	7.76	0.61	0.71
Yenikend	0.1	0.37	4.15	4.19	0.058	0.158
Mingechevir	0.06	0.018	2.81	2.88	0.054	0.182
Yevlakh	0.09	0.024	2.85	2.88	0.077	0.205
Piraza	0.09	0.011	2.67	2.68	0.075	0.175
Zardab	0.06	0.011	1.68	1.7	0.081	0.258
Mollakend	0.06	0.008	1.41	1.45	0.062	0.239
Surra	0.06	0.048	2.81	2.88	0.036	0.181
Alibayramly	0.12	0.021	2.98	3.04	0.067	0.183
Salyan	0.1	0.009	3.35	3.71	0.059	0.202
Banke	0.09	0.028	3.08	3.14	0.076	0.209
Mayak	0.08	0.015	2.82	2.84	0.055	0.189

Table 2. Concentrations (μg/l) of various elements along the River Kura, Azerbaijan in 1999 (< = less than detection limit).

Location	Fe	Si	Cu	Zn	Al	Mn	Ti	Bi	Hg
Shikhly	0.18	8.7	11	7	10.4	17.8	15.8	5	<
Yenikend	0.18	7.4	18	10	15.8	11.2	5	5.8	<
Mingechevir	0.19	5.6	18	12	10	7.4	13.4	2.2	<
Yevlakh	0.28	4.3	14	7	6.4	6.4	6.9	<	<
Piraza	0.26	6	12	7	10.8	10	7.9	<	<
Zardab	0.17	6.6	20	13	11.2	10	15.8	3.2	<
Mollakend	0.23	5.8	12	14	15.8	11.2	11.8	3.2	<
Surra	0.43	7.8	10	10	13.2	12.6	8	4	<
Alibayramly	0.2	7.4	19	14	10	7.2	14.2	<	<
Salyan	0.33	7.4	14	14	13.6	15.8	11.2	3.2	<
Banke	0.26	8.1	13	12	15.8	14.7	13.4	11.8	0.7
Mayak	0.23	6	23	13	8.6	10.7	12.6	3.6	1.4

More than 72% of the established water resources in Azerbaijan are formed by inflows from bordering countries and, as such, presents the republic with a complex situation, ecologically and hydrologically. For example, the annual discharge of sewage waters into the basin of the River Kura and its tributaries (e.g. the Araz, Okhchichay, Razdan, Agstafachay, Alazan (Ganykh) and Iori (Gabbry)) causes ecological damage. This has lead to social, sanitation, and water consumption problems.

The monitoring network of the Ministry of Ecology and Natural Resources provides regulatory control and prevention of contamination of surface and groundwater resources in the republic. Analysis of surface waters typically provides data on dissolved oxygen, nitrite, nitrate, phosphate, silicon, calcium, magnesium, sodium, potassium, sulphate, chloride, bicarbonate, oil products, detergents, phenols, heavy metals, pesticides, and suspended particles, as well as on the biochemical consumption of ammonium and the total mineralization of the water (Tables 1 & 2, and 3 & 4).

Table 3. Concentrations or organic chemical groups in surface runoff of the River Kura, Azerbaijan in 1999 (SSAS – synthetic surface-active substances (surfactants)).

Location	Dichromate COD (mg/l)	BOD (mg/l)	Oil products (mg/l)	Volatiles (mg/l)	Non-volatiles (mg/l)	SSAS (mg/l)
Shikhly	18.1	3.71	0.15	0.08	0.002	0.03
Yenikend	25.1	3.95	0.17	0.008	0.004	0.03
Mingechevir	19.4	2.91	0.18	0.01	0.005	0.05
Yevlakh	11.3	2.17	0.24	0.04	0.004	0.05
Piraza	19.8	1.83	0.07	0.006	0.003	0.03
Zardab	13.5	2.07	0.03	0.01	0.005	0.04
Mollakend	14.9	2.0	0.11	0.005	0.004	0.03
Surra	22.6	4.1	0.25	0.01	0.003	0.06
Alibayramly	18.9	3.34	0.15	0.10	0.006	0.08
Salyan	21.6	3.43	0.24	0.007	0.004	0.08
Banke	23.2	4.0	0.30	0.01	0.009	0.08
Mayak	18.7	3.2	0.19	0.011	0.011	0.06

Table 4. Specific organic compounds concentrations in surface runoff of the River Kura, Azerbaijan in 1999 (DDT – Dichlorodiphenyltrichloroethene; HCCH – hexachlorocyclohexane (also known as lindane)).

Location	DDT (μg/l)	HCCH (μg/l)	Benzene acid (μg/l)	Furfural (μg/l)
Shikhly	<	0.006	<	<
Yenikend	<	0.013	1	<
Mingechevir	<	0.037	<	<
Yevlakh	<	0.015	<	<
Piraza	<	0.011	<	<
Zardab	<	0.013	<	<
Mollakend	<	0.005	<	<
Surra	<	0.031		<
Alibayramly	<	0.025	<	0.54
Salyan	<	0.031	<	<
Banke	<	0.048	2.4	<
Mayak	<	0.031	2	<

The analysis and statistical treatment of results obtained from laboratory studies over many years has enabled the following classification of river water in Azerbaijan to be produced (using the methodology of the CIS Hydrometeorological Service):

1. Conditionally clean waters, with the total concentration of contaminants not exceeding 1 maximum allowable concentration (MAC);
2. Moderately-contaminated waters, with total concentration of contaminants from 1 to 3 times the MAC;
3. Contaminated waters with total concentration of contaminants from 3 to 5 times the MAC;
4. Waters with increased contamination, with total concentration of contaminants from 5 to 10 times the MAC;
5. Highly-contaminated waters, with total concentration of contaminants exceeding 10 times the MAC.

It should be noted that increased contaminant concentrations are typically observed in rivers during low water periods, when river drainage is composed mainly of anthropogenic discharges and low groundwater baseflow. Assessment of surface water contamination in Azerbaijan has been undertaken via the above classification and indicates:

- Waters of the River Kura: have increased contamination (5-10 MAC) at the Georgian border (Shikhly post); are moderately contaminated (1-3 MAC) from the border to the town of Mingechevir (including the water reservoir with the same name) due to self-cleaning processes and an absence of large facilities in the transit zone; are contaminated (3-5 MAC) in the middle section of the river (between Yevlakh and Surra); and, have increased contamination (5-10 MAC) in the downstream region (after the inflow of the River Araz).
- Waters of the River Araz: are highly-contaminated (10-17 MAC) at the Armenian border; and contaminated (3-5 MAC) at the town of Saatly.
- Contamination of water in the rivers Okhchichay and Akerachay, which have their sources in the territory of Armenia, is in the range from increased (5-10 MAC) to high contamination (10-17 MAC).
- Water from rivers on the southern slopes of the Great Caucasus are conditionally clean to moderately contaminated (1-3 MAC).
- River water on the north-eastern slopes of the Lesser Caucasus (Tovuzchay, Shamkirchay, Gyandjachay, and Goshgarchay) range from contaminated (3-5 MAC) to increased contamination (5-10 MAC).
- Water from rivers on the north-eastern slopes of the Great Caucasus and Talysh Mountains range from conditionally clean to moderately contaminated.

It is concluded from the water-quality assessment that river waters in Azerbaijan are suitable as sources of domestic water supply only within mountain areas. There are significant river water-quality problems throughout other areas.

Upstream sections and tributaries of the River Kura in Armenia and Georgia transport a large quantity of chemical compounds, some harmful contaminants, into Azerbaijan. According to the Hydrometeorological Service of the Azerbaijan Republic, these inflowing river arteries carry over 7662 tonnes of dissolved chemical compounds, 6060 tonnes of suspended substances, 4-5 tonnes of oil products, 350 tonnes of phenols, and up to 300 tonnes of metal compounds (Tables 1 & 2, and 3 & 4). More than 60% of these harmful substances are found in the River Kura, 25% in the River Araz, and the remaining 15% in the rivers Ganykh, Iori, Akstafachay, and Okhchichay. These values demonstrate the ecological strain placed on the catchment basin of the River Kura and show that its drainage is not suitable for meeting sanitary and potable requirements of towns and populated rural areas in Azerbaijan. Furthermore, an increasing number of acts of sabotage and terrorism suggest that there is little prospect of this situation improving.

5. A CASE FOR GROUNDWATER SUPPLY

To some extent, all the factors discussed above are prevalent in the larger river basins of the world, such as the Volga, Ural, Dnepr, Syr-Darya, Danube, Amazon and Nile. The problem of contamination can be solved only by the collaboration of bordering countries and international authorities. Hence, in addition to new technologies and projects, a new ecological monitoring system needs to be established. This requires implementation of mutually relevant controlling conditions, mechanisms for risk evaluation and minimization, and facile exchange of monitoring materials/methods and data collated. No individual country is able to, and nor should it be able to, accept ecological damage to the environment, especially in respect to contamination of water supply sources. In recent years, Azerbaijan has been unable to remedy the long-established situation, rendering its surface water resources to an almost unsuitable condition.

Within this present context of strained surface-water resources, it is pragmatic, along with water quality (derogation) and volume (depletion) protection initiatives, to implement a complete transition to the use of groundwater resources in that they are much more protected from contamination. Groundwaters may form a significant supply of water to urbanized areas that are under significant strain.

Uncontaminated groundwater has been one of the main water supply sources in Azerbaijan since ancient times. Prior to construction of the water-intake systems for the Shollar water-supply network (1909-1917), the first such network in Baku, recovery and use of groundwater was carried out, along with the capping of springs and construction of karizes (subterranean canals) and wells. The total number of karizes and wells in the republic exceeded 1700 and 25,000 respectively. Necessarily, the rates of urban development required the construction of centralized water-supply and canal systems, especially in towns and urban villages. Unfortunately, not all towns in Azerbaijan are provided with sufficient water volumes per head at present, and the quality of the water consumed does not always meet sanitary requirements. In many towns, only 30-60% of residential buildings are covered by the canalization system.

The Government of Azerbaijan have adopted programmes for the improvement of urban water supply and canalization systems. Over 60% of the population of Azerbaijan live in urban areas. Besides the large urbanized territories, such as Greater Baku, Sumgayit, Gyandja, Mingechevir, and Alibayramly, with respective populations of 1,839,800, 290,700, 303,100, 95,100, and 72,100, there are 23 towns with a population of between 20,000 and 100,000, and 25 towns with a population of between 5000 and 20,000.

One of the most difficult problems in the selection of water supply sources is convincing officials and investors, who do not always take decisions appropriate to guaranteeing water quality within the lifetime of a water-intake system. In Azerbaijan, there are many who advocate the use of water from the River Kura and other rivers to supply water for towns such as Baku and Sumgayit. River systems play a regional role in the drainage of surface waters and groundwater and, if territories adjacent to catchment basins are developed, the rivers are likely to become contaminated. In the case of a potential oil-spill emergency along the Western Export Pipeline or Baku-Tbilisi-Ceyhan Main Export Pipeline, caused by natural and/or anthropogenic factors (e.g. sabotage), it is not guaranteed that the waters of the River Kura would not become contaminated. Water supply to populations by centralized water-intake systems, especially in towns, must be based on sources that are reliably protected from both long- and short-term contamination.

All large towns in Azerbaijan are situated in piedmont or lowland zones. As mentioned above, the recoverable groundwater reserves, which are found in Quaternary sedimentary deposits, have been studied in detail

(Aliyev et al., 1983), both regionally and within individual groundwater basins (Table 5). They may form a major supply and significantly alleviate current difficulties faced.

As in other arid regions of the world, most towns and villages in Azerbaijan are situated alongside rivers and often occupy the upper parts of their alluvial fans. Over time, these towns have expanded and become potential sources of regional contamination of pure groundwater. Local groundwater resources confined within boulder and gravel deposits (see Table 6) are presently the main sources of water supply to these towns. Rocks forming the overlying beds have good infiltration properties (Figure 2) and their protection from contamination is very difficult under current natural and anthropogenic conditions. In addition to taking certain measures in the construction of facilities producing solid or liquid wastes and their storage and transportation, it is necessary to establish a special-regime monitoring network within zones of their perceived geographic impact.

Table 5. Recoverable groundwater reserves of pure (up to 1 g/l) and low-mineralized (1-3 g/l) water, Azerbaijan (as of 01.01.2004)(thousands of m^3/d).

Aquifer	Expected Resource	Resource approved by SCR	Demand for sanitary and drinking	Demand for irrigation and industrial
Greater Caucasus mountains	1,008.87	31.0	7.8	12.4
Absheron Peninsula	241.92	0.3	0.4	0.5
Gobustan	Not assessed	9.8	9.8	1.9
Samur-Devechi Plain	3,470.72	1,686.1	375.6	31.1
Ganykh-Agrichay Valley	3822	2,000.0	32.7	263.3
Gyandja Plain	4,218.6	4,218.6	91.3	751.3
Shirvan Plain	517.7	517.7	20.5	14
Garabakh Plain Milsk Plain	7,909.9	1,857.9 408.7	63.9	1,212,3
Mugan Plain (Talysh zone)	130	76.0	7.2	5.8
Lesser Caucasus mountains	989.4	98.9	23	N/A
Djebrail Plain	344	234.6	occupied by	Armenia
Lenkoran Plain	209	86.0	13	48.3
Nakhchyvan Plain	902.2	902.2	56.1	85.9
Total	23,764.3	1,2127.8	701.3	2,426.8

Episodic studies, carried out during searches for and exploration of groundwater resources, showed that there were no local areas where the water content of individual chemical components exceeded permissible levels. Even so, the establishment of a strictly enforced first sanitary protection zone is a necessity at every water-intake system. Within Azerbaijan, only water-intake systems and the route of the first (Shollar) and second (Khachmaz) Baku Water-Supply Networks have a sanitary protection zone; in the other cases (e.g. Gyzyl-Gaya, Malbinasi, Fizuli) there is only wire fencing at best.

Local studies determined that, under natural conditions, groundwater within piedmont plains is not generally protected from the migration of contaminants from the surface. This is a consequence of the lithological composition of rocks in the aeration zone, which are typically gravels and shingles, well-sorted sands, and sandy loams, while clay layers, which are encountered in the peripheral zones of fans, are of insignificant thickness. However, the thickness of impermeable clay deposits between groundwater aquifer units and confining horizons exceeds 10-20 m, and, in almost all confined areas of development, piezometric levels are such that upward flows in monoclinal structures, are established leading on occasion to artesian (spontaneous outflow) conditions (Figure 3).

Table 6. Centralized water-intake systems in Azerbaijan, operating by using river underflows designed to meet town needs for sanitary and potable water.

River Valley	Number of drainage systems	Towns using water	Capacity of drainage systems (l/s)
Gusarchay	2	Gusar	44
Gudialchay	4	Guba	52
Girdimchay	1	Kyurdamir	45
Gurmukhchay	1	Gakh	81
Kishchay	2	Sheki	18
Gumbulchay	2	Belakan	9
Geychay	2	Geychay	85
Gyandjachay	1	Gyandja	78
Gargarchay	2	Agdam	52
Gargarchay	1	Gadjibedi	35
Shamkirchay	1	Shamkir	35
Terterchay	2	Agdere	7
Turianchay	2	Agdash	81
Pirsaatchay	1	Shamakhy	110
Kendalanchay	1	Fizuli	8
Nakhchyvan	4	Nakhchyvan	110
Lenkoranchay	3	Lenkoran	24
Astarachay	1	Astara	19

Active unconfined groundwater flow systems therefore play an important role in protecting confined aquifer units. Nevertheless, penetration of contamination still occurs to some degree. Predominant groundwater flow is limited to the head regions of fans, where groundwater flow systems are not yet divided by clay units to unconfined and confined conditions. Maps of natural groundwater protection have been issued (Aliyev, 2000; and others) on a scale of 1:200,000 for all piedmont plains, and for the Samur-Devechi and Lenkoran plains the maps were drawn to a scale of 1:100,000. Such maps are the basis for establishing a system of hydrochemical monitoring of groundwater.

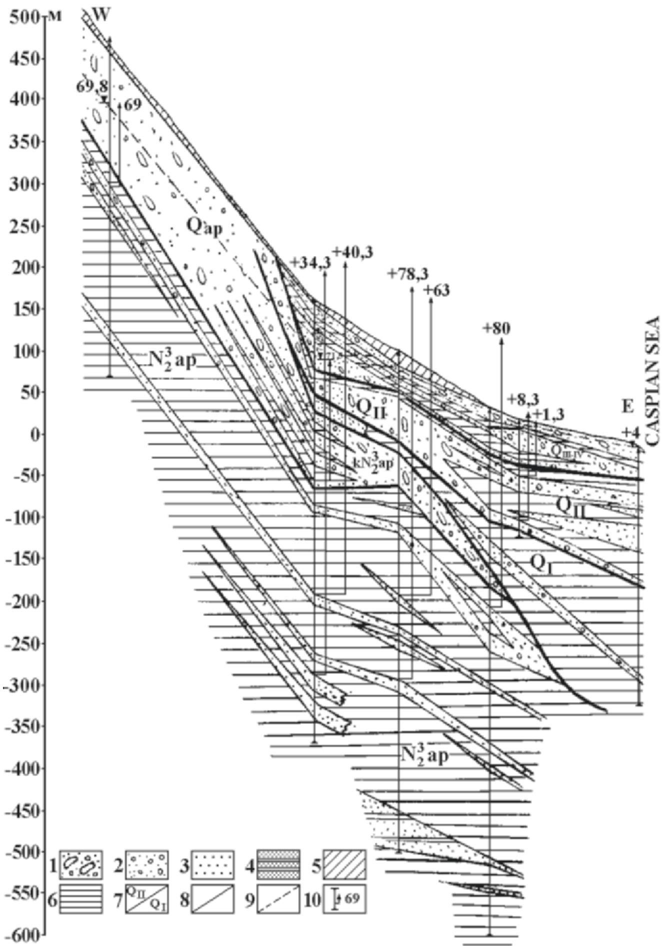

Figure 2. Hydrogeological profile along a line from Gusar through Khudat to the Caspian Sea, Azerbaijan. Lithologies as follows: 1. Gravel and shingle deposits with sand inclusions; 2. Shingles with sand inclusions; 3. Sands; 4. Alternating sands and clays; 5. Loams; 6. Clays; 7. Stratigraphical units and boundaries; 8. Lithological boundaries; 9. Groundwater levels; 10. Wells. Triangles signify groundwater depth, in metres below sea level; crosses indicate piezometric water level, in metres above sea level.

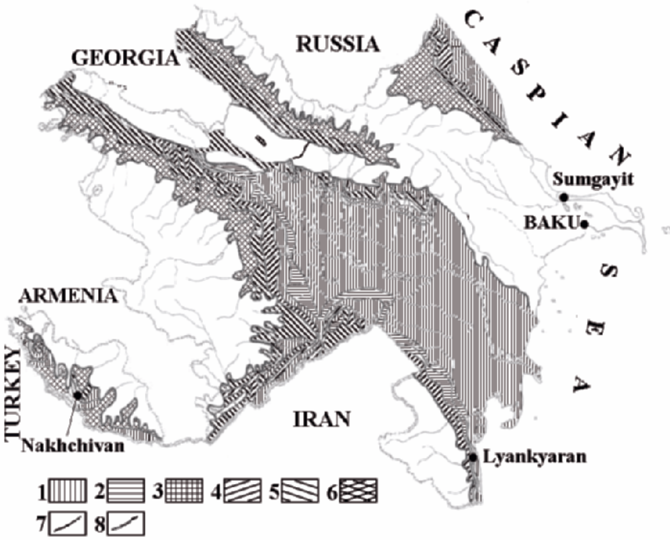

Figure 3. Lithological composition of aquifers in continental deposits of low-lying regions, Azerbaijan. Variations in gravel content of the water-containing rocks is as follows: 1 = practically absent; 2 = up to 25 %; 3 = 25-50%; 4 = 50-75%; 5 = 75-90%; 6 = 90-100%. Boundary keys as follows: 6 = areas with different shingle content in the section; 7 = alluvial-proluvial and lowland plains).

Under conditions of intensive urbanization, the risk of negative impacts on the environment and its constituent part, the underground hydrosphere, increases with the greater demand for sanitary and potable water in towns. Moreover, rational production and use of uncontaminated water requires a careful approach to ensure preservation of not only the resources, but also their quality. This is necessary in Azerbaijan as, in the near future, groundwater resources will potentially become the only reliable source of water supply to towns and populated rural areas. Historical groundwater use in Azerbaijan involved water recovered by karizes in the 14th and 15th centuries and by wells in more ancient times.

From the 20th century, major well fields were developed for the construction of water-intake systems for the First Baku Water-Supply Network in 1909, at the north-eastern margin of the Samur-Devechi Plain, near Shollar. Many wells were drilled during the last 50-70 years. The number of wells used for the recovery of groundwater exceeds 15,000 at present. Regular monitoring of the production and use of both natural ground and surface sources, has been carried out in Azerbaijan since 1976, while reserves of pure and low-mineralized waters have been approved since 1969.

Studies to forecast water reserves have been undertaken in the piedmont plains (Aliyev et al., 2001), mountain and other piedmont zones. The expected groundwater reserves (as of 2004) are 23.8 million m^3/day, while the reserves approved by the State Committee for Reserves are 12.1 million m^3/day (Table 5). Only 17-18 % of recovered groundwater is used for sanitary and drinking needs, with the remainder used for irrigation (Figure 4). High groundwater use for irrigation and other industrial requirements is a direct consequence of the inability of local surface-water resources to meet demand. Analysis of water supply to urban populations shows that large periods have developed where no water can be supplied, and in many cases water is supplied according to a timetable, with a maximum 2-3 hours supply in the morning and evening.

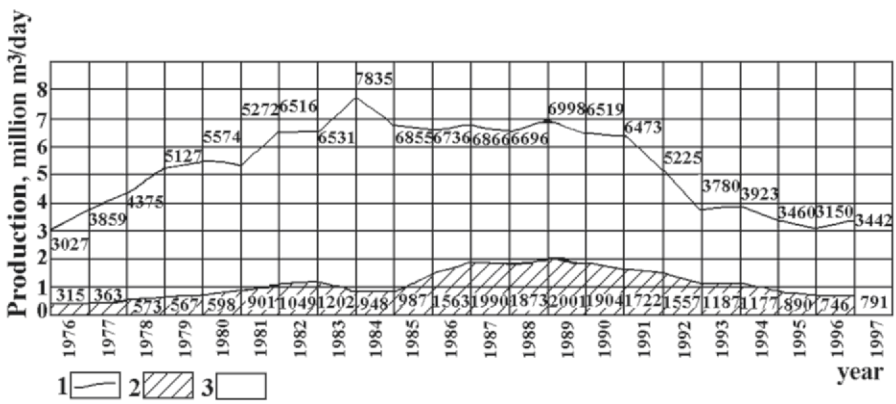

Figure 4. Changes in production and use of groundwater (million m3/year) in Azerbaijan, 1976-1997. 1 = quantity of groundwater extracted; 2 = quantity used for drinking and industrial needs; 3 – quantity used for irrigation.

The groundwater reserves explored and exploited are sufficient to meet sanitary-potable water requirements using centralized water-intake systems (Table 7), even considering population increases and urbanization. The reserves given in Table 6 consider water requirements of populated areas until 2015. Since the collapse of the USSR, design and construction of new water-intake systems has terminated. Moreover, existing designs and documentation (including those on the exploration and assessment of groundwater reserves) were lost upon liquidation of the design institutes (Bakvodokanalproyektand Yuzhgidroselkhozvodosnabzheniye). Centralized water-intake systems which recover groundwater from fans within piedmont plains, such as Shollar, Khachmaz, Gusar and Fizuli, are not the only such systems operating in Azerbaijan. The republic also has a number of water-intake systems that function using river underflows (Table 7).

Table 7. Prospective areas for centralized groundwater systems for recovery of groundwater, where respective recoverable reserves are approved by the State Commission for Reserves.

Aquifers	Supply for	Reserves, 1000 m³/day	Current Status
Quaternary sedimentary deposits Samur-Gusarchay area	Baku, Sumgayit, and Absheron	1,686	The Water-Intake System III of the Baku Water-Supply Network suspended from 1980.
Quaternary deposits of the Ganykh-Agrichay Valley – fans of the Rivers Dashagylchay, Talachay, Gyungyutly, Garachay, Akhokhchay, and Gurmukhchay	Towns Balaken, Zakatala, Sheki, G Rural areas: Gazakh District, Sheki District, Oguz District abala, and Ismailly	Respectively: 19.3; 35.5; 80; 15.2, and 11.3. 13, 48.7, and10.5	Centralised water-intake systems have not been designed, and waters are recovered by decentralised wells (towns) or individual wells (rural).
Quaternary deposits of the Gyandja Plain: Molla Djafarly, Ashagy Ayibly, Garayeri, Dalimamedly, Muzdurlar.	Population of rural districts: Gazakh-Agstafa, Tovuz, Khanlar, Geranboy	Respectively: 33.4, 31.2, 44.9 and 20.5	Water-intake systems have not been designed, and individual sub-artesian wells are functioning
Quaternary deposits of the fan of the River Gyandjachay – Gyandja	Population of the town Gyandja	200.8	More than 200 decentralised sub-artesian wells and karizes functioning
Underflow waters of alluvial deposits of the River Dzegamchay	Villages Yapykhly and Alakel of the Tovuz District	23.8	Water-intake systems have not been designed
Jurassic fracture-matrix flow deposits around town of Khankendi	Population of the town Khankendi	9	Operated by individual wells
Underflow waters of alluvial deposits of the River Gargar	Population of the towns Khankendi and Khodjaly	39.3	Water-intake systems not yet designed.
Quaternary deposits of the fan of the River Gargarchay	Population of the town Agdam	38.4+14.7 = 52.4	Operated by individual wells
Quaternary deposits of the Garabakh Plain in Gasanchay, & Kalantarly.	Population of Agdere town; rural population of Terter and Barda districts	42.1 45.5	Water-intake systems have not been designed.
Quaternary deposits of the Milsk Plain	Population of the town Beylagan	72	Partly by individual sub-artesian wells
Quaternary deposits of the Milsk Plain (Araz zone)	Population of the town Fizuli	37	Operated below full capacity
Groundwaters (Quaternary) of the Batabad Urotchishche	Population of the town Nakhchyvan	24.3	Water-intake systems not yet designed.
Quaternary deposits of the fan of the River Eastern Arpachay, Nakhchyvan Plain	Population of the town Nakhchyvan, populated areas of Sharur and Babek districts	63.8	Centralised intake systems not yet designed. Individual wells are functioning.

The Government of Azerbaijan, with local specialists, has commenced design of water-supply systems and the construction of canals in the towns of Geychay, Agdash, and Nakhchyvan. Funding is from the Asian Bank of Development. Good quality groundwater reserves in fans of the rivers Turanchay and Geychay has been selected to supply the towns of Agdash and Geychay, with River Nakhchyvan alluvial deposits selected for the town of Nakhchyvan. The water supply scheme to towns in Azerbaijan is outlined in Table 8 that forms the basis for selection of water supply sources by the Government of Azerbaijan and foreign investors.

Table 8. Requirements (until 2010) and potential sources (by use of groundwater) of water supply to towns and villages in Azerbaijan.

Towns/ Villages	Water Use 2004 (m^3/d)	Requirement to 2010 (m^3/d)	Proposed potential sources of water supply
Greater Baku, Sumgayit	306,849 145,000	1,784,054	Samur-Gusarchay interstream area (9 m^3/s) and Oguz; Gabala District, Ganykh-Agrichay Valley (15 m^3/s)
Gusar	1,560	31,050	Underflow waters of the River Gusarchay and Samur-Gusarchay interstream area
Guba	7,640	26,600	Gesher springs and underflow waters of the River Gudialanchay
Khachmaz	12,100	29,960	Underground waters of the Samur-Gusarchay interstream area
Devechi	N/A	25,125	Underground waters of the Samur-Devechi Plain
Siyazan	N/A	66,060	Underground waters of the Samur-Devechi Plain
Khyzy	N/A	12,000	Capping of springs in the Khyzy zone and underground waters of the town territory
Shamakhy	4,000	158,450	Underflow waters of the River Pirsaat and underground waters of the Ganykh-Agrichay Valley
Belakan	1,300	19,300	Underground waters of the Ganykh-Agrichay Valley – 1 km to the south of the village Katsbina
Zagatala	3,700	35,470	Underground waters of the Ganykh-Agrichay Valley, which are situated in the vicinity of the village Ashagy Tala
Sheki	3,400	58,770	Underground waters of the Ganykh-Agrichay Valley – in the vicinity of the village Gabar Zeyzit
Gakh	1,500	19,300	Underground waters of the Ganykh-Agrichay Valley, the northern part of the location of the village Embirdjan
Oguz	1,400	6,875	Underground waters of the Ganykh-Agrichay Valley – the fan of the River Dashagyl (near the village Budjag)
Gabala	1,380	15,185	Underground waters of the Ganykh-Agrichay Valley – the fan of the River Damiraparanchay
Ismailly	8,200	11,275	Underground waters of the northeastern part of the town Ismailly
Gazakh	6,470	37,010	Underground waters of the Gyandja Plain – the fans of the River Agstafachay, 3-4 km to the northeast of the town Gazakh
Agstafa	1,900	8,170	Underground waters of the Gyandja Plain – the fans of the River Agstafachay (the western margin of Agstafa)
Tovuz	3,500	30,600	Underground waters of the town territory (Gyandja Plain) and underflow waters of the River Tovuzchay
Shamkir	1,900	50,500	Underground waters of the Gyandja Plain and territory of the town Shamkir
Khanlar	2,400	35,600	Underflow waters of the River Gyandjachay
Gyandja	114,000	352,515	Underground (fans) and underflow waters of the River Gyandjachay and also surface waters of the River Agsu
Geranboy	3,500	10,125	Underground waters of the fan of the River Kyurakchay
Naftalan	8,600	28,600	Underground pore waters of the northeastern part of the settlement
Mingechevir	4,657	193,145	Underground waters of the Ganykh-Agrichay Valley
Agdash	7,800	32,300	Underground waters of the fan of the River Turianchay (territory of the town)
Geychay	10320	43,850	Underground waters of the fan of the River Geychay (territory of the town and village Potu)
Agsu	4,320	2,170	Underflow waters of the rivers Agsu and Girdymchay
Udjar	6,000	72,700	Underground waters of the fan of the River Geychay and Ganykh-Agrichay Valley
Zardab	N/A	11,320	Underground waters of the Ganykh-Agrichay Valley
Kyurdamir	3,890	31,130	Underflow waters of the River Girdymchay and underground waters of the Kyurdmashin Valley (Kyululli section)
Yevlakh	12,500	62,350	Underground waters of the Garabakh Plain (fans of the River Terter)

Towns/ Villages	Water Use 2004 (m^3/d)	Requirement to 2010 (m^3/d)	Proposed potential sources of water supply
Terter	2,800	28,200	Underground waters of the fan of the River Terter (near the village Verikchayly)
Barda	10,400	27,675	Underground waters of the fan of the River Terter
Agdam	11,100	44,150	Underflow waters of the River Gargar and the fan of the same river
Agdjabedi	6,030	22,685	Underground waters of the fan of the River Gargar
Beylagan	2,800	16,100	Underground waters of the fan of the River Araz (section of the village Duyamalylar)
Fizuli	2,200	20,425	Underground waters of the fan of the River Guruchay (section of the village Beyuk Bakhmanly)
Djebrail	3,500	6,695	Underflow waters of the River Chakhmagchay (section of the River Veysally)
Zangilan	2,800	6,025	Underflow waters of the River Okhchichay
Gedabek	1,500	10,660	Capping of springs and underground fracture-pore waters of the suburban zone
Dashkesan	1,400	33,650	Capping of spring runoff and underground fracture-pore waters of the suburban zone
Kelbadjar	1,600	5,010	Capping of spring runoff and underground fracture-pore waters of the suburban zone
Khankendi	9,200	12,245	Fracture-pore waters of the town territory and underflow waters of the River Gargar
Shusha	2,800	15,800	Capping of spring runoff and underflow waters of the River Gargar
Askeran	2,900	2,800	Underflow waters of the River Gargar
Gadrud	3,850	2,668	Fracture-pore waters of the town and suburban territory
Agdere	9,300	6,790	Underflow waters of the River Chaylyg
Khodjavend	10,000	5,715	Underflow waters of the rivers Khonashen and Gargar
Lachin	1,200	11,650	Capping of spring runoff and underflow waters of the River Aker
Gubadly	2,500	4,500	Fracture-pore waters of the northern margin of the village Seytaz
Imishli	1,700	21,270	Underground waters of the fan of the River Araz – vicinity of Bakhram-Tap
Saatly	1,500	18,900	Underground waters of the Araz zone and fan of the river Gargar
Sabirabad	1,820	22,400	Underground waters of the Garabakh Plain or Ganykh-Agrichay Depression
Gadjigabul	3,200	16,500	Underground waters of the Ganykh-Agrichay Valley
Alibayramly	5,200	45,500	Underground waters of the Ganykh-Agrichay Valley
Salyan	1,500	23,100	Underground waters of the Ganykh-Agrichay Valley
Neftchala	5,480	10,850	Underground waters of the Ganykh-Agrichay Valley
Djalilabad	1,600	35,300	Pore underground waters of the suburban zone (3-4 km to the northeast)
Masally	2,700	14,000	Pore waters of the sedimentary series in the northeastern zone of the town (1.5 km)
Yardymly	2,700	4,600	Fracture-pore waters of the north-western margin (1 km) of the urban village
Lerik	800	7,000	Fracture-pore waters of the north-western zone (1.5 km) of the town
Lenkoran	4,700	45,000	Underflow of the River Lenkoran and underground waters of the Piedmont Strip of the Lenkoran Plain
Astara	4,500	12,000	Underground waters of Quaternary deposits of the Lenkoran Plain (4 km to the west of the town)
Sharur	1,260	12,000	Underground waters of Quaternary deposits of the Sharur Plain
Shakhbuz	500	5,500	Underground waters of Quaternary deposits of the southern part of the urban village
Nakhchyvan	11,000	129,850	Underground waters of the Nakhchyvan Plain and Batabad and underflow of the River Nakhchivanchay
Babek	274	4,900	Underground waters of the Nakhchyvan Plain (northeastern zone of the urban village)
Djulfa	5,300	8,500	Underground waters of Quaternary deposits of the Araz zone (station Dasta)

Towns/ Villages	Water Use 2004 (m^3/d)	Requirement to 2010 (m^3/d)	Proposed potential sources of water supply
Ordubad	10,800	12,000	Local (kariz) pore and alluvial-proluvial deposits of the Dasta section in the Araz zone

6. CONCLUSIONS

The surface-water based supplies to Azerbaijan have been under significant strain over recent decades. This is an international problem as major rivers are sourced in neighbouring countries. Problems are expected to continue. As such other options need to be evaluated. It is contended that the indigenous groundwater resources of Azerbaijan are sufficient to provide a sustainable water supply to the republic well into the future.

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OPTIMIZATION OF GROUNDWATER USAGE FOR URBANIZED RURAL SETTLEMENT SUPPLY IN AZERBAIJAN

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Abstract: The discovery of new water sources, including groundwater, acquires great importance in states with a deficit of general water resources. This paper examines the scientific and technical aspects of evaluating the resources of hydrogeological structures, such as the alluvial cone deposits present in some mountain rivers, in the context of Azerbaijan. It also investigates the feasibility of diverting some of the sources of fresh groundwater in the Ganja and Agrichai regions, with the aim of providing a centralized water supply to all cities within the Republic.

Key words: Ganja, Azerbaijan; Agrichai, Azerbaijan; diversion of water; water supply; watertight diaphragm; barrage; centralized water supply; alluvial cone deposits

1. INTRODUCTION

Among all cities and settlements in Azerbaijan Republic only Baku and Sumgayit cities have a centralized supply network of potable and industrial water. A centralized water supply is partially developed in the cities of Ganja, Geokchai, Agdash, Mingechaur, and Evlakh, but in many cities and settlements of the Republic a centralized water supply is absent.

Surface water and groundwater are the sources of water supply in Azerbaijan. At the same time, the hydroeconomic balance of the Republic is characterized by annual and seasonal deficits arising from the implementation of hydroeconomic measures directed to increase the water supply to different branches of the national economy. In states with arid

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climates, approximately 95% of the taken river runoff is used in the national economic production.

The optimal use of fresh groundwater resources acquires great significance in the current situation. These resources are generally characterized by their high quality and better protection against contamination than surface waters. Also, being less dependent upon annual precipitation, they serve as important sources of potable water supply. This paper outlines developmental work that aims to optimize the rational use of fresh groundwater deposits in the water supply to Azerbaijani settlements and builds upon foundational earlier work (Israfilov, 1983; Listengarten, 1983; Israfilov, 1990).

At present, eleven deposits of fresh groundwater corresponding to mountain valleys are being exploited. However, they are unevenly distributed within the Republic. They also have unequal reserves (Figure 1), so their rational use is concerned with their re-distribution and the regulation of groundwater resources in definite natural and economic states. Analysis of the existing hydroeconomic and hydrogeological states has revealed the main trends of optimization of rational use of the Republic fresh groundwater, namely:

- Accumulation of ground runoff in favourable hydrogeological structures;
- Groundwater diversion to water poorly supplied regions.

The results of scientific-technically developmental works on preset issues are presented below.

2. SCIENTIFIC-TECHNICAL FOUNDATIONS

2.1 Ground Runoff Accumulation in Favourable Hydrogeological States

It is known that the mountain valleys of the Republic are composed of fluvial drift cone deposits characterized by so-called ‘comb’ structures. Analysis of numerous hydrogeological profiles, both longitudinal and perpendicular to ground runoff, showed the presence, in nearly all drift cones, of lithological cavities composed of boulder-gravel-pebble deposits favourable for ground runoff accumulation. Three typical structures with characteristic hydrodynamical parameters have been defined (Figure 2). The first structure has significant resource capacity, is unconfined with confined units occurring below (Figure 2a). The second one is characterized by two-layer structure consisted of soil and confined aquifers hydraulically related to each other (Figure 2b). The third one is differentiated by a multi-layered structure and composed of soil and some confined aquifers hydraulically connected to each other (Figure 2c). The thickness of all three typical structures does not exceed 100 m, filtration coefficient (C_f) of the water-saturated series varies within 10-15 m/day, with a transmissivity of about 300-500 m^2/day. Hydraulic gradients are generally range around 0.01-0.008.

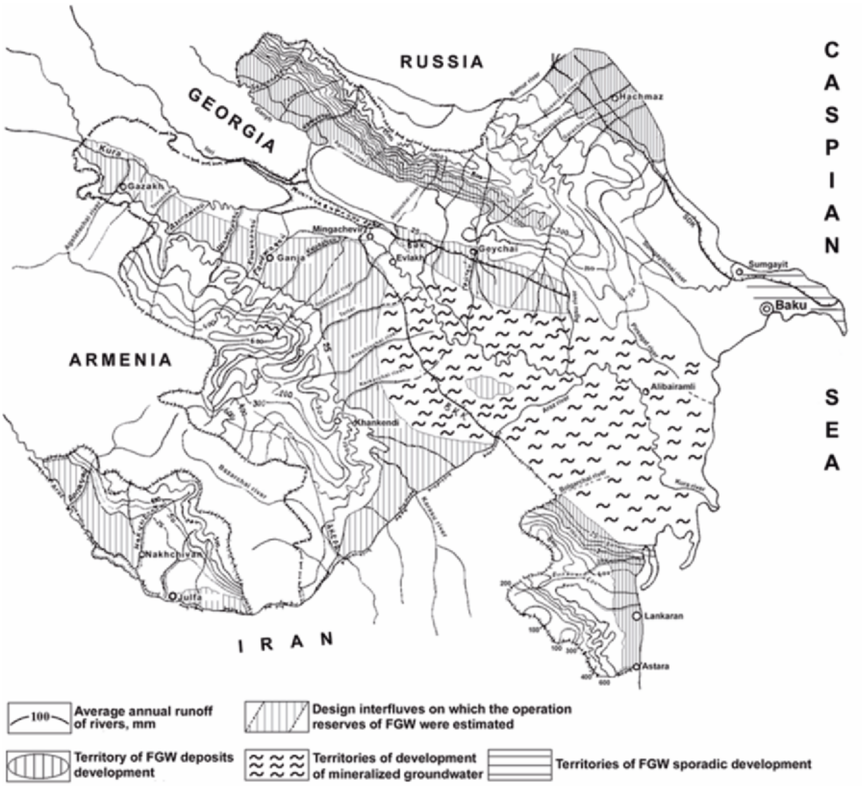

Figure 1. Schematic map showing average annual runoff of rivers in Azerbaijan and sources of fresh groundwater.

The proposed technology of ground runoff accumulation provides for creation of a watertight diaphragm (WTD) across the ground runoff; a so-called “barrage”, in favourable hydrogeological states. WTD construction will be performed in two ways: pumping of a paraffin-rich mixture into aquifers by high-pressure booster pumps in conjunction with an aerated water mixture. In the first case a solid diaphragm of long-term durability is formed. In the second case, an elastic diaphragm of short-term longevity is formed. The technology of WTD should be considered separately.

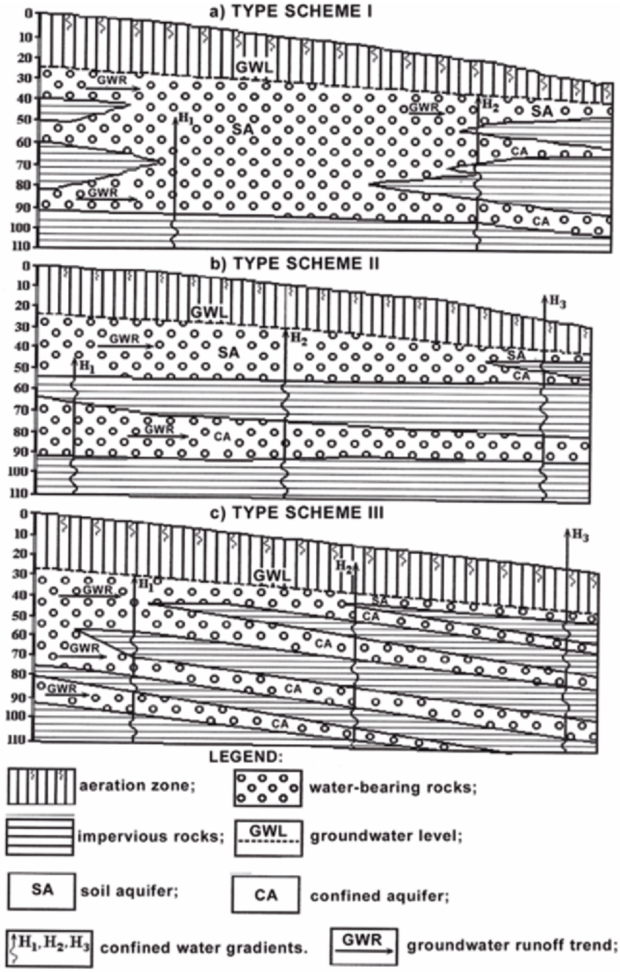

Figure 2. Type schemes.

At the same time, to develop optimal and economical technology for accumulating ground runoff, it was necessary to determine the dependence of groundwater runoff on WTD permeability. With this aim, the mathematical modelling of WTD functions in typical structures has been undertaken. The modelling was performed with the program "MORFLOT".

Initially accumulations of groundwater under various WTD permeabilities were modelled. Model values were: 50 to 80 % of initial permeability decrease of WTD screen for the Figure 2a structure where the head gradient was 0.008; coefficient of aquifer filtration at 12 m/day and a transmissivity at 350 m^2/day. Graphs showing the dependence of dynamics of the groundwater level rise and its accumulation upon the degree of WTD permeability in stationary and transient conditions are shown in Figure 3a. Modelling of the Figure 2b structure was conducted with head gradients at 0.01; coefficient of aquifer filtration 11-13 m^3/day, transmissivity of 300-350 m^2/day. Results are shown in Figure 3b. Modelling of the third typical structure (Figure 2c) used similar parameters.

The following conclusions were drawn:

- In the soil aquifer the rate of groundwater accumulation is directly proportional to degree of WTD permeability; the soil-pressure structure has a linear dependence.
- For the "elastic" WTD of water-and-air pulp the inclinations of ground runoff should not exceed 0.018.
- In the first typical structure the WTD should be constructed over the full saturated depth plus the projected rise of the groundwater level (ΔH).

In the second and the third structures together with soil aquifer plus ΔH it is necessary to mud-confine aquifers with WTD permeability reduction to 70%. There are possible choices without mudding of pressure aquifers, but the rate of ground runoff accumulation decreases by a factor of ~1.2 times. Quantitative estimation of the ground runoff accumulation was solved for the case of the drift cone of the Ganjachai River in the city of Ganja. Having a great deficit in potable water, the city supply is based on groundwater, produced by ~ 200 wells located directly on favourable drift cones corresponding to typical structures II, III. Accumulation of runoff permits increased groundwater production and further development of exploratory wells. Groundwater contour, transmissivity, chemical composition and total mineralization have all been used in the analysis.

WTD was set on groundwater contour 280 m, where the groundwater levels occur on depth 20-25 m, pressure gradients exceed groundwater levels for 2-3 m (Figure 4a). The water-enclosing rocks are gravel-pebbles with sandy fillers, the filtration coefficient being 9-12 m/day. Taking into account the natural and hydrogeological states the projected value of groundwater level rise (ΔH) from the present level (+12 m) was predicted and a total WTD perimeter of 6500 m (Figure 4a). After the initial model calibration (by solution of inverse problems), the dynamics of rise and period of stabilization of groundwater levels were predicted using a time step Δt=10 day. An artificial ground reservoir of about 26 million m^3 of groundwater accumulation for a period of 109 days, radius of WTD influence of 4000 m was predicted (Figure 4b). That groundwater volume may more than double water supply to Ganja city.

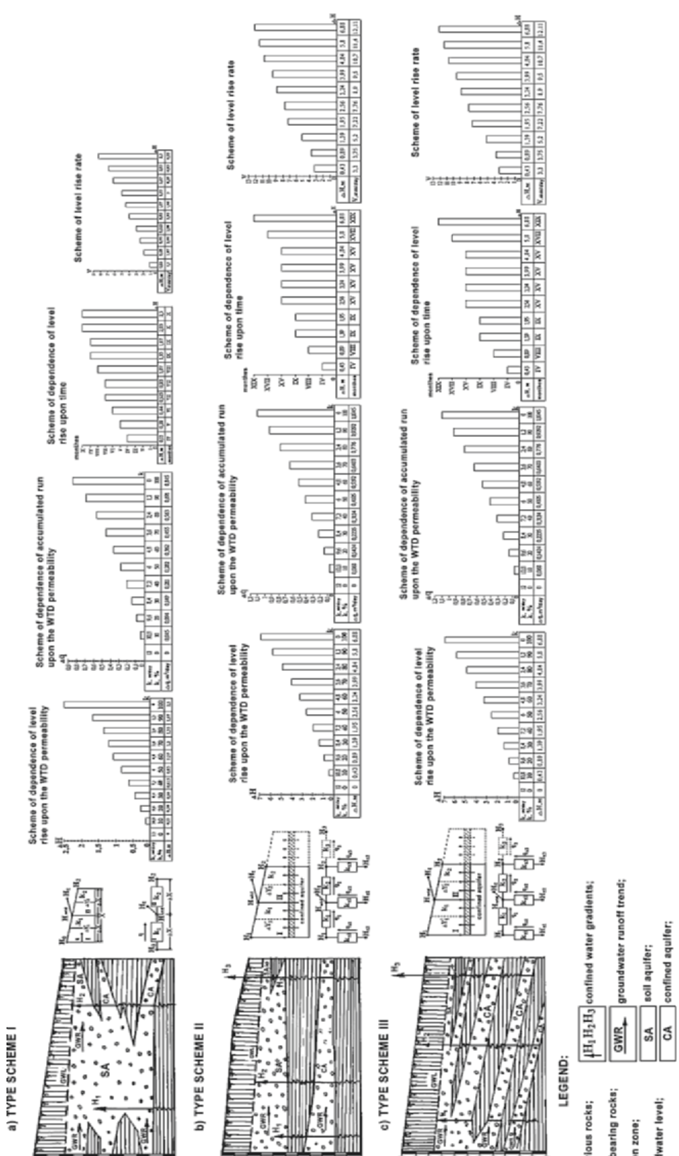

Figure 3. Groundwater levels and WTD permeability.

2.2 Diversion of Groundwater Resources to Water Deficient Regions

Evaluation of the natural-economical and hydroeconomic states has permitted definition of three deposits of fresh groundwater that have sufficient resources for their partial diversion to water-deficient regions of the Republic. These deposits are: Ganyh-Agrichai, Ganja and Gusar. The latter has been exploited since 1905 as an exclusive water supply to Absheron, an agro-industrial complex. There are hence two choices for diversion: from Ganyh-Agricgai and Ganja deposits. These options have been evaluated via the following:

- substantiation of the residual operating reserves of fresh groundwater for diversion;
- evaluation of technical-economical indices of diversion;
- confirmation of rational water intake of groundwater (their types, locations, productivity, etc.);
- optimization of routes of diversion and points of water feed are identified.

Volumes of groundwater recommended for diversion were calculated based on data from the Central Statistical Department, general scheme of complex use of water resources of Kura river basin and former Bureau of Geology of the Republic. The design parameters are given in Table 1. Thus, the guaranteed volumes of diversion of fresh groundwater from Ganyh-Agrichai and Ganja deposits are determined to be a volume of 700,000 m^3/day.

Economical analysis of the diversion options (optimal schemes, productivity of water intakes, distances, routes and points of water feed) has been undertaken. The competitive ability of groundwater diversion in comparison with river waters has also been determined (Israfilov, 1983). The basic indices of the comparison are the capital investments and expenditures with specific investments and prime cost plotted in Figures 5-7. Capital investments into the water supply system, of various productivity, are characterized by an inverse proportional dependence to the productivity value. The specific capital investments into water intakes of groundwater lies below the surface water intake predicted curves. Under productivity of various systems of water supply to 80,000 m^3/day, the water cost from surface-water sources is two times greater than that from the groundwater ones. Overall, the specific indices of the cost of the ground water intakes are always lower than surface water schemes of comparable capacity.

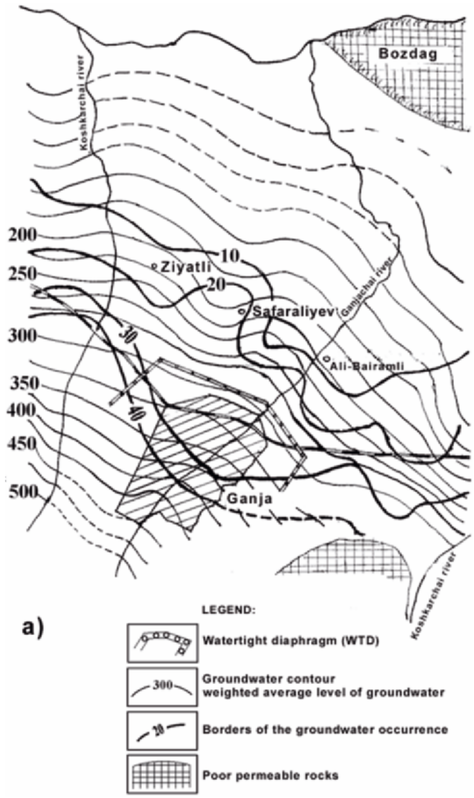

Figure 4. a) Scheme of border states of the drift cone of the Ganjachai river.

Optimal distances of groundwater diversion, competitive with surface water, are revealed from the comparison of capital investments and prime costs (Figure 7); the distance is about 53-59 km. Data on the developmental works has also permitted determination of more economically located water intakes (based on their productivity). If it is necessary to add 700 thousand m^3/day of groundwater the schemes are advisable consisted of 14 water intakes accordingly with productivity 50,000 m^3/day. The proposed location of water intakes, watershed, routes of diversion and final points of water feed are given in Figures 6 and 7.

3. CONCLUSIONS

1. The high-quality groundwater resources present in the deposits of the mountain valleys of the Republic of Azerbaijan completely satisfy the present and forecast requirements of the local settlements. However, the

uneven development of aquifers in the area requires consideration in order to optimize their exploitation.

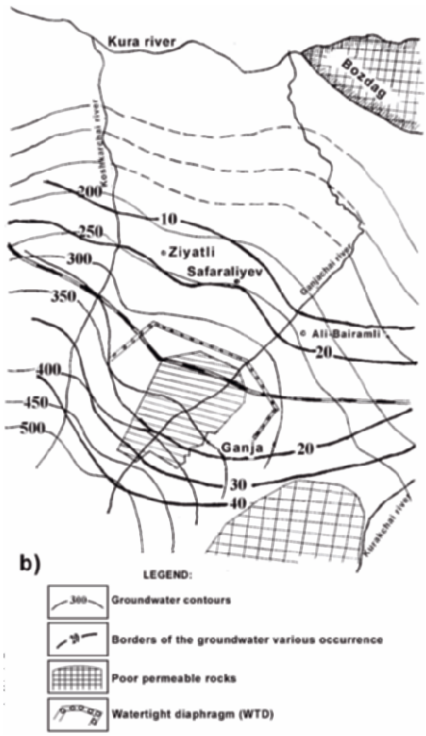

Figure 4. b) Prognostic scheme of the hydrogeological states of the drift cone of Ganjachai river after commencement of the watertight diaphragm function.

2. The ‘barrage’ technology developed encourages accumulation of the natural groundwater runoff within the favourable hydrological structures. The present structures are nearly on all talus trains of mountain valleys where the majority of the cities of the Republic are located. This increases the effectiveness of the optimization measures. The boundary conditions and optimal parameters for accumulating groundwater runoff were determined using a mathematical modelling method. The investigation of the talus train of the River Ganjachai illustrates the approach. The city of Ganja is developed on this talus train: based on population and industrial production; Ganja is the second city in the Republic. The study showed that it is possible to accumulate an extra 26 million m^3/day of fresh groundwater within the artificial groundwater reservoir. Thus, it will be possible to more than double the city’s supply of drinking water, and completely satisfy its present

requirements. Similar measures on other talus trains will allow substantial increases in the supply of drinking water for many cities and settlements of the Republic.

Table 1. Design parameters for groundwater diversion proposals (Ml/d).

Interfluve	Operational resources	Actual maximum production	Forecasted maximum production	Forecasted sufficient resources	Volume guaranteed for diversion	Operational resources after diversion
Ganyh-Agrichai deposits of fresh groundwater						
Border – Garachai	769	55	249	520	150	370
Garachai - Kurmukhchai	804	20	357	447	150	297
Kurmukhchai - Kishchai	907	31	370	537	200	337
Kishchai – Alijachai	881	52	345	536	250	286
Alijachai – Damiraparanchai	467	22	128	338	150	188
Damiraparanchai – Akhokhchai	294	11	64	229	100	130
Total:	4121	191	1513	2608	1000	1608
Ganja deposits of fresh groundwater						
Kura river left bank	130	46	60	70	50	20
Border - Agstafachai	242	73	100	142	100	42
Agstafachai - Tovuzchai	726	278	380	346	200	146
Tovuzchai - Shamkirchai	1564	750	950	614	350	264
Shamkirchai - Kyurakchai	1529	688	1150	379	200	179
Kyurakchai – Injachai	596	238	360	236	100	136
Total:	4787	2072	3000	1787	1000	787

3. The diversion of 700 thousand m^3/day of fresh groundwater from the Ganyh–Agrichai deposit to the Shirvan hydroeconomic province (350 thousand m^3/day to Haldan, 200 thousand m^3/day to Agdash and 150 thousand m^3/day to Ujari) will practically satisfy the requirements for drinking water supply of all settlements of the present region, provided a united system of water distribution is constructed. The diversion of 600 thousand m^3/day of fresh groundwater from the Ganja deposit to Jeiranchel province will satisfy drinking water needs and allow a significant increase in cattle-breeding and other agricultural production.
4. Due to the current new socio-economic situation in the Republic, and as a consequence of the hydroeconomic state at present, it is necessary to develop a new general scheme of rational use of water resources as a whole, and groundwater in particular. This will allow the more complete development of optimization measures that is necessary to ensure the forecast demands are met.

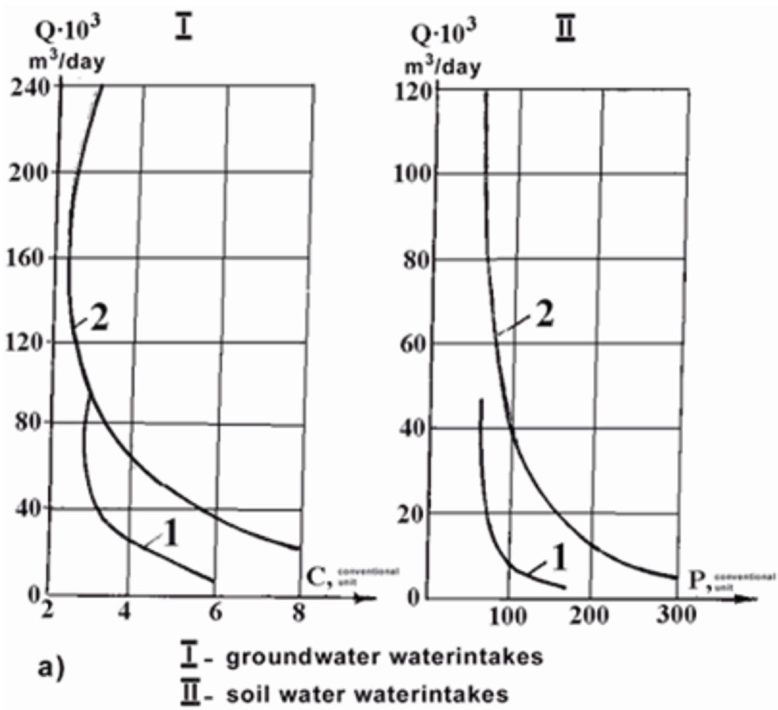

Figure 5. Schemes (for various considered options, i.e. different graph lines) of prime cost dependence (I) and specific investments (П) upon the water-intake production.

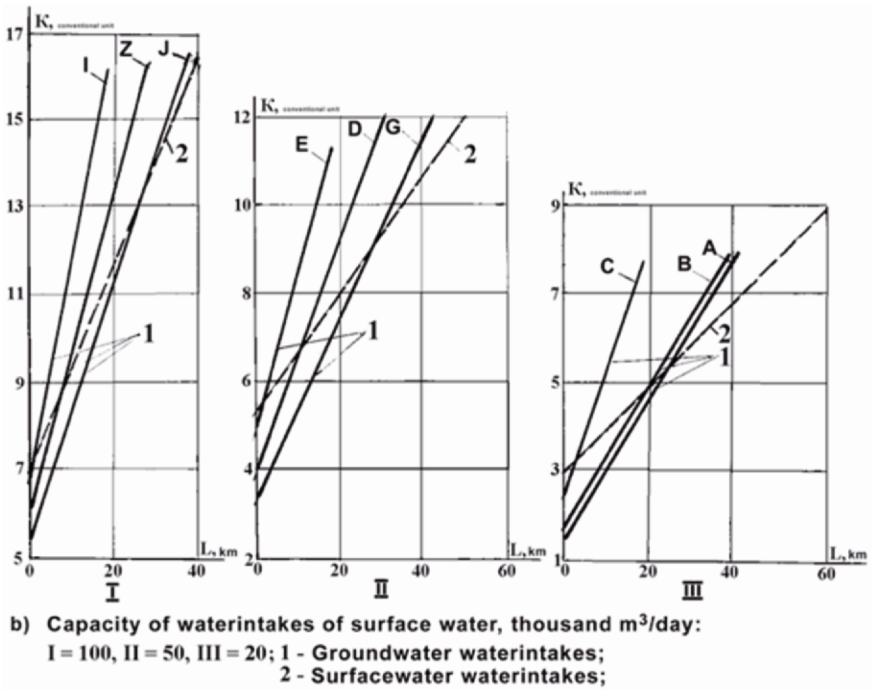

Figure 6. Schemes of dependence of water diversion distance (L).

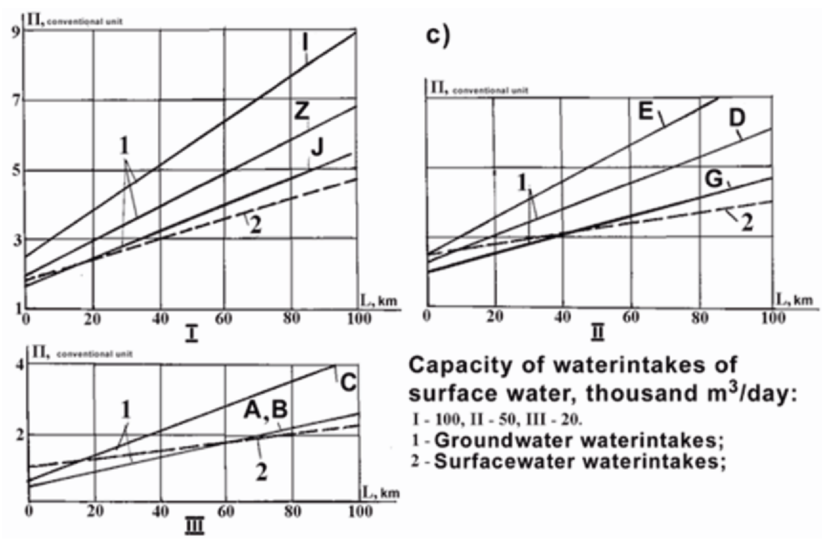

Figure 7. Schemes of dependence of expenses (П) upon the water diversion distance (L).

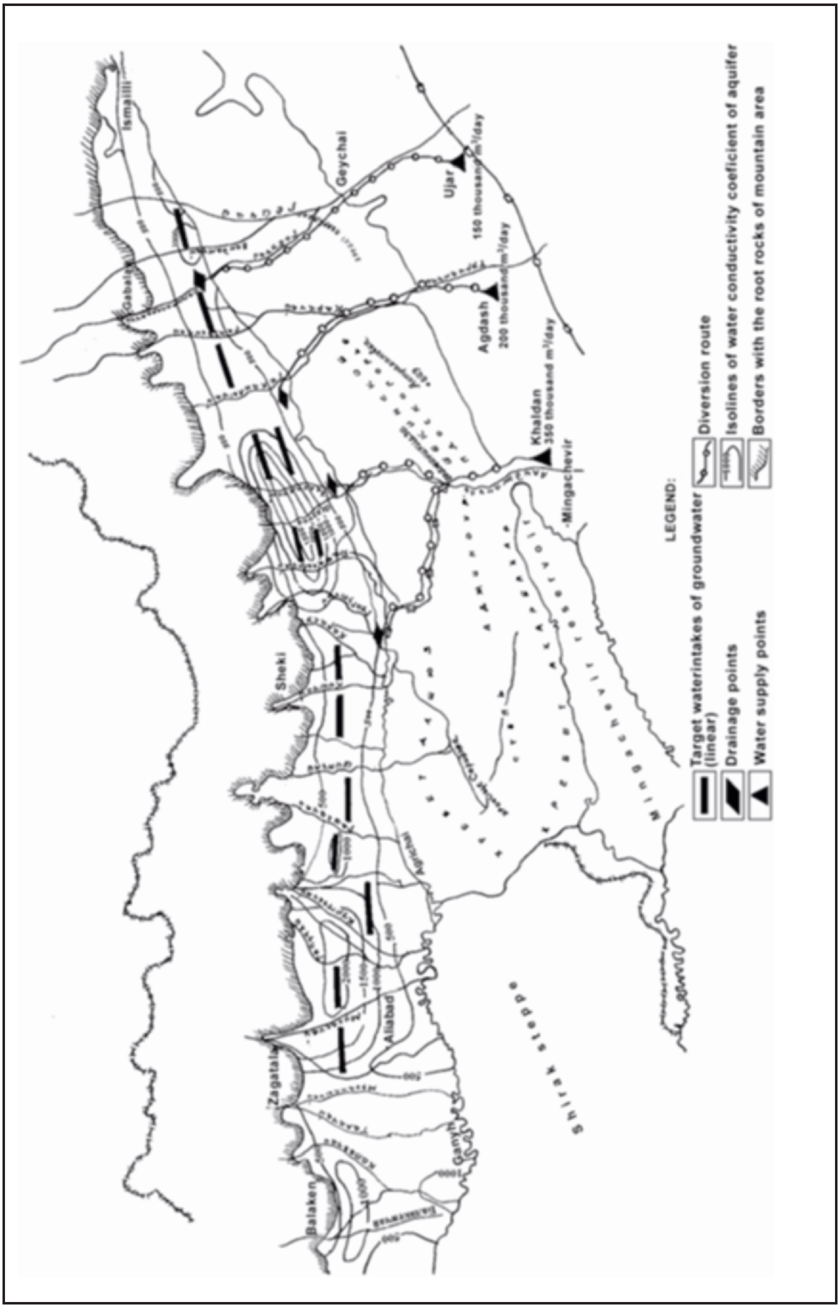

Figure 8. Scheme of diversion of some resource of groundwater of Ganyh-Agrichay deposits of fresh groundwater.

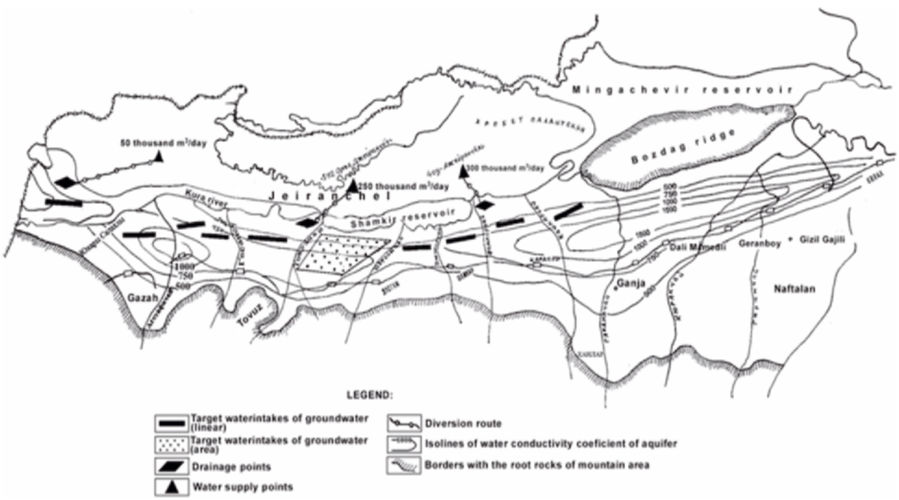

Figure 9. Scheme of diversion of some of the Ganja groundwater reserves.

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URBAN GROUNDWATER POLLUTION IN TURKEY

A National Review of Urban Groundwater Quality Issues

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Abstract: Groundwater pollution in Turkey is examined. Important natural sources of groundwater pollution identified include seawater intrusion, discharges from contaminated lakes and streams, geothermal waters, and dissolution of minerals. The major sources of anthropogenic groundwater contamination identified are: agricultural pesticides and fertilizers; mining waste products; industrial waste; on–site septic tank systems; and pollution from poorly constructed wells. Although industrial waste and on–site septic tanks are important sources of anthropogenic pollution, because agricultural activities are very significant contributors to the Turkish economy, pollution from pesticides and fertilizers poses the larger threat.

Key words: Turkey; pollution; pesticides; septic tanks; industrial wastes; natural pollutant sources; geothermal waters

1. INTRODUCTION

For a water source to be usable, it has to be of sufficient chemical and microbiological quality. However, due to rapid technological advances and population increases in recent years, sources such as streams, lakes, and groundwater are often in danger of severe pollution. This problem is becoming manifest in the republic of Turkey. Located where three continents – Africa, Europe and Asia – meet, it is bordered by the Black Sea to the north, the Mediterranean Sea to the south, and the Aegean Sea to the west (Fig. 1). Its geographical location means that Turkey has an important geo–political status in the region, acting as a centre of trade between Europe and Asia. Turkey covers a total area of 780,000 km^2, of

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which 14,300 km^2 is covered by water. According to the Turkish General Directorate of State Hydraulic Works (DSİ), the annual surface water potential of Turkey is about 234 billion m^3. In addition to its surface water resources, the groundwater potential of Turkey has been a focus of numerous studies since 1956. It is estimated that Turkey's total annual groundwater resource is approximately 12.3 billion m^3. The total usable annual surface and groundwater potential of Turkey is 110.3 billion m^3. This total amount of water includes 95 billion m^3 from internal rivers, 3 billion m^3 from external rivers, and 12.3 billion m^3 from groundwater resources (DSI, 2004).

Turkey is one of the fastest growing countries in the world and, due to its rapid industrialization and urbanization, domestic, medical and industrial waste products have become an increasing threat to the country's water sources (Afsin, 1997; Kaçaroğlu, 1999; Karagüzel *et al*., 1999; Baba *et al*., 2001; Karlık and Kaya 2001). Groundwater contamination can be classified as having either natural or man–made (anthropogenic) sources. Natural groundwater contamination is primarily caused by infiltration from low quality streams, rivers, lakes, or seawater, or due to geothermal field effects. Anthropogenic groundwater contamination is generally attributed to the excessive use of agricultural pesticides and fertilizers; mining waste products; disposal of industrial waste; waste disposal sites; and poor well construction. The aim of this paper is to provide an overview of urban groundwater pollution in Turkey highlighting both natural and anthropogenic issues of significant concern.

2. NATURAL GROUNDWATER CONTAMINATION

In some Turkish regions, groundwater has been polluted seriously by natural sources (Aslan and Akkaya, 2001). Salinization from seawater incursions, heavy metals from mines, increased water hardness from dissolved limestones, and boron from geothermal fields are some examples of natural groundwater contamination observed and are examined below.

2.1 Naturally Poor Quality Surface Waters

In the Lakes Region of Turkey, the drilling of wells has caused salty and acidic lake waters to penetrate local aquifers (Burak *et al*., 1997). Studies in 1997–98 showed that the groundwater of the Eskisehir plain, especially around the city centre, did not meet drinking water quality standards. Reasons for the groundwater contamination vary, but the most important is the mixing of the Porsuk Stream and irrigation channel waters with the groundwater (Ozcelik and Sarıiz, 2001). A further example is provided by Karamanderesi and Helvaci (2001) who investigated groundwater pollution from the Buyuk Menderes River. They reported that unregulated, excessive abstraction of groundwater caused polluted surface water from the Buyuk Menderes to infiltrate aquifers around the river. Water contamination in the Buyuk Menderes River also impacted the ecology of the local geothermal systems.

Figure 1. Location map of study area.

2.2 Seawater Intrusion

Groundwater in some areas has become polluted via seawater intrusion arising from excessive groundwater abstraction from coastal wells. In the coastal plains of regions such as Çeşme (İzmir), Bodrum (Muğla), Marmaris (Muğla) and Çanakkale, groundwater has been either partially or completely affected by seawater (Burak *et al.*, 1997; Gemici and Filiz, 2001; Baba and Deniz, 2004; Gürçay, 2004). Around the Çeşme peninsula (İzmir), high groundwater abstraction in summer has caused groundwater levels to drop below sea level and seawater penetration landwards. Çeşme's peninsula location, its tectonic structure oriented towards the sea, and the development of many wells due to increasing water demand has lead to increasing groundwater salinity problems. Some wells have indeed become abandoned due to their quality derogation.

The aquifers in the region are karstic Mesozoic limestones. Both karstification and the seaward orientation of faults can encourage seawater incursion. This is particularly true in areas where no intervening clay units are present between the sea and coastal aquifers. Karstification is one of the most important factors affecting seawater penetration into aquifers. In the central parts of the Çeşme region, Mg, Ca, and HCO_3 are the dominant ions, while in areas close to shorelines, Na and Cl dominate. In general, the Cl concentration in groundwater varies from 100 to 200 mg/L, but it can be as high as 4000 mg/L (18% seawater) when seawater intrusion occurs (Gemici and Filiz, 2001). Therefore, groundwater from karstic coastal aquifers tend to be both saline and hard and unsuitable for drinking water supply.

Groundwater in the Lake Van coastal aquifer has also been subject to salinity problems due to saline intrusion from the lake (Ozler, 2003). The aquifer is one of the main sources of potable, industrial and irrigation water since surface water is limited due to the semi–arid climate. However, due to a growing population and increased agricultural and industrial activities during the last 20 years, groundwater abstraction has exceeded groundwater replenishment. Ozler (2003) concluded that the main processes influencing groundwater chemistry were salinization from salt–water incursion, silicate mineral dissolution, cation exchange, and anthropogenic pollution.

Further studies confirming the widespread nature of the salinity problem include the following. Groundwater in Bodrum was also observed to have higher salinity due to seawater incursions (Filiz *et al*., 1998a). Bodrum's peninsula location and karstic geologic nature means where excessive pumping of groundwater is present saline intrusion typically occurs. In some parts of the region, the proportion of seawater in pumped groundwater can be as high as 28% (Filiz *et al*., 1998b). The Mersin–Erdemli region studied in 1999–2000 also exhibited seawater incursion (Degirmenci and Altin, 2001).

Clearly continued groundwater abstraction from the above aquifers will inevitably lead to exacerbation, rather than resolution, of the salinity problems already evident.

2.3 Geothermal Field Effects

Geothermal waters are present in some deep aquifer systems in Turkey. As thermal waters flow through rocks, they dissolve and transport many chemical elements. Such dissolution causes groundwater quality deterioration (Baba, 2004). Geothermal waters contain high boron concentrations that are particularly hazardous for irrigation waters. Boron in trace levels is useful for plant growth, but at high concentrations it has adverse impacts (Uğurluoğlu, 2004) with the gradual boron accumulations lowering the quality of agricultural soils. Geothermal waters may also contain heavy metals such as arsenic, mercury, cadmium, lead, or chromium that at elevated concentrations may be fatal to both humans and animals.

Several geothermal water systems are present in Turkey, and they have been used for numerous industrial and recreational activities. Many groundwater quality studies have been carried out around the Gediz and Asagi Menderes Plains geothermal fields. In the geothermal fields of the Afyon–Akarcay Basin, including the Omer–Gecek, Gazligol, and Heybeli fields, geothermal pollution of cold groundwater has been examined. Hydrochemical analyses indicate that the thermal waters are of Na–Cl type, whereas the cold groundwaters are of Ca–HCO_3 type. Using Na, K, Cl, Li, B, temperature, and conductivity as indicators of thermal water mixing, it was shown that cold groundwater pollution is occurring around geothermal fields in the Afyon–Akarcay region (Doğdu and Bayarı, 2002).

2.4 Geological Factors

Great damage to urban infrastructure can be caused by earthquakes. Recent hydrogeological and hydrological investigations regarding the impact of urbanization have been undertaken for the city of Burdur (Turkey) (Davraz *et al*., 2003). To evaluate the effect of earthquakes on groundwater, groundwater isohypse and isopach maps were prepared showing the buildings in areas of Burdur where groundwater depth is less than 10 m. This depth is considered to be the critical depth for liquefaction during an earthquake and the groundwater table should be lowered below this depth to eliminate earthquake hazards. The chemical makeup of groundwater was also assessed to determine possible effects on foundations. Streams flowing across residential areas in the city of Burdur pose a flooding risk during earthquakes (Davraz *et al*., 2003).

Geological formations in Turkey containing halite, gypsum or anhydrite result in salt and sulphate pollution exceeding the limits permitted for irrigation and drinking water. Some parts of the Iskenderun–Ulucinar–Arsuz plain, the central part of the Asi Basin, and areas around Sivas and Corum are at risk of this type of pollution. Groundwater in some parts of Aydin–Soke is also of low quality because of geological pollution. The solubility of gypsum is very high when compared with many other minerals. Surface waters and groundwater in direct contact with gypsum formations can attain significant total dissolved solids (TDS) calcium and sulphate contamination. This may prevent usage for drinking, domestic, industrial and irrigation purposes. Gypsum formations containing halite layers crop out in a large area of the Upper Kızılırmak Basin, Sivas. Kaçaroglu *et al*. (2001) studied the effects of the lithological composition of the catchment area on water chemistry and quality in the Upper Kızılırmak Basin. Surface waters draining gypsiferous areas, and the Göydün and Seyfe springs, create high TDS concentrations in the River Kızılırmak

(EC = 1100–5200 μS cm^{-1}). The Göydün and Seyfe springs, which issue from gypsum–rich deposits, have very high TDS values (EC = 12825 – 13900 μS cm^{-1}). Surface and groundwater resources in non–gypsiferous parts of the basin, specifically the River Yıldız, and the Kaynarca and Gaziköy Springs in the Tavra Valley, have lower TDS (EC ranges between 495–630, 795–995, and 530–575 μS cm^{-1}, respectively). Water from the Göydün and Seyfe springs and the Kızılırmak River alluvium is not suitable for drinking, irrigation or industrial purposes, with the TDS, hardness, sulphate and chloride concentrations of these waters exceeding the maximum permissible under Turkish Drinking Water Standards. Their very high salinity prevents their use for irrigation (Kacaroglu *et al*., 2001).

The higher elevation grounds of the Bornava, Yamanlar, and Cigli regions are formed of volcanic rocks. Around Yamanlar and Cigli, these rocks contain mineralized zones and, during periods of heavy precipitation, especially in the spring and autumn, heavy metals are released into the groundwater (Baba *et al*., 2001).

Examining drinking water sources around the villages of Beylikova–Kizilcaoren (Eskisehir), a region well-known in Turkey for its fluoride production, Uslu (1982) and Fidanci *et al*. (1994) found fluoride concentrations of around 3.9–4.8 mg/l and 2.0–9.2 mg/l respectively. Bayindir–Kaman (Kirsehir) is another region famous for its fluoride formations, and groundwater there contains approximately 2.6 mg/l of fluoride (Fidanci, 1997). Fluoride contamination in groundwater has also been observed in the Dogubeyazit (Agri), Muradiye (Van), and Habiller (Edirne) regions (Sendil and Baysu, 1973; Oruc, 1977; Uslu, 1982). Analyses of 13 surface and groundwater samples from Gullu Village in the city of Usak showed fluoride concentrations were between 0.7 and 2 mg/l (average 1.35, median 1.60) (Oruc and Vicil, 2001).

3. ANTHROPOGENIC GROUNDWATER CONTAMINATION

While the degree of anthropogenic groundwater pollution varies from one region to another, it is possible to categorize the fundamental causes of the pollution into a few major types. These categories are: pesticide and fertilizer use in agricultural activities; industrial waste disposal; on-site septic tank systems; mining waste products; and imperfect well construction.

3.1 Agricultural use of Pesticides and Fertilizers

Fertilizers and pesticides are commonly used in modern agriculture. Exposing agricultural fields to hazardous chemicals for lengthy periods increases the risk of groundwater contamination. Groundwater pollution due to nitrate and pesticides has recently become an important issue. The excessive use of chemicals in agricultural activities can cause adverse effects on ecological systems. When compared with developed countries, the use of fertilizers is low in developing countries. However, the presence of pesticides in water sources can be high, due mainly to inappropriate application. Pollutants from agricultural activities can be summarized as follows (Aslan and Akkaya, 2001):

- fertilizers (sodium, nitrate, ammonium, sulphate, chloride, and phosphate);
- pesticides (organic chlorinated compounds);
- animal farming (nitrate, ammonium, chloride, faecal coliforms, organic pollutants).

In the area of the Incesu–Dokuzpinar springs, agriculture and other human activities have had both direct and indirect effects on the rates of contamination of groundwater. Direct effects include the dissolution and transport of excess quantities of fertilizers, and hydrological modifications related to irrigation and drainage. Indirect effects include changes in water–rock reactions in soils and aquifers, caused by increased concentrations of dissolved oxidants, pH and major ions. Agricultural activities have affected directly or indirectly the concentrations of a large number of inorganic chemicals in groundwater, including NO_3, N_2, Cl, SO_4, H, K, Mg, Ca, Fe, Cu, B, Pb, and Zn, as well as a wide variety of pesticides and other organic compounds. The high concentrations of NO_3 and NaCl show that the area around the springs has been continuously contaminated by untreated sewage and agricultural wastes (Elhatip *et al*., 2003).

In the Aegean Region around Urla and Menemen, groundwater contamination has been detected as a result of using fertilizers and pesticides in agricultural activities. Aslan *et al*. (2001) undertook sampling of fourteen wells, and tested for AOX and nitrate. In Menemen, nitrate levels of 53 mg/L and 146 mg/L were observed in two wells. AOX concentrations from three different wells were1.6, 4.8, and 5.4 µg/L, respectively, but these concentrations were all below World Health Organization (WHO) Standards. Samples examined from six wells in the Urla region revealed nitrate levels of between 69 and 127 mg/L were observed. In addition AOX pollution (1 mg/L) was determined in one of the wells but, again, this did not exceed the permitted standard (Aslan *et al*., 2001).

In the Bursa–Nilufer and Ayvali basins, nitrate concentrations were measured on samples from 35 well and spring locations. Results indicated that nitrate concentrations in these regions were above World Health Organization Standard levels (Sen, 1996). Sen (1996) believed that the contamination was strongly related to the use of nitrate fertilizers. In the same study, it was stressed that nitrate levels in groundwater in such regions should be monitored regularly.

In the Manisa region, five wells were drilled to determine nitrate (NO_3), nitrite (NO_2), and ammonium (NH_4) levels from the oxidation of organic-N. Organic-N is dissociated to NH_4, NO_2, NO_3, respectively. Results indicated that nitrate concentrations in the first four wells were below pollution limits but the nitrate concentration was much higher in the remaining well (127−133 mg/L), rendering water from this well undrinkable (Eryurt and Sekin, 2001). Nitrite and nitrate were detected in all wells, though Eryurt and Sekin (2001) reported that the concentrations were all below the maximum permitted by drinking water standards. Nitrite and nitrate in soil and groundwater around the Manisa region are derived mainly from fertilizers used in agricultural activities. When organic fertilizers are used for agricultural activities, nitrite and nitrate can be formed by bacterial metabolic activities and washed out from soil into groundwater (Eryurt and Sekin, 2001). In addition, agriculture pollution of groundwater also occurs in the Eskisehir plain (Kaçaroğlu and Günay, 1997).

In and around the city of Nigde, excessive use of nitrate fertilizers for potato production has caused adverse effects on groundwater (Karadavut *et al*., 1997), and similar problems have occurred around the Cukurova and Amik plains. It should be noted also that phosphate fertilizers contain heavy metals (Haktanir, 1992). Furthermore, groundwater contamination by pesticides has become an important environmental issue. Aydin and Yurdun (1999) tested for nine different chlorinated pesticides in the water sources and tap waters of Istanbul. The observed organochlorine pesticides were α– and γ–HCH and aldrin, both of which had been banned from use. The contents of α– and γ–HCH in raw waters were in the range 0.34–1.7 μg/l and 0–0.077 μg/l, respectively, whilst aldrin was observed at a concentration of 0.03 μg/l in some samples. The levels of organochlorine pesticide residues in the drinking water supplies of Istanbul were found to be significantly below the maximum permissible levels. The effectiveness of potable water treatment processes and the importance of maintenance and backwashing of sand filters in pesticide removal were noted. Improper and delayed backwashing of filters caused an increase in the pesticide residues in the distributed water. In older water distribution lines, higher concentrations of some organochlorines were also noted (Aydin and Yurdun, 1999).

3.2 Industrial Waste Products

Locating industrial facilities on or around plains in many Turkish regions and discharging untreated waste water into aquifers has resulted in serious pollution. Contaminants resulting from industrial hazardous waste disposals can be classified as follows (Aslan and Akkaya, 2001):

- suspended solid and organic compounds;
- textile wastes (suspended solid and alkaline compounds);
- chemical wastes (acids, alcohols, oils, and other toxic compounds);
- petrol refinery wastes (chloride, phenols, sulphur compounds);
- metal–containing wastes (acids, toxic heavy metals);
- thermal energy waste (increases in water temperature).

Groundwater contamination has been observed in and around the Gulf of Izmir and within the remainder of the Izmir region. This contamination has arisen due to improper discharge of groundwater and the rapid development of industry. Industries include textile manufacture, leather, cement-stone production, port facilities, paper manufacture, detergent manufacture, transportation, and food–vegetable–oil production. In the Bostanli, Karsiyaka, Sahilevleri, and Alsancak regions of Izmir, groundwater levels were observed to be unusually high, with soils containing a high concentration of organic matter (Baba *et al*., 2001).

The toxicity of some organic and inorganic chemicals to micro-organisms is an important consideration in assessing their environmental impact. Micro–organisms play an important role in the natural attenuation of contaminants in aquifers. Some organic and inorganic compounds have been detected at toxic levels in industrial discharges, in clear violation of discharge permits. In addition, even though in some cases the effluent wastewater does not exceed the discharge limits, the results of toxicity tests show potential toxicity (Sponza, 2002). Knowledge of the toxicity of effluents can benefit treatment plant operators in enabling optimization of plant operation, setting pre-treatment standards, protecting receiving water quality and in establishing sewer discharge permits. Under Turkish regulations only the toxicity dilution factor (TDF) for the protection of fish is included in the toxicity-monitoring programme for permissible wastewater discharges.

Laboratory studies of different organisms and protocols for toxicity assessment have been conducted in various countries, including Turkey. Sponza (2002) carried out a study to investigate the acute toxicity of textile and metal industry wastewaters in Turkey using both traditional and enrichment toxicity tests, and emphasized the importance of toxicity tests in wastewater discharge regulations. The enrichment toxicity tests are novel and give an idea whether there is a potential toxicity, or a growth-limiting, or growth-stimulating condition. Different organisms were used, including bacteria (*Floc* and *coliform*), algae (*Chlorella* sp.), fish (*Lepistes* sp.) and protozoa (*Vorticella* sp.), to represent four trophic levels. The textile industry results showed acute toxicity for at least one organism in 8 out of 23 effluent samples. Acute toxicity for at least two organisms in 7 out of 23 effluent samples was observed for the metal industry. The toxicity test results were complemented with other analyses such as COD, BOD, colour, and heavy metals. It was observed that, in five cases from the metal industry and four from the textile industries, the toxicity of the effluents could not be explained by using physicochemical analyses. Results from Sponza's (2002) study showed clearly that the use of bioassay tests produced additional information about the toxicity potential of industrial discharges and effluents.

3.3 Mining Waste Products

Major pollutants originating from mining activities are as follows:

- heavy metals, cyanide, chloride, sulphur, and sulphate from coal and metal processing;
- Ca, Mg, Na, K, Cl, phenol, and other petroleum compounds from petroleum and gas fields;
- Na, Mg, Ca, K, Cl, and SO_4 from salinization.

In the city of Balikesir, in the Balya region, rainfall runoff from lead mining wastes has entered the River Kocacay and created serious environmental pollution. In addition, hazardous wastes in this region, containing high concentrations of cyanide and iron, have caused many fish species to become extinct (Aykol *et al*., 2003). Studies carried out in the region have also indicated that groundwaters have acidic characteristics due to mining activities. The effects of the waste–rock dump (WRD) of the underground polymetallic Balya Mine on the Kocacay River and, downstream, on Lake Manyas were investigated by Aykol *et al*. (2003). The geochemical characteristics of various kinds of water (mine, surface and groundwater) were examined and suspended-particle samples were also analyzed. The more polluted mine waters had low pH and high conductivity. High concentrations of Zn, Cd, and Mn tended to be found in both the dry and wet seasons, whereas high concentrations of Pb, As, Cr, Cu and S appeared only in the wet season. The sources of the heavy metal concentration within the River Kocacay were leached waste, surface runoff, and overflow from the spillway of the WRD (Aykol *et al*., 2003).

Water pollution by heavy metals produced by mining processes, such as copper, chromium, lead, cobalt, nickel, and zinc, has also been observed in the Tigris River in Elazig city (Çetindağ and Okan, 2004).

Studying the adverse affects of mine processing on the environment is extremely important because significant urban development is already taking place on the geological units in Turkey that contain gold ore. Such an area is located to the southeast of Izmir, where approximately 2 km-long gold-bearing ore veins occur close to the village of Efemcukuru. The gold-bearing formation is highly weathered and fractured, and these fractures control the permeability and the depth of groundwater in the area (Baba and Gungur, 2002).

A hydrochemical study of the area surrounding the Hisarcık (Emet–Kütahya) colemanite mine by Colak *et al*. (2003) showed extremely high arsenic contamination in groundwater, with concentrations ranging between 0.07 and 7.754 mg/l. This contamination, in and around the village of Igdeköy in the Emet region, was caused by naturally occurring arsenic dissolution from a borate-bearing clay zone due to the leaching of arsenic-bearing minerals. The arsenic concentration in the groundwater varies locally from spring to spring and is related to the mineralogical and geochemical compositions and lithofacies of the aquifer (Colak *et al*., 2003).

Coal-burning power plants in Turkey produce large quantities of coal-related waste, which is collected using various systems. Coal ash is presently accumulating in Turkey at a rate in excess of 10 million tonnes annually, roughly 1% of which is used in a variety of products, such as concrete, and aggregates for stabilizing roadways. When coal is burned, this results in an increased concentration of most heavy metals and radionuclides being found in the waste material. Baba (2002, 2003) reported that the waste disposal site near the Yatagan (Mugla) thermal power plant contained a major enhancement of heavy metals and radioactivity that have affected groundwater. Water leaches through the waste heaps at Yatagan and into karstic marbles overlying schist, adversely affecting the quality of both groundwater and surface waters. The chemical analyses revealed that, in some samples, the concentrations of Ca, Cd, Pb, Sb and SO_4 exceeded the Turkish Drinking Water, U.S. EPA and WHO limits. Isotope analyses were carried out to determine the origin of the polluted waters and showed that contamination is taking place in the vicinity of the waste disposal site (Baba *et al*., 2003). In 1997, wet disposal was improperly exercised in the Seyitomer power plant area, causing ash flow over agricultural fields. Gulec *et al*. (2001) reported that Cr and Co contents released from the Seyitomer power plant were exceedingly high when compared with the regulatory standards for drinking water and irrigation, whilst there was moderate soil pollution by Pb and Cu.

The disposal of power station fly ash results in significant environmental problems. The leaching of coal fly ash during disposal is a concern, especially for the aquatic environment. Ugurlu (2004) examined the leaching behaviour of fly ashes currently disposed of in the Kemerkoy Power Plant fly-ash-holding pond. The Ca and Mn concentrations decreased with increasing temperature, whereas Na and K concentrations increased. The results showed that the most important effects of fly ash leaching were pH, Na, Ca, K, Fe, Mg, Mn and Pb. In particular, use of low quality coal with high ash content results in huge quantities of fly ash to be disposed of. Ugurlu (2004) indicated that the main problem related to fly ash disposal was the heavy metal content of the residue. Numerous experimental studies have shown that toxic trace metals can be leached from fly ash (e.g. Davison *et al*., 1974; Klein *et al*., 1975; Campbell *et al*., 1978; Gehrs *et al*., 1979; Hulett *et al*., 1980; Hansen and Fisher 1980; Burcu *et al*., 1997; Georgakopoulos *et al*., 2002).

Using the European Committee for standardization (CEN) and toxicity characteristic leaching procedures (TCLP) of the U.S. Environmental Protection Agency (U.S. EPA), Baba and Kaya (2004) conducted toxicity tests for several fly ash samples taken from the thermal power plants of Soma and Tuncbilek in the western part of Turkey. The results showed that the ash samples can be classified as `toxic waste' based on the TCLP results, but as `non–toxic' wastes based on the CEN results: this suggests that test results were pH-dependent.

3.4 Septic Tank Systems

Septic tank systems are another important source of anthropogenic contamination. Septic tank systems may be present in homes, offices, or other buildings. Septic tank systems are designed to slowly drain human waste underground at a harmless rate. However, an improperly designed, located, constructed, or maintained septic system can leak bacteria, viruses, household chemicals, and other contaminants into the groundwater, causing serious problems. Currently in Turkey there are thought to be many abandoned and uncontrolled septic disposal waste sites and the numbers grow in different parts of urban areas every year.

Kaçaroglu and Gunay (1997) studied the groundwater pollution in the Eskisehir Plain, an urban area. They reported that the groundwater in the Eskisehir Plain alluvium had been polluted by municipal and industrial wastewater and agricultural activities. The nitrate concentrations at nine sampling points on the River Porsuk, the main watercourse in the plain, ranged from 1.5 to 63 mg/l during the period from July 1986 to August 1988. In the same period, the nitrate concentrations measured in water from 51 wells in the alluvium ranged between 2.2 and 257 mg/l. In 34.2 % of the groundwater samples, the nitrate content was above 45 mg/l, the upper limit for nitrate in drinking water standards in Turkey. High nitrate levels were observed by Karacoglu and Gunay (1997) in water from wells in the central and eastern parts of the urban area.

In the Isparta Plain region, Karaguzel and Irlayici (1998) indicated that sewage water in the city canals and waste storage were the largest polluters. They concluded that almost two–thirds of the aquifer in the Isparta Plain was severely polluted. The groundwater at shallow depths in the alluvium, composed of gravel, sand and clay, is expected to be most contaminated under and around the open waste-disposal site, where both household and industrial wastes are disposed of improperly. In a different study, Karaguzel *et al*. (1999) showed that groundwater in the urban area of Antalya was contaminated by sewage discharge, industrial works, and other activities that created an ever-expanding impact on the region's only available aquifer.

Oztas (1997) studied groundwater contamination around the catchments of the Bakirkoy Basin (Istanbul), and concluded that groundwater had been severely polluted due to leakage water from domestic septic tank systems. Oztas (1997) reported also that, at the northern part of Lake Kucukcekmece in Istanbul, groundwater had been seriously contaminated due to the presence of septic tank waste.

In the Buyuk Menderes Valley, the continuous discharge of untreated domestic sewage water into the river has caused the groundwater to become heavily contaminated (Karamanderesi and Helavaci, 2001). Due to rapid population rise and urbanization, water consumption has been increasing and this will probably lead to more serious environmental problems in the near future.

At the Golbasi waste disposal site in Ankara, located in a small stream valley, sewage water is discharged directly into the stream, so that groundwater around the region is now significantly contaminated. Toxic elements, released from the disposal sites, are then carried into Lake Eymir through groundwater flow.

The disposal of industrial and domestic waste onto karstic environments poses particular dangers for groundwater. The carbonate rocks of Turkey, which underlie about one third of the country, hold major water resources with great potential for electricity generation and water supplies. The ongoing development of karst aquifers, especially in the southern part of Turkey, demonstrates specific environmental problems that occur in many karst water supply projects (Elhatip, 1997; Elhatip and Gunay 1998). Karst environments are used also for the disposal of liquid and solid domestic, agricultural, and industrial wastes, which result in karst groundwater pollution. Karst aquifers have specific hydraulic and hydrogeological characteristics that render them highly vulnerable to pollution from human activities. Karst groundwater becomes polluted more easily and quickly than water in non-karstic aquifers (Kaçaroglu, 1999). Scarcity of water, particularly in towns situated along the Mediterranean coast where the main aquifers are karstic, necessitates greater thoroughness in exploiting and protecting the groundwater resources.

Recharge areas tend to be located at higher elevations and in regions remote from urbanized areas. Dolines and ponors (sink and swallow holes) are commonly utilized as injection points for wastewater, while uvalas and poljes (karstic valleys and depressions) are used as solid waste disposal sites. When doing this, people are normally unaware of the connection of such sites with the wells and springs that provide their water supply (Ekmekci and Gunay, 1997).

3.5 Imperfect Well Construction

In aquifers, good and poor quality waters are sometimes separated from each other by low permeability (aquitard) geological structures. This natural balance can be destroyed through poorly completed (imperfectly designed) wells that penetrate the aquitard into the deeper aquifer unit (Aslan and Akkaya, 2001).

In Turkey, because many geothermal wells are not properly designed, aquifers are becoming polluted and the production of good quality fresh water has decreased. Such problems occur especially around the Germecik–Omerbeyli, Aydin–Yilmazkoy, Denizli–Kizildere, Izmir–Balcova, Izmir–Aliaga, and Izmir–Seferhisar geothermal fields (Ozudogru and Babur, 2001). Because Bozcaada–Canakkale does not have sufficient drinking water reservoirs, wells were drilled around the Pinarbasi–Ezine basin. However, samples taken from the wells contained ammonium, methane gas, and other chemical and biological contaminants. Akpinar (2001) indicated that biological and chemical pollution observed in groundwater around the Pinarbasi–Ezine Basin was mainly due to imperfect well designs in the region. Such studies show clearly that the design and construction of wells is extremely important if clean water is to be obtained and pollution is to be prevented.

Pamukkale is a protected site located 20 km from the provincial centre of Denizli in southwest Turkey. This site is unique naturally and culturally, with hot water springs, white travertine terraces, and ruins of the ancient city of Hierapolis. However, the increasing interest in this site has led to an irreversible deterioration of the natural and cultural assets, a contributing factor to which has been poor well construction around the site (Dilsiz, 2002).

4. CONCLUSIONS

A number of occurrences in the urban areas of Turkey have demonstrated that, no matter how efficient the technical work, the protection of water resources is primarily related to the attitude and policies of the local authorities. They must take adequate measures to protect the resources and educate the public in this issue. To achieve this aim, it is important to involve the public administrative and technical sectors in preparing guidelines for integrated environmental evaluation of water resources. Thus, protection measures that consider specifically the vulnerability of the environment are required to preserve the quality and quantity of groundwater. In order to preserve groundwater, the geological, hydrological and hydrogeological characteristics of the urban areas must be investigated and information on polluting activities and sources collected. Then, a comprehensive protection and control system must be developed, including the establishment of a groundwater monitoring system, the delineation of critical protection zones, the control through effective land planning and elimination of sources of pollution, and the development of increased public awareness of the value and vulnerability of aquifers.

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GROUNDWATER POTENTIAL AND HYDROGEOLOGICAL CHARACTERISTICS OF ÇORLU, TURKEY

A Case of Over-abstraction of Good Quality Groundwater Resources

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Abstract: The sustainability of the clastic Tertiary aquifers supplying the Corlu region of Turkey has been examined. It is found that: (i) with recharge of 194 million m^3/year and abstraction of 271 million m^3/year, groundwater levels are declining annually by about 1 m; (ii) serious pollution has not been recorded in the region, and groundwater quality is satisfactory for industrial applications at least. Water quality is therefore less of a concern than water quantity. The latter, however, requires active intervention.

Key words: Corlu, Turkey; Thrace, Turkey; over-abstraction; water levels; Tertiary aquifers; water quality

1. INTRODUCTION

This study covers the eastern part of the Ergene basin, Turkey. Long-term water level measurements from observation monitoring wells in the research area are available from which groundwater yield and ground water quality have been obtained. Ital Consultants undertook initial studies in the 1970s, producing the Ergene Basin Master Water Plan Report**.** The study estimated the groundwater potential of the basin to be 274 million m^3/year. Additionally, Atabay (1977) suggested that underground water availability from the Eocene limestone aquifer in the northern part of the basin was 45.5 million m^3/year.

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A continental climate is prevalent over the Ergene basin, with precipitation occurring mostly in the winter and spring. An abundance of groundwater in the basin has brought textile industries to the region in the last few years, leading to factories being built on many 1st and 2nd class agricultural lands. Industrial water use, heavily reliant on groundwater and insufficiently regulated, has caused a drop in groundwater level. Changes in ground- and surface water quality have also occurred (Aktimur *et al.*, 1994), including along the Çerkezköy-Çorlu reach of the River Ergene.

2. HYDROGEOLOGY

The basement of the Ergene basin is formed of Paleozoic amphibole schists, gneisses, granodiorite, marble, greenschist and calc-schists. In the north of the basin, this is overlain by Eocene limestone, frequently fractured by vertical step faulting of a south-southwesterly orientation (Çağlayan and Yurtsever, 1998). There are some sources of groundwater along these different-sized fault lines, with available reserves of 90 million m^3/year (Ital Consultants, 1970). In the Çorlu-Çerkezköy region, Tertiary sediments lie unconformably above the Paleozoic units, divided into the Ergene Formation and Thrace Formation. The Ergene Formation consists of white, grey, or light yellow, poorly consolidated or unconsolidated medium-coarse sandstone and, in the upper layers, green clay, pebbles and red mudstone and poorly consolidated siltstone. The dominant lithology is a poorly-consolidated fine sandstone, with a clay layer, a red mudstone with sand lenses and scattered pebbles above. Cross-bedding is observed in the white, fine-grained sandstones, and the Ergene Formation is interpreted as the deposits of a braided or meandering river system.

The Thrace Formation is interpreted as a series of alluvial sediments (Çağlayan and Yurtsever, 1998). It consists of unconsolidated pebbles, sand and clay grains. The thickness of the Pliocene deposits reaches 600 m in the basin centre; in the central and southern regions of the basin, there are volcanic outcrops of augite-olivine basalt (Şentürk and Karaköse, 1998). In the south, Quaternary deposition is represented by alluvium. The beds of the River Ergene and its tributaries are formed of pebbles, silt, and clay, and are frequently terraced. The thickness of the bed materials can reach 25 m at some sites (Figure 1).

The geological formations of the Ergene basin can be separated into two main aquifers: Lüleburgaz-Ahmetbey and Çerkezköy-Havsa-Hayrabolu (Çengel *et al.*, 2001). Located in the south-southeastern part of the basin, the Çerkezköy–Havsa-Hayrabolu aquifer (Figure 2) is the subject of this study.

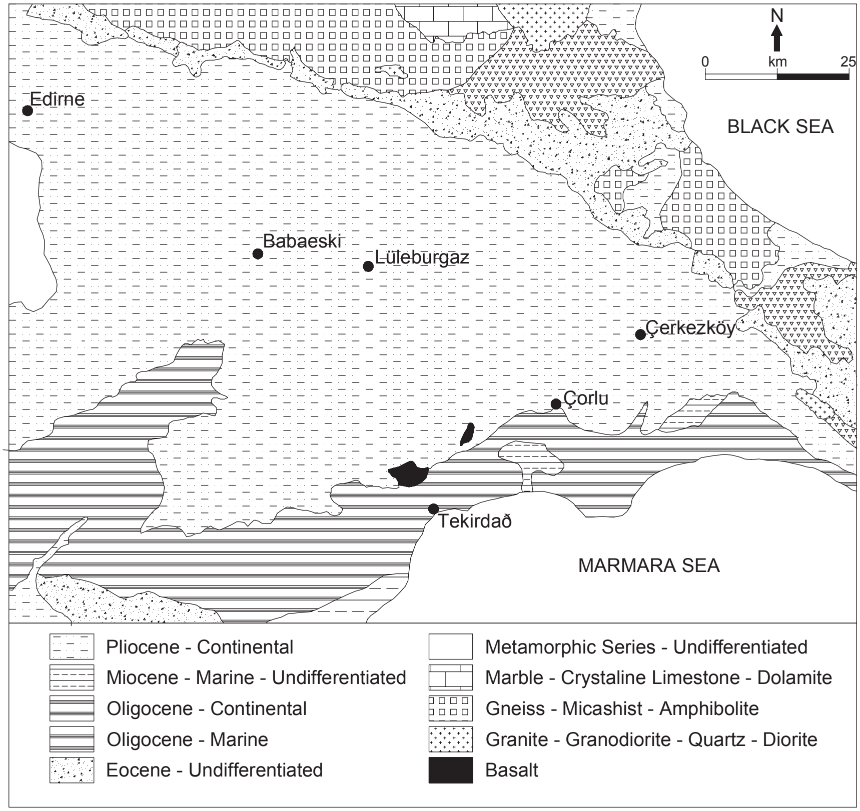

Figure 1. Geology of NW Turkey based on Erentöz and Pamir (1987).

Both older Ergene Formation and younger Thrace Formation outcrop in the area. The unconsolidated Thrace Formation is transitive within the Ergene Formation. Logs of the Ergene Formation reveal that it contains clay lenses that are apparently continuous laterally and vertically thick. Within the Ergene aquifer, the presence of laterally and vertically discontinuous clay lenses decreases the efficiency of wells. Flow rate of the wells is controlled by the occurrence and distribution of sand, silt, clay and pebble pockets, with coarse-grained sediments (sand, pebbles) increasing well efficiency and fine-grained sediments (clay, silt) decreasing the efficiency.

The available operating reserve of groundwater in the Ergene basin is 329.2 million m^3/year; 0.02% of Turkey's groundwater reserves (Çengel *et al*., 2001). In the Thrace region, total surface water potential is 2461 million m^3/year. The total irrigated area within the region is 156,672 hectares. The total drainage area of the basin (Çorlu and its vicinity) is 210 km^2 (S_T), total river length within the basin is 444.45 km (L_T), and total drainage density (L_T / S_T) is 2.11. The basin is classified as having a high drainage density.

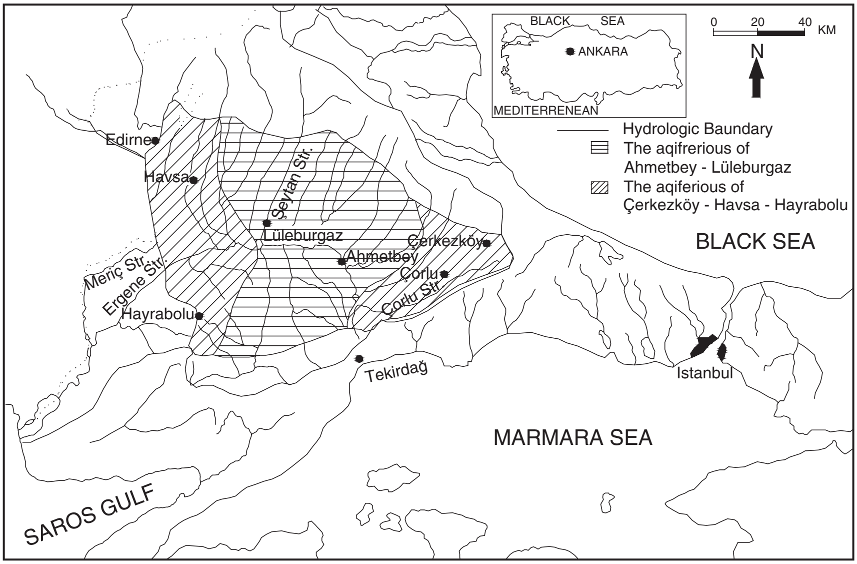

Figure 1. Hydrological map of the Thrace (Trakya) region.

The hydraulic conductivity of the Ergene Formation, the main aquifer in the study area, is 10-15 m/day, the storage coefficient $3x10^{-3}$ and transmissivity 500 m^2/day. The conductivity of the Thrace formation is 10-50 m/day and transmissivity 800 m^2/day (Kırsaç, 1999). The Ergene basin has around 3000 deep wells with depths ranging from 100 to 350 m.

The study area is in the transition zone of the Mediterranean and Black Sea climatic zones. Summers are hot and moist usually, with winters mild and rainy, and annual precipitation is 571 mm. In terms of monthly average precipitation, November to January are the wettest months, and June to September is the dry season. The hottest month is July, with an average temperature of 22.3^{o}C, whilst the coldest month is January (2.8ºC average). Average evaporation is 918 mm/y (Ulusoy, 1989).

Most of the land surface in the study area is permeable. The coefficient for rainwater infiltration is taken as k = 15%. With average annual precipitation taken as P_m = 0.570 m, precipitation recharge (Q_p) = A x P_m x k = 210 x 0.570 x 0.15 = 1.7955 x 10^7 m^3/y. There is also input from surrounding drainage basins. Bahçıvan (1995) indicated that this total area is approximately 1161.2 km^2. If this value is taken as the input area, then from Q_p = A x P_m x k, Q_p = 1161.2 x 0.570 x 0.15 = 9.928 x 10^7 m^3/y. Discharge from the study area occurs as natural springs, and surface water and groundwater extractions. Groundwater pumping from water supply wells is estimated to be around 2.05 x 10^7 m^3/year (Bahçıvan, 1995).

All available groundwater derives from the recharge of basin rainfall. Input flow from the basin boundaries, surface water flow, springs, and output from the boundaries can be considered as the groundwater operating reserves. Due to water volumes withdrawn from the system via artesian wells, springs and pumping wells now being greater than natural recharge, very significant declines in water levels can be observed in the aquifers (Table 1). Such withdrawals can clearly not be sustained in the long term.

Table 1. Average annual groundwater levels in observation wells in the aquifer at Çorlu and surrounding region (D.S.I. XI. Region observation wells measurement values).

Year	Çerkezköy Well 806	Marmaracık Well 12626	Dambaslar Well 5592	Ahmetbey Well 52282
	Groundwater level (depth from ground surface) (m)			
1969	42.8	17.0	5.2	
1972	42.9	17.0	5.5	
1975	43.1	17.3	6.0	
1978	44.4	18.6	9.5	
1981	45.1	19.4	8.8	36.9
1984	46.3	20.6	9.9	41.2
1987	46.1	20.5	16.4	43.6
1990	50.0	26.0	18.7	47.9
1993	54.4	29.5	21.4	50.8
1996	64.6	33.3	24.7	
1999	72.8	33.8	25.7	
2002	76.4	33.8	5.2	

3. HYDROCHEMISTRY

Briefly reporting on hydrochemical data, groundwater taken from these wells has been physically and chemically tested and the analysis shows the order of cations in the water to be $Ca^{2+}>Na^{+}>Mg^{2+}>K^{+}$ and the anion order $HCO_3^{-}>Cl^{-}>SO_4^{2-}$. Due to the underlying geology, the groundwater is of a calcium-bicarbonate type. The pH value of water for industrial uses is normally required to be between 7 and 8, with chlorine concentrations of under 200 mg/l and organic substances <30 mg/l. For groundwater used in concrete manufacturing, the upper limit for sulphate (SO_4^{2-}) is 150 mg/l and for free HCO_3^{-} is 20 mg/l. The groundwater in Çorlu and surrounding area satisfies these conditions and is thus utilized by the concrete industry. No organic pollutants were detected in the chemical analyses; this is interpreted as a consequence of the groundwater level being very deep.

4. CONCLUSIONS

Concerning the sustainability of the groundwater resource of the Çerkezköy-Havsa-Hayrabolu area of the Thrace region the following is concluded. The area recharge has been determined to be 194 million m^3/year. However, since withdrawal is greater (by 76.7 million m^3/year), groundwater levels are declining by about 1 m per annum. This is not sustainable requiring active intervention. Serious pollution has not been recorded in the region's groundwater, and it satisfies the water quality requirements for at least industrial applications. Chemical and physical analyses indicate a calcium-bicarbonate type water. In terms of resource sustainability, it is water volume, rather than quality that is the key concern.

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EVALUATION OF GROUNDWATER OVER-ABSTRACTION BY INDUSTRIAL ACTIVITIES IN THE TRAKYA REGION, TURKEY

A Case of Urban Groundwater Resource Over-abstraction

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Abstract: This study examines the effects of rapidly increasing industrial activities on groundwater levels in the Trakya Region of Turkey where groundwater is the main resource for industrial, agricultural, and domestic water supply. The Ergene River Basin (11,325 km^2) is the most important catchment, around half of which is underlain by the unconfined 600 m thick Çorlu/Ergene aquifer of Miocene / Pliocene age. Using water-table monitoring data from 13 monitoring wells the relation between potential pumping and water table fluctuation was evaluated. The average aquifer reserve and the total discharge of the area per year were determined as 340 and 460 Mm^3 respectively. The annual recharge was calculated as 123 Mm^3. The results show that the total amount of discharge (outflow + withdrawal) is much higher than the groundwater recharge, i.e. the system is not sustainable.

Key words: Trakya, Turkey; Thrace, Turkey; Turkey; water level; over-abstraction; Tertiary aquifers.

1. INTRODUCTION

To achieve sustainable management and development of groundwater systems, natural groundwater discharge and pumped volumes should not exceed the recharge rate of the groundwater basin. The latter depends on many factors, including soil, topography, vegetation, and climate.

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One of the most important problems in the region of Trakya, Turkey (Figure 1) is that groundwater management has never been properly considered during the planning or processing stages of industrial activities. The General Directorate of State Hydraulic Works (DSI) is the only Turkish authority responsible for such matters. It has, however, had insufficient employees to check for and determine illegal groundwater consumption. There are numerous wells without legal permits since the low fines for illegal groundwater usage are an insufficient deterrent. Significant reductions in the groundwater levels have been attributed to aquifer over-abstraction.

In 1970, Italconsult reported that the total groundwater potential of the basin was around 274 Mm^3/y. The northern side of the basin, Sazlıdere, has a potential of 45.5 Mm^3/y (Atalay, 1977). However, water demand for soil/water cooperatives and domestic usage in the region was determined as 210 Mm^3/y and 170 Mm^3/y, respectively (DSI, 2001).

The Trakya region has a soil structure suitable for agricultural activities. Of the eight-level soil classification used in most countries, the region has first and second-class soils. The first class soil is suitable for regular cultivation where no special conservation measures are necessary. The second class soil is also suitable for regular cultivation but requires simple (non-structural) soil conservation measures. However, since the 1990s, economical activities in the region have changed from agriculture to industrialization. Textile industries have grown significantly in the surrounding area of Lüleburgaz-Çorlu-Çerkezköy, constituting 72% of total industrial establishments in the region. Such industrial establishments have therefore developed large areas that were previously agricultural land.

2. GROUNDWATER OVER-ABSTRACTION STUDY

2.1 Study Setting and Monitoring Networks

The data used in this study include water level-monitoring data from 13 wells installed in water-table aquifers at national groundwater monitoring stations, and daily precipitation data from the Beyazköy Meteorological Association. All calculated values were obtained from Trakya University Ergene River Basin Project, Hydrological Report of Northern Ergene (Atalay, 1977) and Ergene River Basin Groundwater Development Report (DSI, 2001).Uncontrolled industrialization has led to over-exploitation of groundwater resources becoming a significant problem. Figure 1 shows the distribution of industry and other land use across the Trakya region.

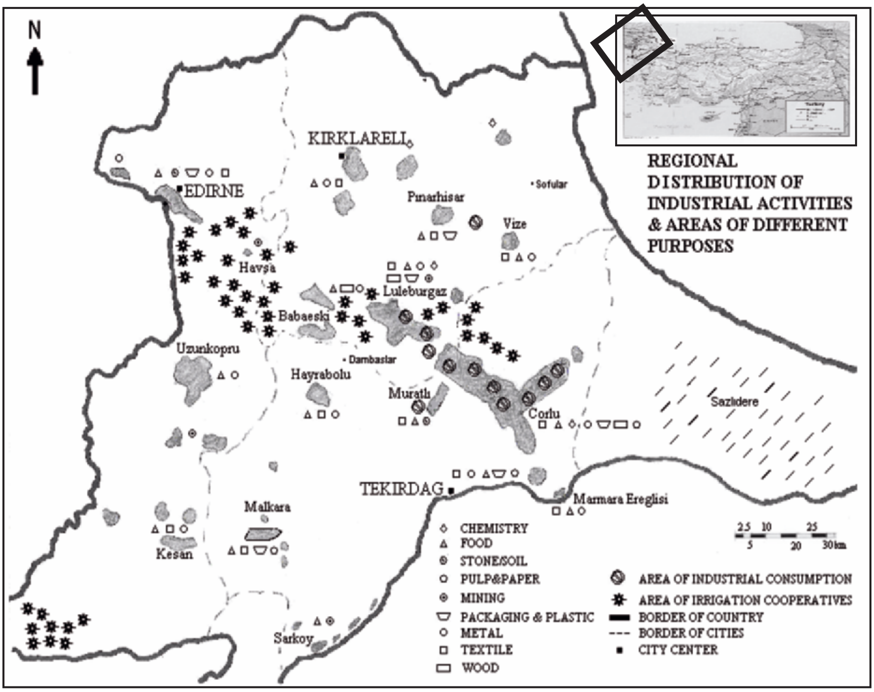

Figure 1. Distribution of industrial and other land use across the Trakya region, Turkey (study area is marked in the inset map of Turkey).

Previous studies examining rainfall and recharge suggest a simple modelling approach to estimate recharge from precipitation records, water balance values and water-table fluctuations (Bear, 1979; Sophocleus, 1991). Similar approaches have been applied since the 1960s to create a general groundwater potential profile for the Trakya region. There are 13 main controlling wells and a number of research and observation stations (ROS) for groundwater level and quality measurements monitored by DSI (Figure 2). There are 321 water supply wells allocated to Soil/Water Cooperatives (agricultural activities) and approximately 3000 wells for other activities. Well depths varies between 100-250 m and all extract water from the Mio-Pliocene aquifer. The permeability and transmissivity of the Miocene rocks are 10-15 m/d and 500 m/d. The Pliocene strata have a transmissivity of 800 m^2/d and a permeability value of 10-50 m^2/d (DSI, 2001).

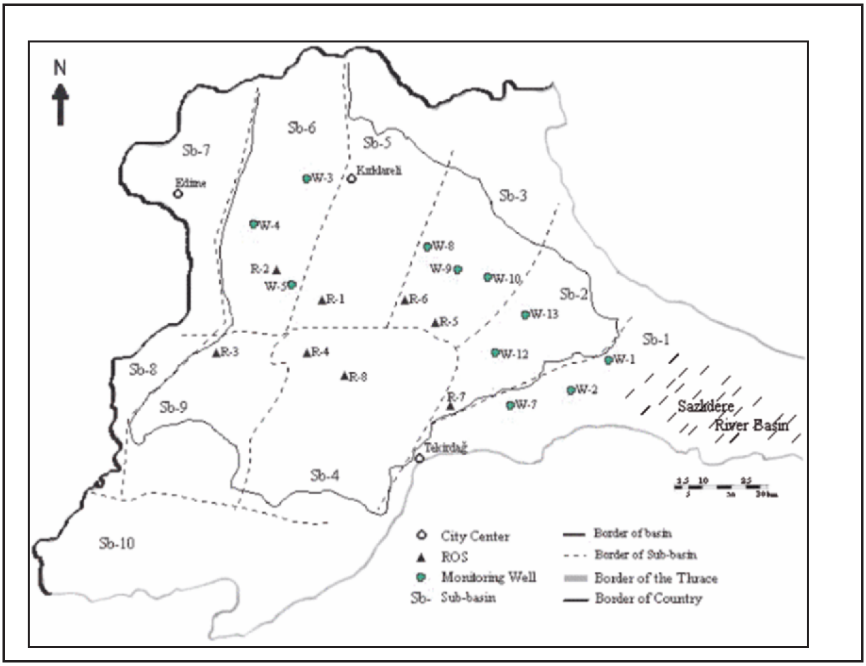

Figure 2. Monitoring wells and research observation stations in Trakya region study area.

2.2 Rainfall Data

A reduction in rainfall in the region since the 1980s has led to reduced recharge during rainy seasons with an arid period since 1983. Groundwater levels have declined and recharge has reduced. Industrialization has grown since the 1990s with an increase in water demand. Figure 3 shows the rainy and arid seasons with rainfall data (Beyazköy Meteorological Station). The annual average rainfall is 504.6 Mm^3 (1996-2000). Beginning with 1966, the difference between average rainfall (AR) and the total average (TA) rainfall was calculated (Table 1). The quantity of rainfall in each year was added to the budget of the following year to calculate the accumulated rainfall budget for every year between 1966 and 2000 (Figure 4).

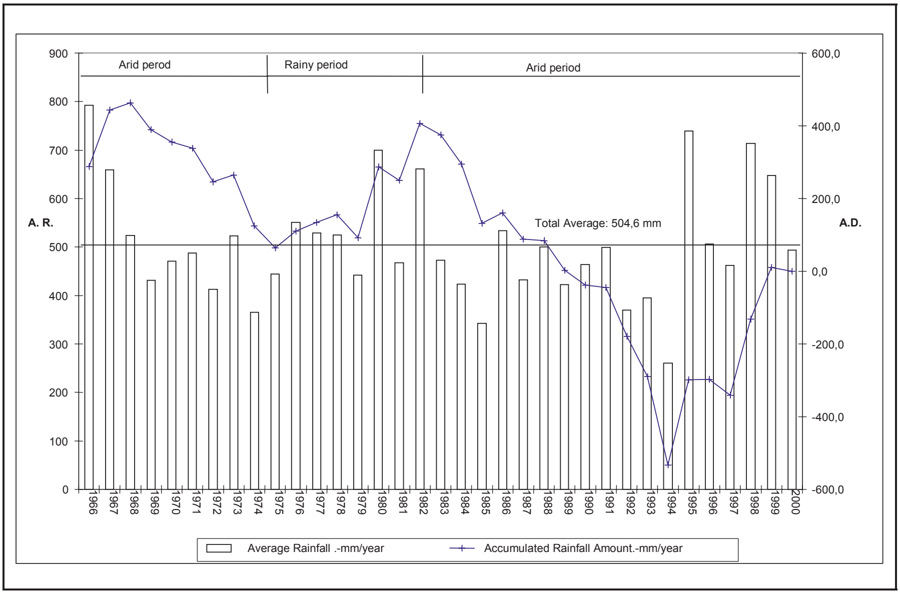

Figure 3. Average rainfall (A.R.) with Rainy and Arid periods shown.

Table 1. Rainfall measurements (mm/y) between 1966-2000 (AR = average rainfall; AD = accumulated average deviation).

BEYAZKÖY METEOROLOGICAL STATION											
Year	AR	AD	Year	AR	AD	Year	AR	AD	Year	AR	AD
1966	792.9	288.3	1975	443.8	64.6	1984	423.7	294.1	1993	394.9	-288.9
1967	659.6	443.3	1976	550.3	110.3	1985	342.1	131.6	1994	260.2	-533.3
1968	524.2	462.9	1977	529.2	134.8	1986	533.9	160.9	1995	739.1	-298.8
1969	431.7	389.9	1978	524.6	154.8	1987	432.5	88.8	1996	505.8	-297.7
1970	470.3	355.6	1979	442.2	92.4	1988	500.5	84.7	1997	461.9	-340.4
1971	487.2	338.2	1980	699.6	287.4	1989	422.3	2.4	1998	713.3	-131.7
1972	412.1	245.7	1981	467.5	250.3	1990	463.7	-38.6	1999	647.1	10.8
1973	523.4	264.5	1982	661.5	407.2	1991	498.9	-44.3	2000	493.8	0
1974	365.5	125.4	1983	472.5	375.1	1992	369.7	-179.2	TA	504.6	

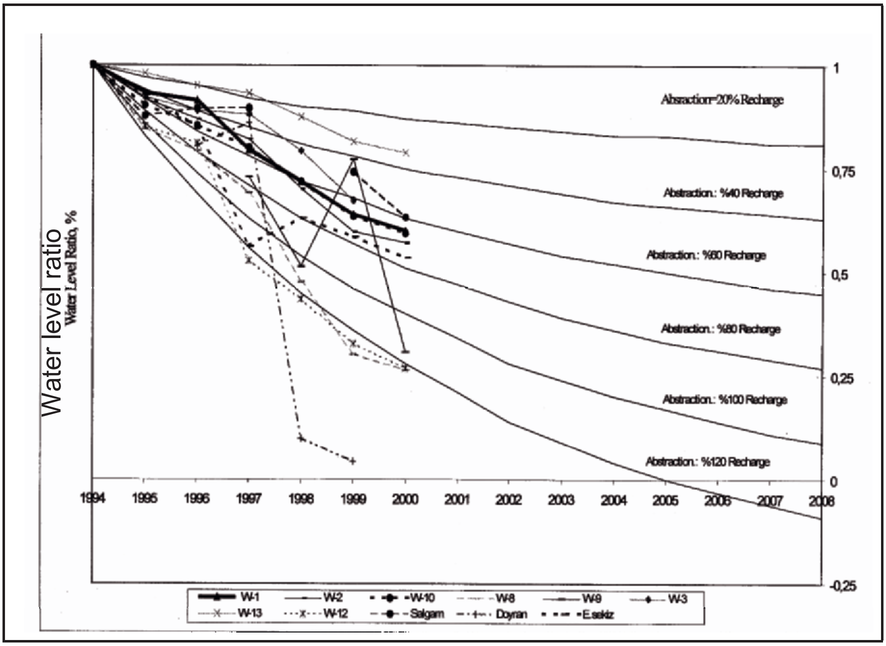

Figure 4. Comparison of observed groundwater level data with dynamic reserve predicted trendlines for abstraction set at varying percentage of recharge (data are plotted relative to 1994 data).

2.3 Groundwater Budgets

Based on daily rainfall data, November - May were determined as the recharge period and April - September as the irrigation period. Taking into account industrial demand and drinking water allocations of previous years, the amount of groundwater permitted for abstraction was determined at monthly intervals and given in Table 2 along with the 1994 aquifer groundwater budget. Although abstraction was unrestricted and groundwater resources were used freely until 1994 (for agriculture and drinking), the amount of water consumed was not more than 20% of the natural recharge. Tables 3, 4 and 5 show the measured and calculated amounts of dynamic reserve, which is the quantity of groundwater reserve remaining after water abstractions. Groundwater abstraction and groundwater flow are two major source of aquifer drainage. In order to estimate dynamic reserve changes within the Ergene aquifer, the DSI made a series of calculations based on different assumed levels of abstraction (A) and recharge (R), as follows: A=20%R, A=40%R, A=60%R, A=80%R, A=100%R, and A=120%R.

Table 2. Groundwater budget, 1994, A=20%R (where: volumes are in Mm^3; A = level of abstraction; R = recharge) (DSI, 2001).

z Drainage Coefficient : 0.01185 month^{-1}				
Months	Dynamic Reserve	Recharge	Abstraction	GW Flow
October	2340.3	0.0	5.0	28.1
November	2359.8	52.2	5.0	27.7
December	2431.3	104.5	5.0	28.0
January	2424.9	27.4	5.0	28.8
February	2395.1	4.0	5.0	28.7
March	2387.5	25.8	5.0	28.4
April	2407.3	54.1	5.9	28.3
May	2442.3	69.6	6.1	28.5
June	2406.7	0.0	6.7	28.9
July	2371.3	0.0	6.9	28.5
August	2336.7	0.0	6.4	28.1
September	2303.6	0.0	5.5	27.7
Total		337.5	67.5	339.8

The relationship between dynamic reserve and groundwater level is summarized in Figure 4 (based on Tables 3-5). Demand from industrial activities, the extraction of groundwater for natural gas research and production activities since 1977, and the effect of aquifer geologic boundaries have created local variations in groundwater level within the aquifer. Figure 4 suggests the quantity of groundwater withdrawn varies locally around wells. The ideal ratio of recharge to water abstraction for the aquifer was taken to be 100%.

As industrial activities have grown rapidly, so has the demand for water. If the reductions in groundwater level and the calculated dynamic reserve lines are compared (Figure 4), unless the ratio of dynamic reserve and reserve consumption is assumed to be 1:1, the demand for water will not be met. The ratio of groundwater levels pre 1995 (no industrial groundwater abstraction) and the change in these levels over time were used to calculate the average value of drainage coefficient for the aquifer (DSI, 2001).

Table 3. Groundwater budget of the Ergene aquifer from 1994 to 2008 based on A=20%R and A=40%R (DSI, 2001). The water level ratio is expressed relative to 1994 being unity.

Abstraction : 20% Recharge				Abstraction : 40% Recharge			
Year	Dynamic reserve		GW Flow /Drainage	Year	Dynamic reserve		GW Flow /Drainage
	Mm^3	Water-level ratio	Mm^3		Mm^3	Water-level ratio	Mm^3
1994	2373.42	1.00	339.82	1994	2373.42	1.00	339.82
1999	2105.72	0.89	304.14	1999	1854.59	0.78	304.14
2000	2071.56	0.87	299.58	2000	1788.39	0.75	299.58
2001	2041.96	0.86	295.64	2001	1731.02	0.73	295.64
2002	2016.30	0.85	292.22	2002	1681.29	0.71	292.22
2003	1994.07	0.84	289.25	2003	1638.19	0.69	289.25
2004	1974.79	0.83	286.68	2004	1600.84	0.67	286.68
2005	1958.09	0.83	284.46	2005	1568.47	0.66	284.46
2006	1943.61	0.82	282.53	2006	1540.41	0.65	282.53
2007	1931.06	0.81	280.86	2007	1516.09	0.64	280.86
2008	1920.19	0.81	279.41	2008	1495.01	0.63	279.41

Table 4. Groundwater budget of the Ergene aquifer from 1994 to 2008 based on A=60%R and A=80%R (DSI, 2001).

Abstraction : 60% Recharge				Abstraction : 80% Recharge			
Year	Dynamic reserve		GW Flow /Drainage	Year	Dynamic reserve		GW Flow /Drainage
	Mm^3	Water-level ratio	Mm^3		Mm^3	Water-level ratio	Mm^3
1994	2373.42	1.00	335.86	1994	2373.42	1.00	333.89
1999	1603.42	0.68	233.23	1999	1352.25	0.89	197.78
2000	1505.18	0.63	220.14	2000	1221.96	0.79	180.41
2001	1420.03	0.60	208.79	2001	1109.04	0.71	165.36
2002	1346.23	0.57	198.95	2002	1011.17	0.63	152.32
2003	1282.27	0.54	190.43	2003	926.34	0.57	141.01
2004	1226.83	0.52	183.04	2004	852.82	0.51	131.21
2005	1178.78	0.50	176.63	2005	789.10	0.47	122.72
2006	1137.14	0.48	171.08	2006	733.88	0.43	115.36
2007	1101.05	0.46	166.27	2007	686.01	0.39	108.98
2008	1069.77	0.45	162.10	2008	644.53	0.36	103.45

Table 5. Groundwater budget of the Ergene aquifer from 1994 to 2008 based on A=100%R and A=120%R (DSI, 2001).

Abstraction : 100% Recharge				Abstraction : 120% Recharge			
Year	Dynamic reserve		GW Flow /Drainage	Year	Dynamic reserve		GW Flow /Drainage
	Mm^3	Water-level ratio	Mm^3		Mm^3	Water-level ratio	Mm^3
1994	2373.42	1.00	331.91	1994	2373.42	1.00	329.93
1999	1101.08	0.46	162.32	1999	849.91	0.36	126.87
2000	938.75	0.40	140.69	2000	655.54	0.28	100.96
2001	798.05	0.34	121.93	2001	487.07	0.21	78.51
2002	676.11	0.28	105.68	2002	341.05	0.14	59.04
2003	570.82	0.24	91.59	2003	214.50	0.09	42.18
2004	478.82	0.20	79.38	2004	104.81	0.04	27.56
2005	399.43	0.17	68.80	2005	9.75	0.00	14.88
2006	330.62	0.14	59.63	2006	-72.65	-0.03	3.90
2007	270.98	0.11	51.68	2007	-144.06	-0.06	-5.62
2008	219.29	0.09	44.79	2008	-205.95	-0.09	-13.87

2.4 Sub-basin Evaluation of Groundwater Reserves

Water potential measured at each observation well and the quantity of water withdrawn are given in Table 6. Sub-basin Sb-6 (Figure 3) with its 10.8 Mm^3/y reserve, sub-basin Sb-1 with its 12.4 Mm^3/y reserve, drainage area R-4 of sub-basin Sb-4 with its 12.6 Mm^3/y reserve and sub-basin Sb-2 with its 8.1 Mm^3/y reserve (total = 21.8 Mm^3/y) could supply water for the activities requiring water in those areas.

Since the values measured by R-3 (328.2 Mm^3/y), placed at the outlet of the river basin, and those collated from stations across the river basin (320.1 Mm^3/y) are similar, groundwater drainage of the Ergene basin was accepted as equal to the R-3 value. Adding the value (9.8 Mm^3/y) measured from Sazlıdere drainage area, the total groundwater drainage for the system was calculated approximately as 337.5 Mm^3/y.

The average ratio of water abstracted to total reserve was 93%, supporting the assumption that a 1:1 ratio of recharge to withdrawals was suitable for this region (Table 6). Calculated annual groundwater outflow was 146.6 Mm^3/y, assuming the quantity extracted was 93% of operational reserve. The annual change in groundwater reserve is given in Table 7.

Table 6. Remaining reserve values for each sub-basin of the Ergene aquifer system.

ROS	Sub-basin Sb-	Operational Reserve/Baseflow (Mm^3)	Abstraction/ Recharge %	Gwater Withdrawn (Mm^3)	Reserve Remaining (Mm^3)
R-1	5	30.3	120	36.4	-6.1
R-2	6	53.9	80	43.1	10.8
R-4	4	63.0	80	50.4	12.6
R-6	3	48.1	120	57.7	-9.6
R-8	4	22.6	120	27.1	-4.5
R-5 / R-7	2	40.3	80	32.2	8.1
R-7	1	61.8	80	49.4	12.4
Sazlıdere	7	*9.3*	120	11.2	-1.9
Total		329.3	93	307.5	21.8
R-3	9	*328.2*			
Sazlıdere + R-3		337.5	93	313.9	23.6

Table 7. Summary of Groundwater Budget (GW = groundwater).

GW Recharge (Mm^3/y)	Baseflow (Mm^3/y)	Dynamic Reserve (Mm^3/y)	Discharge Coeff (day^{-1})	GW Outflow (Mm^3/y)	GW Drawn (Mm^3/y)	Total Drainage (Mm^3/y)	Change in Reserve (Mm^3/y)
364.1	337.5	2373.4	0.000395	146.6	313.9	460.5	123.0

3. CONCLUSIONS

The effect of over-exploitation by industrial activities on the groundwater level in the Trakya region has been evaluated in this study. Industrial activities are not the only reason for the reduction of groundwater level: natural gas production and research activities since 1977 have also caused loss of groundwater reserves in the region.

According to the measured values taken from ROS stations (No R-3), average aquifer reserve was determined as 337.5 Mm^3/y. The change in the reserve per year was calculated as 123 Mm^3/y. It was observed that the quantity of water extracted was higher than the reserve in the sub-basins Sb-3, Sb-5, Sb-7, and in drainage area R-8 of sub-basin sSb-4, due to over-exploitation and geological boundary limitations. These critical areas, where withdrawals are higher than recharge, should be taken under control and excessive pumping prohibited.

The results show that the total amount of drainage (outflow + withdrawal) was much higher than the groundwater baseflow. Total drainage was observed as 460.5 Mm^3/y and base flow was 337.5 Mm^3/y in the region. The change in the aquifer reserve was determined to be 123Mm^3/y. In the region, the ratio of reserve recharge to groundwater abstraction must be controlled, and should not exceed 100% in order to prevent over exploitation (mining) of the aquifer.

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A LONG-TERM PERSPECTIVE ON THE SUSTAINABLE DEVELOPMENT OF URBAN GROUNDWATER RESOURCES IN ROMANIA

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Abstract: Romania is the beneficiary of an important complex of water resources. The underground waters are estimated to contain about 9.6 x 10^9 m^3 as a resource that can be exploited, such that supply is greater than demand. However, as various forms of water pollution are present in Romania, some sources of water have become unusable, giving the demand/supply ratio a more complex significance. Generally, fertilizers, livestock industry waste, and household effluents can be regarded as the main sources of groundwater pollution by nitrate. The Romanian environmental authorities have established provisional guidelines concerning the controls on discharge of three substances, including trichloroethene. The principles for the sustainable management of water resources, recommended at the Rio de Janeiro Conference (1992), also lie at the foundation of the concept of integrated water management in Romania, where the problems of water usage have to be balanced with the need to protect natural ecosystems.

Key words: Romania; pollution; nitrate; trichloroethene; environmental management; sustainability; water quality.

1. INTRODUCTION

As a resource, water is necessary to every socio-economic activity and its management is an important branch of the national economy. Unlike other resources however, water is both irreplaceable and permanently renewed in a natural process, the hydrological cycle. The quantity and quality of water resources is a pivotal factor in the general development of the national economy and of land management.

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Romania is the beneficiary of an important complex of water resources, comprising groundwaters, drainage waters and natural lakes internal to Romania, the Danube River, and the Black Sea.

2. CHARACTERISTICS OF GROUNDWATER RESOURCES IN ROMANIA

Romania's water resources are made up of groundwaters and surface waters. In Table 1 the estimated quantities for each of the resource types are given: the resource estimates from the interior rivers and the Danube are based on their average discharge.

Table 1. Water resources in Romania.

Resource	Theoretical resource (Gm^3)	Technically usable (Gm^3)	Technically usable resources under present conditions (Gm^3)
Groundwater	9.6	6	3
Interior rivers	40	25**	13
River Danube	85*	30	10
TOTAL	134.6	61	26

* Represents 50% of the annual flow of the Danube River in the Bazias sector, at the entrance in Romania;

** Collects circa 5 Gm^3 of secure resources under normal circumstances.

Underground waters are found in medium and deep aquifers and are estimated to contain about 9.6 Gm^3 as a resource that can be exploited. In relation to the country's population, it can be said that there is more supply than demand, as the average annual water consumption in Romania is 27 m^3 per capita (compared with the European average figure of 40 m^3 per capita).

However, there is an increasing problem of groundwater quality. As various forms of water pollution occur in Romania, some sources of water have become unusable, so that the demand/supply ratio acquires a more complex significance.

In groundwaters, a reduction in quality can be observed due to insufficient protection of deep strata from contaminated or waste water, especially in large industrial areas, and also due to the incorrect application of chemical fertilizers. The areas with a high degree of pollution are situated in the following basins: Ialomita - the fan-talus of Prahova-Teleajen; Arges - the Dimbovnic Valley; Sasar - Baia Mare area; Jiu - Targu Jiu area; Barcau - downstream Suplacu de Barcau.

3. GROUNDWATER POLLUTION IN ROMANIA

3.1 Status of Groundwater Pollution in Romania

At present, about 30% of the water used in Romanian cities originates from groundwater. However, since the second half of the 1970s, groundwater pollution due to organochlorine compounds (including trichloroethene) has become a major concern. In most cases, this pollution has arisen from improper management of hazardous substances in effluents and waste water.

In recent years, as a consequence of the revised Water Pollution Control Law, both local and national water and environmental agencies have carried out groundwater quality monitoring. A survey is divided into three types. Firstly, a general condition survey assesses the regional groundwater conditions. Then, any pollution detected by the general survey is identified via surveys on the areas around polluted wells, and its scope confirmed. Finally, periodic monitoring surveys are carried out, including continuous monitoring of the pollution in the vicinity of a polluted well.

When groundwater pollution is detected, besides carrying out guidance with respect to businesses that use hazardous substances, and guidance on the method of using a well, measures are carried out designed to improve the water quality of the polluted groundwater. The results of the 2001 national general condition survey are shown in Table 2.

Groundwater pollution by nitrate is a problem that arose in European Union countries and the United States in the 1960s due to the use of large amounts of nitrogen-based fertilizers in intensive agriculture. Groundwater pollution by nitrate has recently come to light in Romania: a survey of nitrate levels in water conducted in 11 of the regions of Romania in 2001 showed that 4.9% of wells surveyed exceeded the index value (10 mg/l) and thus required observation. Generally, fertilizers, waste from the livestock industry, and household effluents are the sources of groundwater nitrate pollution. Since it has been reported that nitrate is an environmental influence on infants, it is a problem that cannot be overlooked. Furthermore, with areas displaying high levels of pollution now present in almost every part of Romania, control of the situation, including counter measures in polluted areas, is becoming increasingly important.

3.2 Measures to Control Groundwater Pollution in Romania

Once groundwater is polluted, it is very difficult to restore its original water quality. Therefore it is important to take preventive measures before the pollution advances. The environment authority has established provisional guidelines concerning the controls on discharge of three substances (including trichloroethene) and since 1995 provided guidance to factories and other business establishments handling these substances. The Ministry of Health, the Ministry of Trade, and the Ministry of Industry have instructed the industries concerned, and the Ministry of Construction has conducted investigations of the groundwater. Despite these measures, however, the state of groundwater pollution by substances such as trichloroethene has not improved.

Table 2. Results of 2001 ground water quality survey1 in Romania (general conditions survey; Compania Nationala "Apele Romane", 2004).

Item	Number of wells examined	Number of wells > evaluation standards	% wells > evaluation standards	Evaluation standard [2]
Lead	1,313	3	0.22	≤ 0.01 mg/l
Hexavalent chromium	1,338	1	0.07	≤ 0.05 mg/l
Arsenic	1,280	18	1.40	≤ 0.01 mg/l
Total mercury	1,313	2	0.15	≤ 0.0005 mg/l
Tetrachloromethane	1,191	1	0.08	≤ 0.002 mg/l
1,1-dichloroethene	505	1	0.19	≤ 0.02 mg/l
Cis-1,2-dichloroethene	505	4	0.79	≤ 0.04 mg/l
Trichloroethene	2,240	7	0.31	≤ 0.03 mg/l
Tetrachloroethene	2,240	12	0.53	≤ 0.01 mg/l
Benzene	454	1	0.22	≤ 0.01 mg/l

Notes: 1. Levels of substances not listed in table were all below evaluation standards; 2. Evaluation standards used here are for potable waters. In recent revision of environmental quality standards, organic phosphorus was deleted and 13 substances (including dichloromethane) added (the evaluation standards were already set for trichloroethene and tetrachloroethene). Evaluation standards for lead and arsenic have been strengthened.

The Water Pollution Control Law promotes measures to prevent pollution caused by hazardous chemicals. With this law, measures such as the prohibition of the infiltration of water containing hazardous substances (e.g. trichloroethene) into the ground, and regular surveys of groundwater quality by the environmental agencies, have been implemented (Figure 1). Financial aid has been provided by the Ministry of Environment toward the expenses incurred in the water quality surveys, which are conducted in accordance with the groundwater quality measurement programme.

In coping with groundwater pollution, it is essential to make efforts to prevent pollution by strictly enforcing the Water Pollution Control Law.

Simultaneously, assessments of how the pollution was generated and measures for cleaning up groundwater must be implemented. The environmental authority has set provisional guidelines for the surveys, established general soil and groundwater survey procedures, and defined measures for cases where groundwater pollution has been confirmed. It also carries out research and investigations, such as on-the-spot verification surveys of pollution restoration methods, in order to further encourage the establishment of relevant technologies.

4. SUSTAINABLE DEVELOPMENT AND MANAGEMENT OF GROUNDWATER RESOURCES IN ROMANIA

The total water demand in Romania grew from 1.4 Gm^3 in 1950 to over 20 Gm^3 in 1989, as a result of population growth and industrial and agricultural development. After 1989, demand decreased to around 12 Gm^3, and water exploitation varied between 9.8 and 10.5 Gm^3/year (Guvernul Romaniei, 1999). The reasons for the decreased exploitation of the water resources are:

- Decreased industrial production, which produced a corresponding decrease in water consumption, now around 70-75% of the corresponding amount used in 1989.
- Sharp and steady decrease of water used for irrigation, now around 12 - 22% of the corresponding amount in 1989, caused primarily by difficulties experienced in ensuring the proper working of the irrigation systems and a decrease in the requirement for irrigation.

At the Dublin Water and Environment Convention of 1992, and at the UN Conference for the Environment and its Development, held at Rio de Janeiro in July 1992, the international community recommended governments adhere to the following principles for the sustainable management of water resources, all of which apply to Romanian groundwater:

- *The basin principle*, according to which water resources are formed and managed in hydrographical basins. The proper management of water resources (including groundwater resources) requires a global approach taking into account social problems and economical development as well as protection of natural ecosystems. A sustainable management of water resources can only be achieved at hydrographical basin level by integrating all water users.

- *The unitary quantity-quality management principle*. These two aspects of groundwater management being closely connected, a unitary approach is necessary in order to produce best technical-economical solutions in both areas.
- *The solidarity principle***.** The planning and development of groundwater resources requires the collaboration of all participants involved in the water sector: the state, local communities, users, water managers and non-governmental organizations (NGOs).
- *"The polluter pays" principle***.** All costs resulting from groundwater and environment pollution are to be paid by the polluter.
- *The economic principle – "the user pays"*. Groundwater has economic value in all its usable forms and has to be seen as an economic commodity. The management of groundwater as an economic commodity is an important way of achieving an efficient and equitable exploitation and of conserving and protecting the groundwater resources.

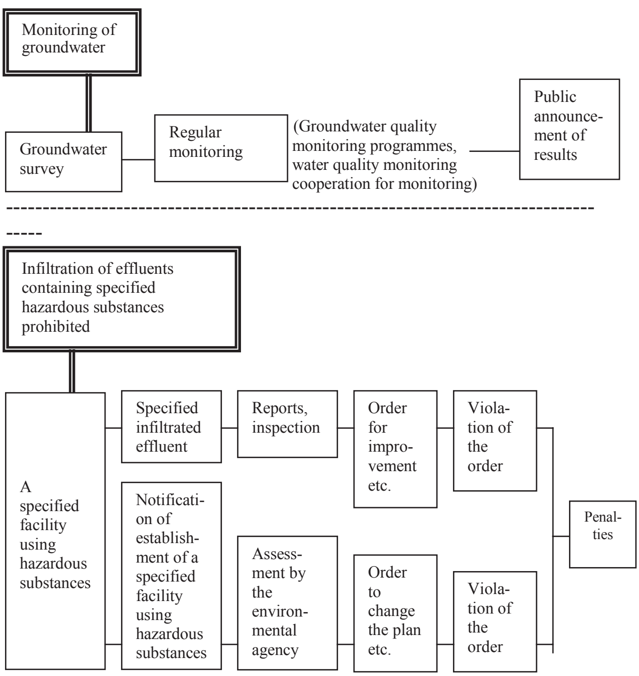

Figure 1. Groundwater pollution control system under Water Pollution Control Law in Romania.

These principles lie at the foundation of a concept of integrated water management, which combines the problems of water usage with the protection of natural ecosystems. In the sustainable management of water resources the targets to be pursued are:

- a) Continuous supply of water to users, especially to domestic users by:

Creating new sources of water, especially large storage basins and lakes with multiple uses in areas with water deficits. Deep subsoil water resources will be used especially to supply villages and towns with water.

Saving of water and reducing losses from distribution networks in towns, economic units and housing complexes.

Creating separate water supply systems for industry and for domestic users, in order to reduce the costs of making it potable.

- b) The improvement of the quality of water resources by:

Modernizing production processes by using clean, non-polluting technologies.

Creation of new water treatment facilities and updating of the existing ones in order to reduce the amount of polluting substances released into subsoil and surface waters.

Setting up a framework for the creation of hydrologically and hydrogeologically protected zones in some sensitive areas.

Improvement of the economic system of incentives in the field, by updating prices, tariffs and penalties for the products and water management services concurrent with the introduction of heavier fines for violations of the law. The implementation of a system of bonuses, granted to those users that show a constant interest in the field of groundwater protection.

Implementing of new methods of prevention, limitation and minimization of the effects caused by accidental pollution.

Improvement of education, in order to arouse the concern for a clean aquatic environment. This education process should begin in upper school levels and continue during employment.

- c) Ecological reshaping of rivers.
- d) Reduction in the risk of floods.
- e) Creation of basin committees:

Basin committees should involve all responsible or affected parties: the state, local communities, groundwater users, and managers.

The need to manage resources responsibly is set by law. The necessary conditions to achieve this are as follows:

- taking the best technical, scientific, and objective decisions during the planning of quantitative and qualitative management of groundwater exploitation by all interested parties;

- a specific legal system as regards the rules and laws connected with water and the economy of the country as well;
- the acknowledgment of the fact that managing groundwater resources is a dynamic activity and it is subject both to socio–economic constraints and to the requirements of the environment.

5. CONCLUDING REMARKS

There is an increasing problem of groundwater quality in Romania due to insufficient protection of the deep strata from an influx of polluted and used waters, especially in large industrial areas. Recently, groundwater pollution due to the use of large amounts of nitrogen-based fertilizer in intensive agriculture has also begun to come to light. Currently, under the revised Water Pollution Control Law, both local and regional water and environmental agencies are carrying out monitoring of groundwater quality. It has become essential to enforce this law strictly to prevent pollution, whilst investigating the sources of current pollution and devising measures for cleaning up groundwater. Meanwhile, Romania applies to its groundwater use the five principles recommended by the international community at the UN Conference held at Rio de Janeiro in 1992, considering them the foundation of integrated underground water management necessary for sustainable development.

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SECTION III:

GROUNDWATER FLOW

INTEGRATED HYDROLOGICAL MODELLING FOR SUSTAINABLE DEVELOPMENT AND MANAGEMENT OF URBAN WATER SUPPLIES

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Abstract: This paper describes the application of integrated surface-groundwater (ISGW) models in regard to the planning of urban water supplies, basin water management, sustainability of natural resources, and ecosystem preservation. A brief description of the integrated models is provided to illustrate their flexibility in assessing various schemes used in the management of urban water supply systems and basin water resources. A more detailed discussion of one of the most functional and widely used integrated models (MIKE SHE) and its applications at a rapidly urbanizing site in southwest Florida is also presented.

Key words: integrated hydrologic models; surface water; GIS database; MIKE-SHE; modelling; water supply;Florida, USA.

1. INTRODUCTION

1.1 Integrated Surface and Ground Water Model (ISGW) – A Brief Overview

ISGW is a first generation, integrated surface and ground water model that was developed by combining an existing surface water hydrologic model (HSPF) and a groundwater flow model (MODFLOW). ISGW has been applied extensively in west central Florida to the assessment of water supply permits, regional water resources, and the impact of groundwater withdrawals on local ecosystems. It has also been used as part of an operational wellfield rotation scheme for urban water supply management

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and water supply source optimization (Hosseinipour, 2002). Although ISGW was, in the late 1990s, the only functional model available for such applications, experience gained from its use revealed a number of issues that needed resolving to optimize performance. An important issue was that the model operated only in the Windows DOS environment but was not a truly Windows-based code, and also that it had limited pre- and post-processing capability. There were also a number of issues with the model's handling of vadose zone processes and the exchange of fluxes between the saturated and unsaturated zones, and with its accessing of the hydrologic databases that underpin the model's execution. A revised version of the model – the Integrated Hydrologic Model (IHM) – is being developed that will replace ISGW in the near future. A synopsis of the IHM due to be released in mid-2005 is given below.

1.2 Integrated Hydrologic Model (IHM) – An Overview

The Integrated Hydrologic Model (IHM) integrates the significant surface and subsurface processes of the hydrologic cycle into a single software package. Through the coupling of surface water and groundwater process models, represented by the HSPF V12 (Bicknell *et al.*, 2004) and MODFLOW-96 (McDonald and Harbaugh, 1996) models respectively, IHM provides a state-of-the-art, public domain, Windows-based capability to simulate the interaction between surface water and groundwater. IHM is designed to provide an advanced capability to predict the complex interactions of surface water and groundwater features in shallow water-table environments. Model components explicitly account for all significant hydrologic processes including precipitation, interception, vapotranspiration, runoff, recharge, irrigation flux applied to land, streamflow, wetland hydroperiod, baseflow, groundwater flow, and all the component storages of the surface, vadose, and saturated zones. Input requirements include precipitation and potential evapotranspiration time series, surface topologic features (i.e. land use, soils, topographic elevations, and derived slopes), irrigation fluxes, hydrographic characteristics, rating conditions, hydrogeological parameters of the groundwater system and information about well pumping and surface-water diversions. Its output provides detailed water balance information on all major hydrologic processes, including surface water and groundwater flows to wetlands, streams and lakes, evapotranspiration losses from all storage regions, reach stage, soil moisture, recharge to the groundwater system and storage, heads and fluxes in the groundwater system.

Aggregated, hydrologically similar response units are used in IHM by defining hydrologic land segments (HLS) within each surface basin. For

IHM, the hydrologic response of a basin contributing to a stream or wetland is the aggregated response of the various HLS elements within the basin. All hydrologically similar HLS elements are assigned the same values for infiltration, surface runoff, and ET parameters based on an areally-weighted average for the HLS computational element. However, variations in depth to the water table and groundwater ET characteristics are retained at the HLS scale in a very efficient scheme unique to IHM. Thus, while the location of HLS elements is not necessarily contiguous and while their size may also vary, the important characteristics of unique runoff, ET, and recharge behavior are retained at individual scales within a very efficient computational framework. In IHM, recharge and ET fluxes use both regular grid cells and irregular land segments within sub-basins. Therefore, the results from each discretization domain must be manipulated prior to transfer to the other domain.

Temporal discretization also varies between HSPF and MODLOW as a characteristic of the different timescales of surface and groundwater processes. The integration time step of IHM (the time interval over which time-averaged model results are transferred from one component model to the other) is specified to be the same as the stress period length of MODFLOW. Surface water features, such as lakes, wetlands, and streams, have a characteristic timescale different to that of rainfall/runoff processes. Therefore, HSPF reaches can use a different time step length than that used for the land segments. To provide time step compatibility, IHM integrating software aggregates HSPF results (e.g. in 15 minute, hourly or daily increments) into MODFLOW stress periods, and MODFLOW results are partitioned into appropriate periods for HSPF. Within a MODFLOW stress period, a time step length of less than the stress period length can be specified for MODFLOW simulation. Integration and component model time step lengths are variable and user-specified in IHM. Updated values for recharge, baseflow, stream-stage relationships, soil moisture, depth-to-water table, specific yield, and remaining potential evapotranspiration are passed to various integrating software components throughout the numerical integration time step.

Intersection of HSPF land segments with MODFLOW cells forms individual land fragments. It is not necessary for integration to maintain the individual land fragments of any particular land segment within a cell as there is no intra-grid detail for depth-to-water table. Within a cell, the number of unique land segments, including those that lie in different sub-basins, is equal to the number of aggregated land fragments. Each HSPF land segment is associated with one or more MODFLOW cells. By using land fragments to dynamically modify model variables for HSPF and MODFLOW, IHM provides 'semi-distributed' discretization of the model domain. Intra-cell variability in hydrologic behavior is facilitated by this approach. The advantage is computational efficiency provided by performing only one set of computations for all hydrologically similar fragments distributed over the sub-basin domain.

Surface water bodies, including streams, lakes, and wetlands, are distinctly discretized in HSPF and MODFLOW for integrated simulations with IHM. The hydrologic response of all wetland and open water features is simulated by coupling HSPF reaches and MODFLOW river package elements. An HSPF reach comprises multiple segments of streams and individual lakes and wetlands. The corollary to hydrologic response units for surface-water bodies is a segment of a stream or an individual lake or wetland. The individual lakes, wetlands, and stream segments are aggregated in a consistent manner to stay within the defined limit for HSPF reaches for the model application. An HSPF reach is comprised of one to multiple elements of the MODFLOW river package.

A three-layer soil moisture model is used to integrate transfer of flux and storage between HSPF and MODFLOW. The three layers represent the upper gravity zone, the capillary fringe zone (which is near saturation and is located in the lower portion of the capillary zone, just above the water table), and the upper portion of the capillary zone located above the capillary fringe. Due to temporal changes in depth-to-water table, soil moisture storage (HSPF) and specific yield (MODFLOW) vary and are dynamically modified by IHM using the soil moisture model at each integration step.

HSPF is used to simulate the distribution of ET among principal surface-water storages, including interception, depression and the vadose zone. Both vadose zone and saturated groundwater ET are dictated by vegetative cover characteristics including plant coefficient and root-zone thickness, soil conditions including soil water retention characteristics and soil moisture, and depth to saturated groundwater. The groundwater ET extinction depth and elevation at which groundwater ET reaches a maximum rate are explicitly defined in IHM. The IHM ET model provides a smooth transition in the simulated response, as the source of ET changes from predominantly surface storages to predominantly saturated groundwater storage and vice versa. The ET model also provides a smooth transition into direct evaporation from the soil surface when the water table nears the land surface.

IHM uses the standard text, binary, and time series files of HSPF and MODFLOW as input and output file formats. In addition, a Microsoft Access database is used to store, manage, and display data that are needed for pre and post-processing and for model integration during the simulation. Geographic Information System software is essential to manage spatial data during model application setup and for the visual display of simulation results. Public release of IHM, its documents, and the source and executable codes is anticipated to be imminent.

2. MIKE SHE

MIKE SHE is a deterministic, distributed and physically based modeling system for simulation of hydrological processes in the land phase of the hydrological cycle. The model is applicable to a wide range of water resource and environmental problems related to surface water and groundwater systems, and the dynamic interaction between the two regimes. The modeling package comprises a number of pre- and postprocessors to facilitate the input of data and the analysis of simulation results. Among others, these include spatial interpolation routines, graphical editing, and plots of the variations in space and time of any variable, as well as animation tools. MIKE SHE simulates the variations in hydraulic heads, flows and water storage on the ground surface, in rivers and in the unsaturated and saturated subsurface zones. The spatial variations of meteorological input data and catchment characteristics are represented in a network of grid squares. Within every grid square the soil profile is divided into a series of vertical layers.

The governing partial differential equations for the flow processes are solved numerically by efficient and stable finite difference methods in separate process components. All process descriptions operate at time steps consistent with their own most appropriate temporal scales. Hence the processes may be simulated using different time steps that may be updated during the simulation and coupled with the adjoining processes as and when their time steps coincide. This facility allows for a very efficient operation, making it possible to carry out simulations extending over long periods. Furthermore, the modular structure of the Water Movement Module of MIKE SHE allows the components to assume different levels of complexity according to local needs and preferences. A frame component coordinates and controls the operation of the individual modules; further details are provided in the MIKE SHE manual (2004).

The application of MIKE SHE to the assessment of flooding issues in urban areas with a high water table, the management of urban and agricultural water supply, and the sustainability of resources for developments, all in the region of South Florida, is discussed. Modeling results are presented to illustrate the model's capacity for conjunctive use/management of regional water resources. Among many differences between IHM and MIKE SHE is that of integration between surface water and groundwater; in IHM it is sub-basin-based while in MIKE SHE it is grid-based.

2.1 Applications of MIKE SHE in Southwest Florida

MIKE SHE has been used in many regions of South Florida. In most areas of its application, all components of the integrated model were used to study watersheds with a high water table and a complex hydrography (networks of interconnected streams, canals and estuaries, various hydraulic structures such as weirs and culverts, and control structures such as gates). Across the application areas, groundwater is used for domestic as well as agricultural supply. Reclaimed water, surface water, and shallow groundwater are also used for lawn irrigation. All of these sources are included in the irrigation component of the model as separate time series. Spatially distributed rainfall time series and Potential evapotranspiration time series are also included in the model database (for complete details refer to DHI, 2001). One particular case study is discussed in detail below.

3. APPLICATION OF MIKE SHE TO THE TIDAL CALOOSAHATCHEE RIVER BASIN (TCRB)

The tidal Caloosahatchee River basin Integrated Surface and Groundwater Model (ISGM) covers the section of the Caloosahatchee River basin downstream of a hydraulic structure (the Franklin Lock: S-79), where the water is saline. The model includes a suite of components capable of simulating flow on the overland plane, flow in rivers and canals, in the unsaturated and saturated zones, and evapotranspiration losses to the atmosphere, with an extension that describes the use, spatial and temporal distribution of water used in irrigation (Petersen et al., 2002).

The basin is relatively flat with little or no topographical relief (Figure 1). Larger depressions and sloughs with the capacity of retaining large volumes of storm water have been partially drained as part of agricultural development activities. A large number of wetlands and retention ponds are found scattered across the basin. The soils are generally coarse and sandy with high infiltration capacity. Horizons of low permeable finer sediments are found locally, especially in depression areas. The upper aquifer system consists of shells, sand and limestone with a relatively high hydraulic conductivity. Shallow water tables are found in most parts of the basin. The observed water table and stream stage response to rainfall indicates a close link between rainfall, surface water and shallow groundwater. The following components of the hydrologic cycle are represented in the TCRB model.

- Overland sheet flow and depression storage;
- Infiltration and storage in the unsaturated zone;

- Dynamic exchange between unsaturated zone-groundwater (recharge);
- Dynamic exchange between aquifers-rivers/canals (seepage);
- Groundwater flow, storage and potential heads;
- River/canal flow and water levels;
- Evapotranspiration;
- Drainage;
- Irrigation water allocation from various sources;
- Groundwater abstraction.

As a compromise between detailed model output and computational efficiency, a 457.2 m (1500 ft) computational grid was applied. Parameters and input data are combined to represent the average conditions within the computational cells. Because the timescales of the surface water and groundwater regimes are different, the model allows use of different time steps for calculation of river/canal flow and groundwater flow. The river hydraulics model time steps are between 2 and 5 minutes, while overland flow is solved in 6 hour time steps and groundwater flow calculations are based on a 12 hour time step. The area modeled is the lower part of the Caloosahatchee River and is located west of the Caloosahatchee Basin. The catchment is approximately 2,427 km^2 (938 square miles) and, as shown in Figure 1, consists of 11,600 square elements for each layer (about 11,000 internal elements).

3.1 Rainfall

Rainfall distribution in the basin is highly variable over both time and space. Local thunderstorms account for much of the rainfall volume. Accumulated rainfall measured at stations in the basin and surrounding areas does not show a clear geographical pattern and the total rainfall at the stations is generally determined by local weather phenomena. The rainfall distribution used in the model was originally based on Thiessen polygons generated from 19 rain gauges. Time series were gap-filled using the inverse distance method from other rain gauges. In the latest revision of the model, NEXRAD-generated rainfall data from SFWMD is used across the model domain, giving greater consistency, both temporally and spatially.

3.2 Geological Interpretations and Hydrogeological Parameter Assignments

Geostatistical techniques (ordinary kriging) were used to define top and bottom surfaces for the geologic units of the revised model, as well as aquifer parameters (horizontal hydraulic conductivity, leakance, and specific storage). The top and bottom surfaces defined for the geologic units were then directly incorporated into MIKE SHE models. In the case of aquifer parameters, the zones were based on raw data contained in the database developed to generate the interpolated aquifer parameter surfaces. This was considered necessary and appropriate since the initial parameter values may require adjustment during calibration of the model. The modification of an interpolated surface is not straightforward without using sophisticated parameter estimation software with built-in interpolation routines (e.g. the pilot point method). Use of sophisticated parameter estimation software was not considered because of the large number of parameters that are typically modified during the calibration of integrated surface water and groundwater models, and the long run times typical of these types of models.

3.2.1 Definition of Initial Aquifer Parameter Zones

The Theissen polygon interpolation method (polygon interpolation method – Isaaks and Srivastava, 1989) was used to define the initial aquifer parameter zones for the area. This method consists of generating polygons using perpendicular bisectors between data points and assigning the observed aquifer parameter value to the entire polygon area. In contrast to distance-weighted interpolation methods, discontinuities exist between adjacent zones. Although the discontinuities resulting from the interpolation method may not reflect the actual spatial distribution of aquifer parameters, it should be a reasonable approach given the sparseness of the available aquifer parameter data. Furthermore, it is recognized that the zonation used in the final calibrated models may differ significantly from the initial zonation for some parameters (except in the vicinity of the available aquifer parameter data where final values should correspond closely with the initial values).

3.2.2 Assignment of Observed Leakance Values in the Model Area

The leakance values included in the data set were unprocessed in a vertical sense (i.e. leakage in the tests may be from overlying or underlying aquifer units). Leakance values were not resolved into vertical hydraulic conductivities (K_v) because the data was insufficient to resolve the source of vertical leakage in many of the aquifer performance tests (APT) performed in the model area. In converting leakance to K_v values it was assumed that leakance occurring during the APT resulted from leakage through the thinner of the upper or lower confining to semi-confining unit.

It is recognized that this assumption may fail to accurately represent the K_v values of the confining to semi-confining units, especially when the thicknesses are similar, but it is a reasonable approach to take given the lack of appropriate data necessary to address this issue more rigorously.

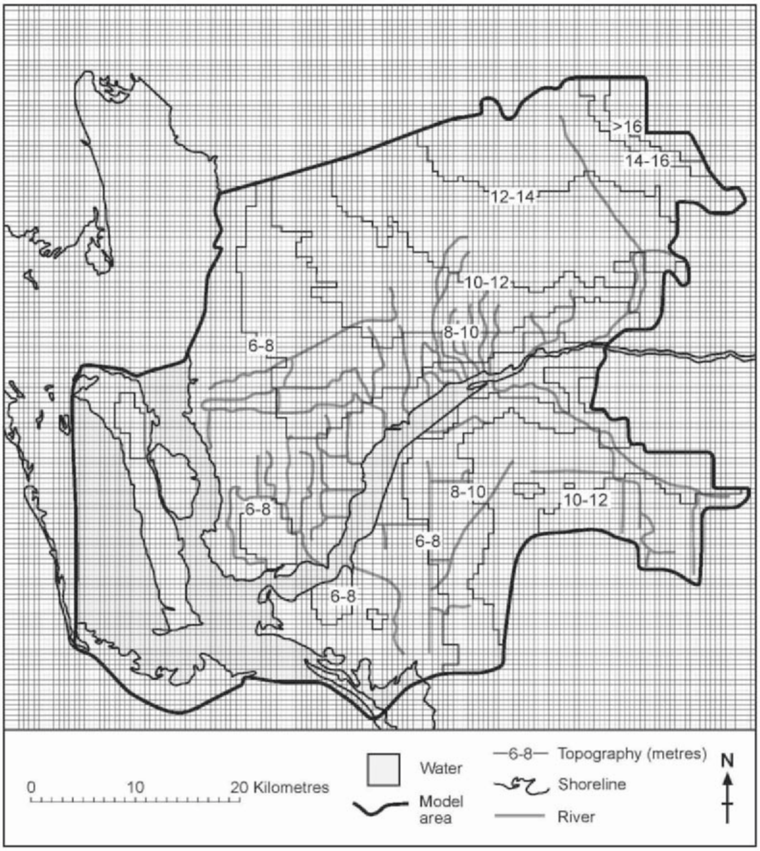

Figure 1. TRCB model boundary, grid, and surface hydrography network.

After assigning an APT leakance value to a particular geologic unit, leakance values were converted to K_v values by multiplying the leakance value by the closest interpolated aquifer thickness in the aquifer geometry data sets. In areas where the thickness of the confining or semi-confining units was less than one foot (0.3 m), the horizontal and vertical properties of the aquifer units below these formations were used for the confining unit (in effect making the thin units part of the underlying aquifer).

3.2.3 Assignment of Horizontal Hydraulic Conductivity Values in the Model Area

The horizontal hydraulic conductivity (K_h) data was used directly for the Holocene-Pliocene (HP), Lower Tamiami (LT), and Sandstone (SS) aquifer units. As indicated above, the K_h value of the confining units was assumed to be equal to the calculated K_v value of the units. The initial K_h/K_v ratios of the aquifer units were based on the general characteristics of the aquifer materials. An initial K_h/K_v ratio of ten was used for the HP and SS aquifer units because they are predominantly composed of clastic material. An initial K_h/K_v ratio of thirty was used for the LT aquifer since it is composed of carbonate materials that typically have strong horizontal to vertical anisotropy.

3.2.4 Assignment of Observed Specific Storage Coefficients in the Model Area

The specific storage coefficient (S_s) for the aquifer units was derived from the interpolated raster data provided by the SFWMD for the LT and SS aquifers. The interpolated values are considered appropriate since it is anticipated that this parameter will not be adjusted during model calibration. An initial default S_s value of 3.3 $x10^{-5}$ 1/m was used for all confining to semi-confining units and the HP aquifer unit.

3.3 Parameter Adjustment During Calibration of the MIKE SHE Model

The initial parameter zonation was altered significantly during calibration to achieve a better fit of simulated results with observed groundwater levels, surface water stages, and/or surface water discharges. The following methodology was used to modify parameters during calibration:

K_h values in the vicinity of APT locations were maintained within a factor of 5 of the reported values unless calibration data was determined to be inconsistent with the reported APT values.

K_v values in the vicinity of APT locations with leakance values were maintained within a factor of 10 of the reported values unless calibration data was determined to be inconsistent with the reported APT values.

The orientation of initial aquifer parameter zones may be adjusted during calibration to be consistent with observation data or to better replicate observed/inferred groundwater flow directions.

Additional zones could have been added during calibration to improve calibration at observation locations.

3.4 Groundwater Abstractions (Water Use)

All major well fields with an abstraction rate greater than 40 m^3/day were included in the model database. Public water supply (PWS), industry and other water consumers are included in the water use, and a total of 300 major wells are included in the model. For the largest well fields a time series of monthly abstraction rates was used and for smaller wells where no information on pumping was available, a constant abstraction rate was applied. Since no information on individual well abstraction rates within a well field was available, a constant level of abstraction was assumed. This is a valid assumption since most wells are clustered in the same grid cell or neighboring cell(s).

Domestic wells not connected to the public water system were also taken into account. Lee County provided a well permit database that contained information including depth, casing size, water use, well yield, and parcel identification number. The well database was geo-referenced with the parcel shapefile using the parcel identification number. In the parcel shapefile, some parcel identification numbers were listed in more than one parcel polygon. In such cases, the parcel with the largest area was used to join the well database information.

3.5 Drainage

For the Caloosahatchee River basin, drainage patterns are determined partly by hydraulic control (pumping) and partly by gravity. In densely drained areas most drainage flow is discharged into nearby canals. Drain codes (Drainage Options) have been used to overrule the routing by levels and direct local drainage to nearby canals. The drainage component was found to affect both river discharges and groundwater levels strongly. The drainage model parameters were estimated from available data and general knowledge of the field drainage operation (storm water controls).

3.6 Channel Flow (MIKE 11)

Stream flows and water levels were simulated within all major creeks and drainage canals in the basin using MIKE 11, the river hydrodynamic component of the MIKE SHE model. MIKE 11 is a full unsteady river hydraulics model, dynamically coupled to MIKE SHE such that the integrated modeling system dynamically accounts for the exchange of water between the river and groundwater components. In each time step the exchanged volumes are updated for all computational points of the model.

Inflow and outflow from the river may take place as groundwater seepage (calculated from simulated water level differences between the aquifers and the river reaches), overland flow (surface runoff driven by actual overland water depths and surface slope) and drainage flow (groundwater drainage flow routed to rivers/canal network).

The model uses a MIKE 11 river system that includes the Caloosahatchee River and all major canals, creeks, and tributaries. A total of 63 branches were included in the MIKE 11 river networks system. Natural cross sections from previous HEC-2 models were applied to 20 of the branches; survey data are used for the remainder. For the Caloosahatchee River, cross sections were based on a soundings map from the SFWMD. Figure 2 shows the surface water network and calibration points.

A total of 145 culverts, 167 weirs, seven operated structures with 2-4 gates each, and two tabulated structures (Amil gates) were applied to the model. The MIKE 11 setup is coupled via links to MIKE SHE for most branches. However, some branches are located too close to each other or other branches and, because of the grid resolution, it was necessary to exclude some of these branches from coupling. The MIKE 11 setup uses a global bed resistance of Manning's M = 20 $m^{1/3}/s$ (Manning's n = 0.05), and leakage coefficients between groundwater and surface water in the range 1×10^{-7} to 1×10^{-6} s^{-1} for the reaches found by calibration.

3.7 Boundary Conditions

Depending on location and data availability, groundwater boundary conditions of fixed head, no flow, and variable head time series were used. Along coastal areas, fixed heads were typically applied, whilst on the landward sides either no flow or fixed head was assigned. In areas where a number of good quality observation wells were available, time series of water levels were used. For some scenarios simulated results from the adjacent models were used. For surface water, boundary conditions are specified at upstream and downstream ends of the river network. A constant upstream flow boundary condition of 0.001 m^3/s (0.035 cfs) was applied for streams and canals to avoid numerical instability. This boundary flow is not significant with respect to the water balance of the basin. The total flow introduced through the boundary conditions is only 0.06 m^3/s (2.1 cfs) or 0.8 mm/year (0.03 inches/year), which is negligible in terms of the basin's total water balance.

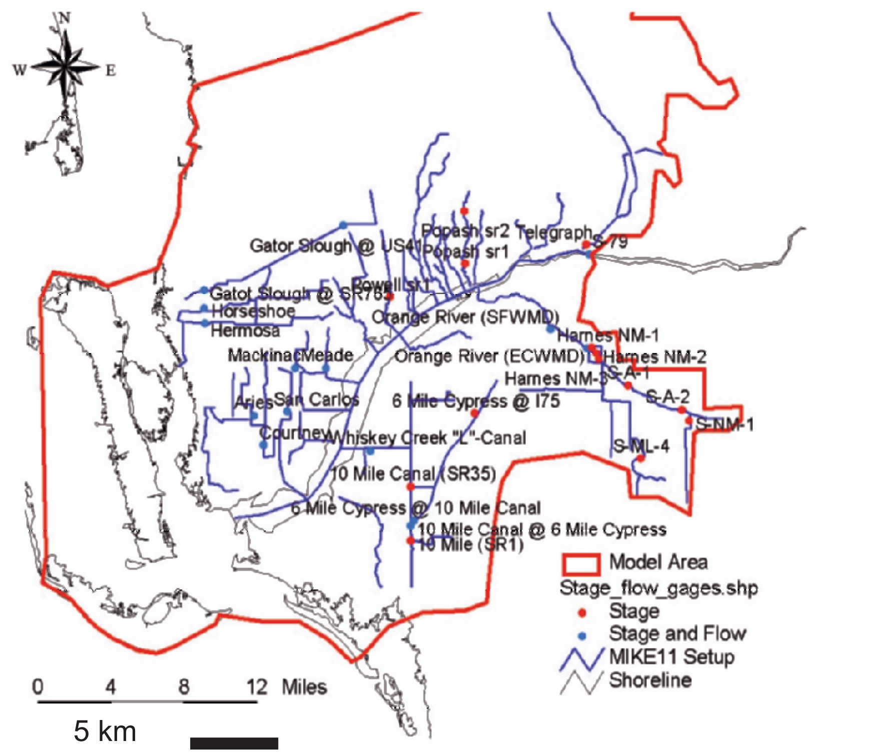

Figure 2. Surface water hydrography of the Caloosahatchee River basin, as represented in the MIKE 11 network.

A time series of measured discharge from the Franklin Lock (S-79) was used as an upstream discharge boundary condition for the Caloosahatchee River. Tidal boundary conditions were calculated using measurements for all downstream rivers/canals discharging into the Gulf of Mexico. Three time series were generated: Caloosahatchee River/Cape Coral Canals, Gator Slough/Horseshoe Canal, and 10 Mile Canal/Hendry Creek. Finally, a fixed water level of 8.1 m (26.5 ft) was applied in Bullhead Strand when simulating the bypass flow from Telegraph Swamp to upstream of S-79. Flow in Bullhead Strand is not lost from the model area, but shows up as a discharge at S-79 since, as described above, an observed discharge time series is used here.

3.8 Basin Land Use

A land use map was created based on vegetation types (2000 GIS-coverage) and a vegetation database with crop characteristics from the

Caloosahatchee Basin ISGM. The database contains 22 types of land use, including Citrus, Pasture, Sugar Cane, Urban, Forests, Mangrove, Marsh/Wetland, Grass, Truck Crops (melons, vegetables, etc.), and 'Water' (wetlands, marshes, small lakes). Some landuses are further broken into sub-categories (e.g., low and high density urban, etc.). The volume of water used for irrigation is important to the water budget of the tidal Caloosahatchee basin. The purpose of simulating irrigation is to quantify demand and describe the effects of irrigation water allocation. The irrigation module (MIKE SHE IR) was applied in order to generate spatially and temporally varying irrigation demands based on simulated field conditions. The module also simulates allocation of water from groundwater wells, irrigation canals, or from sources outside the model area, to meet the irrigation demand and its effects on the basin water balance.

3.9 Irrigation Areas

The irrigation module was applied focusing on the objectives of the irrigation, namely to provide sufficient water for crop transpiration. The water resources in the watershed are to a large extent controlled to achieve optimal conditions for the various crops. Certain crops, such as citrus and truck crops (vegetables), depend partially or entirely on irrigation and must be irrigated as part of a profitable agricultural production. Improved pasture areas may also be irrigated in parts of the basin, but in general it is assumed that pasture is not irrigated. Golf courses and areas with domestic wells are also considered for irrigation. The domestic wells are simulated by first abstracting the water from the shallow domestic wells, and secondly, irrigating the areas with water from an external source.

A total of 85 irrigation command areas and 12 domestic well polygons were included in the model, although some irrigation areas are only represented by a single grid. Irrigation areas smaller than 50% of a grid cell, (i.e. smaller than 0.209 km^2/1,250,000 square feet), were not included. Land use was checked against the permit coverage and is consistent with the irrigation map, so that modeling of irrigation takes place only where the land use is as specified.

3.10 Irrigation Demand

Irrigation demand depends on many factors, is highly variable in time, and relies on estimates of actual evapotranspiration. Such estimates are based on available meteorological data and aim to calculate the supplemental water required to maintain potential rates of evapotranspiration for the respective crops. The use of an integrated hydrologic model, simulating the water content in the root zone and the actual evapotranspiration rates, offers the opportunity of an alternative approach. By focusing on the purpose of irrigation (the consumptive requirements of crops), rather than either describing fixed rates of supply or attempting to describe the actual operation of structures at field level, it is possible to formulate irrigation targets and determine the irrigation required to meet the targets. In the MIKE SHE irrigation module, irrigation demand can be given for each agricultural area prior to the simulation or it can be regulated from a set of management criteria. Some of the criteria are:

- Maximum allowable soil water deficit in the root zone;
- Maximum allowable crop water stress (E_{act}/E_{pot});
- Prescribed time series of crop water requirements.

The first option has been used for the tidal Caloosahatchee River model. In the vegetation database the targeted upper and lower limits of the average root zone soil water content are specified as:

$$\Theta_{fc} - 0.1(\Theta_{fc} - \Theta_w) < \tilde{\Theta} < \Theta_{fc}$$

where Θ is the actual mean soil water content of the root zone, and Θ_{fc} and Θ_w are soil water content at field capacity and wilting point, respectively. The total soil water volume available for transpiration through plant uptake is $(\Theta_{fc} - \Theta_w)$. During the simulation, whenever the actual soil water content drops below 90% of this volume the maximum allowable water deficit of the root zone is exceeded and irrigation water is supplied. The demand is calculated as the difference between water content at field capacity and actual water content (i.e. $\Theta_{fc} - \tilde{\Theta}$). If available, the water is allocated from associated irrigation sources and supplied at the rate $(\Theta_{fc} - \tilde{\Theta})/\Delta t$ to the unsaturated zone in the following time step of the simulation. The allowable deficit is given relative to soil properties to account for the differences in soil properties in the model area.

The soil water content is kept close to field capacity to prevent any reduction in actual evapotranspiration rates due to soil water availability. To keep soil water content within this narrow range, irrigation water is supplied at a high frequency corresponding to an optimized operational schedule. Using field capacity as the soil moisture reference level for irrigation is considered an appropriate approximation for citrus and truck crops.

3.10.1 Irrigation water allocation

Irrigation water is supplied by conjunctive use of surface water and groundwater. The availability of surface water and groundwater varies both seasonally and geographically within the basin. As a general rule, the use of surface water is less costly than the withdrawal of groundwater. Groundwater is used mainly in areas without an irrigation canal network or when surface water resources are scarce. A link between irrigation command areas (fields) and specific locations for water allocation is established by means of GIS pre-processing. Each irrigated area is associated with a prioritized list of river locations (defined by branch name and chainage) and/or groundwater wells from where the required irrigation water volume is allocated (if available). Limits are specified for each source in terms of a minimum river flow/stage, groundwater potential head, or maximum pump capacity. At a given time, the generated demand is met by exhausting the available resources at specified locations in order of priority until sufficient water has been provided. If the total volume of available water does not cover the demand, shortage occurs.

A general assumption was adopted for water allocation: first and second priority is given to nearby irrigation canals while third priority is given to shallow groundwater wells.

4. MODEL CALIBRATION

Calibration and verification were performed for two periods applying both a relatively dry, and wet conditions. These periods were: 1994 to 1996 and 1997 to 1999 respectively. Calibration and verification results included:

- 70 observed and simulated potential heads in Water Table aquifer wells;
- 3 observed and simulated potential heads in deeper aquifer wells;
- 18 observed and simulated river stage in various creeks and canals;
- 15 observed and simulated discharge associated with discharge measurements;
- Duration curves and accumulated discharge for some creeks and canals when continuous flow measurements were available;
- A water balance for the entire catchment and calibration statistics.

A large quantity of model calibration plots have been produced with one such example being shown in Figure 3 that shows observed and simulated groundwater levels for an observation point. Calibrations of aquifer water table and stream stage and discharge data have all been successfully undertaken.

4.1 Calibration Statistics

A statistical analysis of the calibration and verification was performed for the groundwater wells in the aquifer water table. Both calibration and verification periods have significantly lower mean errors than the specified criteria, although individual wells can have a higher discrepancy. The statistical criteria are formulated as:

$$\frac{ME}{\Delta h_{\max}} \leq \beta_1 \qquad ME = \frac{1}{n}\sum_{i=1}^{n}\left(H_{obs,i,j} - H_{sim,i,j}\right)$$

$$\frac{RMS}{\Delta h_{\max}} \leq \beta_2 \qquad RMS = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(H_{obs,i,j} - H_{sim,i,j}\right)^2}$$

where: n is the product of total number of observations at all observation times; ME is the mean error; RMS is the root mean square error; $H_{obs,\mathrm{i,j}}$ is the observed value at time i and at a observation location j; $H_{sim,\mathrm{i,j}}$ is the simulated value at time i and at a observation location j.

The β_1 and β_2 -criteria are fulfilled in all cases, indicating that the potential heads in the aquifer water table are well simulated.

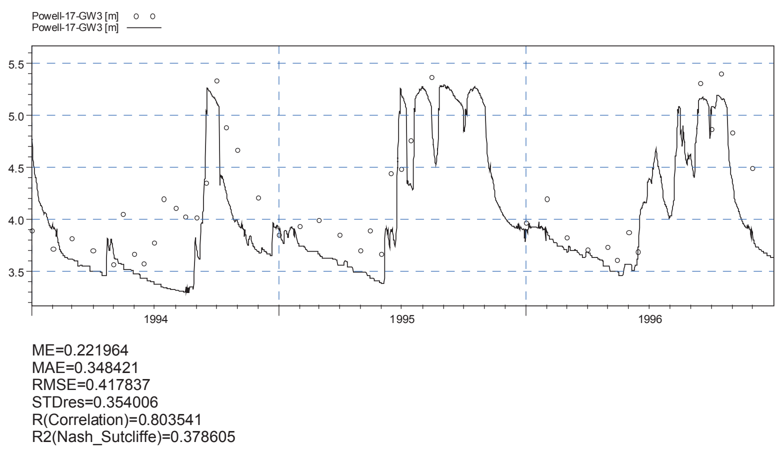

Figure 3. Example simulated and observed groundwater level data for point GW3 with model fit statistics recorded below.

5. CONCLUSIONS

5.1 Summary

An integrated hydrological model has been developed for the tidally influenced part of the Caloosahatchee River estuary. The model includes subsurface flow in terms of groundwater and unsaturated zone flow, surface water in terms of overland and canal flow, and a full dynamic coupling of the model components. Furthermore, a fully distributed irrigation module is applied which links irrigated land with irrigation sources in the basin. Meteorological, topographic, soil physical, land use, vegetation, hydrogeological, canal, hydraulic structure and irrigation permit data have been used to construct the model.

While attempting to maintain the major flow processes of the basin, the model has been developed to simulate the fully dynamic flow and exchange comprising:

- Groundwater (3-layer geological and computational model);
- Unsaturated zone (5 characteristic soil columns);
- Evapotranspiration (10 characteristic vegetation cover classes);
- Overland flow (fully distributed controlled by overland water level and slope);
- Rivers and canals (including all major hydraulic structures);
- Irrigation (irrigated areas and the conjunctive use of surface water and groundwater).

The model is calibrated and validated against multiple measured data in the rivers and aquifers. On the whole, the model is well-calibrated in terms of water balance, potential heads in the aquifers, river stages, and river flows.

The calibrated model has also been used to extract values of seepage flow and tributary flow to the Caloosahatchee River in order to estimate a total water balance for the entire estuary. Similar analysis could be performed for other areas in the catchment, whilst sub-models could be extracted from the larger model if more detailed studies are needed in some areas. Simulation results from the tidal Caloosahatchee River model can then be used as boundary conditions for local models, typically with a finer grid resolution. Distributed maps of the soil type and land use distribution were made with a 114.3 m (375 ft) grid, so local models could benefit from the more detailed information in these maps. For other parameters new maps may be needed, such as a Digital Elevation Model (DEM) with a finer grid resolution than the 457.2 m (1500 ft) used in this model.

Calibration has been challenging in terms of matching simultaneously the groundwater and surface water at various gages distributed across the entire model area. In some areas the amount of data was adequate: a total of 70 time series of potential head observations in the aquifer water table is absolutely sufficient to calibrate water level in the surficial aquifer. For the deeper aquifers (and shallow aquifer wells in the Cape Coral area) more data would probably have improved calibration of the potential head. For the rivers and canals a total of 18 stage gages (14 of which have calculated flow) were used in the calibration of the river stage and flow. Some of the time series are incomplete with large data gaps and others only have data for one year or less. In some cases data were only available during the verification periods and not the two calibration periods, making calibration almost impossible.

5.2 Keeping the Model Up-to-date

Keeping an integrated groundwater/surface water model up-to-date may be important for its future use. In addition to the data most obviously prone to fluctuation, such as rain, evapotranspiration, and abstraction time series, some of the apparently constant data may also change. Land use and paved areas are parameters that may be considered constant over short periods but which certainly cannot over a longer period, especially in areas where considerable urban development is taking place, such as in southern parts of Florida. Land use and paved area maps can be easily updated when necessary by converting shape files in ArcView to distributed maps in MIKE SHE format.

Other important parameters that may need updating are changes in the rivers and canals. These changes can be accommodated in the MIKE 11 setup by adding or changing canal dimensions, weirs or culverts, for example. While it is relatively straightforward to make changes to the river setup, one must bear in mind that this new setup file would only be applicable until the date of the next change. New setup files, therefore, should only be made when major changes have occurred to a canal or a structure, otherwise the file will be valid for a short period only, making it difficult to perform long term simulations.

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ESTIMATING EVAPOTRANSPIRATION IN URBAN ENVIRONMENTS

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Abstract: Quantifying evapotranspiration (ET) rates in urban environments is paramount for understanding and modeling other hydrologic fluxes such as runoff and recharge. Large impermeable fractions of land in urban areas may only experience evaporation following rainfall events. Pervious fractions are believed to support most of the ET burden. Point measurements of land-cover ET in pervious areas can provide better estimates of the overall urban ET budget. These can be made by examining changes in the total soil moisture above the seasonal low water table or the ET extinction depth. Soil moisture can be determined by summing the average soil moisture content measured at various depths with capacitance sensors on a vertical probe. Graphs of total soil moisture above the water table for a riparian area in west-central Florida display two distinct slopes, a flattened slope during the overnight period and a steeper slope between approximately 9:00 am and 6:00 pm. The overnight slope is believed to correspond to a process removing moisture continually from the soil, such as gravity drainage. The daylight portion of the slope corresponds to ET plus the continuing downward gravity drainage. Daylight ET is the cause of the difference between the two slopes. Different plant communities exhibit measurably different ET rates and can be estimated using this methodology.

Key words: evapotranspiration; groundwater modelling; surface water modelling; recharge assessment; vadose zone; Florida, USA; riparian area; capacitance sensors; integrated hydrologic modelling

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1. INTRODUCTION

Hydrologic modeling has become a useful tool in predicting the effects of stresses on hydrologic systems. Predictions from such models may include storm-water runoff, river stages following droughts or heavy rainfall events, expected groundwater yield from a wellfield and the environmental effects of wellfield pumping, maximum sustainable wellfield pumping, and the fate and transport of water-borne contaminants. In areas where the water table is high enough to interact with the surface water system, integrated surface water and groundwater modeling is particularly useful. Integrated models such as Mike-She (DHI, 2004) and the Integrated Hydrologic Model (Ross *et al*., 2004; Ross *et al*., 2005) are necessary to simulate accurately water-table fluctuations in high water-table environments and to model the complete hydrologic budget.

Of course, models are only as good as the data with which they are constructed, and among the most problematic are those concerning evapotranspiration (ET). After precipitation, ET is the second largest water budget term in hydrologic modeling. Brooks *et al*. (1997) estimated ET for the United States at more than 70 percent of annual precipitation and Sun *et al*. (2002) estimated ET for a coastal flatwoods site in Florida at 85 percent of annual precipitation. Considering the magnitude of the ET flux, the ET budget must be well-constrained for a physically based hydrologic model to provide reliable results. Unfortunately, because ET cannot be measured easily, ET estimates are among the least reliable of the water budget terms and, until recently, have received relatively little attention.

Due to the difficulty in measuring ET directly, a number of researchers have suggested indirect methods of estimation. Maidment (1992) listed several methods of estimating evaporation and evapotranspiration, including Penman-Monteith (Monteith, 1965), using measurements including temperature, humidity, wind speed and plant cover. Shuttleworth *et al*. (1988) introduced eddy current measurements of vertical water vapor flux which is now used by the U.S. Geological Survey in their suite of measurements to estimate ET. Several researchers have used numerical models to simulate and estimate ET (Sun *et al*., 1998; Kustasa and Norman, 1999; Lu *et al*., 2003; Mackay *et al*., 2003; Mo *et al*., 2004). Others have adapted ET estimation to data obtainable from remote sensing (Bastiaanssen *et al*., 1998; Mauser and Schadlich, 1998).

Urban environments present even greater challenges for estimating ET: much of urban areas is impervious, storm-water runoff is collected and removed from the area, a significant amount of evaporation takes place on impervious surfaces immediately following rainfall, tall buildings prevent instrument clusters from collecting data over more than a few hundred meters, and wide-scale or remotely sensed data will ‘see’ primarily impervious surfaces. A better approach is to collect point ET measurements in the different types of land cover present in the urban area. The ET measurements for the different land covers can be area-weighted, averaged and combined with storm evaporation estimates to arrive at a useful urban ET budget.

In this paper a method of obtaining point measurements of ET is introduced. The method is based on the total soil moisture above either a point below the seasonal low water table or the ET extinction depth. By noting the rate of change in total soil moisture, plant transpiration and evaporation from the soil column can be estimated. It is recognized that this method will slightly under-predict total ET as surface evaporation (e.g. dew) and evaporation from intercepted rainfall will be missed.

2. MATERIALS AND METHODS

2.1 Soil Moisture Profiles

An equilibrium soil moisture profile shows the vertical distribution of moisture within the soil column from the water table to the land surface (Fig. 1) .The moisture content of the soil at equilibrium varies from field capacity (θ_{FC}) at land surface to near saturation (θ_S) at the capillary fringe just above the water table. The shaded area to the left of the curve represents the total moisture content of the soil above the water table.

As the soil dries, the upper portion of the moisture profile moves to the left, representing a decrease in total soil moisture. During rainfall events, the moisture content of the soil at land surface increases. The moisture in the upper soil column then propagates downward (Fig. 2), shifting the profile to the right and eventually, with sufficient infiltration, raising the water table.

2.2 Soil Moisture Equipment

The soil moisture equipment described in this paper is manufactured by Sentek Sensor Technologies of Australia (Sentek, 2004). Individual moisture sensors are mounted on an aluminum rail at various intervals. The position of the sensors on the rail determines the depth at which soil moisture measurements are taken (Fig. 3). The rail is inserted into a PVC access tube installed vertically in the soil (Fig. 4). Each sensor measures the average moisture content of the surrounding soil within a sphere of influence of 10 cm vertically and 5-10 cm from the outer wall of the access tube.

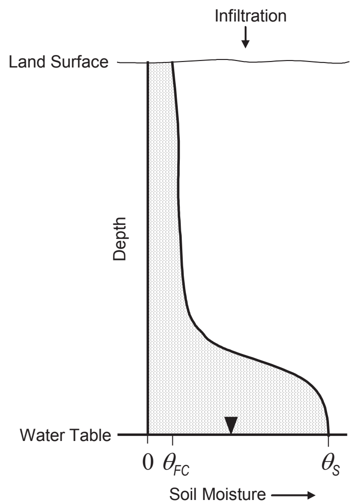


Figure 1. Equilibrium soil moisture profile.

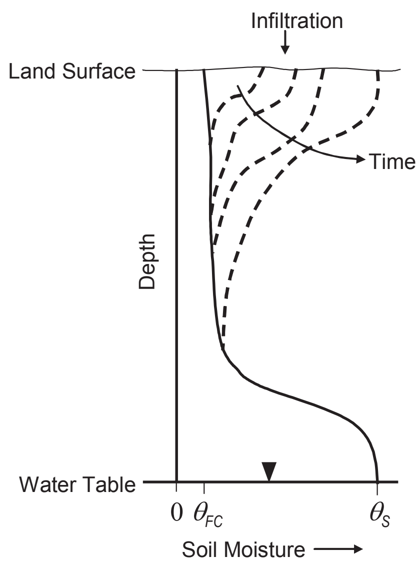


Figure 2. Wetting soil moisture profile.

The individual sensors use capacitance to measure soil moisture. The capacitance of a material is a function of the dielectric constant of that material. Water has a relatively high dielectric constant and, assuming the soil and organic matter remain constant, the capacitance of the soil varies with the water content.

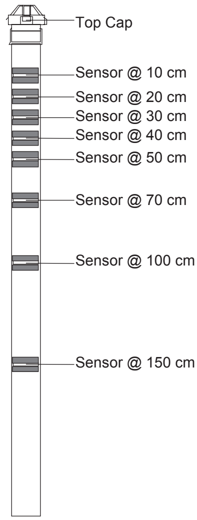


Figure 3. Rail for soil moisture probe with eight sensors.

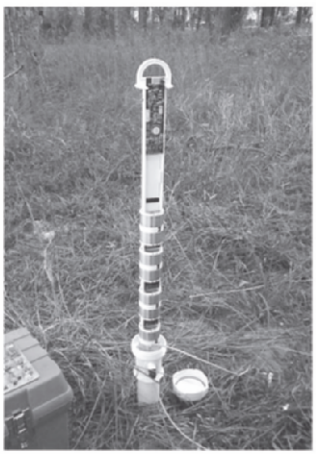


Figure 4. Rail with sensors being inserted into the access tube.

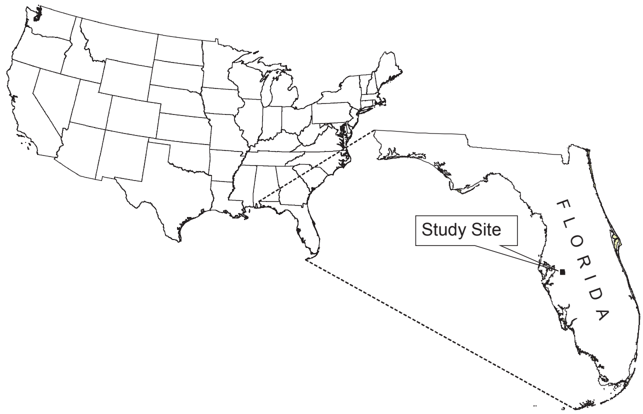


Figure 5. Location of the study site.

2.3 Study Site and Measured Profiles

As part of an in-depth study of surface water and groundwater interaction, soil moisture probes were installed near the city of Tampa in west-central Florida, USA (Fig. 5). The study site is an open grassed area adjacent to a small stream that is surrounded by a riparian forest. The forest is primarily made up of oak trees. A moisture probe and water-table well were installed at two sites within the study area, separated by approximately 200 m. Each probe had sensors mounted such that soil moisture was recorded at depths of 10, 20, 30, 40, 50, 70, 100, and 150 cm below land surface. One moisture probe and well was installed in the upland grassed area (PS-43) and another moisture probe and well was installed near the stream just within the riparian forest (PS-41). Figure 6 shows the locations of the moisture probes within the study area and the stream, which is obscured by the riparian forest, running from the southeast to the northwest.

Figure 7 illustrates two measured soil moisture profiles: one drying and one wetting. The drying profile (Fig. 7a) begins with the solid line at 8 am on 4/24/02. The upper portion of the soil profile moves to the left through 4/26/02 at 8 pm as moisture is lost from the soil column. During these three days, most of the drying took place between 20 and 60 cm below the land surface in the root zone. The water table depth declined from 77 cm to 84 cm below land surface. The wetting profile (Fig. 7b) begins with the solid line at midnight on 2/22/02. A gentle rainfall began shortly after midnight and continued until 8:30 pm on 2/23/02. The total rainfall over the 2-day period was 5 cm. The dashed line, which begins at slightly less than 20% water content, is the moisture profile at 8:00 pm on 2/22/02. Consistent with the theoretical profile (Fig. 2), moisture content increased significantly in the upper 20 cm of soil and increased somewhat in the next 20 cm of soil. Infiltration had not occurred below 40 cm at this time. The dotted line is the moisture profile one day after the rainfall event. The soil moisture in the upper 20 cm of soil is reduced from the previous period but infiltration has proceeded to the water table. The water table rose from 139 cm to 96 cm below land surface during this time period.

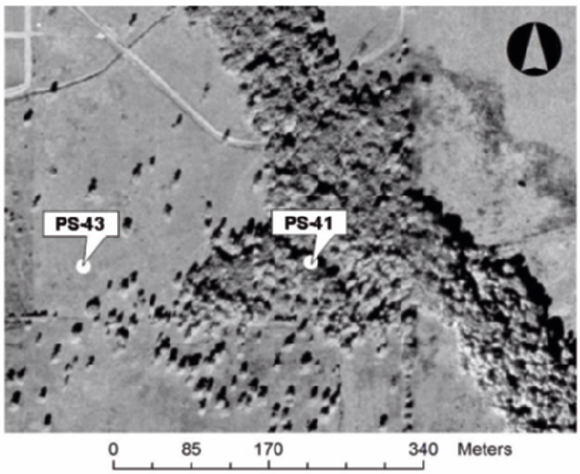
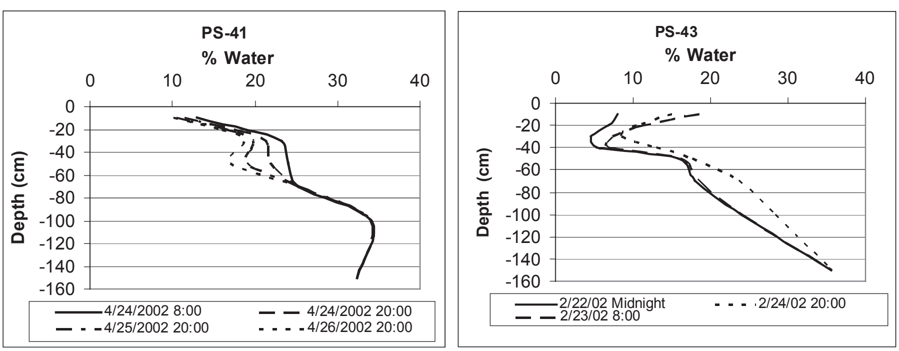


Figure 6. Locations of soil moisture probes at the study site.



a b

Figure 7. Measured drying profile (a) and wetting profile (b).

3. DERIVATION OF EVAPOTRANSPIRATION

By plotting the total moisture content of the soil over time above a fixed datum below the water table, in this case 150 cm below land surface, the response of soil moisture to evapotranspiration and rainfall can be observed. Figure 8 shows the soil moisture variation from the end of March 2002 until July 2002. The dots indicate hourly total soil moisture and the solid gray line the cumulative rainfall. Soil moisture responds quickly to rainfall events, then declines. The loss of soil moisture is due to direct evaporation, plant transpiration and downward gravity drainage.

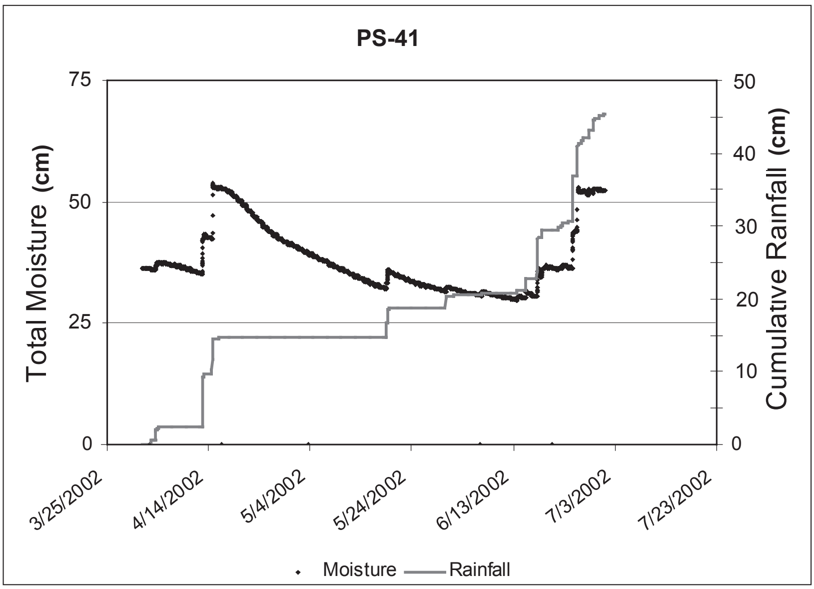


Figure 8. Three-month total soil moisture and cumulative rainfall.

Evaporation and transpiration may be assumed to occur primarily during daylight hours, whereas gravity drainage should occur continuously. By separating the gravity drainage component of soil moisture loss from the total change in soil moisture, evapotranspiration from the soil can be derived. Figure 9 displays the variation of soil moisture on a daily time scale in the forested portion of the study area at PS-41. For this period, a time of high ET losses, there are two distinct slopes that are apparent in the soil moisture plot. The most rapid soil moisture decline occurs between approximately 9:00 am and 6:00 pm, the period when all three components of soil moisture loss are present. During the evening hours, when gravity drainage is the primary factor in soil moisture loss, the decline is much less pronounced. Subtracting the slope of the soil moisture curve during the evening from the slope of the soil moisture curve during the steep daylight decline yields the evapotranspiration rate.

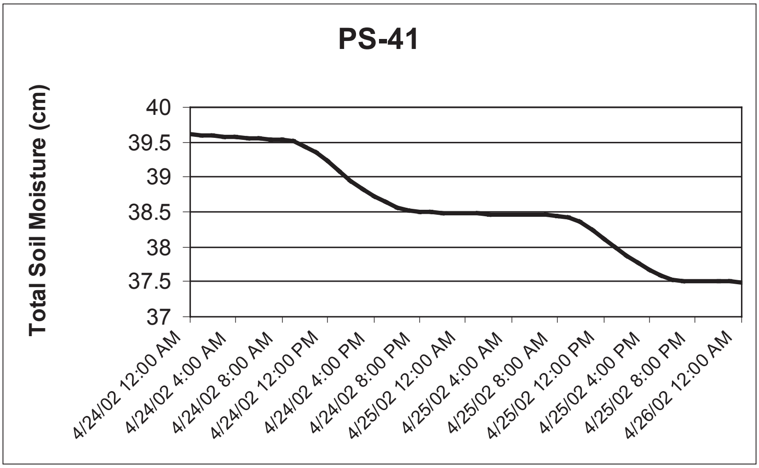


Figure 9. Soil moisture curve for PS-41.

The same processes occur in the grassed area at PS-43. Figure 10 displays the soil moisture curve at PS-43 for the same time period. The curves for both PS-41 and PS-43 are scaled at 3 cm total soil moisture above the datum on the y-axis, but the daylight portions of the curves are substantially different. As one would expect, the evapotranspiration of the grass is significantly less than that of the forest and, although the two probes are located at different elevations, the evening gravity drainage is similar.

The soil moisture curves presented thus far are for April 2002, a period of high plant transpiration in Florida. The soil moisture curves in Fig. 11 are for October 2002, when transpiration is slowing and grasses are approaching dormancy. Both the forested and grassed areas show substantially lower ET rates for the October period than for the April period. In fact, a significant portion of the ET losses from the grassed area may be direct evaporation. These real-time soil moisture measurements can provide estimates of ET rates on the time basis of interest to the modeler, either seasonally, monthly, weekly or even daily.

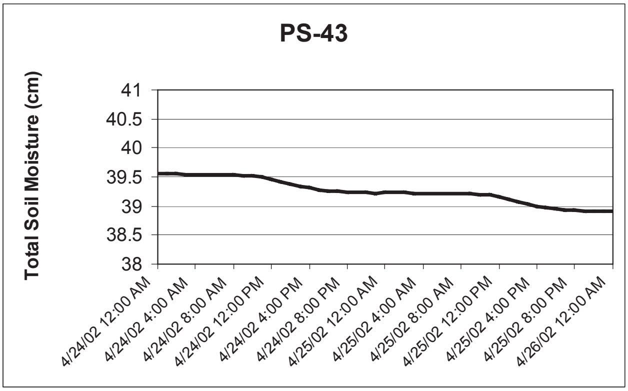


Figure 10. Soil moisture curve for PS-43.

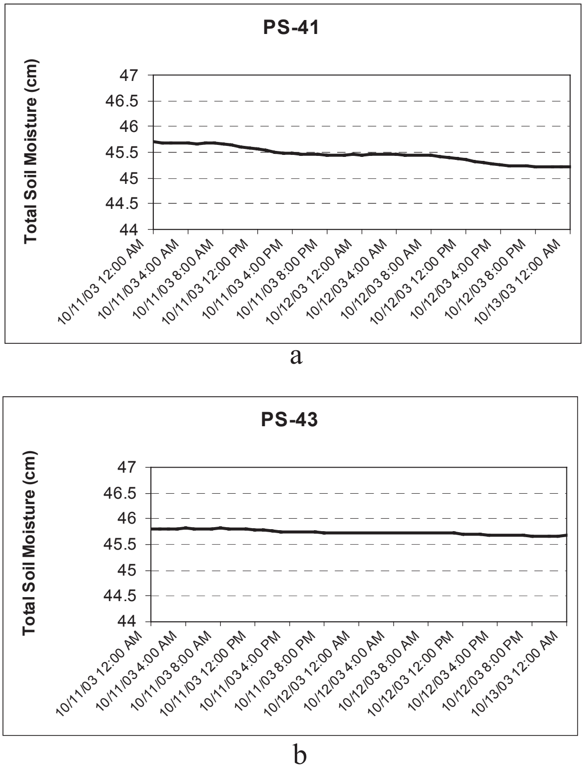


Figure 11. October 2002 soil moisture curves, a) forested area at PS-41, b) grassed area at PS-43.

This method of point measurements for ET rates works well when the water table is below the land surface. However, when the water table is at the land surface or when water is ponding on the ground, the evaporation and transpiration burden may be completely satisfied by the surface water. In this case, water removed from the soil by transpiration is replenished from the surface water and there is no net soil moisture loss. Figure 12 shows the soil moisture curve for PS-41 during the first week of September 2002, toward the end of a wet summer. During this period when transpiration should still be moderately high, there is little change in soil moisture during the daylight hours, and a slight increase in moisture during the evening as surface water infiltrates to offset the daylight losses.

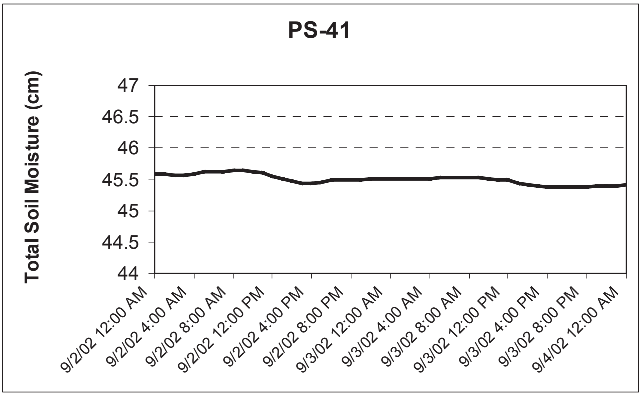


Figure 12. September 2002 soil moisture curve for PS-41.

4. CONCLUSIONS

Evapotranspiration (ET) measurements in urban areas need to be made at the point scale. Large impervious surfaces make ET rates difficult to interpret from remotely sensed data or large-scale measured data. By combining storm evaporation estimates with area-weighted ET rates from the different land covers present, an estimate of the total ET rate in an urban environment can be achieved.

A new method, which uses the change in total soil moisture above the seasonal low water table or above the extinction elevation to derive point measurements of ET, is presented in this paper. Loss of soil moisture is caused by three primary processes: plant transpiration, evaporation, and gravity drainage. The gravity drainage process is always present.

Evaporation and transpiration occur primarily during the daylight hours. Subtracting the rate of change in soil moisture that occurs during the evening, which is primarily gravity drainage, from the rate of change in soil moisture during the daylight hours, when all three processes are present, yields the ET rate. This method does not, however, work well when the water table is at or very near the land surface.

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RELIABILITY IN ESTIMATING URBAN GROUNDWATER RECHARGE THROUGH THE VADOSE ZONE

Managing Sustainable Development in Arid and Semiarid Regions

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Abstract: Reliance on vadose zone models to estimate groundwater recharge in arid and semiarid regions of the United States is increasing due to limited groundwater supplies and continued urbanization. The success of vadose zone models in providing reliable estimates of urban recharge and other fluxes depends on the information content used to constrain the calibration process. In this study, a numerical experiment of artificial recharge through a three-layered vadose zone system revealed several findings related to coupled model calibration. First, the extension of vadose zone model calibration to three dependent variables added information content that enhanced parameter sensitivities. Second, predictive analysis using the calibration-constrained Monte Carlo approach was time-prohibitive because of the tendency toward local minima when using the gradient algorithm. Third, despite a perfect match to historical data, the affect of alternative starting calibration parameters sets on condition number illustrated the limitations of information quality on model uncertainty. Fourth, perfect observed / simulated profiles correlation coefficients for all calibration parameter sets, were, by themselves, poor indicators of model success. Fifth, the range of predictive recharge uncertainty, and uncertainty in energy and solute mass entering the groundwater system, estimated using the likelihood-type approach, is attributed to parameter non-uniqueness due to limited calibration information. For a calibration of a field system, the estimated range of predictive uncertainty would be larger because of additional uncertainty from errors in measurements and/or the conceptual model.

Key words: recharge; vadose zone; parameter non-zuniqueness; uncertainty; modelling; Monte Carlo modelling; heat fluxes; solute fluxes

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1. INTRODUCTION

Since 1900, urban growth has increased globally some fifteen-fold, with fifty percent of the world's population now living in urban areas. In the United States, urbanization typically occurs at rates that are similar to groundwater withdrawals. The concomitant increase in ground-water withdrawals has resulted in regional water level declines exceeding several decameters in many places (Leake, 1999). In these urbanized areas, regional delines in hydraulic head were often associated with groundwater mining that resulted in subsidence, and affected many ecosystems through reduced baseflow, particularly in arid and semiarid regions of the United States.

In arid and semiarid regions of the United States, nearly all surface water is appropriated, and rapid growth continues to increase the demand for already diminishing groundwater supplies (Leake, 1999). It is hoped presently that augmentation of natural ground-water recharge by artificial recharge, using treated wastewater and storm-water runoff, will play a key role in achieving urban groundwater sustainability in arid and semiarid regions of the United States. In these regions, groundwater recharge (hereafter referred to as recharge) is temporally focused at high and low elevations. For example, natural recharge to groundwater at high elevations typically occurs through alluvial fans. At lower elevations, ephemeral streams are the primary source of natural recharge to groundwater (Burkham, 1970). The ability of alluvial fans and ephemeral streams to permit appreciable natural and artificial recharge to the groundwater system is attributed to localized, high-intensity precipitation that falls on highly permeable, stratified, coarse soils (Hendrickx *et al*., 1991).

To manage water resources effectively in arid and semiarid regions, the various water budget components, such as recharge, must be quantified accurately. Unfortunately, the intermittent nature of stream flow and variably saturated subsurface (vadose zone) conditions in arid and semiarid regions make it difficult to quantify recharge using hydrograph separation techniques and saturated groundwater flow models (Constantz, 1996). Other techniques for examining the vadose zone have been investigated, such as zero flux plane, isotope and solute profiles, and groundwater level fluctuation, but assumptions, time, and costs limit their usefulness (American Petroleum Institute, 1997). For these reasons, coupled flow/solute/heat vadose zone models are being investigated by hydrologists for their applicability in estimating groundwater recharge and related fluxes in arid and semiarid regions (Silliman *et al*., 1995; Constantz, 1996; Bailey, 2001; Le Cain, 2002; Nimmo, *et al*., 2002; Stamos *et al*., 2002). However, the ultimate success of these coupled vadose zone models in providing reliable estimates of groundwater recharge and other fluxes depends largely on the adequacy of the model calibration process.

Because the simulated vadose zone transport response depends on many model parameters that are not known *a priori*, an objective mathematical approach, such as nonlinear regression, or genetic algorithm, is often used to estimate optimal (in the least squares sense) model parameter values. This is calibrated by comparing the simulation results with a measured sequence of dependent variables. A related issue to this, and one that is not often reported in the literature, is quantifying the uncertainty associated with recharge predictions made by a coupled vadose zone model once it has been calibrated (Friedel, 2002). The fact that most sets of calibrated vadose zone field model parameter values are non-unique, despite their optimality (Friedel, 2002; Pang, 2003; Friedel, 2005), suggests that recharge predictions made by a calibrated field model may also be non-unique and therefore uncertain. Because accurate recharge predictions in urban settings are fundamental to resource management and economic development, quantifying the uncertainty associated with those predictions is important.

In this paper, the problem of parameter non-uniqueness during model calibration is investigated with regard to predictive recharge uncertainty in arid and semiarid urban settings. Two predictive methods – calibration-constrained Monte Carlo and likelihood-type estimation – are evaluated for their suitability for quantifying the range of predictive recharge uncertainty associated with model parameterization and the calibration process. A numerical experiment is used also to facilitate exploration of uncertainty associated with parameter non-uniqueness, while avoiding uncertainty due to conceptual model and measurement errors. The coupled flow and transport processes are simulated using the Variably Saturated coupled water-heat-solute Transport simulator (VST2D) (Friedel, 2000). Model calibration is achieved by applying a nonlinear-parameter estimation gradient-based algorithm (PEST) (Doherty, 2000) to the VST2D model.

2. PARSIMONY, IDENTIFIABILITY, AND VALIDATION

The principle of parsimony implores a hydrologist to seek the simplest model that is consistent with available measurement and other information. In contrast to a simple vadose zone laboratory model (Simunek, 1998), groundwater hydrologists often argue for the development and use of a complex vadose zone field model (Constantz, 1996; Izbicki *et al*., 2000; Pang *et al*., 2000; Bailey, 2001; Le Cain, 2002). The argument for using a complex vadose zone field model is based often on recognition of the coupled interaction between multiple dependent variables, spatial and temporal heterogeneities, and a desire to perform predictions beyond the calibration conditions. In using a coupled model to simulate a complex vadose zone field system, the number of parameters requiring estimation is greater than that of simple models, as a consequence of the additional coupled governing equations and because attempts are made to represent spatial heterogeneity and/or temporal variation in model structure (parameterization).

Unfortunately, attempts to calibrate more complex field models under steady-state (e.g. Hughson and Yeh, 1998) or transient conditions (e.g. Friedel, 2002, 2005) often result in estimated parameter values with a high degree of uncertainty (poor parameter identifiability). One documented means of resolving the problem of poor parameter identifiability in groundwater (Woodbury and Smith, 1988; Sun and Yeh, 1990; Medina and Carrera, 1996) and vadose zone (Abbaspour *et al*., 1997; Friedel, 2002, 2005) models is to estimate parameter values while calibrating a coupled model against multiple types of coupled measurement information (dependent variables). Composite-scaled sensitivities provide a means to evaluate the relative quantity of information available for estimating parameter values, and the likely importance of each parameter for predictions of interest (Mehl and Hill, 2001). If the information quality of coupled measurements is comparatively poor (i.e. limited in one or more of the following: number, type, space, and time), the calibration process may become numerically intractable. To overcome the potential ill-posedness of a more complex vadose zone model represented by an under-determined system of equations (i.e. over-parameterization), regularization is often applied during the calibration process (Carrera and Neuman, 1986; Friedel, 2002, 2005).

Despite the stability afforded by a regularized inversion, parameter non-uniqueness is often encountered during the model calibration process (*Friedel, 2005*). Parameter non-uniqueness reflects the fact that it is possible to calibrate a model against a set of measurements with optimal yet alternative sets of parameter values. The four primary reasons for parameter non-uniqueness occurring during the model calibration process are as follows: 1) precision of the numerical solution, 2) numerical dispersion, 3) local minima in parameter space, and 4) correlation among parameters. These potential reasons and solutions for avoiding parameter non-uniqueness are explained in more detail below.

In general, if a published code is used during vadose zone model calibration, the precision of a numerical solution will probably not promote parameter non-uniqueness because of the standard validation process (comparing whether simulated dependent variables are the same as analytical and/or other numerical solutions [Friedel, 2000]). By contrast, because the nonlinear inverse algorithm used to estimate the vector of parameter values relies on a sensitivity matrix represented by derivatives of observations with respect to parameter values (Doherty, 2000), the significance of values used in the sensitivity calculation must be sufficient to avoid round-off errors. Round-off in sensitivity calculations is typically a concern when an independent parameter estimation algorithm is applied to the vadose zone transport model. One way to test the significance of sensitivity calculations is by comparing results of the calibration process using a synthetic model (Friedel, 2002). Assuming that the precision of the numerical model is satisfactory, the next factor of concern in promoting parameter non-uniqueness is numerical dispersion. Numerical dispersion can contribute to parameter non-uniqueness when the velocity of mass and/or energy are of sufficient magnitude to prevent accurate calculations at a given time step (Friedel, 2002). In general, the investigator can avoid numerical dispersion by ensuring vadose zone model calculations incorporate either an adaptive time step or an adaptive grid (Friedel, 2000). In many models containing both intrinsic and extrinsic nonlinearities, such as vadose zone transport, the calibration process terminates after becoming trapped in a local rather than global minimum (see below). Termination of the calibration process in local minima is particularly problematic for nonlinear parameter estimation algorithms based on a gradient type approach. One test ensuring that the inverse algorithm located a global minimum is to evaluate objective function (typically a least squares fit criterion) values following subsequent calibrations using alternative trial starting parameter values (Friedel, 2002).

To avoid parameter non-uniqueness due to numerical precision, numerical dispersion, or local minima, the solutions discussed above are implemented in this study. However, the most difficult and elusive factor affecting parameter non-uniqueness during model calibration is correlation between multiple parameters. Parameter correlation contributes to parameter non-uniqueness because two or more parameters can be varied in such a way as to have virtually no effect on the calibration objective function (Friedel, 2002, 2005). The degree of correlation between parameters is dependent on the information content used to constrain the solution (Friedel, 2005).

3. MODEL QUALITY, VALIDATION AND PREDICTIVE UNCERTAINTY

The quality of a calibrated vadose zone model is often evaluated using multiple types of indicators that include the objective function, mean of weighted residuals (bias), standard error of weighted residuals, and correlation coefficient (Friedel, 2002). Because the least-squares objective function essentially represents the model error variance, significant reductions in the objective function are analogous to reductions in overall model uncertainty. Whereas comparatively large values of the final objective function often indicate poor model construction, insensitive parameters, or poor-quality information, the final objective function values are not used in formal comparisons between models unless the number of parameters between models stays the same.

Following assessment of model calibration quality, the conventional approach is to perform model validation. Model validation is a process commonly used to evaluate the accuracy of calibrated model predictions. Model validation differs from model calibration in that the parameter values are not adjusted and performance is evaluated against a different data set than that used to calibrate the model. The most common model validation approach used is the so-called split-sample approach. In this approach, a portion of the measured data are withheld from the calibration process in favor of comparing these data to simulated responses that arise when using the calibrated model. Despite favorable validation results, the post-audit analysis of many field predictions using validated groundwater models demonstrate mixed results -- some are relatively poor while others are reasonably good (Anderson and Woesner, 1992). These mixed post-audit analyses point to the presence of model uncertainty due to unaccounted system stresses, problems in model conceptualization and parameterization, and parameter non-uniqueness. Collectively, these unaccounted sources of model uncertainty underscore the inherent weakness of model validation based on the split-sample approach. For these reasons, quantifying the uncertainty associated with predictions made by the calibrated vadose zone model is a stronger test of the model's predictive ability.

Composite-scaled sensitivities provide a means of evaluating the relative amount of information available to estimate parameter values and the likely importance of each parameter in predictions of interest (Mehl and Hill, 2001). In general, the smallest composite-scaled sensitivity values tend to share the predominance of elements (correlation) associated with the largest eigenvalues. For this reason, the ratio of largest to smallest eigenvalues (condition number) provides a relative and useful measure of total model uncertainty (Friedel, 2002, 2005). In calibrating a coupled model against multiple types of measurement information, the intrinsic coupling between multiple equations facilitates crossover of information between dependent variables that oftentimes enhances parameter sensitivities, enhances estimation of parameter values, and reduces parameter and model uncertainty (Friedel, 2005).

In this study, calculations were performed to represent the predictive uncertainty in using the calibrated vadose zone model. This was determined using two different methods: the calibration-constrained Monte Carlo method and likelihood-type estimation. The calibration-constrained Monte Carlo predictive method is an inverse approach that focuses on determination of uncertainty associated with parameter non-uniqueness, as opposed to the conventional forward Monte Carlo assessment of parameter uncertainty on predictions. To ensure that the maximum range of predictive uncertainty is explored, alternative starting calibration parameter sets are derived by random sampling of uniform parameter distributions bounded by physical- or knowledge-based constraints. Values from individual parameter sets then are used to initialize the nonlinear calibration process. Providing that a statistically significant number of final calibrated parameter sets can be estimated, the range of recharge uncertainty and probability of recharge occurrence can then be evaluated. One alternative to the calibration-constrained Monte Carlo predictive uncertainty is the likelihood-type estimation approach.

The likelihood-type approach is used to estimate the uncertainty limits associated with a key prediction. This type of predictive analysis, first described by Vecchia and Cooley (1987), is incorporated into PEST-ASP (Doherty, 2001) and used herein. The key prediction evaluated in this study is the cumulative amount (per unit area) of water that recharges the water table at 14 days. Like the nonlinear parameter estimation approach used to calibrate a coupled vadose zone model, predictive analysis is iterative and uses the calibrated (optimal) parameter set of values as an initial starting point that ends at the maximum (or minimum) critical point. In this numerical study, the final parameter sets associated with these critical points reflect the presence of parameter non-uniqueness due to parameter correlation.

4. NUMERICAL EXPERIMENT

The artificial recharge model was constructed to represent recharge basins that are operated in southern California (Munevar and Marino, 1999). The coupled transport of water, heat, and solute is conceptualized to occur over a vertical and variably saturated two-dimensional section. The total depth from the streambed surface to the water table is 14 m and includes three layers: a three meter-thick, highly conductive sand and gravel deposit; a two meter-thick, less conductive, silt deposit; and a nine meter-thick, fine-grained sand deposit with intermediate conductivity. The assignment of mass and energy properties in the artificial recharge model reflects findings from the studies by Fogg *et al*. (1995) and Izbiki *et al*. (2000).

The computational mesh used to represent the geologic framework is uniformly spaced at one meter intervals in both vertical (15 nodes) and horizontal (3 nodes) directions. Boundary conditions applied to the streambed surface (upper boundary) reflect an initially dry stream that receives instantaneously treated wastewater of known stage, temperature, and solute concentration. For this case, a constant hydraulic head (equal to a stream depth of two meters) and surface-water temperature (30 ^{0}C) were applied along the upper boundary together with a convective solute flux (Cauchy) condition, where the stream concentration (1 mol/kg H_2O) is considered to be the ambient concentration. For this case, a constant pressure head (0 m) and groundwater temperature (10 ^{o}C) are applied along the lower boundary together with a convective solute flux condition where the groundwater concentration (0.001 mol/kg H_2O) is considered the ambient concentration. Cauchy boundary conditions are applied for water, heat, and solute along the vertical side boundaries with similar ambient conditions used, and the initial conditions are applied uniformly with depth.

The simulated measurement profiles reflect field measurements that are associated with nests of tensiometers (pressure head), thermocouples (temperature), and suction lysimeters (pore-fluid chemistry) located at 15 depths spaced 1 m apart from the streambed surface to the groundwater table. Simulated sampling at these locations occurs over a 2-week period at the following times (in days): 0.01, 0.02, 0.03, 0.04, 0.05, 0.06, 0.07, 0.08, 0.09, 0.1, 0.2, 0.3, 0.4, 0.5, 0.6, 0.7, 0.8, 0.9, 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, and 14. By simulating 27 measurements at 15 locations, 1215 measurements (405 measurements per state variable) are available for use in calibrating the vadose zone transport model. This number of measurements satisfies the general need to have many more (nominally 10 times) measurements than parameters being estimated (Friedel, 2002). All sampled measurements are at the same locations and nonlinear regression weights were applied to form a single population with a uniform variance (homoscedastic) (Gailey *et al*., 1991).

5. RESULTS AND DISCUSSION

The composite scaled sensitivities computed during calibration of the artificial recharge model indicate that there is sufficient measurement information to estimate 30 of the 60 water, heat, and solute transport parameters (Fig. 1). The crossover effect associated with coupling of the governing equations is evident by the multiple types of measurement information that contribute to the total sensitivity of parameters. The estimated water, heat, and solute transport parameters include: saturated hydraulic conductivity (kxs1, kxs2, kxs3), van Genuchten moisture retention parameters (alpha1, alpha2, alpha3, n1, n2, n3), saturated and residual moisture contents (ts1, ts2, ts3, tr1, tr2, tr3), specific heat capacity of solids (cps1, cps2, cps3), thermal conductivity of silt (lambdas1, lambdas2, lambdas3), sand fraction (xsnd1), silt fraction (xslt2, xslt3), longitudinal dispersivity (al1, al2, al3), distribution parameter (kd1, kd2, kd3), and bulk density (rhob1, rhob2, rhob3). The numbers following the parameter variable indicate the lithologic unit layers; for example, 1 upper, 2 intermediate, and 3 lower layers.

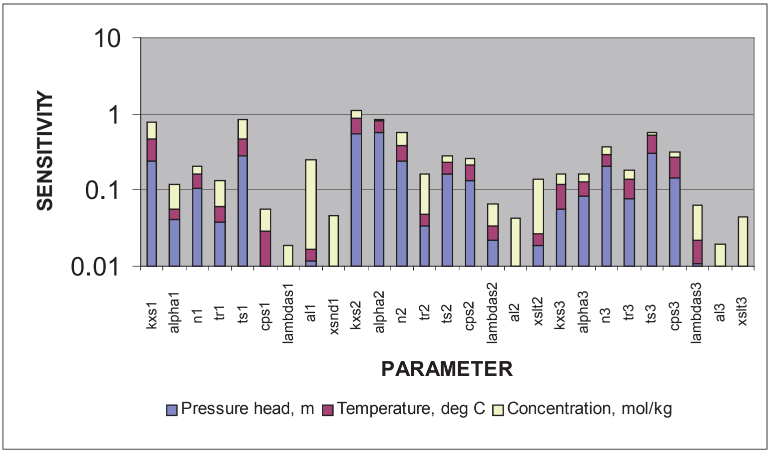

Figure 1. Composite scaled sensitivities plot showing the contributions of measurement information in the calibration of the artificial recharge model parameters.

During application of the calibration-constrained Monte Carlo prediction approach, model calibration using alternative sets of starting parameter values resulted in different final objective function values. This calibration behavior is characteristic of algorithms that find a local rather than global minimum in nonlinear parameter space. It is interesting that to note that even though the parameter estimation algorithm terminated in various local minima, all of the trials resulted in a nearly perfect match (correlation coefficients > .99) to the calibration information (Fig. 2). In contrast to near perfect correlation, the calibrated model parameter sets resulted in variations in model uncertainty as indicated by the variations in condition numbers (Fig 3). For example, the absolute value of the median percent difference from the actual value increased together with the model uncertainty indicated by the condition number. For these reasons, the perfect model correlation describing observed and simulated profiles in this study is by itself a poor indicator of model predictive ability.

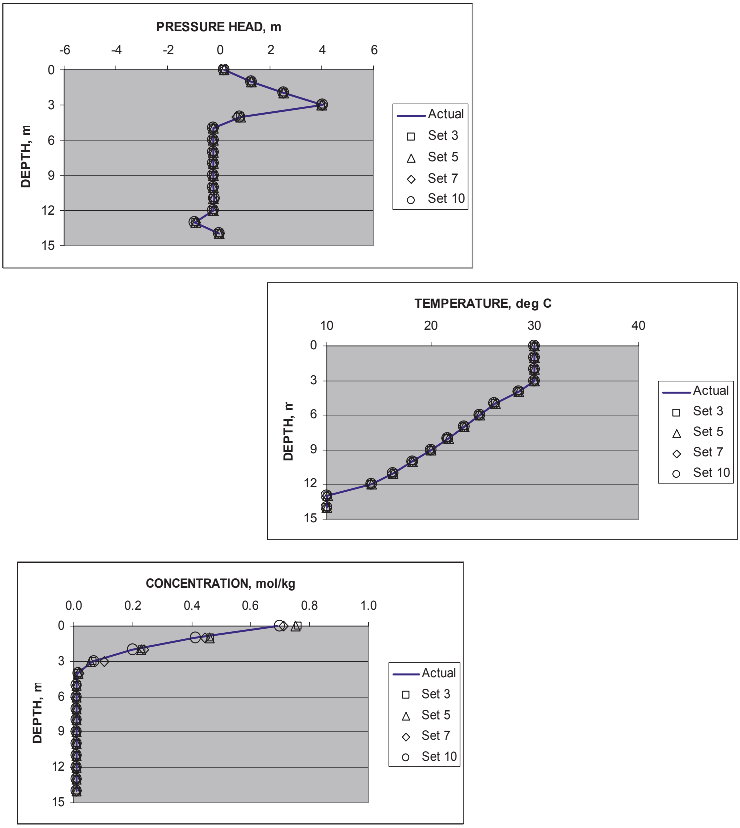

Figure 2. Comparison of actual and calibrated dependent variables (pressure head, temperature, and solute concentration) using alternative sets of starting parameters.

Aside from the fact that the gradient algorithm tended to terminate in local minima while using the calibration-constrained Monte Carlo approach, other issues further limit the usefulness of this approach. For example, it is not clear how many parameter-estimation runs are necessary to compute suitable statistics and(or) threshold probability for the coupled transport problem. A related concern is the actual amount of time required to estimate a single set of parameter values; for example, depending on the starting parameter values, estimation of a single set of model parameter values can take anywhere between 3 hours to 3 days to reach completion using a 1-GHz personal computer. Even if a suitable number of model calibration runs are conducted, the issue of what statistics and distributional shape to use for probability assessment remain a problem.

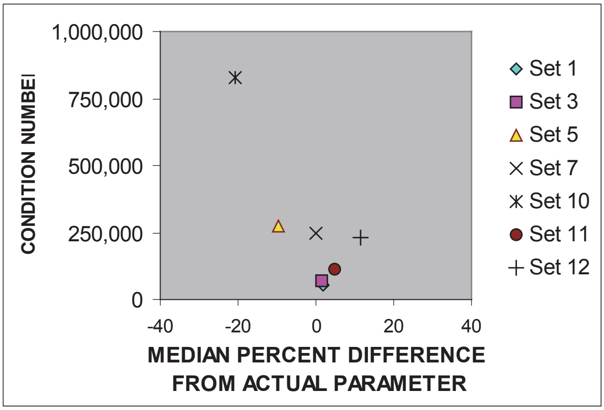

Figure 3. Variations in model uncertainty following termination of model calibration in local minima using selected randomly derived starting parameter sets.

Of the alternative starting parameter sets evaluated, parameter set 3 resulted in the minimum amount of uncertainty and estimated parameter values that were closest to the actual parameter values. In performing model calibration using these starting parameter values, the total measurement objective function decreased from a maximum dimensionless value of 4197 to a minimum dimensionless value of 2.7. Similar decreases occurred in the respective objective function contributions by measurement type. For example, the objective function contributions associated with pressure head, temperature, and solute concentration decreased from their respective maximum dimensionless values of 1282, 1688, and 1226 to minimum dimensionless values of 0.47, 0.96, and 1.3. The respective beginning and final contributions from the regularization group were 1 and 31.7 indicating that the final geologic setting differed from the initial assumption of homogeneity. An indicator of unbiased model quality includes the low mean weighted residual of 10^{-3}. Visual corroboration of the unbiasedness of the calibrated model are provided collectively and by individual dependent variables in Fig. 4.

In addition to the evaluation of general model quality, eigenvector analysis was applied to demonstrate the relation between parameter nonuniqueness, parameter correlation, and model uncertainty under optimal model conditions. For example, the final respective minimum and maximum eigenvalues associated with the calibrated model (starting parameter set number 3) are on the order of 10^{-7} and 10^{-2}. These eigenvalues represent respective reductions from the initial (first iteration) minimum and maximum eigenvalues that are 2 and 4 orders of magnitude, indicating an overall reduction in model uncertainty as a result of the calibration process. Because of the limited number of Monte Carlo trials that converged to a global minimum during the time allowed for study, the actual prediction limits could not be computed.

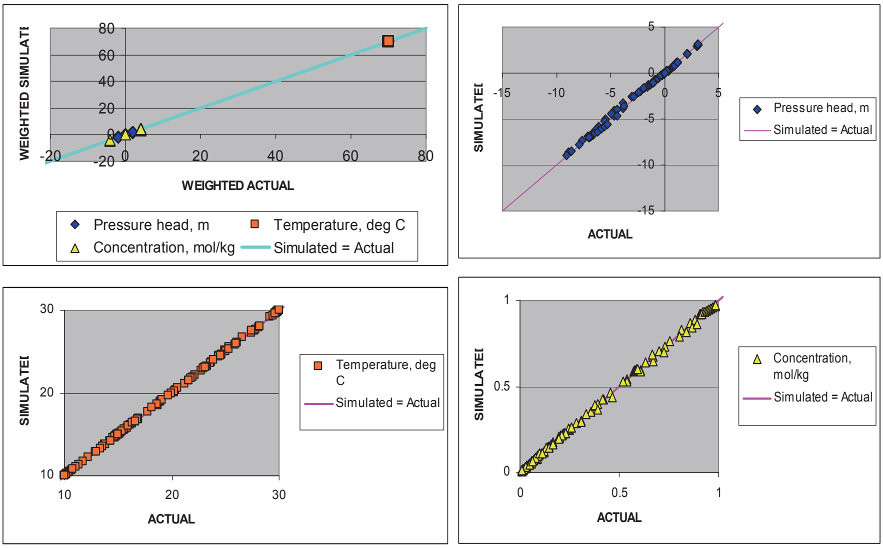

Figure 4. Evaluation of bias for artificial recharge model calibrated using starting parameter set 3.

Table 1. Summary findings for model prediction analysis.

Parameter estimates	Number of measurements	Condition number	Range predictive uncertainty		
		dimensionless	Water, m^3	Solute, mol m^3/kg	Heat, J^8
30	1,215	52,136	1.14	1.15	1.07 $x10^9$

As an alternative to the calibration-constrained Monte Carlo predictive method, the likelihood-type method was used. In using the likelihood method, parameter sets were estimated that maximize (and minimize) the cumulative recharge (per unit area) at 14 days (key prediction) while maintaining the vadose zone model in the calibrated state previously determined. The estimated prediction limits for artificial recharge uncertainty were about 1.14 m^3 (per unit area). In addition to estimating the range uncertainty associated with recharge water, the respective ranges of uncertainty were estimated for temperature and solute concentration entering the groundwater system at 14 days to be 1.07 x10^9 J and 1.15 mol m^3/kg (i.e. concentration x water volume)(Table 1). For this study, the range of predictive uncertainty was due to parameter nonuniqueness because of limited calibration information.

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URBAN WELL-FIELD CAPTURE ZONES DELINEATED USING FLOW STRUCTURE MODELLING

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Abstract: The town of Domodedovo, Russia, is used here as a case study to test a recently developed flow structure modelling method for the delineation of urban well-field capture zones. A three-dimensional capture zone was determined for the well fields used for the town's water supply. Significant differences in the size of the capture zone intersecting the ground surface and the size of the entire sub-surface capture zone projection were found, suggesting that accurate understanding and prediction of contaminant movement within urban systems cannot ignore the third dimension. The modelling procedure used here performed well.

Key words: capture zone; flow structure modelling; Domodedovo, Russia; solute transport modelling; flow fields; contaminant migration

1. INTRODUCTION

Many aspects of groundwater quality assessment and forecasting, and groundwater contamination risk management, require the determination of well capture zones. In principle, ultimate capture zones indicate the possibility of breakthrough to a pumping well from an arbitrary point in space.

2. METHODS

Firstly, a groundwater flow model was constructed using MODFLOW. From this, heads in each layer are known and Darcy's Law makes it

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possible to obtain the integral flow rate through the model's cell faces. The next stage is to reverse the groundwater flow directions. As a result of the reversion, signs of all groundwater flows reverse, and pumping wells become injection wells. Next, water samples from wells under study are labelled, and a mass-transport problem is solved for the new, back-tracked system. The propagation zone of water from an injection well in the reversed model coincides exactly with the capture zone of the same pumping well in the original model.

3. FLOW STRUCTURE MODELLING

The recently developed flow structure modelling method (Kuznetsov and Roshal, 2002) is a good alternative to conventional mass-transport modelling methods that are used for capture zones determination. The flow structure modelling method makes it possible to determine ultimate capture zones directly, and to avoid some of the undesirable computational effects inherent in many mass-transport modelling methods (Kuznetsov and Roshal, 2003).

After reversing the flow directions, the focus of the problem becomes the determination of the spatial area occupied by the labelled sources. A convenient means of identifying liquid from different sources is the use of virtual liquid types. 'Liquid type' is a virtual label, which can be imaged by assigning different colours to liquids from different sources.

For any given source (e.g. a pumping well, or part of an outer inflow boundary), its surface is bounded by a closed contour. The set of all streamlines passing through a closed contour make up the stream surface. The spatial area bounded by a stream surface is called a stream tube and this stream tube contains liquid solely from the source under consideration. In this study, stream tubes are defined as such and hence, each stream tube contained liquid of a single type only.

If all the sources are assigned the same liquid type, then this liquid type will fill the model space completely and continuously, and an-assigned liquid will not exist within the modelled area.

However, each source in the model is normally assigned its own liquid type. As a result, we can obtain stream tubes with total flow rate equal to that of their associated source and containing liquid only from that source. If the spatial distribution of all stream tubes is known, then for an arbitrary point in space the source will be known.

The spatial location of a stream tube can be described in terms of its location as it crosses model cell faces. A set of cross-sections belonging to one face can be described by a cross-section matrix. Formal operations at cross-section matrices were introduced; these operations make it possible to construct flow structure and contents continuously in the whole modelling domain. Thus, for ultimate capture zone delineation, it is sufficient to reverse the groundwater flow solution and assign liquid types for those wells under study that are different from the background liquid type. As a result of the flow structure modelling, the area from which water can reach wells of interest can be determined. The area obtained is an ultimate capture zone.

4. FLOW MODEL DESCRIPTION

The technique described was applied for the delineation of well-field capture zones in the Domodedovo district, Moscow region, Russia. Well-fields in this area are used for the supply of water supply to Domodedovo town and the numerous settlements in the vicinity.

The region under consideration has a complex system of interacting aquifers. Cumulative groundwater recharge, which is the main source of groundwater abstracted within the region, was estimated at 3.95×10^5 m^3/day. Total groundwater withdrawal is about 2.27×10^5 m^3/day. The high ratio of artificial withdrawal to groundwater recharge characterizes the considerable anthropogenic impact on groundwater in the Domodedovo district.

In the model, the aquifer complex was combined into five permeable layers and four low-permeability dividing layers. Two auxiliary layers for recharge and surface-groundwater interaction were used also. The vertical dimension of the model domain is about 200 m. The horizontal discretization comprises 176 columns and 201 rows, with a constant grid spacing of 250 m, and results in a total of almost 2.1×10^5 grid cells in the model. The model working domain covers an area of 1.37×10^3 km^2.

A no-flow boundary is defined at the bottom of the model to represent the impervious layer. For the lateral boundaries, a constant head is mainly specified. Some parts of the boundary pass along streamlines and a no-flow boundary is implemented at these locations. Groundwater recharge is implemented as a type II boundary condition: $Q = W\,\Delta x\,\Delta y = \text{const}$, where W is recharge intensity [$L\,T^{-1}$], and $\Delta x\,\Delta y$ is the area of the model cell. Surface water is implemented with type II and III boundary conditions because of its dependence on groundwater head.

Permeability is heterogeneously distributed in the study area. For the Podolsko-Machkovskiy aquifer, which is the most variable, transmissivity varies from a few tens of m^2/d to 3-4 x 10^3 m^2/day. Regions with maximum transmissivity values are associated with karst zones. The leakage parameters for the dividing low-permeability layers vary in the range 10^{-7} d^{-1} to 10^{-4} d^{-1}.

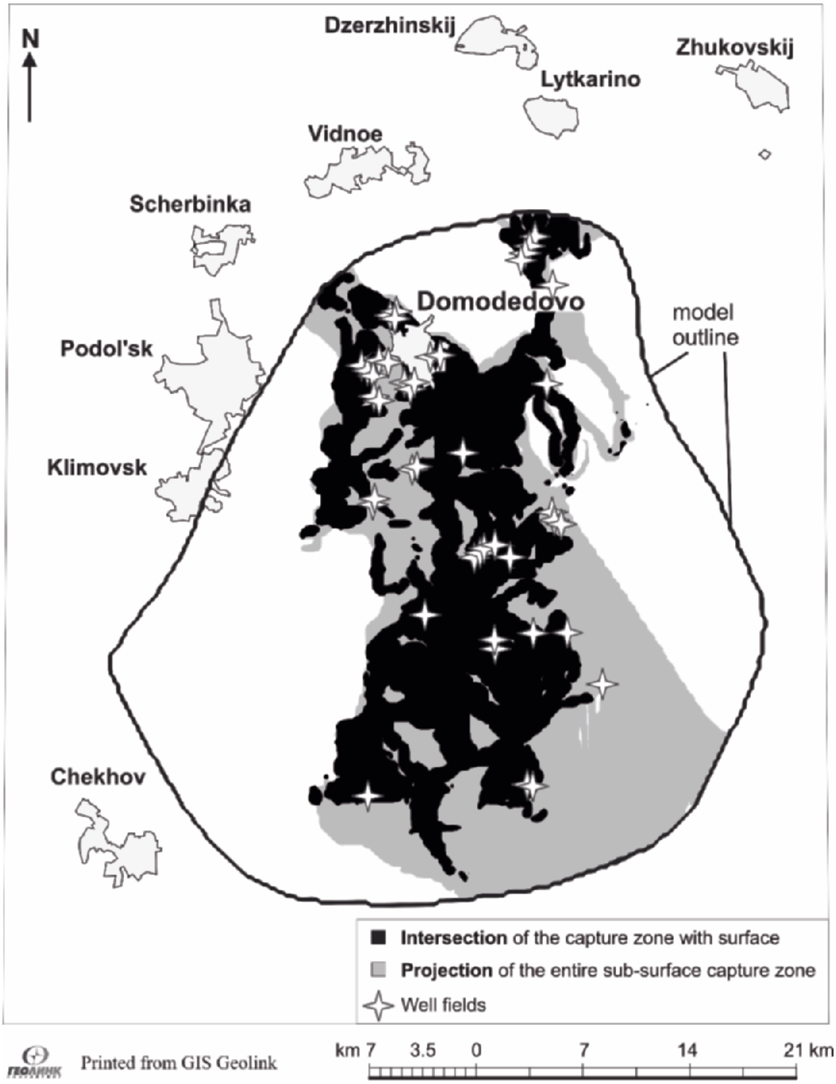

Figure 1. Projection of a three-dimensional well field capture zone to ground surface, and areas where the capture zone emerges at ground surface.

5. MODELLING RESULTS

Flow problems for the described model were solved by the Chebyshev iterative chess algorithm. For the group of wells studied, the same liquid type was assigned. As a result of flow structure modelling, a capture zone common to all studied wells was obtained.

For data processing, groundwater flow problem solving and flow structure modelling, ModTech modelling software was used.

Capture zones in multilayered groundwater systems are three-dimensional in nature. Projection to ground surface of the well-field capture zone is shown in Figure 1 in grey. The intersection of the well-field capture zone with the ground surface is shown in Figure 1 in black. This area coloured black shows the location where, according to the model, surface contamination could reach the supply wells.

It is apparent that there are significant differences in the size of the capture zone intersecting the ground surface and the size of the entire sub-surface distribution when projected to the ground surface. These results indicate the necessity of a three-dimensional zone analysis in the application of groundwater capture zone determination in multilayered systems.

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SECTION IV:

CHEMICAL WATER QUALITY

IMPACTS OF SEWER LEAKAGE ON URBAN GROUNDWATER

Review of a Case Study in Germany

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Abstract: Leaky sewers have to be considered as potential sources for groundwater contamination in urban areas. The medium-sized city of Rastatt in SW-Germany with 50,000 inhabitants was subject of a series of investigations on sewer leakage which are summarized and discussed in this paper. Amongst others factors, the degree of pollution depends on the chemical composition of the wastewater and the amount of exfiltration. The groundwater underneath the city area showed anthropogenic influence resulting in elevated concentrations especially of boron, potassium and sodium as well as a generally increased electrical conductivity. Groundwater in the close vicinity of broken sewers showed typical sewage indicators such as iodated X-ray contrast media and microbiological parameters. Contamination with pharmaceutical residues and gadolinium could not be found in groundwater, despite significant concentrations of pharmaceuticals in wastewater. In autumn, the contents of boron and microbiological pollution were higher than in spring, indicating a higher ratio of wastewater in groundwater during times of less natural recharge. Groundwater monitoring wells near prominent leaks showed short-time fluctuations of EC and groundwater levels which could be correlated to changes of the wastewater composition and the flow regime in the pipe.

Key words: wastewater; sewer leakage; new pollutants; marker substances; exfiltration; Rastatt, Germany; pharmaceuticals; boron; microbiological pollution; iodated X-ray contrast media

1. INTRODUCTION

A defective or insufficient sewer system is a reality for many cities worldwide that may ultimately cause a series of ecological and economic problems. In many cases the reason for sewer failures can be attributed to

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increasing age. According to the German Association for Water, Wastewater and Waste for example, almost 30% of German sewers are more than 50 years old (ATV-DVWK, 2001). Alternatively, leakage may be related to poor materials and negligence during construction. For example a large number of house connections are known to be only improperly connected to the municipal pipes. In terms of water quality, a variety of substances, hazardous or not, are entering the groundwater through exfiltrating sewage with potential to impact drinking water supplies. The volume of sewage entering groundwater can account for an increased urban groundwater recharge and locally lead to elevated groundwater levels.

Defect sewer pipelines are not only subject to wastewater exfiltration, but, if lying below the groundwater table, might also function as drainage systems discharging high volumes of clear groundwater to the waste water treatment plant (WWTP) and keeping groundwater elevations depressed. From the economic point of view, this is increasing the costs of the wastewater treatment plants due to the high amounts of additional water and can even result in the capacity overload of the wastewater treatment systems. In terms of groundwater protection and cost efficiency of wastewater treatment, a good condition of the sewer network including maintenance and rehabilitation should be ranked a high priority. On the other hand the rehabilitation of the pipe network can result in a rise of the groundwater level, often leading to flooded cellars and deep construction works.

For a feasible and sustainable urban water management, the effects of sewage exfiltration have to be carefully weighted and potential risks have to be assessed. This paper reviews the results from studies recently undertaken in Rastatt on environmental impacts of sewer leakage as well as addressing questions concerning emerging novel pollutants that may be related to such leakages.

2. ASSESSING THE RISK OF SEWER LEAKAGE

A condition assessment of sewer pipes is generally accomplished with the help of CCTV inspections. However, these CCTV records and resulting sewer defect databases do not reveal the potential for exfiltration as visually eminent defects do not necessarily account for high exfiltration rates. Furthermore, besides the absolute volume of exfiltrating sewage, the wastewater composition is of great importance concerning a risk assessment for the groundwater. Whereas industrial wastewater often contains a great number of substances hazardous to groundwater such as heavy metals, halogenated hydrocarbons, oils, etc., domestic wastewater may be less critical from its pure composition, but may play an important role because of its large quantities compared to industrial sewage. The largest fraction of the networks is generally domestic containing a high number of defects and therefore resulting in high exfiltration volumes. Finally, private house connections are typically of very poor condition. Critical effects of mixing of groundwater with domestic sewage are organic pollution, increase of biological oxygen demand and the entry of microbial contaminants, nutrients such as nitrate and phosphorous and novel pollutants.

2.1 Determining Factors for Exfiltration

DeSilva *et al*. (2005) studied the methods of quantifying exfiltration and infiltration at pipe level. The extent of sewage exfiltration and therefore the risk for groundwater pollution is ruled by several parameters. The first variable is the area and position of the leak. The exfiltration rate then depends on the depth of wastewater in the pipe. Besides higher pressure on the leak, a high water level in the pipe can affect defects that are not covered by water during low levels. On the pipe walls, but also in the leak area, a clogging layer typically develops from microbiological colmation and unsuspended solids, which leads to a certain self-sealing of the leak. Investigations showed that this colmation layer is not steady but subject to regular changes and can be damaged or destroyed easily during extreme flow conditions e.g. after storm events or hydraulic cleaning of the pipe. Finally, the surrounding geologic material is of great importance. Its permeability and saturation effects the amount of exfiltration, the seepage distance to the groundwater as well as soil characteristics in the pipe environment are important in terms of adsorption or degradation of potential contaminants from sewage.

2.2 Pollutant Potential of Sewage

A vast number of chemical pollutants in groundwater are regulated by various international or federal laws. However, these represent only a small fraction of chemicals that occur in the natural or human environments that can enter the groundwater through urban activities. A variety of these contaminants can be found in sewage, industrial rather than domestic, and leaky sewers therefore have to be considered a possible cause of risk for anthropogenic pollution in urban areas. Although in terms of acute toxicity and source concentration these chemicals constitute the biggest risk to the aquatic environment and if ingested through drinking water to human health, a number of new pollutants have attracted public interest in the last couple of years. However, most of these pollutants are not new to the environment but became recognized due to developments in chemical analysis (Daughton and Ternes, 1999). Counted within such is the search for contaminants that were not considered in previous analytical campaigns, non-target identification and the continuous lowering of detection limits for existing pollutants. To these "old" new pollutants add chemicals which are newly introduced (e.g. new drugs).

2.3 New Pollutants

From the group of so called "new" pollutants, Daughton and Ternes (1999) collectively termed pharmaceuticals and active ingredients in personal care products as PPCP, including not just prescription drugs and biologics, but also diagnostic agents, fragrances, sun-screen agents and others. Leaky sewers have to be considered as one major potential source of new pollutant release. The substances enter wastewater mainly through consumer end use such as receipts and excretion of medicine or personal hygiene (showering, bathing) as well as disposal of unwanted or expired materials into the sewer system. Stiftung Warentest, a German institute for comparative investigations on goods and services, estimated in 2000 the amount of improper disposal of pharmaceuticals in private households in Germany to reach up to 4500 t/a.

One common characteristic of most new pollutants is that they are highly persistent and highly soluble. Many of them are continually introduced to surface waters (through treated sewage in outlets of wastewater treatment plants) and groundwater (through untreated wastewater via leaky sewers). In many areas, they can be found in groundwater and surface water posing a risk for the drinking water supply as the majority is only insufficiently or not degraded by drinking water production (Scheytt *et al.*, 1998). The concentrations, in which these pollutants can be found in groundwater are generally far below the concentrations that affect human bodies, however little to nothing is known about the impact of long-time ingestion in sub-therapeutical doses. An extensive risk assessment of potential impacts of pharmaceutical residues on the environment is not yet possible, as there is too little valid data about ecotoxic effects of pharmaceuticals and their metabolites, nor the toxicity of complex mixtures of several chemical constituents in terms of additive or interactive effects from multiple agents.

In Germany, approximately 3000 different pharmaceutical agents are in use, the annual production ranging from 30,000 to 35,000 t (Heberer, 1999). Among the most important pharmaceutical agents relevant for a potential groundwater risk are antibiotics (penicillin, sulfomethoxazole, erithromycin), anti-epileptics (carbamezepine), antiphlogistics and analgetics (diclofenac, ibuprofene, naproxene, phenazone), antihypertonics (verapamile), psychostimulants (caffein) and lipid regulating agents (clofibrate, bezafibrate, fenofibrate). A study of the Austrian Federal Environmental Agency (Scharf *et al*., 2002) analysed in- and outflows of 12 wastewater treatment plants in Austria and apart from penicillin, detected all above named substances in the sewage. Highest concentrations showed bezafibrate (620-5560 ng/l), naproxene (200-2030 ng/l), ibuprofene (166-3470 ng/l), carbamazepine (212-914 ng/l), diclofenac (<40-1380 ng/l) and caffein (20800-58300 ng/l) (concentrations for WWTP inflow). Scheytt *et al*. (1998) found a series of drugs and drug metabolites at concentrations up to µg/l-level in groundwater samples, among those clofibric acid, the active metabolite of blood lipid regulators clofibrate and etofibrate, phenazone, diclofenac, ibuprofen and fenofibrate.

3. CASE STUDY INVESTIGATIONS

A typical, medium-sized city (Rastatt: 50,000 inhabitants) in the SW of Germany was chosen to investigate anthropogenic influence on groundwater underneath the urban area with focus on the impacts of wastewater exfiltration out of leaky sewers. The main aquifer is a quaternary gravel layer which in the city area has a thickness of approximately 30 m. The aquifer consists of coarse sand and gravel which is generally covered by protective layers of sand, silt and clay less then 1-3 m thickness. Groundwater levels are usually 2-8 m underneath the ground surface. Public drinking water in Rastatt is 100% reliant upon local groundwater whose pumping facilities lie 1-5 km out of the city area. The water is usually fed into the public mains unchlorinated and without any artificial supplements (Eiswirth *et al*., 2003).

First attempts to study sewer leakage in Rastatt have been conducted and summarized by Eiswirth and Hötzl (1997). These involved the construction of a sewer test site in Rastatt-Plittersdorf where the effects of wastewater exfiltration were monitored under the controlled conditions of several floodings of the test sewer. This test site demonstrated the effects of sewage exfiltration on soil-air composition and the fingering of flow paths. In following years an in-situ test site beneath an active part of the sewer network was constructed at Rastatt-Kastanienweg as part of a research project on the risk potential of leaky sewers on soil and groundwater funded by the German Research Foundation (DFG). The in-situ test site Kastanienweg showed changes in the composition of soil water but no major influence on groundwater quality due to clogging effects and the low permeability of the surrounding silty sediments (Forschergruppe Kanalleckage, 2002). An initiative to balance urban water and solute fluxes for the city of Rastatt has been brought forward by Eiswirth (2002) and resulted in the ongoing EU-project AISUWRS (Assessing and Improving the Sustainability of Urban Water Resources and Systems). Attempts to specifically quantify the wastewater exfiltration rates based on sewer condition monitoring on the city scale have been documented in Wolf *et al.* (2006). The next step was to do a city scale sampling programme and to screen for marker substances (Wolf *et al.*, 2004).

3.1 Urban Water Infrastructure

The total length of the pipe network in Rastatt is 442 km, of which stormwater pipelines account for 254 km (Figure 1). Of the 188 km of sewer pipes, separated sewers comprise 47 km, with the remaining 141 km of the network being combined sewers. Using CCTV, a condition assessment of 90 % of the city's sewer network revealed 31,006 defects, the greatest proportion of defects (44 %) being damaged or improperly installed house connections, 24 % joint displacements, 13 % cracks, 5 % obstacles, root intrusions and corrosion problems respectively and a small number of others (Eiswirth *et al.*, 2003). This number might sound very high, but the pipe network of Rastatt is well maintained and in rather good condition compared with other cities in Germany.

As groundwater levels are rather high, parts of the pipe network are lying in the groundwater fluctuation zone or underneath the groundwater table. Where the pipes are underneath the groundwater table, the high number of defects leads to groundwater ingress into the pipes. In all other areas, wastewater is subject of exfiltration with the passage through the unsaturated zone typically being less then 1 or 2 m.

3.2 Groundwater Monitoring

A dense network of groundwater observation wells covers the urban area of Rastatt and its rural surroundings. About 50 wells are used for sampling on urban scale twice a year to document the condition of the urban groundwater and the degree of anthropogenic impacts. For detailed monitoring eight so called focus observation wells have been specifically built in the close vicinity of major sewer defects identified with the help of CCTV inspections. Groundwater monitoring wells showing marked quality deterioration are sampled each month and analysed on specific sewer marker substances. Additionally some of the focus observation wells are equipped with multi-parameter probes for on-line monitoring. Investigations on ground, surface water and wastewater complete the studies in Rastatt.

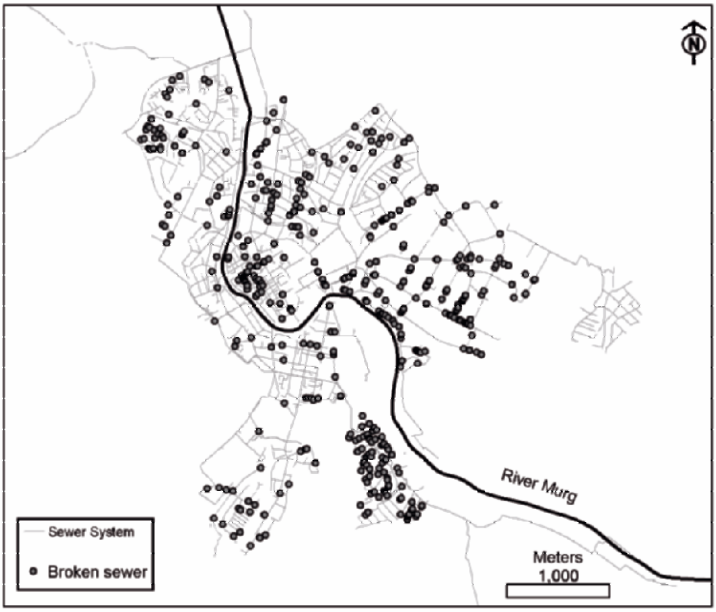

Figure 1. Spatial distribution of major sewer defects in the case study city (after Wolf et al., 2004)

3.2.1 General groundwater condition underneath the urban area

The groundwater underneath the urban area in the case study city features a general influence of human activities showing for example in elevated electrical conductivities (>900 µS/cm). Eiswirth (2002) found elevated concentrations of sodium (maximum 74 mg/l), potassium (maximum 28 mg/l), chloride (maximum 156 mg/l) and sulphate (maximum 511 mg/l).

During area-wide sampling campaigns in 2003, highest concentrations of sodium, potassium, phosphorous and ammonia were monitored in the focus observation wells close to sewer defects whereas oxygen contents were lowest compared to other groundwater monitoring wells elsewhere in the urban area or outside the city (Table 1; Wolf *et al.*, 2004).

The sources for these substances can be multiple, ranging from seepage water from dumping grounds, road salting, excessive use of fertilizers, contaminated lands to sewer leakage. To assess the impact of sewer leakage boron, iodated x-ray contrast media, pharmaceutical residues, rare earth elements and microbiological parameters (*E. coli*, enterococci, coliform bacteria) were analysed as indicators which could be related to wastewater origin with high plausibility.

Table 1. Chemical characteristics of groundwater according to the position of the groundwater monitoring well. (after Wolf et al., 2004).

Group	Number	O_2 median (mg/l)	Na median (mg/l)	K median (mg/l)	NH_4 median (mg/l)	P median (mg/l)	B median (mg/l)
Out of Town	4	4.76	8.90	1.56	0.05	0.05	0.02
City Limits	8	2.43	11.82	2.45	0.07	0.05	0.02
City Area	22	3.34	14.58	4.19	0.05	0.08	0.06
Sewer Focus	7	1.56	16.69	5.70	0.13	0.30	0.07

3.2.2 Sewage marker investigation

3.2.2.1 Boron

As an ingredient of many detergents and cleaners, boron is commonly found in wastewater. Boron concentrations in groundwater monitoring wells in Rastatts city centre show generally higher values compared to monitoring points in the outskirts or in the more rural environment. Outside the city, boron concentrations usually do not exceed 0.03 mg/l, urban concentrations reach up to 0.16 mg/l. Wastewater samples at the inflow of the wastewater treatment plant varied between 0.69 and 2.48 mg/l which is in the typical range for boron concentrations of around 1 mg/l (Mentzner *et al*., 1999). The high boron content in wastewater is reflected in the seepage water, samples from underneath a sewer leak that were around 0.35 mg/l.

In most of the monitoring wells where boron showed elevated concentration compared to the rest of the samples, temporal variations between samples from March and October 2003 could be observed with marked higher concentrations in the autumn samples (43% increase in mean values) (Wolf *et al*., 2004). Considering that this trend cannot be found in the other anions and regarding boron as a typical constituent of wastewater, it can be assumed that the fraction of wastewater in groundwater decreases during the winter months due to high amounts of natural recharge.

3.2.2.2 Pharmaceutical residues

Prior to the drilling of the new focus observation wells, groundwater from monitoring wells which showed anomalies in the groundwater composition or where one could suspect the influence of wastewater as they were located in the vicinity of defect sewers were analysed for 15 pharmaceutical residues. At the same time wastewater samples from the inflow of the wastewater treatment plant and out of one sewer in the city centre were analysed as well as soil-water from suction cups directly positioned underneath an artificial leak in a sewer test site. Despite high concentrations in the wastewater (Table 2) and positive detection in the soil water, none of the substances were encountered in groundwater.

Although there is evidence that some of the pharmaceutical residues are removed to a certain extent during their passage through unsaturated soil (Hua *et al*., 2003), the fact that not a single substance could be found in groundwater rather indicates a dilution with unaffected groundwater below the detection limit.

Table 2. Pharmaceutical residue concentration in wastewater and seepage water.

Substance (ng/l)	Soil Water Suction Cup 7	Wastewater Inflow WWTP	Sewer
Bezafibrate	440	1900	6000
Carbamazepine	42	-	970
Clofibric acid	<20	340	<50
Diazepam	<20	310	120
Diclofenac	260	4100	8400
Etofibrate	<20	<50	<50
Fenofibrate	<20	-	<50
Fenofibric acid	94	740	190
Fenoprofen	<20	<50	<50
Gemfibrozil	81	190	<50
Ibuprofen	120	3800	2100
Indometacin	<20	220	76
Ketoprofen	<20	<50	<50
Naproxene	<20	540	<50
Pentoxifillin	<20	<50	<50

3.2.2.3 Iodated X-ray contrast media

From the new pollutants iodated X-ray contrast media proved to be a very good marker substance as concentrations in urban groundwater were high enough to allow a reliable analysis.

Iodated X-ray contrast media are taken by or injected to patients prior to X-ray for visualisation of soft tissues or organs. By weight, iodated X-ray contrast media are the most often prescribed pharamceuticals in hospitals (Moreau *et al*., 1989). Among other things, this is due to their very high amounts of application, which can reach up to 300 g per X-ray examination (Speck, 1992). The application amounts of iodated X-ray contrast media for Germany (Steger-Hartmann *et al*., 1999) estimated the annual consumption in Germany to be 500 t; Jekel *et al*. (2000) quoted 15 t for Berlin alone) give an idea about the eminent amounts permanently entering our sewage system as excretion makes up almost 100 %.

As the application is carried out in hospitals or medical practices, the sources on the one hand are punctual (hospitals) and on the other hand, non-point via surgeries and out-patients. With only poor elimination in wastewater treatment plants, they enter surface water and finally reach groundwater through bank infiltration or leaky sewer systems. As they are very persistent, they can remain in the environment for long times and have been detected in drinking water in the µg/l range (Jekel & Wischnak, 2000).

Within the urban area, 35 groundwater monitoring wells were tested for iodated X-ray contrast media with eight showing positive evidence. From a suite of nine single substances (amidotrizoic acid, iohexol, iomeprol, iopamidol, iopromid, iothalamic acid, iotrolan, ioxithalamic acid and iodipamid) except one (iodipamid) every substance could be detected in at least one sample.

Table 3. Iodated X-ray contrast media in groundwater samples (ng/l)(*: 285 ng/l the next day) (October 2003).

GW Monitoring well	Amido-trizoic acid	Io-hexol	Iomep-rol	Iopam-idol	Iopro-mid	Iotha-lamic acid	Iotrol-an	Io-xitha-lamic acid	Iodi-pamid
155/211-7	<10	<25	<25	27	<10	<10	<100	<10	<25
90/211-3	31	<25	<25	<10	<10	10	<100	<10	<25
ELF P1	<10	53	<25	25	24	24	<100*	36	<25
Garten str	13	<25	<25	<10	<10	<10	<100	30	<25
GWM3Kast	<10	<25	<25	<10	<10	<10	<100	12	<25
Danz/Moz	20	<25	<25	19	22	16	<100	<10	<25
Ottersdorfer	31	<25	710	18	30	21	<100	33	<25
Zaystrasse	210	<25	<25	<10	<10	<10	<100	<10	<25

Due to prominent fluctuations in the concentrations in wastewater, the entries of these pharmaceuticals are very diffuse. For examples, wastewater concentrations from the inflow of the wastewater treatment plant ranged between 400 ng/l and 21000 ng/l for iomeprol and 120-5900 ng/l for ioxithalamic acid. Some groundwater samples showed specific substances which could not be detected at the same location on another day. Other groundwater monitoring wells showed constant proof for specific parameters but had strong concentration changes in very short intervals (Table 4). The highest concentration (360 ng/l amidotrizoic acid) that could be measured in several sampling campaigns was in a groundwater monitoring well (Zaystrasse) close to the sewer which is collecting the sewage from the local hospital. From the hospital sewage, mainly amidotrizoic acid seems to enter groundwater, as only one other agent could be found once.

Table 4. Temporal concentration changes of iodated X-ray contrast media in several groundwater monitoring wells (ng/l; blank = not detected).

Location	*Day in Oct 2003 (time)*	*Amidotrizoic Acid*	*Iohexol*	*Iomeprol*	*Iopamidol*	*Iopromid*	*Iothalamic Acid*	*Iotrolan*	*Ioxithalamic Acid*
Danz./Moz	*15th (15:40)*								
	16th (15:30)	*20*			*19*	*22*	*16*		
	17th (14:45)								*27*
	18th (15:500								
	27th (16:30)								
Ottersdorfer	*15th (17:55)*			*504*					
	16th (17:30)	*31*		*710*	*18*	*30*	*21*		*33*
	17th (16:30)			*784*					*19*
	18th (15:00)			*580*					*24*
Zaystrasse	*15th (17:35)*								
	16th (17:00)	*210*							
	17th (16:35)	*47*							
	18th(15:00)	*30*							
	27th (15:15)	*77*							*69*
ELF P1	*15th (15:16)*		*53*		*25*	*24*	*24*		*36*
	16th (14:50)							*285*	*20*
	17th (15:20)								
	27th (16:50)								

3.2.2.4 Gadolinium

A second contrast medium which has often been detected in surface and groundwater and which is attributed to sewage pollution is the rare earth element Gadolinium (Möller *et al.*, 2000). It is used as contrast agent for computer tomography. Anomalies in the pattern of rare earth elements with increased gadolinium concentration compared to other rare earth elements in groundwater have been interpreted as sewage influence. However, in Rastatt, from 32 groundwater samples, only three showed a slight anomaly which could not be identified as wastewater pollution but were due to elevated geogenic background. As in Rastatt, tomographic examinations and therefore the application of gadolinium as contrast medium is limited to one single hospital, the content of gadolinium in wastewater is very little and a potential impact on groundwater to small to be detected.

3.2.2.5 Microbiological indicators

Several groundwater observation wells in the urban area revealed coliforms, *E. coli* and enterococci as indicators for faecal contamination

(Paul et al., 2004). Regarding the coliforms and *E. coli* the bacterial counts showed a strong correlation with the distance to the leakage, all three bacteria were found in highest concentrations in the focus observation wells close to major sewer leaks. Similar to boron, the concentrations of germs were higher in October than in March indicating the different ratio between natural recharge and recharge due to sewer leakage. Whereas the horizontal distance from the groundwater monitoring well to the sewer is of great importance for bacterial occurrence the seepage distance between the groundwater table and the bottom of the pipe seems to play a secondary role.

In laboratory tests with homogeneous sand columns constantly percolated with sewage, Hua *et al*. (2003) showed that 99.9 % of the total coliforms and 98.6 % of the faecal coliforms were eliminated within the first 25 cm of the column. Nevertheless a significant number of bacteria still passes the soil columns as the bacteria are so abundant in the wastewater (10^5 – 10^6 CFu/l). It has to be expected, that preferential flow paths and changes in the clogging layer, leading to sudden flushes of exfiltrating wastewater rapidly increase the bacterial transport and prevent the elimination of the bacteria.

3.2.3 Contamination through industrial sewage

In earlier investigations the Environmental Agency of Rastatt found widespread distribution of volatile halogenated hydrocarbons in the case study city groundwater. Two plumes could be detected, mainly containing perchloroethylene (PCE) and trans-1,2-dichloroethylene (trans-DCE).

Table 5. Bacterial counts in groundwater samples according to the location of the groundwater monitoring well (Paul et al., 2004).

Classification	No. of samples	Median (counts/l)	Average (counts/l)
Coliform bacteria			
Reference	5	275	568
Urban background	6	432	103950
Focus well	14	1000	233889
E.coli			
Reference	5	3	14
Urban background	6	30	79
Focus well	14	95	902
Enterococci			
Reference	5	0 (CFU/l)	
Urban background	6	383 (CFU/l)	
Focus well	14	822 (CFU/l)	

As no obvious originator could be identified and the plumes extend along one of the main sewers it was considered that the contamination results from sewer leakage. Furthermore, trans-DCE in groundwater is only stable for a very short time as it is further degraded to vinylchloride so findings of trans-DCE in groundwater suggest a recent or permanent entry.

Wastewater samples from the effected areas showed concentrations of volatile halogenated hydrocarbons at 1.8 - 144.4 µg/l. The main component was trans-DCE with PCE only occurred in minor concentrations (Tropf, 2001). PCE was also detected in air within the sewer (Gutekunst, 2003).

Although several details indicate wastewater as contaminant source, a clear-cut connection remains unclear. Similar to the iodated X-ray contrast media, one of the main difficulties are the huge concentration variations in wastewater. As samples represent only punctual information, it is very hard to correlate concentrations in wastewater and groundwater and to assess the degree of sewage influence on groundwater.

3.2.4 Groundwater near major leaks

Three of the focus observation wells were selected for more detailed investigations. For each of them, wastewater influence could be proven through one or several marker substances. The monitoring wells have been equipped with multi-parameter probes for on-line monitoring of pH, specific electrical conductivity, temperature and water level. The same parameters were monitored in wastewater in the closest sewer together with the wastewater flow. Two of the monitoring wells exhibited elevated specific electrical conductivity (~900-1000 µS/cm compared to 500-600 µS/cm urban background). Besides the generally higher mineralization, diurnal variations of the conductivity were observed (up to 50 µS/cm and >100 µS/cm respectively), reflecting different contaminant loads due to variations of the exfiltration rate or of the sewage composition. No such behaviour could be monitored in pH or temperature.

The amount and the chemical composition of wastewater are subject to constant change depending on the actual production (minimum at nights and during weekends, industrial production during the working hours, etc.). In combined sewers, storm water occurs in irregular intervals on top of this. High fill levels in the pipe as well as thin wastewater with only a small amount of suspended solids favour exfiltration.

The investigations indicated a clear correlation between the composition and the amount of wastewater in the pipe and the electrical conductivity and even the water level of the groundwater. During rainfall events, the amount of water in the pipe multiplied with the storm water heavily reducing the dirt load in the wastewater resulting in an increase of the exfiltration rate. The strongest response in groundwater could be observed near a DN600 sewer with only a couple of decimetres distance between the pipe bottom and the groundwater level. As a result of the enhanced exfiltration, a decline in electrical conductivity could be observed in groundwater (1080 to 1020 µS/cm) together with a slight rise of the groundwater level of half a centimetre a couple of hours after the rainfall event. This indicates that a high rate of exfiltration is not necessarily linked to strong pollution or heavy entry of objectionable substances into the groundwater.

4. CONCLUSIONS

Groundwater investigations have been performed in a typical medium-sized city in Germany to investigate the impacts of wastewater exfiltration from leaky sewers on urban groundwater (Eiswirth & Hötzl, 1997; Eiswirth, 2002; Eiswirth *et al*., 2003; Wolf *et al*., 2004, 2006). The finding of substances exclusively originating from sewage proved wastewater impacts in groundwater in the urban area, especially in proximity to sewers known to have major defects. Among those substances were iodated x-ray contrast media and microbiological indicators. Elevated boron concentrations and contaminations with CHC are very likely to result from sewer leakage but other sources cannot be ruled out to date. Gadolinium was not found in conspicuous concentrations. Pharmaceutical residues could not be detected in groundwater so far despite marked concentrations of several substances in wastewater samples and some detections in soil water. But analyses of the most affected groundwater monitoring wells close to major sewer leaks on pharmaceutical residues are still outstanding.

Boron concentrations and the number of faecal bacteria were higher in early autumn than in spring which could be due to the fact, that the ratio of wastewater in groundwater in early autumn is greater than in spring because of low or missing natural recharge. Focus groundwater observation wells drilled close to known sewer defects showed very elevated concentrations of ammonia, boron, potassium, iodated contrast media as well as elevated electrical conductivity. Beside a generally increased SEC some places showed prominent diurnal variations. Reactions of SEC and groundwater level could be correlated to enhanced sewage spell in the pipe during storm events which resulted in an exfiltration increase. Due to the high fraction of storm water the enhanced exfiltration led to a dilution of groundwater.

The urban aquifer underneath the case study city is showing extensive anthropogenic influence. Wastewater related quality deterioration can mainly be observed in groundwater monitoring wells in the close vicinity to defect sewers. The finding of pharmaceutical residues in wastewater in the case study city in high concentration leads to the assumption, that there is a permanent entry of these substances into the groundwater- to small to be detected yet (although no analyses has been undertaken in the focus observation wells), groundwater flow velocities seem to be high enough to distribute and dilute these contaminants below any detection limits. Nevertheless as there is still not much profound knowledge about the effects of many of the new pollutants and in regard to their persistency, further observations especially in the focus observation wells seem to be prudent.

5. OUTLOOK

Exfiltration is a very unsteady process. The exfiltration rate is subject to changes in the clogging layer, fluctuating fill levels in the pipe and, in combined systems, the diluting influence of storm water. In addition, the composition of the sewage itself is constantly varying. Both effects influence concentrations of sewage markers in groundwater resulting in large uncertainties in mass-balance approaches. To investigate the temporal variations of exfiltration under normal operating conditions, a new test site has been built for qualitative and quantitative measurements. The volume exfiltrating out of two transversal trenches is permanently recorded together with soil moisture in different depths. The results can be associated with flow measurements in the pipe. Suction cups deliver seepage water for chemical analyses, the wastewater composition is monitored with multiparameter probes. Those data together with the data from the focus observation wells will go into numerical modelling designed to calculate the long-term impact to urban aquifers from leaky sewers.

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CONTAMINATION AND DEGRADATION OF DE-ICING CHEMICALS IN THE UNSATURATED AND SATURATED ZONES AT OSLO AIRPORT, GARDERMOEN, NORWAY

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Abstract: Oslo Airport is located on a large unconfined aquifer. It follows strict governmental regulations regarding spills of de-icing chemicals. During winter, de-icing of aeroplanes and runways is necessary for safety reasons: propylene glycol (aeroplanes) and potassium formate (runways) are used. During melts releases to the subsurface may potentially occur. The Pollution Control Authority allows Oslo Airport to use the unsaturated zone for remediation. Biodegradation potential although significant is strongly dependent on retention time. If necessary, pump & treat is implemented. Chemical analyses of the groundwater demonstrate capacity for microbial biodegradation. Locally this sometimes changes the composition of inorganic compounds in the groundwater. A simple box-model was created to estimate oxidation potential in the groundwater. The model indicates that the degradation potential is exceeded west of the western runway, but this is not supported by chemical analyses. Estimates of spatial distribution zones of electron-acceptors and estimates of physical spread of contamination and its residues are not fully supported by the box-model.

Key words: airport; aeroplane; de-icing; biodegradation; formate; propylene glycol; vadose zone; Oslo, Norway.

1. INTRODUCTION

Oslo Airport opened in 1998. It is located 50 km northeast of the Norwegian capital Oslo, on a large glaciofluvial deposit. This deposit forms

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the largest Norwegian groundwater resource in unconsolidated material: the Gardermoen aquifer. At present, the public does not use the groundwater, but strict governmental regulations protect it from being contaminated by airport activities, in order to preserve it for future use. Detection of de-icing chemicals in groundwater outside the airport regulated area results in a $300,000 fine. The groundwater volumetric balance is also regulated because of a landscape vulnerable to gully erosion in groundwater discharge areas to the west of the airport. The official requirement is to maintain the water balance, which makes it impossible to place an impermeable membrane liner in the areas between taxiways and runways to prevent infiltration of water possibly contaminated by de-icing chemicals. By agreement with the Norwegian Pollution Control Authority, Oslo Airport is permitted to use the unsaturated zone as an attenuation zone in case of de-icing chemical release. It was recognized that biodegradation of the chemicals used should be significant and it was expected sufficient for remediation of chemicals and protection of the underlying groundwater.

In the cold Norwegian winter climate, de-icing of airplanes and runways is necessary for safety reasons. The de-icing chemicals used at Oslo Airport are propylene glycol (airplanes) and formate (runways and taxiways). Propylene glycol (PG) has a high chemical oxygen demand (COD), while formate has a relatively low COD. During melting periods, releases to the subsurface of either chemical may potentially occur. This paper reviews results and experiences gained through six winters of airport management regarding spreading and spill of de-icing chemicals, and responses to this in both the unsaturated and saturated zones. The underlying aim is to assess the potential for attenuation and remediation in the unsaturated zone and hence protection of groundwater below.

2. SPREADING OF DE-ICING CHEMICALS

The Gardermoen aquifer is a glaciofluvial deposit. It is the largest unconfined aquifer in Norway, and hence it is important to maintain the groundwater balance and quality, even if it is not presently used as drinking water. The airport is under strict governmental regulations; the area regulated by official requirements is shown in Figure 1. PG and formate in groundwater outside the regulation area is not tolerated but, within the airport area, short time detections of 15 mg PG equivalents per litre are allowed for. Violation of these restrictions results in a $300,000 fine. Propylene glycol (propan-1,2-diol) is soluble in all proportions in water (CCOHS, 2004a) and potassium formate has a solubility in water of 331g/100ml (Kraft and Roseth, 1999; CCOHS, 2004b). The de-icing chemicals may thus be transported easily in melt-water or groundwater. Their structures are given in Figure 2. The oxygen atoms present permit hydrogen bonding with water and hence high solubility (miscibility) in water of these chemicals. Formate is an anion with potassium as the counter cation.

At Oslo Airport three de-icing platforms for de-icing of airplanes are regularly used throughout the winter (Figure 1). During a season, a total of 1000-2000 tons of 100% PG is used in de-icing airplanes. Excess PG from the de-icing platforms is drained to a local treatment plant (treated as sewage) or tanked and later used as a carbon source in nitrification of sewage, dependent on concentration. Some chemicals unavoidably remain on the aircraft, and during take-off this results in the spreading of PG along the runway.

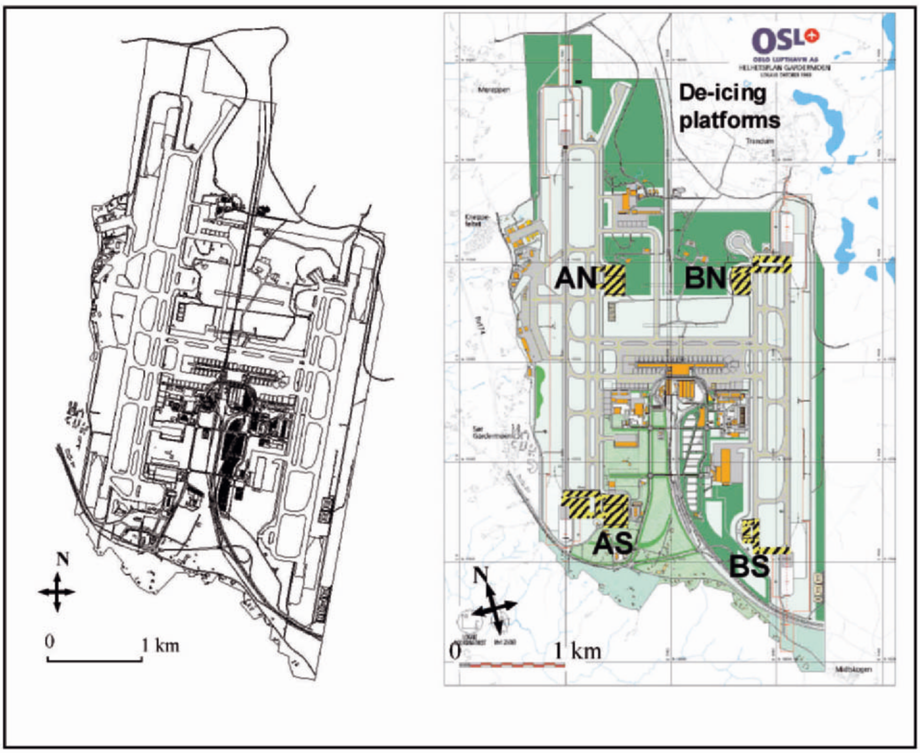

Figure 1. Regulated area at Oslo Airport (left panel) and de-icing platforms (hatched areas, right panel). At the time of this study, platform BS was not being used.

In combination with warmed sand, formate is used to clear snow and ice off runways/taxiways. During a season, approximately 200 tons of 100% formate is used on the runways/taxiways. Clearing of snow is found to result in spreading of de-icing chemicals up to a distance of approximately 30 m from the runway (Kraft and Roseth, 1999).

For most of the de-icing season, the air temperature at Oslo Airport is close to or below zero degrees Celsius, and the ground surface frozen and covered by snow. Instead of immediately infiltrating the unsaturated zone, the de-icing chemicals dispersing from a plain thus mix with snow. During snow melt, melt water contaminated with de-icing chemicals infiltrates in a short period of time (French, van der Zee and Leijnse, 2001).

Snow boxes are used to monitor the spreading of de-icing chemicals during de-icing seasons. As shown in Figure 3, they are placed at different distances laterally from the runway and the start of the runway. The snow boxes contain a mixture of natural snow, snow blast and snow from the clearing of the runway. After a pre-defined period of time, the snow is taken inside to melt, and then analysed for PG and formate.

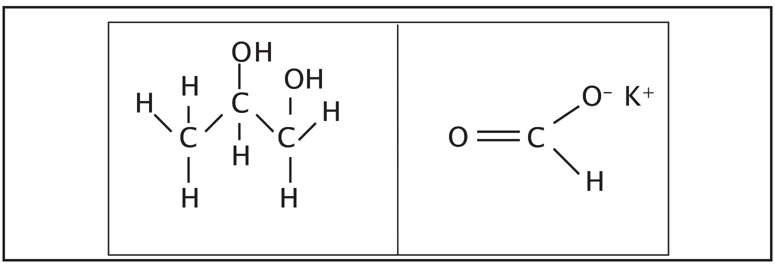

Figure 2. Chemical composition of PG (left panel) and potassium formate (right panel).

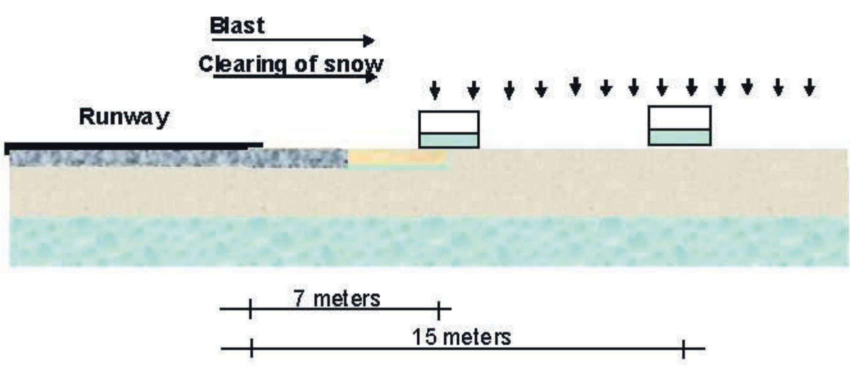

Figure 3. Snow boxes used for sampling of snow for PG and formate analyses.

Figure 4 displays a plot of these data and shows a linear correlation between consumption of PG on de-icing platform AS and surface load (kg COD/m^2) by snow box W600W (600 m north of runway starting point, west of western runway) for the season 2000/2001. The surface load at W600W represents a considerable organic load (1.5 kg COD/m^2), but this is an expected load for a normal winter season in hot-spot areas, the areas of the airport receiving the highest load of chemicals. Through snow-box experiments these areas were found to lie between approximately 400 m and 1000 m from the beginning of the runway. During extreme winters, the accumulated organic surface load may be as high as 3.4 kg COD/m^2.

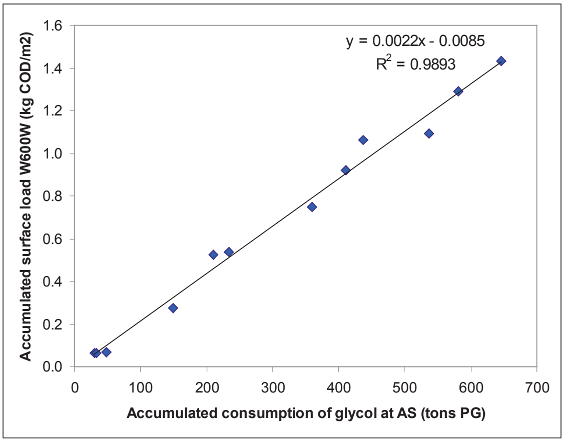

Figure 4. Correlation between glycol consumption at platform A-south and surface load at W600W, winter season 2000/2001.

3. UNSATURATED ZONE

Oslo Airport is located on the groundwater divide (northwest-southeast direction) at Gardermoen. The aquifer recharges through precipitation, and using an impermeable membrane along the runways to avoid infiltration of contaminated water would detrimentally impact the water balance to an undesirable extent and violate regulations.

The Gardermoen deposit is dominated by sandy layers above a clay aquitard (Watn and Solheim, 1993; Tuttle, 1997), the foreset beds having a low clay content (Tuttle, 1997). From boreholes, the unsaturated zone is shown to be thicker in the eastern (10-25m) than the western (5-10m) parts of the deposit. The sediment properties and thickness of the unsaturated zone were judged sufficient to make the area suitable for the natural attenuation approach, principally via biodegradation of water contaminated by de-icing chemicals. In agreement with the Norwegian Pollution Control Authority, Oslo Airport decided to use the unsaturated zone as a natural attenuation remediation zone.

PG has a high chemical oxygen demand (COD), while formate has a relatively low COD. During melting periods, releases to the subsurface of either chemical may potentially occur. Degradation of de-icing chemicals in the unsaturated zone may be described by a first order degradation rate expression (McGahey and Bouwer, 1992; Sabeh and Narasiah, 1992; see equation 1), where pollutant concentration in the soil at time t (*C(t)*) depends on the initial concentration ($C(t_0)$), soil and bacterial properties expressed within the degradation coefficient (*k*) and retention time (*t*) in the unsaturated zone.

$$C(t) = C(t_o)\exp(-kt) \tag{1}$$

The degradation coefficient at Oslo Airport was obtained via local-soil batch experiments and varied from 0.15-0.45 day^{-1} (Roseth and Søvik, 2001). Experiments showed little difference in degradation coefficient between PG and formate (Østeraas, Bakken and Sørheim, 2001). The *k*-value was obtained after mixing the same de-icing chemical with nutrients at least twice. Assuming that this *k* is representative, the degradation occurring in the unsaturated zone and contaminant travel distance is controlled by retention time.

Figure 5 is based on Equation 1 and uses the mean value of $k = 0.3$ day^{-1} for PG and formate. With different initial concentrations of ($C(t_0)$) the graph shows the expected concentration in water reaching the saturated zone (C(t)) as the retention time varies. When no smaller melting periods have occurred before the spring melt, the concentration of PG equivalents accumulated in snow before the melting starts has been observed to be around 1000 mg/l. At the end of extreme winters, prior to the onset of melting, concentrations in snow of some thousands of mg/l have been observed. During a normal winter, a couple of smaller melting periods will occur, lowering the concentration in snow at spring melting. During melting periods, the commonly observed retention time of melt-water in the unsaturated zone is approximately 20 days.

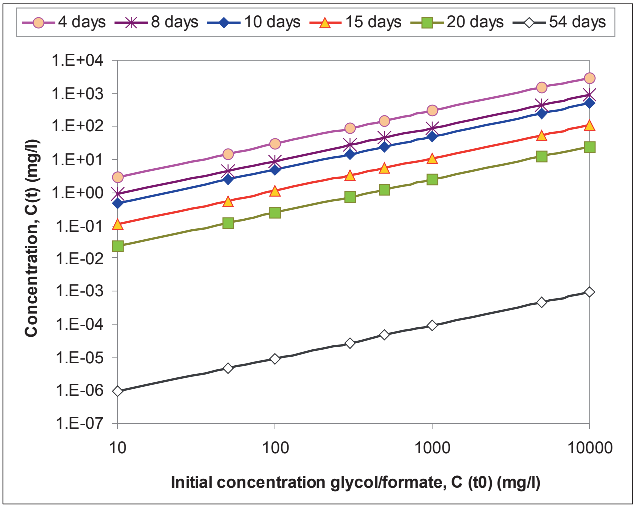

Figure 5. Concentration of de-icing chemicals reaching the saturated zone (C(t)) as a function of PG/formate load to the unsaturated zone ($C(t_0)$) and various retention times. (Degradation coefficient, $k = 0.3\ d^{-1}$.)

Figure 5 shows an increase in retention time in unsaturated zone results in a decrease in C(t). With an initial concentration of 1000 mg/l of PG or formate infiltrating the unsaturated zone, a retention time of ten days gives a C(t) of 49.79 mg/l, while a retention time of twenty days gives C(t) equal to 2.48 mg/l. This is a reduction in C(t) of 95%, while the retention time was increased by 50%.

Local measurements in one area of the airport have shown that more than 90% of the de-icing chemicals that dispersed during take-off are infiltrated through the ground surface. The percentage of water infiltrating the same surface area is observed to be smaller (30-80%) and the difference is believed to be a result of de-icing chemicals melting before the snow. A greater proportion of de-icing chemicals is thus believed to have infiltrated the upper parts of the unsaturated zone as the snow melts.

Snowfall, and its subsequent rearrangement, via runway clearing for example, alters the natural topography in areas between taxiways and runways. During melting periods this has been observed to affect the transport of melted snow, with mildly contaminated water reaching drains designed for storm water run-off. These drains comprise gravel basins leading the water rapidly to the saturated zone. Because de-icing chemicals melt more rapidly than snow, the melt-water has a much lower organic load than would be anticipated. Walls of sand have now been built to prevent contaminated water reaching the storm water drains. When frozen soil thaws, the ice in some zones of the soil melts more easily than in other zones due to differential heating of the subsurface and spatially variable infiltration. As a result of this heterogeneous melting, the melt water can follow preferential flow paths through the unsaturated zone. This increases the risk of low retention time and incomplete degradation in the unsaturated zone, and also the potential for de-icing chemicals to be detected in the groundwater. During one day of spring melting, an extreme infiltration rate of 100 mm/day was observed. According to steady state modelling by Kitterød (2001), retention times in the unsaturated zone vary between four and 126 days for water infiltrating at this rate during 16 days. Experiences from the Gardermoen winter climate shows that it is very unlikely that an infiltration rate of 100 mm/day can be maintained for 16 days.

Using Kitterød's (2001) modelling process, several scenarios were run for the two areas with the highest expected chemical load at the airport (hot-spot areas in the south west, SW, and north east, NE). The thickness of the unsaturated zone was modelled at 4 m in SW and at 11 m in NE. Three different geological sections were used. The scenarios were run with all parameters included, but with one being selected as the most likely, the last parameter varying between a maximum and minimum value. The variable parameter was different in different scenarios. Hydraulic conductivities are given in Table 1. Three different ratios between horizontal and vertical hydraulic conductivity were used, 1:1, 10:1 and 100:1.

Figure 5 shows that a retention time of four days, even with very low initial concentration infiltrating the unsaturated zone, is too little to avoid PG and formate transport to the saturated zone. With a retention time of 54 days, C(t) is expected to be lower than laboratory detection limits.

Table 1. Saturated hydraulic conductivities, K (m/s), used by Kitterød (2001).

Geological unit, unsaturated zone	Mean	Minimum	Maximum
Topset	5.49×10^{-4}	1.64×10^{-4}	1.37×10^{-3}
Foreset bed sand	2.83×10^{-4}	2.66×10^{-5}	1.18×10^{-3}
Foreset bed silt	2.38×10^{-8}	4.46×10^{-9}	7.55×10^{-8}

4. SATURATED ZONE

Micro-organisms can biodegrade organic contaminants in the unsaturated and groundwater zones (Bakken and Swensen, 1998). In order to gain energy, inorganic compounds are utilized as electron acceptors in oxidising the contaminant. Oxygen is preferentially used as its energy yield is greatest. Next, alternative electron acceptors, such as nitrate, iron, manganese and sulphate, are used until finally methanogenesis occurs under the most reducing conditions. Typically, aerobic degradation in the unsaturated zone is higher due to the presence of a ready supply of air to the porous network. In groundwater, maximum dissolved oxygen is around 10 mg/l but is temperature dependent with slightly higher values at low temperatures. During biodegradation in the groundwater, degradation zones develop downstream of the pollution source (Figure 6).

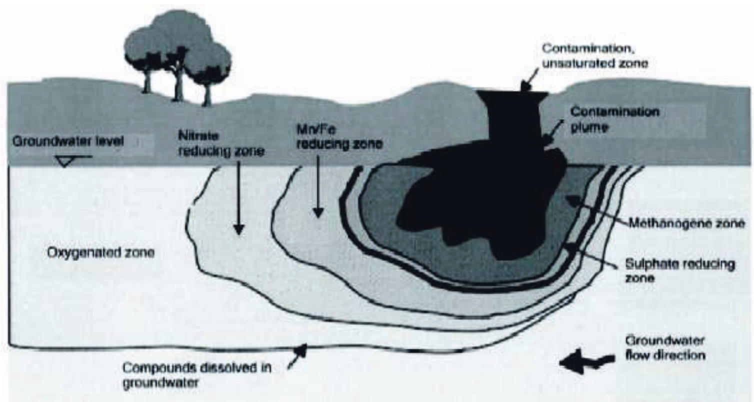

Figure 6. Degradation zones in the groundwater zone, downstream an organic pollution (after Breedveld, 2001).

Oslo Airport has no official requirements to monitor inorganic compounds, but it was decided that inorganic changes would be monitored to assess the potential occurrence of spills reaching the saturated zone. This programme, which utilizes data from 55 wells, also monitors PG, formate and COD/TOC. More wells exist and are available for sampling where required. If PG or formate is detected in the groundwater, the water is removed via extraction wells. It is then either re-infiltrated through the ground surface together with nutrients, or else collected and sent to a local treatment plant, dependent on the season and on the contaminant concentration. The airport's own requirement is that if a spill occurs and changes the composition of inorganic compounds, the concentration of different electron-acceptors should be returned to background levels. No long-term effects on groundwater composition are permitted.

Chemical analyses show that the saturated zone under Oslo Airport has some capacity to degrade de-icing chemicals through microbiological oxidation. Well M8 (Figure 7) is an example of a short term contaminant event and recovery of inorganic compounds, exemplified by manganese (Mn). A spill was observed in M8 early in 1999. This can be seen in Figure 7 as a high concentration of total organic carbon (TOC). Figure 7 shows increased values of Mn in groundwater compared to measured Mn background levels of 0.05 mg/l (Holm, 2000) in the Gardermoen aquifer. This is interpreted as a response to the increased organic load that a spill of de-icing chemicals gives and the processes taking place as a consequence.

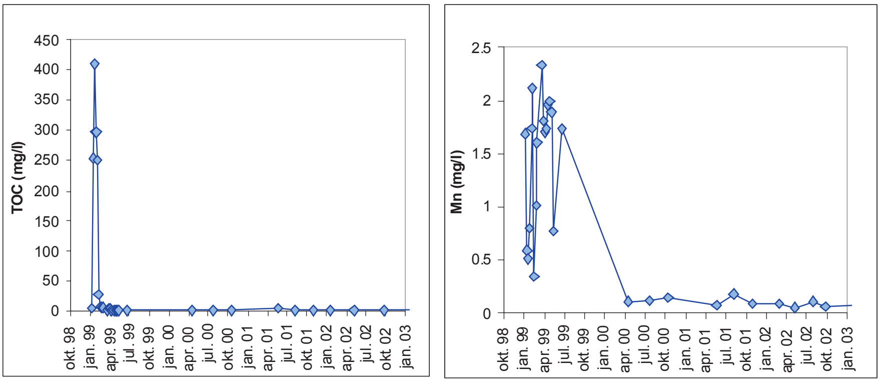

Figure 7. TOC-concentration (left panel) and Mn-concentration (right panel) in well M8.

A wide variety of Mn-reducing organisms exist, including aerobic and anaerobic bacteria, as well as fungi (Gounot, 1994). The reduction of Mn(IV) can thus be a direct result of micro-organisms using Mn(IV) as an electron-acceptor, reducing it to Mn(II) which is more soluble. Mn may also be reduced as a result of micro-organisms producing reducing organic or inorganic compounds, or by a change in physical parameters as pH and Eh (Di-Ruggiero and Gounot, 1990). Such conditions may occur as a result of biodegradation via other electron acceptors, and the observed increase in Mn(II) concentration may thus be an indirect result of reduction of the contaminant.

The background levels of Mn in the Gardermoen aquifer equals the highest concentration allowed in drinking water in Norway, so it is important that any increased concentrations are returned to background levels. A pump and treat programme was started in M8 to reduce the environmental effects and, as a consequence, the TOC-concentration in groundwater returned to background concentrations in March 1999. Mn returned to background levels in May 2000.

A worst-case scenario is pollution of the groundwater to an extent where the inorganic compounds do not return to reference levels. The groundwater capacity may potentially be exceeded and, due to lack of other electron-acceptors, micro-organisms may use CO_2 in the methanogenic process and release methane, an explosive gas. Such a process cannot be permitted to occur beneath an airport.

A simple box-model (Figure 8, after Breedveld, 2001) was created to estimate oxidation potential in the groundwater at Gardermoen, and was based on a mass-balance calculation of compounds and their chemical reactions during a year. The model consists of three boxes: the unsaturated zone, the saturated zone groundwater and the saturated zone solid phase. The box-system is assumed to be in the centre of the spill and each box is defined as a mixed volume.

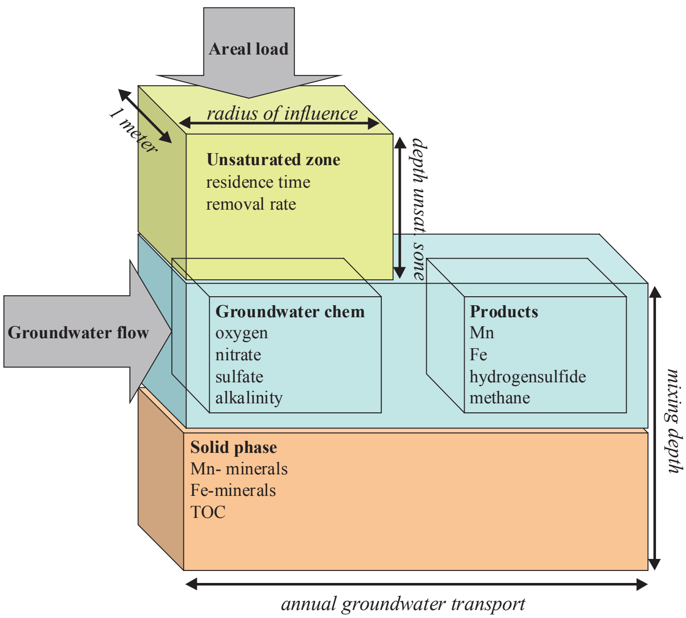

Figure 8. Schematic illustration of the Box-model (after Breedveld, 2001).

The model was used to calculate the annual degrading potential of de-icing chemicals in groundwater at Gardermoen. This was compared to the observed load of de-icing chemicals from snow-box analyses. The western runway is used more than the eastern runway, so the quantity of de-icing chemicals infiltrating the unsaturated zone is higher along the western runway. As can be seen from Table 2, the total annual load of de-icing chemicals to the groundwater in the hot-spot area west of the western runway is higher than the modelled total annual oxidation capacity of the groundwater. East of the eastern runway the chemical load to the groundwater is lower than the modelled oxidation capacity. The modelling results thus indicate that the degradation capacity along western runway is exceeded. However, analyses of the groundwater chemistry show no long term effects on the chemical composition. During a de-icing season though, the chemical composition of groundwater may be locally affected in the short term by small spill events, as shown in Figure 7.

Table 2. Results from the box-model estimations in hot-spot areas west of western runway and east of eastern runway (after Breedveld, 2001).

	West of Western runway	East of eastern runway
Total annual groundwater load (mg PG equiv/l)	66	7
Total annual oxidation capacity (mg PG equiv/l)	39	36
Degradation time (years)	1.7	0.2
Travel distance (m)	211	97

The unsaturated zone is thicker and the groundwater velocity is modelled four times higher along the eastern runway than along the western. The general unpolluted water chemistry is slightly different between the east and west, but the biologically available Fe-oxides are more than twice as high in the east as in the west. These and other input data are based on field observations and measurements taken from groundwater wells along the runways. As shown in Table 2, these differences in input data balance each other and make the modelled annual oxidation capacities east of the eastern runway and west of the western runway approximately equal.

5. CONCLUSIONS

The use of de-icing chemicals at Oslo Airport is a necessity because of the Norwegian winter climate. Dispersion of the de-icing chemicals is found to be strictly dependent on consumption of PG (used to de-ice airplanes) on de-icing platforms, despite the drainage systems in place. Formates used on taxiways and runways are spread by the wind, snow clearance, and blasts from planes. It has been shown that even with strict follow-up of airport activity it is very difficult to completely avoid small short-term spills of de-icing chemicals. Retention time in the unsaturated zone is the most important factor determining whether or not contamination will reach the saturated zone. The occurrence of biodegradation in that zone is critical to groundwater protection. If contamination reaches the groundwater, the field data indicate inorganic electron acceptors are affected due to PG and formate biodegradation. Monitoring to date has shown that the groundwater recovers and concentrations return to background levels, with no long term effects detected.

The box-model created in this study is steady state, so neither estimates of spatial distribution zones of electron-acceptors, nor estimates of the physical spread of contamination and its residues are fully supported by the model. Oslo Airport intends to improve the modelling of degradation of de-icing chemicals in the saturated zone and address the limitations indicated above.

Water soluble organic contaminants may be transported with the same velocity as pure water. If reaching the saturated zone, there is a risk of hydrophilic de-icing chemicals undergoing significant transport due to their low hydrophobicity and sorption. This is countered, however, by their significant biodegradation potential. More studies are required to assess that potential under high de-icing chemical load conditions. Mitigation by pump-and-treat will remain a back-up option in cases where attenuation is deemed insufficient. . It is emphasised that biodegradation of water soluble organic contaminants may still decrease drinking water quality by reducing oxygen content, and increasing the amount of iron, manganese and hydrogen sulphide. This may again increase the costs of water supply because additional purification is required. More studies are also required on the fate of aromatic hydrocarbons from fuels in airports where de-icing chemical load is high as this may influence the biodegradation rates of these hydrocarbons that are frequent risk drivers at urban sites due to their health concerns.

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AROMATIC HYDROCARBON CONTAMINATION OF CLAY STRATA BELOW A PETROCHEMICAL SITE, UK

Organic Contaminant Migration in Clay Aquitards

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Abstract: Clay units have often been assumed to provide good protection for underlying aquifers from contamination. This assumption, however, is rarely proved. The capacity of clays to resist contaminant invasion in urban areas is of significant interest, especially where underlying aquifers are used for supply. We report the initial findings of an on-going field study to investigate organic contaminant penetration of clay strata below a former petrochemical site in the UK. The clay is a thin, relatively continuous unit 6-7.5m below ground surface within a shallow sandy aquifer contaminated with aromatic hydrocarbons (benzene, toluene, ethylbenzene, and styrene). Concentration/ depth profiles were obtained from cores at four locations and demonstrated transport of aromatic hydrocarbons into the clays with significant penetration in places. Preliminary modelling suggests that the clay penetration cannot always be explained by simple diffusion; advective transport through preferential pathways appears probable. Clay thin sections indicate the presence of palaeo-root holes and worm burrows. These potentially explain the enhanced solute penetration, and suggest that the presence of unconformities should be considered when assessing the protection afforded by thin clay beds.

Key words: clay; aquitard; aromatic hydrocarbons; solute transport; UK; preferential pathways

1. INTRODUCTION

Conservative dissolved solute movement in aquifers is controlled by advection and dispersion. Within low permeability deposits such as clays

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and mudstones, the advective component of groundwater flow may be small, and in such cases, depending on the geometry, diffusive transport can become important (*cf.* Johnson *et al.*, 1989). In aquifers, non-conservative solutes will also often be attenuated by sorption and degradation, and in particular by biodegradation. In many argillaceous units, sorption may be considerable because of the large proportion of clay minerals and organic matter often present: biodegradation in the clay matrix, however, may be limited by the inability of bacteria to enter the small pores.

Field and other evidence from various countries suggests that solutes can penetrate clays in some cases (e.g. Parker *et al.*, 2004). Fractures are common in clays, and may form preferential flowpaths. They may be produced during deposition, by tectonic movement, by shrinkage, or by freeze-thaw cycles. Millimetre- to centimetre-scale sand laminae ("stringers") may also provide higher permeability pathways. Contaminant migration through such features is particularly important for virus and bacterial transport, as size exclusion may prevent entry to the matrix. However, lateral diffusion into matrix blocks is often an important attenuation mechanism for solutes which do not have such size limitations, but in this case biodegradation may be limited within these blocks.

Non aqueous phase liquids (NAPLs) may migrate in the subsurface as separate immiscible phases. Given sufficient pressure head, dense NAPLs (DNAPLs) may migrate preferentially through the larger pore aperture pathways thereby penetrating clay horizons much more rapidly than dissolved-phase solutes, perhaps reducing residence time by orders of magnitude. Most hydrocarbons, however, are light non aqueous phase liquids (LNAPLs). Compared with DNAPLs, there is less likelihood of LNAPLs being in a situation to be driven through clay units below the water table, and often for LNAPL compounds solute penetration of clays is the main concern. This LNAPL conceptualization applies to the site described here.

Recent research on clays has focused on penetration of dense organic compounds in both NAPL and solute forms. Feenstra *et al.* (1991) calculated that 1-2 m of trichloroethene (TCE) penetration could occur into unfractured clay in a few decades by diffusion. Kueper and McWhorter (1991) and Parker *et al.* (2004; 1996) found much more rapid migration into clays via sand stringers and fractures. Fractures may only comprise a small volume in clays (0.001-1%) compared with the matrix porosity (30-70%), but form the main flow conduits (Freeze and Cherry, 1979). Rapid movement in fractures enables invasion to a much larger extent than in a non-fractured equivalent (Slough *et al.*, 1999). Relatively low DNAPL heads are required to enter small fractures, e.g. a head of 30 cm is necessary to enter fractures of 17μm aperture (e.g. O'Hara *et al.*, 2000).

This paper presents preliminary on-going research at a hydrocarbon-contaminated site. The aims are: (i) to examine the degree to which contaminant penetration has occurred in a shallow 1-2 m thick clay unit within a permeable sand aquifer; and (ii) to elucidate controlling processes such as the roles of diffusion, preferential advective pathways, and attenuation due to sorption and degradation. The site contains a shallow LNAPL source and the clay stratum studied some way below the water table has only been exposed to dissolved-phase concentrations. Thus migration of dissolved solutes, rather than NAPL, within the clay is the focus of the study.

2. STUDY SITE

2.1 Setting

The precise location of the site cannot be disclosed. However, it is in the UK, and was formerly occupied by a large petrochemical works that is currently undergoing phased remediation. Topographically, the site is flat lying, occupying coastal plains approximately 1 km from the sea at 6-10 m AOD [above Ordnance Datum (~sea level)]. Shallow geological deposits in the study area comprise 0.2 m of made ground underlain by about 6 m of fine to medium windblown sand with occasional peat horizons. Grey clay of 0.5-2 m thickness underlies these deposits. This clay is composed of illite and chlorite with some visually identifiable organic-rich zones. Its exact lateral extent and continuity is unknown, but appears laterally continuous over several hundred metres at least. The geological history suggests that the clay is of lacustrine origin. Discontinuous thin clay lenses (up to 5-10 cm thick) were found above and below this clay bed. The deeper aquifer consists of fine to medium grained marine sand with occasional lenses of silt and clay. It occurs down to about 25 m bgs (below ground surface), below which till and upper Carboniferous mudstones occur.

The site has a shallow water table at 0.7-2m. Groundwater flow is radial away from an apparent recharge mound, central to the study area. Groundwater therefore flows locally away from the site in all directions, though regionally flow is towards the coast. The hydraulic conductivity of the sands is approximately 35 m/d. Site investigation by consultants (Celtic Technologies) has indicated that the head in the overlying sands is typically around 1m higher than that in the sands below the clay bed at ~6 m bgs, suggesting that the clay is a barrier to vertical flow and is relatively continuous. The hydraulic gradient across the clay is around 0.4 (downwards).

2.2 Extent of Contamination

Prior to study, it was known that the site had been subject to contaminant releases from the former hydrocarbon storage tanks and production plant areas. Site investigations by consultants have focused upon the shallow sand aquifer. LNAPL oils floating on the water table were detected in a number of boreholes with residual LNAPL contamination in the first 3 m bgs. More limited water quality data from occasional wells screened below the ~ 6 m bgs clay layer confirmed some low-concentration dissolved-phase contamination by aromatic hydrocarbons in the deeper sand aquifer. The research study described herein focuses upon the processes controlling the transport of the aromatic hydrocarbon solutes across this 1-2 m thick clay horizon to the deeper sand aquifer unit.

3. METHODOLOGY

3.1 Sampling

Cores of 90 mm diameter were collected from four locations using a sonic bore drilling rig (Drill Corp, County Durham, UK). The cores were recovered in plastic sleeves, which were cut open longitudinally immediately before sampling in the field. Soil samples for water extraction were taken every 10 cm and placed in 40 ml vials. Vials were completely filled with aquifer material from the centre of the core and stored in cool boxes before being sent away for commercial analysis. Soil samples for methanol extraction were taken every 5cm using a stainless steel 1cm ID sub-corer. The 5cm long sub-cores were taken through the centre of the core and were placed in 15ml of pre-weighed methanol in 40 ml vials on site. Vials were stored at 4°C in the laboratory before analysis. Additional core material was wrapped in plastic in cool boxes, then stored at 4°C in the laboratory.

3.2 Soil Core Analysis Method

Contaminant concentrations in soil samples from cores 3 and 4 were analyzed using water-extraction analysis whereas cores 1 and 2 were analyzed using methanol extraction. The former quantifies the dissolved aromatic hydrocarbon contamination and probably some of the initially sorbed phase contamination, whereas the latter is a more aggressive extraction methodology and measures the total aromatic hydrocarbon contamination present in the soil sample.

In the case of the methanol extraction method, samples of approximately 10 g (exact mass later determined) were collected at 5-10 cm intervals from the core and placed in 15 ml of pre-weighed methanol with a surrogate compound in 40 ml vials. The sub-corer was washed with Decon®, organic-free water and methanol between samples. Samples were stored at 4˚C for at least two weeks prior to analysis and were daily shaken mechanically to enhance clay dispersion. Before analysis, samples were shaken and then centrifuged at 2500 rpm for 4 minutes. A 100 μl aliquot was taken and added to 15ml of organic-free water in a 20 ml headspace vial. Samples were analyzed by GC-MS in both the regular scanning mode and also selective ion monitoring mode depending upon the concentrations detected.

Analysis of extracts was by headspace-GCMS. Analysis was performed using an Agilent Model 6890 series gas chromatograph (GC) equipped with a Gerstel MPS2 autosampler and a HP 5973 inert mass spectrometer (MS). Chromatographic data were collected and handled by Chemstation software. The headspace autosampler conditions were set as follows: 2.5ml HS syringe; sample volume, 500μl; incubation temperature, 60˚C; incubation time, 15mins; agitation speed, 750rpm. A 30m x 0.32mm ID DB624 column with 1.80μm film thickness (J and W Scientific) was utilized. The carrier gas was helium at a constant flow rate of 3ml/min. The injector was held at 50˚C and a split ratio of 10:1 was used. The column oven temperature programme began at 60˚C for 0.5 minutes, increased at 35˚C/min until a final temperature of 230˚C. The mass spectrometer transfer line was set at 280˚C.

Standard concentrations of benzene, toluene, ethylbenzene and styrene were made by dilution with methanol from pure-phase chemicals. Headspace standard solutions were prepared by adding 100 μl of the working standard into a screw-capped vial already amended with a volume of organic-free water.

Water extraction samples were analyzed by a commercial laboratory (Severn Trent Laboratories (STL)). Samples were collected in completely-filled 40 ml vials. The analysis method is based on the US EPA Methodology 5021 (USEPA, 2004a). Samples of 2.5 g were mixed with 8 ml of deionised water, sodium chloride and sand, aiding the partitioning of VOCs into the gas phase. Samples were also analysed using headspace GC-MS.

3.3 Partitioning Calculations for Water Concentration

Analysis of the soil samples yields a mass of organic contaminant per unit mass of soil. Concentrations predicted to be present in an equilibrated pore water in the soil sample may be back-calculated from the soil sample analysis results via the use of a partitioning equation (Eq. 1) (e.g. Feenstra *et al*., 1996):

$$C_w = \frac{C_t \rho_b}{\left(K_d \rho_b + \phi_w + H_c \phi_a\right)} \tag{1}$$

where C_w is the concentration in the porewater (mg/l), C_t is the total concentration from the solid sample analyses (µg/g dry weight), ρ_b is the dry bulk density (g/cm^3), K_d is the partitioning coefficient (ml/g), ϕ_w is water filled porosity (volume fraction), H_c is the dimensionless Henry's Law Constant, and ϕ_a is the air filled porosity (volume fraction). The measured, estimated, or calculated values for the above parameters are given in Table 1.

Table 1. Estimated parameters used in the calculation of porewater concentrations [[1] USEPA (2004b); [2] Fetter (1999); [3] Parker et al. (2004); [4] calculated; [5] calculated from field data; [6] Morris and Johnson (1967)]

Chemical	H_c [1]	K_{oc} (ml/g) [2]	f_{oc} [3]	K_d (ml/g) [4]	ρ_b (g/ml) [5]	ϕ_w [6]
Benzene	2.28×10^{-1}	66	0.001	0.066	1.62	0.3
Toluene	2.72×10^{-1}	145	0.001	0.145	1.62	0.3
Ethylbenzene	3.23×10^{-1}	207	0.001	0.207	1.62	0.3
Styrene	1.13×10^{-1}	912	0.001	0.912	1.62	0.3

4. RESULTS AND DISCUSSION

4.1 Shallow Aquifer Profiles

Geological log descriptions from the four cores are given in Table 2. Cores 1, 2 and 3 were taken within an area of 20m^2, whilst core 4 was taken from approximately 165 m away. It is unknown whether the main thick clay beds, appearing between 2.39 and 3.95 m AOD, to a thickness of between 0.35 and 0.65 m, are continuous. In cores 1 and 2, additional thin clay lenses occur around the main clay bed.

Table 2. Geological log descriptions for cores taken.

Core	Depth (m bgs)	Log
1	9.85-3.95	Upper profile, fine SAND
	3.95-3.56	Grey clay
	3.56-3.30	Fine sands
	3.3-3.25	Thin, grey clay lens
	3.25-3.00	Fine sands
2	9.84-3.39	Upper profile, fine SAND
	3.39-3.34	Dense, fine sands with lens of clay
	3.34-3.04	Fine sands
	3.04-2.39	Dark grey clay
	2.39-2.34	Fine sands
3	9.65-3.655	Upper profile, medium SAND
	3.655-3.155	Brown-grey, soft to firm, mottled CLAY
	3.155-2.655	Dark brown, coarse-medium, loose, silty SAND
4	9.584-3.684	Upper profile, medium SAND
	3.684-2.584	Dark grey, firm CLAY. Rare black organics. Thinly laminated.

The highest concentrations of benzene were found at the water table which is known to reside between 1 and 2m (Figure 1). The maximum calculated water concentration of benzene in core 3 at this level was 963 mg/l, which was below the solubility limit of 1750 mg/l, and hence it is assumed to be in dissolved form. Bailer samples showed free-phase LNAPL to be floating on the water table; however this was thought to be comprised of ethylbenzene, styrene, and toluene primarily.

4.2 Clay Profiles

Detailed benzene profiles in the clay bed for the four cores are shown in Figure 2. Benzene penetration occurred into all of the clay units to some degree, with maximum penetration occurring in the clay of cores 1, 2, and 3. Cores 3 and 4 have significantly lower contaminant levels than cores 1 and 2; this is ascribed to the different extraction method used (water for cores 3 and 4, methanol for cores 1 and 2). The partitioning calculations do not suggest the presence of free-phase LNAPL since the greatest equivalent porewater concentrations were calculated to be ~ 40 mg/l (in core 2), significantly below benzene solubility. Cores 1 and 3 show that contaminant mass was highest near the sand-clay interface and decreased with depth from this point.

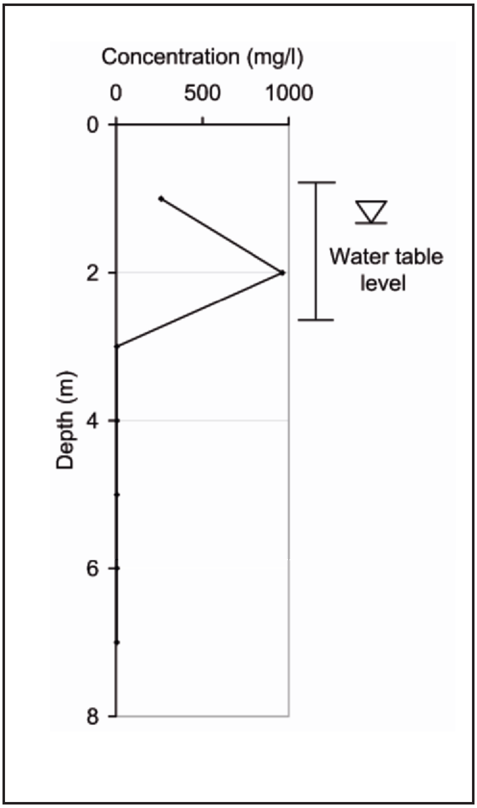

Figure 1. Benzene concentrations in the porewater of core 1 with depth (porewater concentrations based on partitioning calculations on soil concentration data).

Core 1 (Figure 2) shows a profile that is qualitatively consistent with that expected from diffusion-dominated transport, although the clay stratum is thin at this location. Core 2, in contrast, shows higher concentrations extending throughout a much thicker portion of the clay suggesting that advection may be important. Thin clay lenses were also identified in cores 1 (2 cm thick) and 2 (8 cm thick) around the main clay stratum. Benzene concentrations above these clay lenses were found to be higher than those below, suggesting that they may be partial barriers to contaminant migration.

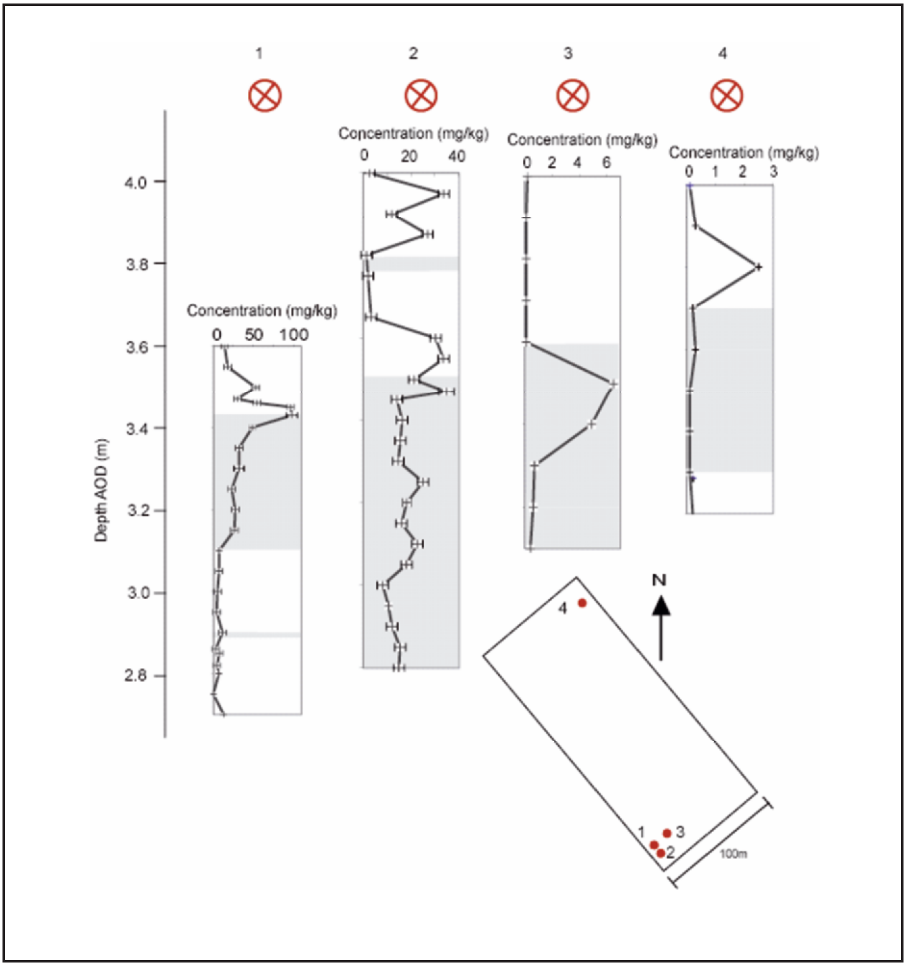

Figure 2. Benzene depth soil-concentration profiles for cores 1-4 (grey bands denote the clay strata).

5. 1-D CONTAMINANT TRANSPORT MODELLING

The movement of non-degrading organic contaminants into the non-fractured clay beds may be expected to be controlled by diffusion and sorption. In more fractured, preferential-pathway clays, advection and dispersion processes would assume greater significance (Parker *et al.*, 2004). A simple 1-D model was used to assess processes controlling contaminant migration into the main clay stratum in core 1 for which the contaminant distributions appeared to be most diffusion-like. Simulated and core data are shown in Figure 3 for benzene.

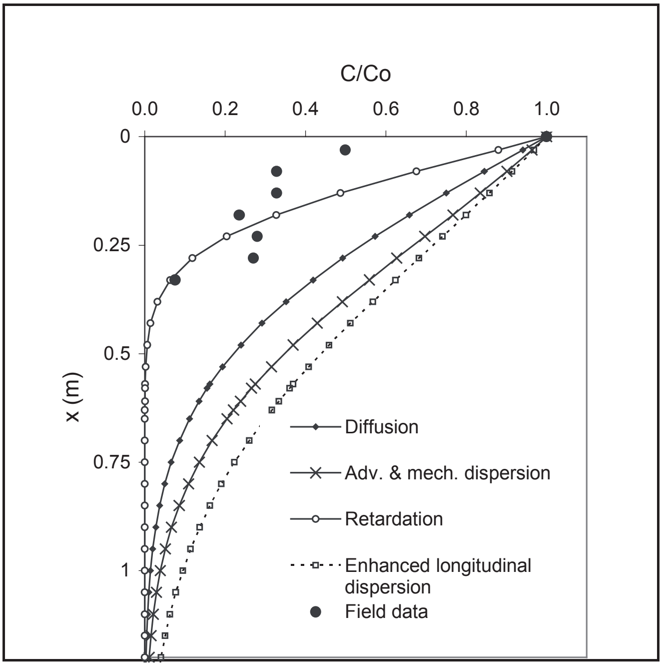

Figure 3. Normalized dissolved-phase concentration profiles for benzene core 1 data and simulated: diffusion-only profile (t = 8.2y, D* = 2.8×10^{-5} m^2/d); advection + dispersion profile (K = 7.1×10^{-6}m/d, i = 2.8, n = 0.4, v = 5.1×10^{-5} m/d, t = 8.2y, α_L = 0.035m, D* = 2.8×10^{-5} m^2/d); advection with retardation and large dispersivity (retardation factor = 6 based on laboratory sorption experiments; α_L = 0.35m).

Figure 3 shows contaminant profiles produced by simulating the following conditions: diffusion only; advection + dispersion; and advection + dispersion + retardation. A further profile was calculated using the dispersion flow solution, but with an increased longitudinal dispersion value to represent in a crude way the presence of preferential pathways. The profiles were calculated assuming 1-D movement into an infinite medium from a constant concentration source, where: for $x = 0$, $t \geq 0$, $C = C_o$; for $0 < x < \infty$, $t = 0$, $C = 0$; $0 < x < \infty$, $t > 0$, $C = C$; and for $x = \infty$, $t \geq 0$, $C = 0$. The time period was estimated from site records; average linear velocity was estimated using Darcy's Law with an hydraulic conductivity (K) of 2.6×10^{-3} m/y, a gradient (i) of 2.857, and a porosity (n) of 0.4; longitudinal dispersivity (α_L) was estimated at 0.035m; and the diffusion coefficient (D*) was estimated as $2.77 \times 10^{-5}$$m^2$/d.

As can be seen from Figure 3, the penetration of benzene into the clay of core 1 can be described by a retarded diffusion profile. The degree to which advection would be expected to occur in the clay matrix and influence penetration is shown to be relatively small, and cannot explain profiles such as that seen in core 2.

6. PREFERENTIAL PATHWAYS IN THE CLAY

Further work is underway to identify possible heterogeneities within the clay which may explain the core 2 where contaminant penetration is greater than can be explained by simple diffusion or matrix advection. Studies of pesticide movement through preferential flow paths by Jørgensen *et al*. (2002) found 96-98% of flow to be along fractures and Broholm *et al.* (1999) showed that the transport of dissolved phase low molecular weight organic compounds through fractured clayey till may occur as rapidly as bromide, a conservative tracer.

Horizontally-oriented thin sections of the clay allowed the identification of possible preferential flowpaths as shown in Figures 4 and 5. These figures include examples of voids surrounded by a layer of black material of possible root wall origin. Also, there is evidence of mineral alteration, both suggesting the presence of root channels formed prior to the deposition of the upper sand. Such features may cause contaminant behaviour changes in terms of both flow velocity and sorption. Organic-rich root-hole linings may increase sorptive retardation of contaminants, much as Turner and Steele (1988) reported for metals in organic-lined fractures. Weathering reactions may also change the characteristics of the clay surrounding the root-holes, for example in terms of the surface area available for sorption.

Evidence of worm activity, in the form of cloud-shaped excreta/bioturbation can also be seen in Figure 5. The properties of such material may vary from that of the bulk clay in respect of sorption, flow, and biodegradation potential. Similarly, intact pieces of organic matter, as also seen in Figure 5, may cause local variation in organic contaminant transport behaviour. The distribution of these features with depth has yet to be established, but it would be expected that they would be more common in the upper parts of the clay. It is also expected that the thinner the clay bed, the greater the chance that the features will fully penetrate.

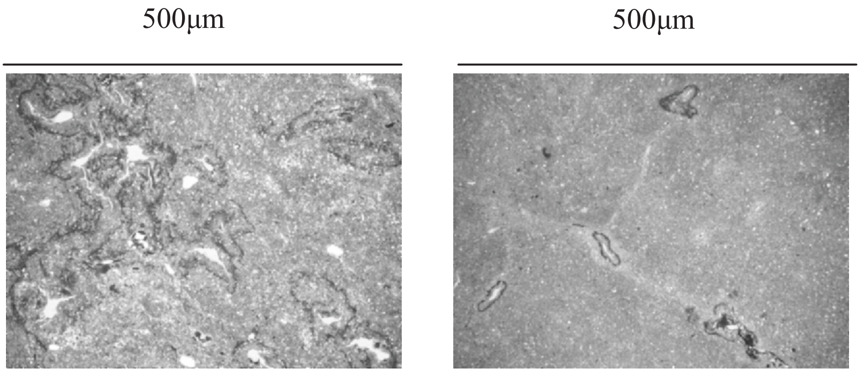

Figure 4. White voids (left) and mineral alteration (right) indicating the presence of root holes.

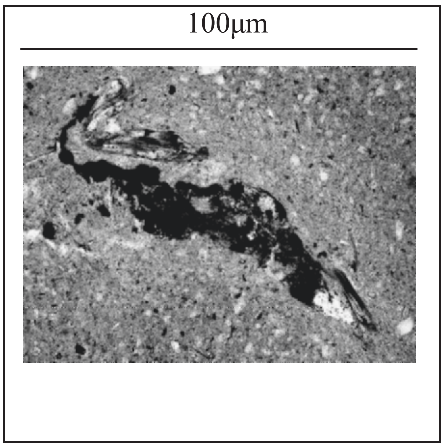

Figure 5. Black cloud-shaped structure indicating worm activity.

7. CONCLUSIONS

The penetration of dissolved-phase aromatic hydrocarbon solutes from a sandy aquifer into a 0.3-1.5 m thick bed of lacustrine clay at 6 m depth was investigated using core-based contaminant profiling. Penetration into the clays occurred to over 0.6 m and could not be fully explained simply by 1-D diffusion or advection-dispersion (-retardation) modelling over the anticipated contaminant contact with the clay.

The contaminant profiles were variable over a small area. Of the two most detailed profiles obtained, one had a profile consistent with diffusion-only transport. The other profile cannot be explained by diffusion alone and it is believed heterogeneities within the clay layer creating preferential flowpaths have influenced contaminant transport. Observations of palaeo-roots in the clay offer a likely explanation. Further field cores and other relevant data (e.g. further hydraulic head gradient analysis over the clay layer) as well as supporting laboratory studies (e.g. sorption) are continuing. It is concluded that unconformity surfaces should be taken into account when deciding whether a clay bed affords protection for an underlying aquifer.

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BASELINE GROUNDWATER QUALITY IN THE COASTAL AQUIFER OF ST. LUCIA, SOUTH AFRICA

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Abstract: A study of baseline groundwater chemistry has been carried out in the pristine areas around Eastern Shores, Lake St Lucia, North-eastern South Africa. The study aims to provide a quality baseline against which anthropogenic (urban) impacts elsewhere may be evaluated. In general, groundwater in the Eastern Shores region is of low conductivity, and a vital freshwater source to both Eastern Shores and Lake St Lucia itself. Two distinct groundwater types are recognized in the aquifer, one dominated by sodium and chloride and the other by calcium and bicarbonate. The two are spatially distinct, corresponding to separate groundwater mounds. Variation in electrical conductivity is thought to be related to evapotranspiration. The study provides valuable background data on coastal aquifer natural quality and a suitable baseline to judge impacts upon that aquifer, both from historic and future urbanization.

Key words: baseline groundwater quality; coastal aquifer; hydrogeochemistry; Lake St Lucia, South Africa

1. INTRODUCTION

In the coastal areas of Maputaland in southeastern Africa groundwater represents an important resource that interacts with both surface and coastal water bodies. The permeable sandy sediments and the relatively high annual rainfall result in a high groundwater recharge rate. Surface water

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forms smaller pans in wetland areas, where the relief is below the groundwater table.

A baseline study of groundwater chemistry has been carried out in the region of Eastern Shores, Lake St Lucia in northeastern South Africa (Figure 1). The study focused upon areas with natural vegetation and some recently deforested areas. As such it represents an area where there has been little historical anthropogenic activity and hence influence on underlying groundwaters that are potentially “natural” or of a pristine nature. Understanding of the groundwater chemistry in natural environments is important in order to interpret the impact of human activity in other more urbanised locations along the coast.

Knowledge of groundwater quality, combined with water quantity studies, is essential in the understanding of hydrogeological and hydrogeochemical processes that affect the ecology of the Eastern Shores area.

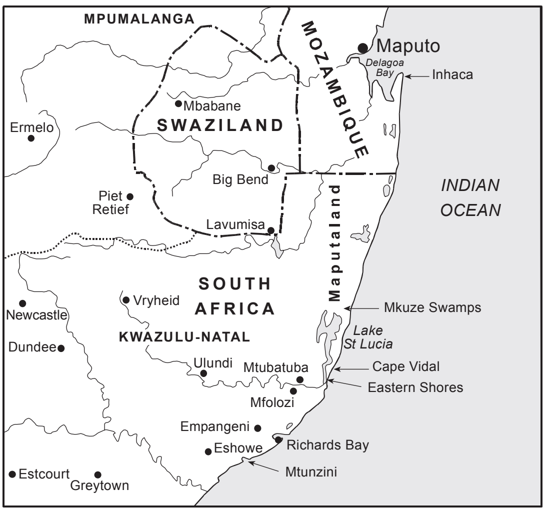

Figure 1. Map showing coastal Maputaland stretching from Mtunzini in the south to Inhaca in the north.

2. AREA DESCRIPTION

Lake St Lucia, the largest coastal lagoon in southern Africa, is located on the Maputaland coastal plain (Fig. 1). The Greater St Lucia Wetland Park was declared a World Heritage Site by UNESCO in 1999, and recognized as a Wetland of International Importance in terms of the Ramsar Convention (Kelbe *et al*., 1995; Porter and Blackmore, 1998; Barnes *et al*., 2002). The St Lucia Wetland Park is located at the interface between tropical and subtropical climates (Porter and Blackmore, 1998). The mean annual temperature is 21°C (Porter and Blackmore, 1998), and rainfall at the coast is 1200-1300 mm per annum. Lake St Lucia has a large surface area (350 km^2) with an average depth of 0.9 m (Taylor, 1998). During droughts, high evaporation rates, in combination with a low supply of freshwater from rivers and precipitation, lead to hypersaline conditions in the lake. The most recent drought started in 2001, and at the end of 2003, the salinity of Lake St Lucia was four times the salinity of seawater (Taylor, 2003).

A high, vegetated coastal dune complex defines the border between the Indian Ocean and the Maputaland coastal plain (Fig. 1). The complex extends from Mtunzini in the south to Inhaca Island, Mozambique in the north (Kelbe and Rawlins, 1992a). Occurring just south of the mouth of Lake St Lucia, these are the highest vegetated dunes in the world, with a peak summit of more than 180 metres above sea level (Taylor, 1991). The Eastern Shores section is located between Lake St Lucia and the Indian Ocean and consists of high coastal sand dunes and low-lying plains that are an important terrestrial component of the Greater St Lucia Wetland Park.

The Eastern Shores can be divided into four topographical units (Fig. 2): 1) the 1-2 km wide and 80-130 m high coastal dune complex, 2) a low-lying plain west of the dunes with a width of 0.5 to 7 km, 3) the Mfabeni Swamp, a depression parallel to the coast, and 4) the Embombveni Ridge, an extensive coastal upland area between Mfabeni Swamp and Lake St Lucia with elevations of 15 – 70 m (Davies *et al*., 1992). Unconsolidated, quartz-dominated aeolian sand forms the major part of the aquifer (Kelbe and Rawlins, 1992a; CSIR Environmental Services, 1993). Calcarenites are observed at depth in the easternmost part of the dune complex (Hobday, 1976; Davies *et al*., 1992; CSIR Environmental Services, 1993). Coastal dune forest, grasslands and wetlands are the dominant vegetation types. In addition there are some recently cleared pine plantation areas. Except for the pine plantations, the area has been relatively undisturbed by human activity in the past 50 years.

The groundwater flow pattern for the Eastern Shores is derived from two shallow mounds separated by the Mfabeni swamp (Fig. 2). A cone-shaped groundwater dome is present under the Embombveni upland area, and from this the groundwater discharges into Lake St Lucia in the north, west and south, and flows into Mfabeni Swamp in the east. From the coastal dune complex, groundwater discharges into the Indian Ocean in the east and into Mfabeni Swamp and Lake St Lucia in the west.

Groundwater is considered to be of particular significance to the St Lucia system, with groundwater seepage into Lake St Lucia from the

Eastern Shores emphasized in several studies (Kelbe and Rawlins, 1992a, 1992b; Kelbe *et al.*, 1995; Været, 2002; Taylor, 2003; Wejden, 2003). The parts of the lake shoreline where there is a particularly high discharge form freshwater refugia. These are places where salt-sensitive organisms can survive droughts and dry periods, when groundwater is the only persistent source of fresh water to Lake St Lucia.

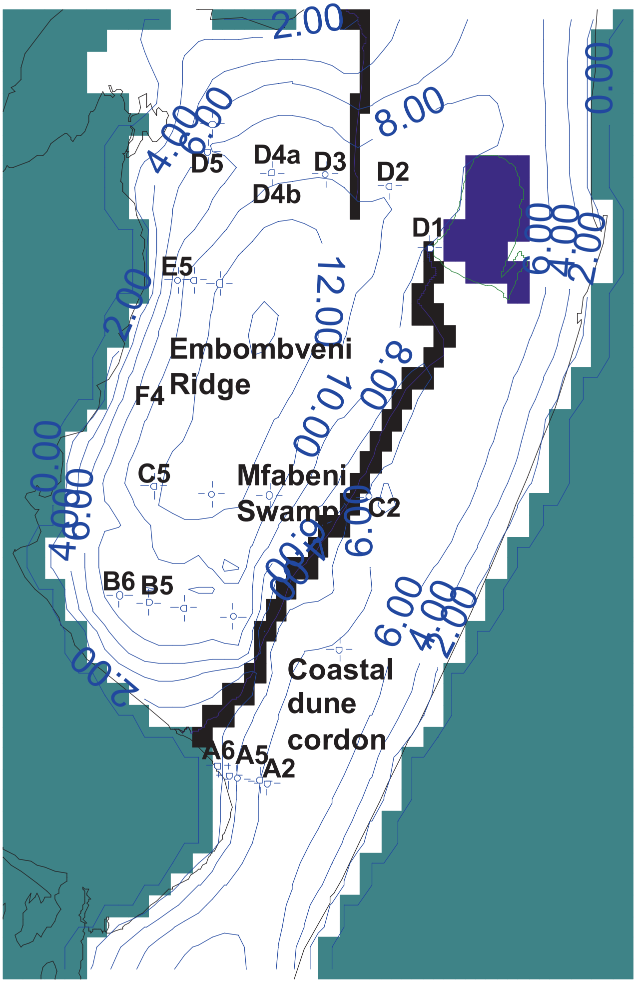

Figure 2. Hydrogeological map of the Eastern Shores simulated for December 1999 (Wejden, 2003). Borehole names are included.

3. SAMPLING AND ANALYSIS

Groundwater samples were collected in the Eastern Shores on four occasions between February 2002 and January 2003 from permanent boreholes and provisional augur holes (Fig. 2). Electrical conductivity, temperature and pH were measured in the field. The samples were filtered in the field at 0.45 μm and stored at 2-5˚C until analyzed. The main water analysis was carried out at the University of Natal. Alkalinity was determined by titration to pH 4.5. Sodium and potassium were analyzed using an atomic absorption spectrophotometer. Calcium, magnesium and silicon were analyzed by ICP. Chloride and sulphate were analyzed by ion chromatography.

4. RESULTS AND DISCUSSION

4.1 Water Quality

The main cations and anions for all groundwater samples were plotted in triangular diagrams. Based on the chemical composition of the groundwater samples, two groups were identified in the Eastern Shores (Figure 3). One group is dominated by the coastal-influenced precipitation parameters sodium (Na^+) and chloride (Cl^-), while the second group is dominated by the geological parameters calcium (Ca^{2+}) and bicarbonate (HCO_3^-). These groups match the groundwater flow pattern, as there is a distinct difference in the composition of groundwater from the eastern part of the Eastern Shores aquifer and that from the western part.

Groundwater from the Embombveni ridge is dominated by the sodium-chloride type, with precipitation carried by south-westerly winds from the southern oceans (Taylor, 1991) contributing Na and Cl ions to the area. The composition is similar in all boreholes from this region, even though electrical conductivity ranges from <100 to >600 μS/cm. This indicates that the ions are concentrated due to evapotranspiration processes. Evapotranspiration varies locally as an effect of variations in vegetation and depth to the groundwater table.

The calcium-bicarbonate type groundwater is associated with flow from the coastal dune complex in the east. The groundwater generally has low electrical conductivities (100-300 μS/cm). The dominance of calcium-bicarbonate indicates that the water percolates down through the unsaturated zone and that the evapotranspiration processes play a minor role for the ionic composition. The source of the calcium is probably the calcarenites in the dune sand. Calcarenite is found in the easternmost part of the dune complex, but not further west (Davies *et al*., 1992). The presence of beach-rocks along the beach from Cape Vidal to St Lucia (Harris, 2002) is also an indication of relatively high calcium content in groundwater from the dune complex.

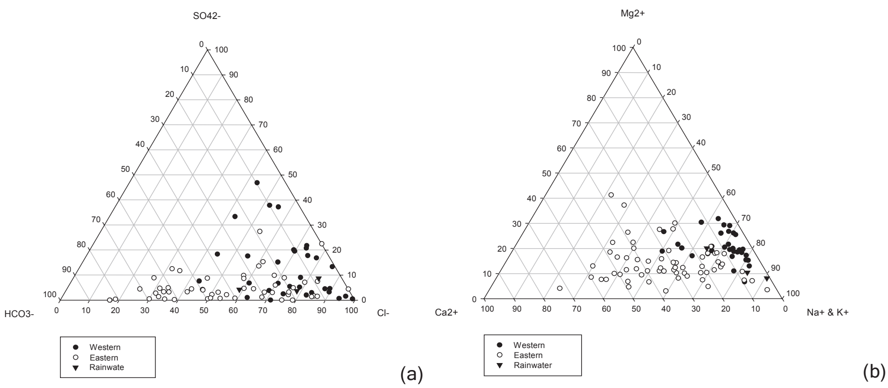

Figure 3. Trilinear plot of (a) anions (Cl^-, SO_4^{2-}, HCO_3^-) and (b) cations ([Na^+, K^+], Ca^{2+}, Mg^{2+}) for groundwater from the Eastern Shores 2002-03. Samples are divided into three groups: the western groundwater system, the eastern groundwater system and rainwater.

4.2 Possible Precipitation of Secondary Minerals

In some other subtropical and tropical wetland areas, extensive precipitation of secondary minerals (e.g. amorphous silica and calcite) occurs. In the Okavango Delta in Botswana, precipitation of secondary silica is observed at a large scale in wetland areas, and is explained by low rainfall, recharge of surface water spread out on a huge wetland plain, and very high evapotranspiration rates (McCarthy *et al*., 1993; McCarthy and Ellery, 1994, 1995). Precipitation of silica is observed in floodplains in the Mkuze Swamps just north of Lake St Lucia, and explained as a consequence of vegetation processes (Barnes *et al*., 2002). The eastern part of the Mkuze system is part of the same dune complex that is present in the Eastern Shores. With these studies in mind, it is hypothesized that precipitation may occur also in the Eastern Shores (Ellery, 2002).

The saturation state for calcite, Ω, was calculated for Eastern Shores groundwater from the ion activity product, *IAP*, and the solubility product constant for calcite, $K_{sp} = 1.0*10^{-8.3}$ (Evangelou, 1998; Appelo and Postma, 1999). The calculations of Ω in the groundwater show that most boreholes in the Eastern Shores are undersaturated with respect to calcium carbonate. Concentrations of silica (Si) in the Eastern Shores groundwater ranged from 0.5 to 20 mg/l, with the majority of samples containing less than 5 mg/l. In consideration of these results it is unlikely that silica precipitates in significant quantities at the studied localities in the Eastern Shores. However, precipitation of calcite and silica may occur in certain areas in the Eastern Shores not included in this study.

4.3 Plantation Logging

Logging of pine plantations in the Embombveni Ridge area has caused increased concentrations of the plant nutrient nitrate and, to a lesser extent, potassium and magnesium in groundwater in the deforested areas.

5. CONCLUSIONS

In general, groundwater in the Eastern Shores is of low conductivity, and provides a vital freshwater source to the Eastern Shores and Lake St Lucia itself. The two different water types recognized may affect the ecology, especially in discharge zones. Their quality is determined by natural conditions, primarily mineralogy, rainwater quality and evapotranspiration. In addition, logging of plantations has increased the quantity of plant nutrients in the groundwater in these areas. However, the groundwater is relatively unaffected by human activities, and provides valuable background data for comparison with other coastal aquifers in south-eastern Africa, particularly those influenced by historic or future anthropogenic activity including growing urbanization.

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HYDROCHEMICAL QUALITY OF GROUNDWATER IN URBAN AREAS OF SOUTH PORTUGAL

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Abstract: This investigation of urban groundwater problems in the city of Évora is one of the first in Portugal, its main objective being to develop a hydrochemical knowledge of the city's groundwater. Thirteen water points, represented by 7 large diameter wells and 6 deep wells, were selected for sampling. The major ions were analysed, as well as temperature, EC, pH and total hardness. The results show that most of the waters are HCO_3-Mg-Ca-Na type, but some of the samples are Cl in type and one is clearly Cl-Na in type. The EC varies between 230 and 2500 μS/cm. The nitrate content varies between 0.5 and 310 mg/L, but over 50 % of the samples exceed the 25 mg/L maximum recommended by Portuguese Law. The absolute permitted maximum of 50 mg/L is also exceeded in 30 % of the samples. When compared with water quality elsewhere in Alentejo, the results are clearly high in terms of EC, nitrates and chlorides. The high content of nitrates and chlorides may be an indication of urban contamination.

Key words: hydrogeochemistry; water quality; contamination; Portugal; Évora, Portugal; nitrate; chloride; water types

1. INTRODUCTION

In recent decades, the processes of urbanization and industrialization have caused severe environmental deterioration, including major changes to hydrodynamics and the quality of groundwater. As a consequence, much

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research has been conducted to further our understanding of the main physical, chemical and biological processes that occur.

In urban areas, the most common concern is groundwater quality, since the remediation of contaminated water to drinking water standards is frequently an impossible task (Howard, 2002). Shallow aquifers suffer most directly from the negative impacts of anthropogenic soil use in urban agglomerates. Considering climatic effects on aquifer recharge, particularly in those areas where recharge comes only from precipitation, impacts are often immediate and direct.

In urban areas, soil impermeability created by man-made constructions has a profound effect on aquifer recharge. In 2002, Duque *et al.* presented a study regarding the recharge-discharge balance in the city of Évora (South Portugal), where conclusions pointed to a reduction of the natural recharge of about 25 % due to this impermeabilization, probably all of them recovered by the losses in the supply and sewerage systems. The main aim of this study is to implement a hydrochemical knowledge of groundwater in the same city (Figure 1).

Évora was a Roman city and, after some centuries under Muslim rule, of major importance in the 16th Century. As result, the city has a walled central district where the typical characteristics of an old city are present: narrow streets and compact buildings with occasional gardens. This area was recognized as a World Heritage Site by UNESCO in 1986. Beyond the wall limits, the new city expanded with a different pattern of buildings and more free spaces. The houses there have generally one to three storeys and are usually separated by small private or public gardens. The city is officially limited by a line (defined in Figure 1) and, inside it, there are also large non-urbanized areas which represent potential construction areas for future decades.

The groundwater points inside the city walls are scarce or inaccessible. This feature is due to the kind of urbanization, where the very narrow streets don't permit the construction of any deep well. Only old large diameter wells are found in some particular gardens or inside the houses, and today they are used mainly as decoration.

Inside and outside the city walls thirteen water points were selected for sampling. Seven of them were large diameter wells and six deep wells, and the major ions were analyzed, as well as the temperature, EC, pH and total hardness in all of them.

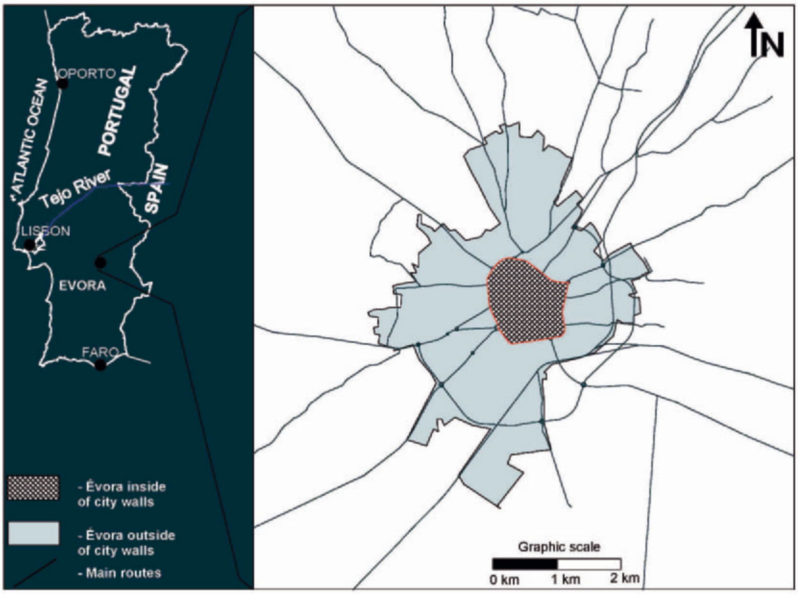

Figure 1. Map of the city of Évora, showing the urban area outside and inside the city walls.

2. GEOGRAPHIC FEATURES

Évora is the main city of Alentejo, a large region of South Portugal. The region is characterized by a large plain approximately 200-400 metres above sea level that corresponds to almost one third of Portugal's total area. However, the population of Alentejo is only about five percent of the total Portuguese population. The region has a semi-arid to sub-wet Mediterranean climate with an obvious Atlantic influence along its western coast.

Évora is in the centre of Alentejo, about 100 km east of Portugal's west coast and about 200 km north of the south coast. It has a population of around 50,000 people, and its main sources of income are from agriculture, tourism, services and light industry.

3. GEOLOGICAL SETTING

The city of Évora is situated on Precambrian and Palaeozoic igneous and metamorphic rocks deformed by the Hercynian Orogeny. Figure 2 illustrates the main geological units.

Two main groups of lithologies occur in the city area:

- The west and south-western parts are underlain by a large batholithic dome, with granodiorites and quartzodiorites the main lithologies; the relief is pronounced in certain areas, due to the high resistance to rock weathering, which permits the presence of some hills; the weathered layer rarely exceeds two to six metres in depth.
- In the north and eastern parts of the city a well defined strip of gneissic and migmatitic rocks occur; these rocks have variable textures and are highly weathered and fractured to a depth of between 20 and 30 metres, giving rise to high rock permeability, allowing the occurrence of a regionally important, unconfined to semi-confined aquifer, the Évora Sector of the Évora-Montemor-Cuba Aquifer System. The main discharge collector around Évora is the Xarrama River, which flows in a southerly direction along the eastern side of the city; the Xarrama valley follows a major fracture of significant hydrogeological importance.

4. HYDROGEOLOGICAL SETTING

As a typical hard rock aquifer, the conceptual model follows a vertical sequence consisting of a shallow, upper unconfined aquifer in a layer of weathered material, a middle double porosity fractured aquifer and finally a lower massive and compact zone with rare deep productive fractures (Chambel and Duque, 1999). The well logs show a wide range of depths for these zones. However the weathered and fractured zones rarely reach 40 metres in depth. Prospective drilling commonly stops if this depth has been reached without success.

The transmissivities on the aquifer area range from 30 to 100 m^2/day. These values are notably higher than those that occur in the granodiorite and quartzodiorite areas, which range from 1 to 10 m^2/day. Yields in the Évora Sector usually range from 3 to 20 l/s. The granodiorite/quartzodiorite wells yield less than 1 to 2 l/s and the rate of unproductive wells in these rocks is very high.

As with almost all shallow hard rock aquifers, the aquifer bottom strongly correlates with the topographic surface. Évora is sited at the top of a granodioritic hill, so the flow system around the city follows a typical radial shape imposed by the topographic conditions.

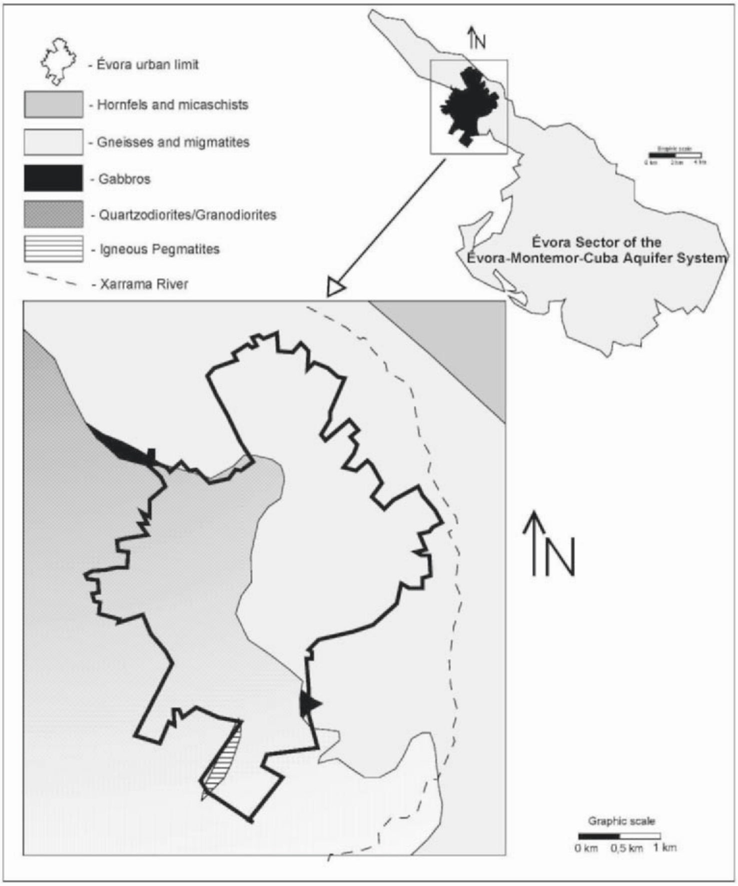

Figure 2. Geologic setting of the city of Évora and position of the Évora Sector of the Évora-Montemor-Cuba Aquifer System.

The main discharge is near the Xarrama River where some flowing wells occur. Secondary discharge zones correspond to a number of ephemeral streams that occur around Évora. However, the flow direction has a primarily south-south-westerly trend, as shown in Figure 3.

Duque et al. (2002) estimated the annual volume of recharge within the city limits to be 874,000 m^3. The annual recharge inside the city walls was determined to be 22,500 m^3 and the annual recharge outside the city walls 851,500 m^3. These values are equivalent to 7.8% of the total annual precipitation (650 mm/year).

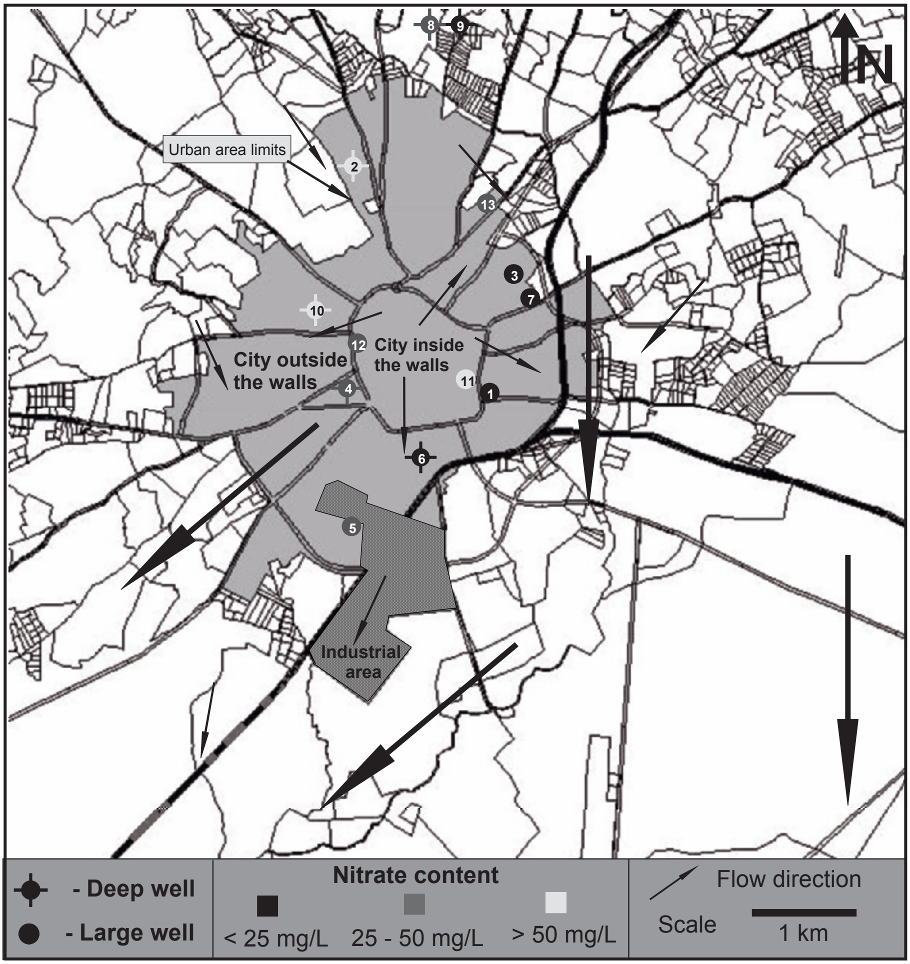

Figure 3. Main flow directions, groundwater sampling sites and distribution of nitrate content around the city of Évora.

5. GROUNDWATER HYDROCHEMISTRY

In order to perform an assessment of the groundwater hydrochemistry of the city of Évora, thirteen water points were chosen in the city area (both inside and outside the city walls). Figure 3 shows the selected sample points, represented by 7 large diameter wells and 6 deep wells.

The major cations and anions were analysed, as well as the temperature, electrical conductivity (EC), pH and total hardness.

Figure 4 represents a Piper diagram for the 13 groundwater samples. The results show that waters are mainly HCO_3-Mg-Ca-Na in type, but some of the samples are Cl in type and one is clearly Cl-Na in type.

The EC ranges between 230 and 2500 μS/cm. The nitrate content varies between 0.5 and 313 mg/l, but more than 50% of the samples contain over 25 mg/l, the recommended maximum value under Portuguese law. The absolute permitted maximum of 50 mg/l is also exceeded in 30% of the samples. Chlorides vary from 10 to 285 mg/l, and sulphates from 10 to 150 mg/l.

When compared with water quality elsewhere in Alentejo, the results are particularly high in terms of EC, nitrates and chlorides. The high content of nitrate, chloride and sulphate may be an indication of urban contamination, the origins of which may be leaking sewerage systems and fertilizer applications in gardens and vegetable gardens, the latter being very common outside the old city centre.

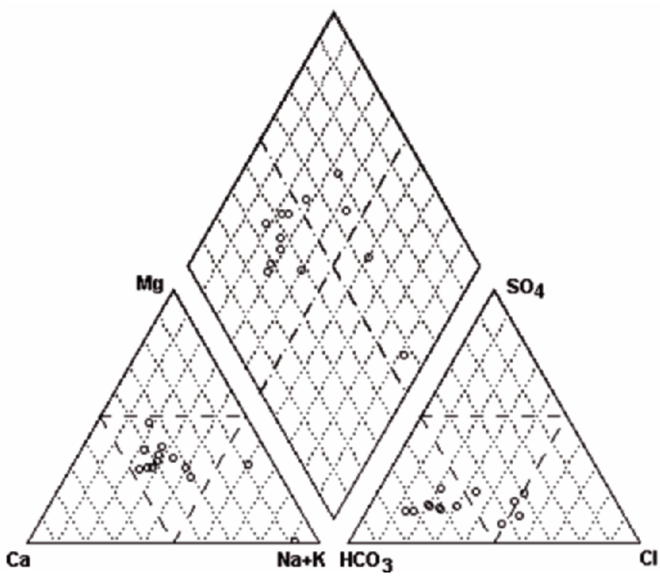

Figure 4. Piper diagram for the groundwater samples collected in the city of Évora.

The large diameter wells seem to have the worst problem with nitrate contents, although the same problem is seen also in some of the deep wells. Furthermore, although the large diameter wells have the highest mean EC values, the highest value is from a deep well sample. Thus, the difference in terms of possible contamination between large diameter and greater depth is not clear, though it is possible that the shallowest groundwater levels are responsible for the highest levels of contamination because, even in deep wells, isolation of the shallow levels is generally ineffective.

The scarcity of wells, the limited groundwater exploration within the city boundary, and the fact that the city is on a topographic high compared to the surrounding region, all suggest that, if it is not used extensively in the future, the groundwater within the city will flow into the Évora Sector of the Évora-Montemor-Cuba Aquifer System, contaminating the aquifer outside the city.

An assessment of the groundwater quality for irrigation was conducted under United States Salinity Labs Staff (USSLS) standards. The Sodium Adsorption Ratio (SAR) was calculated using the HIDSPEC hydrogeochemistry code of Carvalho and Almeida (1989), allowing the evaluation of salinization and alkalinization hazards for soils. Figure 5 shows the results, which indicate no potential hazard of soil alkalinization. However, the potential likelihood of soil salinization is high or very high (above the class C_3S_1) for almost 80% of the groundwater samples.

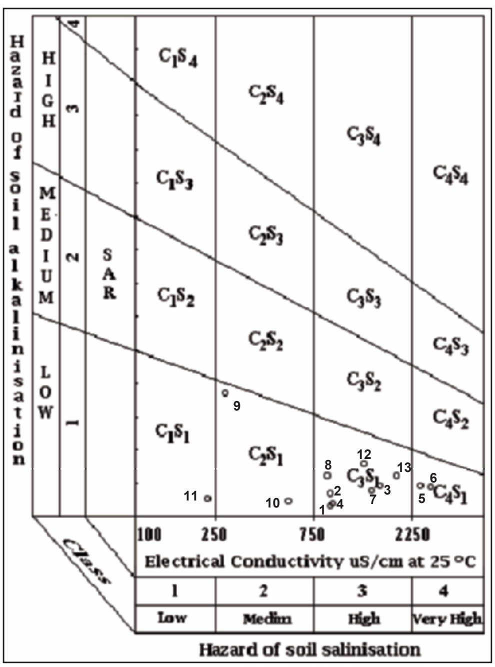

Figure 5. Plot showing risk of soil salinization and alkalinization according to USSLS standards.

The saturation indices (SI) for calcite and dolomite for the 13 groundwater samples were also determined using the HIDSPEC code. They were accordingly classified as subsaturated, supersaturated and in equilibrium, as summarized in Figure 6. In the discharge zones groundwater is supersaturated, precipitating calcite and dolomite, but on higher ground, it has a more aggressive profile, dissolving calcite and dolomite. The only sample displaying equilibrium was collected from between these zones.

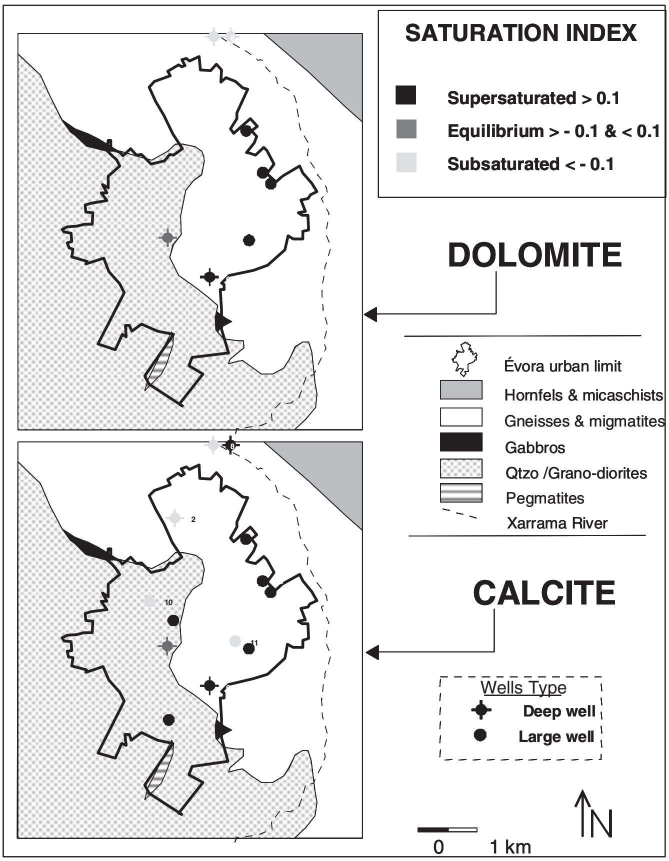

Figure 6. Distribution patterns of the saturation index (SI) for dolomite and calcite in the city of Évora.

6. DISCUSSION AND CONCLUSIONS

The concentration of nitrates in the groundwaters of Alentejo is not a major concern at present. However, the high levels of nitrates detected in some samples from this analysis of the city of Évora show that the potential for contamination should not be overlooked. The same phenomenon occurs in other urban areas within Alentejo, particularly villages where vegetable gardens are present within or very close to the village limits.

Intensive use of fertilizers appears to be much more common in urban vegetable gardens than in areas of extensive agriculture outside the cities, and may be a consequence of higher irrigation rates (most agricultural land within Alentejo is not irrigated) and the low cost of small-scale fertilizer use.

Also the losses from the sewerage system are difficult to control. Information from the Municipality indicates that water loss in the supply system is about 9%, which seems a very low value given that Évora is an ancient city with an ancient supply system. Even accepting this figure, losses in the sewerage system will be considerably greater and could reach 50% in some places. Contamination of the Évora aquifer outside the city limits may occur in future, since the city is located above the highest point of the aquifer and is an area of preferential recharge.

Contamination of the aquifer outside the city limits can be avoided if the nitrate-rich waters are utilized within the city, perhaps in street cleaning, fire fighting, and carwashes. The use of these waters for garden irrigation must be monitored according to the original quality of the groundwater, the nature of the soils and susceptibility of the plants, in order to avoid soil salinization. Of assistance in this regard is the number of areas with wells yielding water of good quality for irrigation that could be intensively used.

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ISSUES OF RADIOACTIVITY AND SUSTAINABLE DEVELOPMENT WITHIN URBAN GROUNDWATER SYSTEMS IN RUSSIA

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Abstract: This paper examines the issue of radioactivity within urban groundwater systems in Russia. Since the atomic industry began, the enterprises associated with different elements of the nuclear fuel cycle have become industrially advanced and helped drive the development of satellite cities. However, the groundwater systems of these cities were affected, especially in terms of water balance and quality. Radioactivity as a specific contaminant is a source of concern for both the public and scientists. This paper examines data concerning radioactive pollution produced by nuclear power plants, research reactors, and the Chernobyl accidents. The main conclusion is that radiation pollution is much less of a risk to the health of the Russian population than social and economic factors.

Key words: radioactivity; Chernobyl; pollution; Chernobyl, Russia; nuclear power.

1. INTRODUCTION

The history of radioactivity in the groundwater of Russian urban areas began in the 1950s with the beginning of the nuclear industry and, in particular, the development of nuclear fuel recycling enterprises. The locations of such highly technical enterprises became industrially advanced, and the enterprises were city-forming. The main activities of the atomic industry in Russia have been: nuclear power plants (Figure 1); enterprises concerned with the operation of nuclear research plants and industrial reactors on land; construction and maintenance of nuclear submarines; manufacture of nuclear weapons; and radioactive waste disposal.

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Nuclear plants are sources of potential threat to the environment from the start of their use right up to the present day. During normal operation, but more especially when accidents occur, the equilibrium of all components of the surrounding environment, including water resources, is disturbed. Of all the components of an environment affected by radioactive pollution, the greatest ecological danger is connected with groundwater pollution. Groundwater requires sustainable development in order to preserve water resources suitable for future generations. According to the World Health Organization (WHO), the quality of public health is defined first of all by socio-economic factors. Ecological conditions also influence public health, for example through the consumption of poor-quality drinking water that spreads disease, exacerbating the effect of social factors.

Groundwater systems supplying the satellite cities serving the nuclear enterprises are affected especially in terms of their water balance and quality. Radioactivity is a constant source of alarm to the public and as such is the subject of enhanced scientific attention.

2. NUCLEAR POWER PLANTS

Nuclear plants are sources of risk to the environment from the beginning of their working lives to long after their decommissioning. The distribution of nuclear plants across Russia is shown in Figure 1.

Figure 1. Nuclear power plants in Russia.

Nuclear power plants influence the groundwater systems of their satellite cities even during normal operation (Table 1). This influence includes thermal pollution, water level rise and flooding of urban areas, and radioactivity pollution as a result of water filtration from reservoir-coolers and other structures.

The radionuclide background levels, caused by natural nuclear processes in the Earth's crust, are less than 0.01 Bq/l for Cs-137 and 0.02 Bq/l for Sr-90 (the radionuclides responsible for long-term radioactive environmental contamination). The Russian Norms of Radiation Safety (NRS-99) have defined the Intervention Level as “a level of a radiation factor, [the] exceeding of which certain requires protective measures to be fulfilled”. For the radionuclides Cs-137 and Sr-90, the intervention levels are 11 and 5 Bq/l respectively. Measurements within the zone around nuclear power plants have shown that the content of these and other artificial radionuclides in surface waters and groundwater frequently exceeds the natural background levels (Table 1). Small radioactive concentrations in groundwater, even if lower than the intervention levels, are cause for concern. However, the irregular nature of the detections means it cannot be concluded that a catastrophic situation exists.

Table 1. Satellite cities associated with nuclear power plants (NPPs).

Nuclear Power Plant (NPP)	Satellite city (population)	Distance from NPP (km)	Radionuclide content in groundwater within NPP’s observation zones (Bq/l) (Petruhin *et al.*, 2001)
Balakovo	Balakovo (200,000)	10	No data
Bilibino	Bilibino (10,000)	4	No data
Volgodonsk	Volgodonsk and Cymlyansk (227,000)	13.5 and 19	Cs-137 = 0.04 Sr-90 = 0.03
Smolensk	Desnogorsk (40,000)	3	Total = 1
Beloyarsk	Zarechny (33,000)	3	No data
Kursk	Kurchatov (49,000)	5	Cs-137 = 1 – 1.6
Novovoronezh	Novovoronezh (39,800)	5	Cs-137 = 0.002 Sr-90 = 0.003
Kola	Polyarny Zori (21,900)	4	No data
Leningrad	Sosnovy Bor: 62,800	4	No data
Kalinin	Udomlya (45,300)	5	Total = 1 – 1.4: 0.01/Cs-137 = 0.01: Sr-90 = 0.003

3. ENTERPRISES CONCERNED WITH THE OPERATION OF NUCLEAR RESEARCH PLANTS AND INDUSTRIAL REACTORS ON LAND

One of the largest nuclear centres in Russia is Dimitrovgrad (population 136,000) where the state Research Institute of Atomic Reactors is located. Results of radiation monitoring in the vicinity of Dimitrovgrad (Gremiachkin *et al.*, 2004) demonstrate that annual caesium activity in water over the last 5 years has remained at a constant level, several orders of magnitude below the intervention level. The population's exposure to radiation due to the presence of the Institute has increased no more than 0.2 % above the natural background level.

In Moscow, many of the close-to 1900 organizations using radioactive substances do not have a significant influence on radioactivity levels in the city's waters: the organizations are responsible for only 0.06 % of the irradiation that Muscovites were exposed to in 2002. The main conclusion made by experts of the state enterprise 'Radon', is that the situation in Moscow is stable and does not represent a health hazard to the population.

4. RADIOACTIVE WASTE ENTERPRISES

Disposal of radioactive waste in Russia has been carried out traditionally by specialized departments of the atomic industry. Since 1963, 47 million m^3 of liquid radioactive waste has been injected into deep reservoir formations (depth >300 m) (Lebedev *et al.*, 2001). The main storage sites are the Siberian Chemical Combine (the satellite city of Seversk having a population of approximately 119,000), the Mining and Chemical Combine (Zheleznogorsk: population 100,000), and the Research Institute of Atomic Reactors (Dimitrovgrad). The travel velocities of the groundwater are slow (3-6, 10-15, and <1 m/y respectively) and provide long-term waste localization. From the time these sites opened to the present day the radioactive components have been localized within predicted boundaries and the possible radiation burden on the general public should not exceed the established norms.

The wastes at the combine 'Mayak' (the nearest towns being Ozersk, with a population of approximately 80,000, and Novogorny, population c.10,000) are stored at the surface. Additionally, there is a large polluted zone in this territory, a consequence of defence programme accidents in the 1950s.

The main pollution is seen in the vicinity of the river Techa. Radionuclide levels in the river water remain elevated due to the washing-in of material from the polluted sites. It has been shown by numerous state experts from various organizations that modern radiation levels do not represent a danger to the living population as long as they observe the established safety rules. Nevertheless, a limited part of the population can receive doses of radiation, which are approaching those permissible under NRS-99, by using water from the river, in infringement of sanitation rules.

The proposed solution to the problem of radioactive waste in Russia is that its disposal should be in a solidified form within geological formations, and that the storage sites should be located in the vicinity of the nuclear power plants.

5. CHERNOBYL FALLOUT ZONE

The Chernobyl accident has influenced the groundwater systems of all cities within the Chernobyl fallout zone. Radionuclide contamination takes place through the further distribution of radionuclides within a geological medium. In this case, the controlling processes are: the infiltration of precipitation recharging groundwater through contaminated soil levels and vadose zones; the filtration of groundwater through water-bearing rocks; and the discharge of shallow groundwater into rivers and other water reservoirs (Rogachevskaya and Zektser, 2003; Rogachevskaya, 2004).

Research in Russia, Ukraine, and Belarus has shown that the conclusion that there is sufficient protection of groundwater from radioactive contamination is incorrect. Higher than background concentrations of radionuclides have been found in groundwaters almost everywhere within the zone affected by fallout from Chernobyl (Rogachevskaya and Dubinchuk, 2001). The main contributors to the pollution of natural waters have been the isotopes Sr-90 and Cs-137, both of which are extremely long-lived. In contrast, regional exceeding of the Intervention Level in potable water outside the Chernobyl zone is practically unknown.

In wells dug to supply water in Novozybkov, the caesium content in the water increased from 0.01 Bq/l in 1989 to 0.04 Bq/l in 1999 (Figure 2). Even confined aquifers may be, in some places, incompletely protected from Chernobyl pollution. Around the water supply wells of Novozybkov and other towns, the cones of depression have radii of up to 2-3 km in diameter, with draw-downs of up to 5-7 m. The small-scale increase in caesium concentration in the water-supply wells is likely to be explained by polluted shallow water being drawn down locally into the main aquifers by an increase in vertical head gradients close to the pumping wells. Similarly, increasing draw-downs have reversed the flow direction at some streams, thus inducing polluted stream water to enter the aquifers.

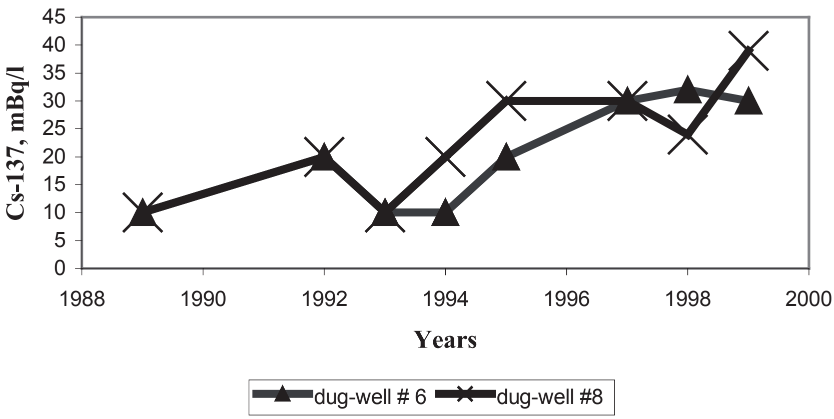

Figure 2. Changing Cs-137 levels in water from dug-wells, Novozybkov town, Bryansk region (Project RF-UNDP RUS/95/004).

The implementation of a complex of measures is necessary for the sustainable development of urban groundwater systems affected by the Chernobyl accident, as follows:

- The provision of regular information and education towards the safer use of local water sources and resources, through public relations, special lessons in schools and universities, and specially run courses;
- The transfer of towns' water supplies from surface water sources to artesian groundwater;
- A technical inspection, inventory and certification of all exploited pumping wells;
- An inventory of unexploited and technically destroyed wells with their consequent remediation using technology that excludes the percolation of radionuclides through the 'shaft' of a remediated well;
- A strengthening of the protection of exploited aquifers against surface contamination and over-exploitation;
- A strengthening of the protection of pumping wells (complete grouting of space beyond the casing, grouting of the well bottom, etc.).

6. CONCLUSIONS

Scientific and technical progress directed towards satisfying the needs of the growing world population (particularly the demand for energy) gives rise to new sources of risk and generates the problem of man-made accidents. Any influence on the environment creates a negative image of nuclear techniques and affects their further development. Furthermore, such effects demand essential expenditures for pollution abatement and indemnity payments to the affected population. Therefore, in the course of their development, nuclear sciences have changed from being purely engineering sciences to sciences combining elements of technical, environmental and socio-political disciplines. In summary, however, it is possible to conclude that radiation pollution is less of a risk to the health of the population in Russia than social and economic factors.

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RISKS POSED BY UNSANITARY LANDFILL LEACHATE TO GROUNDWATER QUALITY

Çorlu (Trakya), Turkey

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Abstract: Poorly regulated landfill disposal is a worldwide problem. This study aimed to assess the risks posed by unsanitary landfill leachates to groundwater quality. The case study presented examines the town of Çorlu in the Tekirdağ Province of Turkey, where various types of waste are disposed into unsanitary landfill sites without any separation or classification of hazardous and non-hazardous waste. The leachate from unsanitary landfill in Çorlu has complex characteristics that are dependent on the composition of solid waste in the landfill. Contamination risks posed to groundwater vary and depend upon the quantity of leachate generated (in this case ~13 m^3/day) and its specific physico-chemical characteristics. Considerable public attention has recently been focused on the environmental and potential human health risks of unsanitary landfill in Çorlu. The potential risk of leachate contamination of groundwater resources in Çorlu Town is assessed from waste quality, soil characteristics and leachate composition data. General recommendations to facilitate improved future protection of groundwater quality are indicated.

Key words: leachate; landfill; Corlu, Turkey; Trakya; Tekirdağ; Thrace; hazardous wastes; non-hazardous wastes.

1. INTRODUCTION

The town of Çorlu is situated in western Turkey, close to Istanbul. It has seen intensive industrial and urban development since 1990. In Çorlu,

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technological developments, increasing population, uncontrolled urbanization and rising standards of living have resulted in a tremendous increase in the quantity and diversity of solid waste.

The most significant problem associated with solid waste in Çorlu Town is the insufficient solid-waste management system. At present, unsanitary landfill is the only management method used and, unfortunately, Çorlu has placed very little emphasis on the proper management of hazardous waste. Not only municipal and non-hazardous industrial solid wastes, but also hazardous wastes from industrial and household activities are disposed into unsanitary landfills. Moreover, medical waste from health-care establishments is also dumped at these sites. Approximately, 4100 tonnes/year of industrial solid waste and 430 tonnes/year of hospital waste are placed into landfill in the Çorlu region, although the additional quantity of illegally dumped waste could not be determined. It is clear that hazardous waste from industrial activities and medical waste present the most important sources of environmental pollution with potential adverse impacts on public health.

Uncontrolled disposal activities threaten the soil, air and water and the populations reliant on these resources. Air pollutants, generated as a result of fires at landfill sites, have adverse effects on human health. The absence of landfill liner systems means that disposal of solid waste often leads to soil and water pollution. In such cases, water percolating through the waste and dissolving various materials is considered the most serious environmental impact of landfill and causes surface and groundwater pollution. In Çorlu, although significant health problems relating to leachate contamination have not been reported, health hazards associated with groundwater utilization for public water supply are undoubtedly present.

The aim of this study is to evaluate water-quality risks posed by unsanitary landfill leachates to groundwater resources in Çorlu Town. The study presents a general overview of current solid waste management practices in Çorlu Town and provides data on solid waste and leachate composition. It considers the vulnerability of the local hydrogeological setting and general recommendations are highlighted to facilitate future protection of groundwater quality.

2. SOLID WASTE MANAGEMENT REGULATIONS AND LEACHATE CONTROL

In Turkey, municipal, medical and industrial waste products are managed according to the Solid Waste Control Regulation of 14th March 1991, the Medical Waste Control Regulation of 20th May 1993, and the

Hazardous Waste Control Regulation of 11th July 1993. These regulations address the principles of collection, transportation, recycling, reuse, recovery and disposal of municipal waste and more hazardous medical and industrial waste. The regulations state that waste products have to be disposed in sanitary landfills. Moreover, it is stated that hazardous waste has to be disposed of separately from non-hazardous waste, again in sanitary landfills. Leachate control practices are also well defined, stating that, in sanitary landfills, a leachate collection system should operate and percolation from the landfill should be prevented by a liner system.

3. METHODOLOGY

A literature-based desk study of the critical issues involved was undertaken. This included:

- Identification of current solid waste management and leachate control practices;
- Review of the current status of unsanitary landfills in Çorlu Town;
- Determination of the quantity and composition of solid waste;
- Characterization of leachate from unsanitary landfill;
- Evaluation of the results and recommendation of remedial measurements.

Some of the information used in this paper was from previous unpublished reports and studies of Çorlu Town and the Trakya region.

4. CASE STUDY SETTING: ÇORLU TOWN

4.1 Geographical Characteristics

Çorlu is in the Trakya (Thrace) region of northwest Turkey, at a longitude of 41°07'30" East and a latitude of 27°41'00" North. Çorlu is the fourth largest town in Tekirdağ Province and covers an area of approximately 991 km^2 at a mean altitude of 193 m. It receives annual rainfall of 545 mm. In terms of groundwater resources, Çorlu is the second-most prominent location for groundwater in the Trakya region. Unfortunately in terms of groundwater protection, industrial activities have increased rapidly in the area (CCIC, 1997).

4.2 Soil Properties

Properties of the soil underlying Çorlu were obtained from the General Directorate of State Hydraulic Works (Anon, 2001). Area soils have widely varying proportions of sand, silt, clay and gravel. Thicknesses of these deposits could be significant, for example: sand (0-14 m), sandy and silty clay (14-30 m), silty sand (30-88 m), silty, sandy and gravelly sand (88-124 m), sandy and silty clay (124-144 m), clayey, silty and gravelly sand (144-280m). Overall mean soil permeability for the region was estimated as 10^{-3}-10^{-4} m/sec, this being designated permeable to semi-permeable.

5. WASTE MANAGEMENT PRACTICE

5.1 Waste Generation

According to existing records, 170 tonnes of waste are collected daily in Çorlu (Anon, 2000a), and solid waste generation for the region has been calculated as 1.150 kg/day/capita. Approximately 4100 tonnes of industrial waste is disposed of annually (Anon, 2000b). The composition of municipal and industrial solid waste is given in Tables 1 and 2, respectively (Tinmaz, 2001). Solid waste from health-care establishments is approximately 1880 kg/day, 33% of which is medical waste, which comprises pathological and non-pathological waste, infectious waste, chemical and pharmaceutical waste, and sharp instruments waste, and 48% of which is municipal waste. The remainder includes recyclable waste, such as glass and packaging (Table 3) (Demircan, 2001).

Table 1. Composition of municipal solid waste, Çorlu.

Waste Component	Percentage by Weight	Waste Component	Percentage by Weight
Cardboard	2.40	Coloured Glass	3.30
Food and Garden waste	54.20	Paper	7.20
Iron	1.00	Plastic	1.50
Aluminium	2.00	Packaging	2.70
Nylon	9.40	Textiles	1.90
PET	2.30	Diapers	3.20
Clear Glass	3.00	Ash and Other	5.90

Table 2. Composition of industrial waste, Çorlu.

Waste Component	Waste Quantity (tonnes/year)	Industrial Waste %
Textile (cloth, cord, etc.)	1284.6	31.4
Metal	130.2	3.2
Plastic	1301.9	31.8
Paper, cardboard	592.4	14.5
Wood	573.9	14.0
Sludge	213.1	5.20
TOTAL	4096.0	100

Table 3. Composition of medical waste, Çorlu.

Type of establishment	Number	Bed Capacity	Waste Quantity					
			Medical		Glass		Municipal	
			kg/d	%	kg/d	%	kg/d	%
Government Hospital	1	216	162	25.9	89.6	26.5	231.1	25.3
Social Security Est. H	1	-	-	-	-	-	-	-
Military Hospital	1	600	450	71.9	249	73.5	642	70.1
Dispensary	11	-	9.35	1.5	-	-	11.99	1.3
Doctor's Office	88	-	-	-	-	-	-	-
Dialysis Centre	1	-	-	-	-	-	-	-
Dentist's Office	48	-	4.80	0.8	-	-	30.2	3.3
Pharmacy	53	-	-	-	-	-	-	-
TOTAL		816	626.15	100	338.64	100	915.33	100

Industrial, agricultural, educational, governmental, health care and household activities all produce hazardous waste. Any hazardous waste generated can be managed either on- or off-site for treatment, disposal or recycling, typically through a commercial hazardous waste facility. The use of steel, aluminium, plastics and other components in automobiles and household items generates abrasives and oils. The coating of goods with attractive and protective finishes generates cyanides, solvents, concentrated acids and paint sludge. Medicine production creates organic solvents and other diverse residues, some of which contain toxic metals, whilst textile production generates heavy metal solutions, dyes and solvents. Sludges can also be produced as a result of waste water treatment, the controlling of air emissions, and from the treatment or recovery of other hazardous waste (LaGrega *et al.*, 1994).

In 1999 there were 398 industrial establishments in Çorlu, a number that has increased subsequently (Anon, 1999). 72% of the establishments were textile industries, which generate waste consisting largely of hazardous paints, solvents and oils. Other industrial facilities included food, metal, paper and chemical industries.

A wide range of household products, when discarded, have the characteristics of hazardous waste. Pesticides, paint products, household cleaners, hobby chemicals and automotive products frequently contain hazardous waste substances (LaGrega *et al.*, 1994). The quantity of hazardous waste in manufactured solid waste varies from 0.01 to 1% by mass, with a typical value of 0.1% (Tchobanoglous *et al.*, 1993). It is also known that medical waste constitutes a large part of the hazardous waste produced.

5.2 Waste Management Practice

In Çorlu, unsanitary landfill is the only solid waste management practice used. The management of solid waste in the city is the responsibility of the local authorities who regulate collection, transportation and disposal. A kerb-side collection method is used for solid waste, with the waste then transported to the landfill site by vehicles belonging to the Municipality. Recycling practices are limited to that of salvagers, There are no data available on their number, or on the quantity of solid waste salvaged. Waste is disposed of, spread and compacted in an uncontrolled manner and cover material is not regularly applied. All types of waste, whether municipal, industrial or medical, are disposed of together. There are no provisions for dust control and there are no leachate or landfill gas collection systems at the landfill sites (Tinmaz, 2001).

There are two unsanitary landfill sites in Çorlu. The first was opened in 1996, but has reached its total capacity. A second site was hence constructed in 2004. Although the Solid Waste Control Regulations stipulate that the distance between sanitary landfill sites and any residential areas has to exceed 1 km, both of the unsanitary landfill sites in Çorlu are less than 1 km from the closest residential area. In this study only the older of the two unsanitary landfill sites was evaluated. The area and total landfill storage volume of that site is 20,000 m^2 and 58,400 m^3 respectively.

5.3 Leachate Characterization

Characterization of the leachate generated at the older unsanitary landfill site in Çorlu Town was taken from a previous study (Töre *et al.*, 2001) and are summarized in Table 4. The leachate with a high BOD_5/COD ratio (0.66) indicated high biodegradability, suggesting that the leachate was from a young landfill. A NH_4-N/TKN ratio of around 0.95 indicated that most of the nitrogen was in the form of ammoniacal nitrogen.

Municipal, industrial and medical waste products are co-disposed (together). Leachate originates from the decomposition of these wastes in contact with any infiltrating water. Since neither legal regulations governing the management of solid waste, nor control practices for leachate are applied in Çorlu, leachate with a high pollution potential infiltrates the groundwater and also discharges into the Sinandede River.

Table 4. Leachate characterization of an unsanitary landfill site, Çorlu.

Parameter	Leachate Characterization in Çorlu	
	Range	Average
pH	8.31-8.47	-
COD (mg/l)	13957-15170	14447
BOD (mg/l)	9097-1016	9584
Alkalinity (mg$CaCO_3$/l)	6200-8810	7505
NH_4-N (mg/l)	923-2656	1193
TKN (mg/l)	985-2800	1262
TSS (mg/l)	528-996	688
VSS (mg/l)	184-218	201
SO_4 (mg/l)	1155-1888	1450
VFA (mg Acetic acid /l)	2842-5053	4072
Zn (mg/l)	0.00-0.18	0.11
Cr (mg/l)	0.00-1.02	0.63
Pb (mg/l)	0.00-0.87	0.46
Cd (mg/l)	0.00-0.00	0.00
Ni (mg/l)	0.00-0.145	0.138
Fe (mg/l)	1.08-4.23	2.46
Mn (mg/l)	1.73-3.49	2.64
Co (mg/l)	0.174-1.80	1.241
Cu (mg/l)	0.02-0.36	0.147

6. RESULTS

Potential pathways for contaminant migration at an uncontrolled waste site are precipitation, infiltration, migration and evaporation. Precipitation at the site either runs on or off, infiltrates the sub-surface, or returns to the atmosphere though evapotranspiration. Precipitation runoff that encounters waste or contaminated soil may transport contaminants or contaminated sediment into the surrounding environment. Precipitation may also infiltrate into and through the waste, generating leachate. Migration of leachate may introduce contaminants into the surface water and the groundwater environment. Contaminants present in leachate migrating into the groundwater environment may then be discharged in surface water. Groundwater and contaminant transport processes include advection, dispersion, absorption, retardation and biological and chemical transformation. Table 5 lists principle releases to surface and groundwater, contrasting the volume of the release, the concentration of contaminants and the types of factors affecting the nature of the release for the site (LaGrega *et al*., 1994).

Table 5. Release of contaminants to surface and groundwater from landfills.

Source	Volume of release	Contaminant Concentration	Type of Factor Affecting Release
Runoff	Possibly large, dependent upon rainfall event	Low; typically contaminated sediment; nil if landfill is capped	Integrity of cap, slope; storm water retention capacity
Surface seeps	Minimal rate; yet could continue indefinitely	Medium to high	Characteristics of cap (slope and permeability); disposal of liquids; removal of leachate
Leachate though base	Minimal to low rate for lined facility; moderate to high rate for unlined facility; continues indefinitely	Medium to high	(Same as above plus permeability of base)

Due to the lack of both liners and collection systems in the unsanitary landfills of Çorlu, leachate can easily infiltrate the soil and migrate to surface water and groundwater environments and is enhanced by the permeable - semi-permeable soils classification. Un-lined sites on such soils rapidly transmit leachate to the subsurface as generated. Leachate deriving from unsanitary landfill sites in Çorlu poses significant potential for infiltration into soil and migration into the groundwater. Leachate discharges to the Sinandede River are already evident. Because groundwater is the main source of drinking water, leachate migration poses a real threat to human health. This risk was also emphasized in the more regional report of the General Directorate of State Hydraulic Works (Anon, 2001). Five areas under threat of organic and inorganic pollution were determined, one of which was Çorlu. According to that report, the cause of pollution was the disposal of petrochemical, industrial and municipal waste in permeable sites.

7. RECOMMENDATIONS

It is clear that leachate from the unsanitary landfill sites in Çorlu poses a huge potential threat to groundwater quality. Suggested remedial and protection measures for preventing groundwater contamination by leachate can be summarized in two parts. The first of these is source control and the second is remediation of the landfill facility itself.

Source control practice is based on integrated solid waste management systems, consisting of separation at source, collection, sorting, recycling, composting and sanitary landfilling. A previous study of waste management systems for Çorlu (Anon, 2000b) suggested a system which encourages maximum recycling and minimum landfilling of municipal solid waste, including:

- separation and storage of wastes at source;
- collection of separately stored wastes;
- sorting of separately collected wastes;
- recovery/reuse of sorted wastes (solid waste occurring from Çorlu is suitable for composting).

The suggested management system would reduce the quantity of waste sent to landfill, and thus the quantity of leachate produced.

Unsanitary landfill is not a suitable method of waste disposal because of its hazards. The first step towards landfill remediation is the closure of unsanitary landfill sites and construction of sanitary lined and capped (engineered) landfill sites. Hazardous wastes and medical wastes should be disposed of at properly regulated sites whilst unsanitary landfill sites should be remediated. Further local ground and groundwaters investigation and supporting research area required in order to determine which remediation strategy is most appropriate to the unsanitary landfill sites of Çorlu. Without such, such landfills represent a significant threat to the highly valued groundwater resources of the area.

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AGRICULTURAL INFLUENCES ON GROUNDWATER USED FOR WATER SUPPLY IN THE CAUCASUS MINERAL WATER REGION

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Abstract: Assessment and mapping of groundwater vulnerability remains foundational to the effective protection of groundwater resources. A combined approach for assessing the vulnerability of groundwater to different types of contamination is suggested. Preliminary results of a groundwater vulnerability assessment for one of the most hazardous pollutants, the organochlorine pesticide lindane, are given. These calculations were based on typical piedmont territory in the Caucasus Mineral Water region, which is characterized by complicated geological and hydrogeological conditions.

Key words: Caucasus; vulnerability of groundwater; agricultural contamination; pesticides; lindane; mineral waters; Russia.

1. INTRODUCTION

In many parts of the world groundwater is the most valuable, ecologically safe and sometimes the only source of potable water supply. Significant anthropogenic impact on groundwater environments, especially shallow aquifers, may however render groundwaters unsuitable for potable water. Deterioration of groundwater quality is observed especially in industrial regions and areas of intensive agricultural development.

When assessing groundwater as a source of potable water supply it is important to remember that groundwater is not only a mineral resource, but also a component of the environment. Any change in one component of the environment (e.g. atmospheric precipitations, river runoff) causes changes in the groundwater regime and vice versa. Thus, the problem of

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groundwater contamination is closely connected with the problem of total environmental protection.

Many potential sources of groundwater contamination are connected with agricultural activity. In such cases the most dangerous sources of groundwater contamination are different mineral fertilizers and chemical pesticides and also waste from cattle-breeding complexes. Groundwater and surface waters are most commonly impacted by nitrates and, or pesticides.

Pesticides can contaminate aquatic systems not only from legitimate application to fields, or transportation routes, but also fallout from aerial sprays, soil erosion, or through the disposal of pesticide containers or effluent from pesticide factories. Water-soluble pesticides may pose a significant threat to the water environment, particularly those that are low sorbing and also of long half-life. Such persistent pesticide may pose significant threats to groundwater supplies and aquatic ecosystems. Assessment and mapping of groundwater vulnerability remains foundational to the effective protection of groundwater resources (Pashkovskii, 2002). The focus of this study is groundwater vulnerability to pesticide contamination in the Caucasus region of the Russian Federation (Karimova, 2003).

2. THE RESEARCH AREA

The Caucasus Mineral Water region is situated in the middle of the seven-hundredth isthmus in the south of the European part of the Russian Federation, between the Caspian and Black Seas. Occupying an area of 6000 km^2, the region is characterized by highly contrasting natural conditions. It is a typical piedmont territory, characterized by high relief in the south and southwest and flat areas in the north and northeast. A volcanic area with isolated laccoliths within an otherwise almost flat area is a notable feature of the region.

The climate of the area is influenced by several factors. The mountainous character of the region and the proximity of the Main Caucasus Range on the one hand, and the proximity of the steppes and semi-arid land of the Caspian Sea depression on the other, determine the continental character of the climate that varies in a northeast – southwest direction. The climate in Pyategorsk (altitude of 576 m) for example, is very continental, in Kislovodsk (altitude of 890 m) it is continental, and in areas with altitudes of about 2500 m it is transitional.

The region is also characterized by irregular distribution of surface water. In the mountainous areas, for example, numerous mountain streams are present, but in the plains numerous irrigation canals complicate the area's hydrologic network. The drainage net of the region is represented by three main rivers flowing into the Caspian Sea basin: Kuma in the north, Podkumok in the south, and Surkul in the northwest (Figure 1). River recharge is formed mainly by precipitation (average annual precipitation amounts to 500-600 mm) and to a lesser extent by groundwater. Moreover, recharge of the rivers Kuma and Podkumok in their upper catchment is formed by snowmelt in mountain areas.

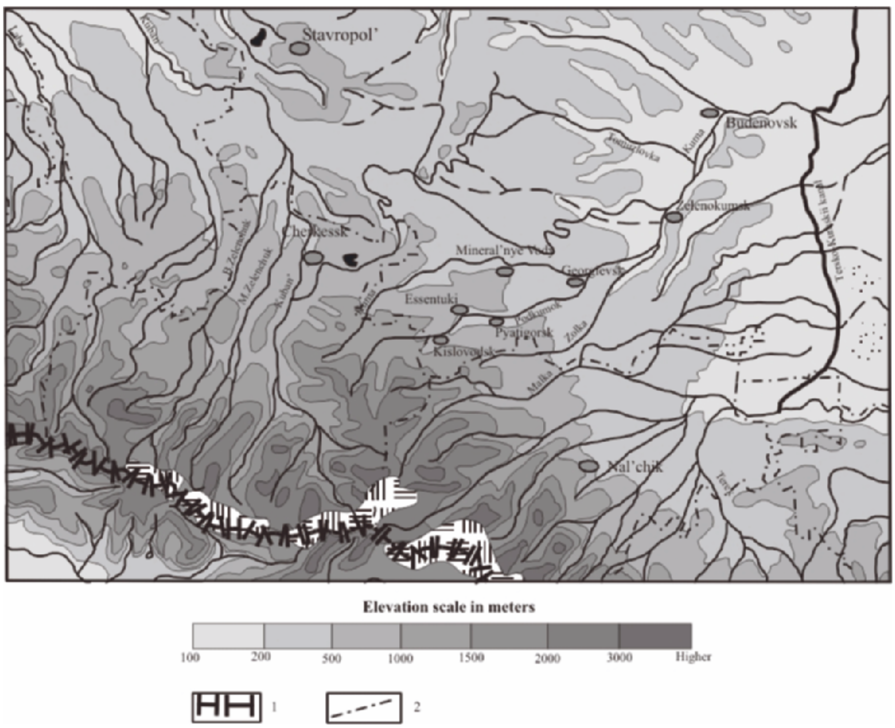

Figure 1. Geographical map of the Caucasian Mineral Water region. 1 = State boundary of the Russian Federation; 2 = boundaries of administrative regions.

Aquifers within Middle and Upper Quaternary deposits are mainly used for water supply of the main cities of the region and some small towns. These aquifers occupy the second, third, fourth and fifth terraces above the floodplain of the rivers, and are composed of gravels with sandy loam intercalations or, in some areas, of clays, loams and sandy loams. The depth to groundwater varies from ground surface to 14 m, and the aquifer thickness varies from 0.5 to 15 m (Figure 2).

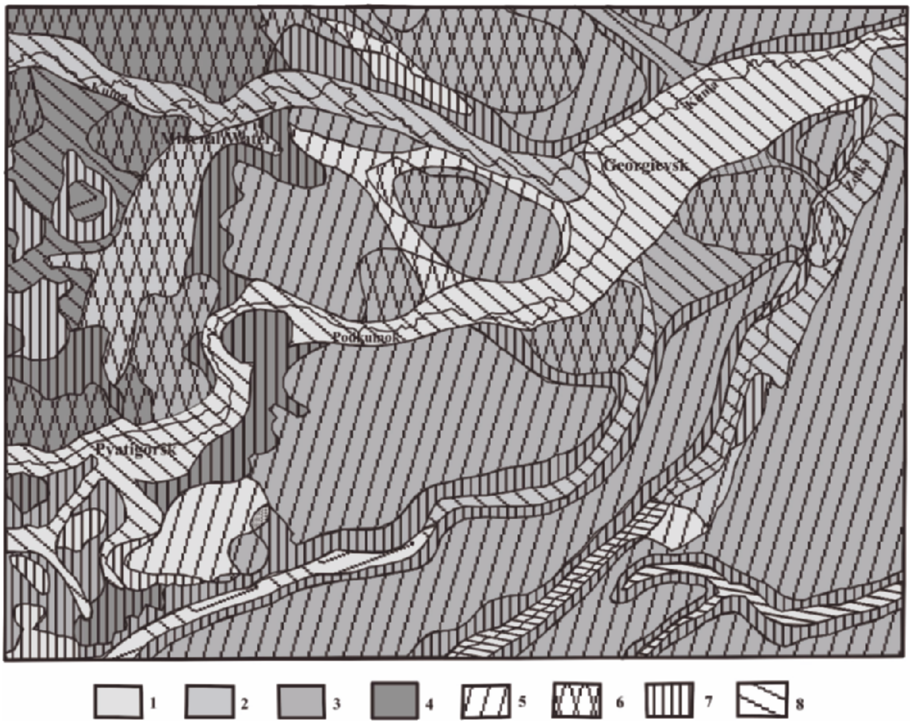

Figure 2. Characteristics of aquifers closest to the surface, Caucasian Mineral Water region. Key: lithological composition (1-4): 1 - gravel, coarse gravel with sandy aggregate; 2 - alternation of clay, loam, sand, and fragmental material; 3 - alternation of loam, sandy loam, clay; 4 - clay, seldom heavy loam; aquifer thickness in m (5-8): 5 = 0-5; 6 = 0-5 (areas of flooding through anthropogenic activity); 7 = 5-10; 8 = >10.

The sources of groundwater contamination in the Caucasian Mineral Water region are summarized in Fig. 3. As noted above, many of these sources are by-products of agricultural activity, with waste from cattle farms comprising 72% of the total contaminants. Pesticides and fertilizers make up a further 11%, with domestic and industrial waste being responsible for only 5% of groundwater contamination. Since shallow groundwater is the main source of potable water to towns and villages in the region, contamination poses a major threat to water supply. Furthermore, many areas of groundwater recharge are situated in areas of intense agricultural activity, further heightening the risk.

Contamination risk is also high in areas where the mineralized waters are used by spas, because they too are dependent on groundwater, which is part of a complex set of geological and hydrogeological conditions.

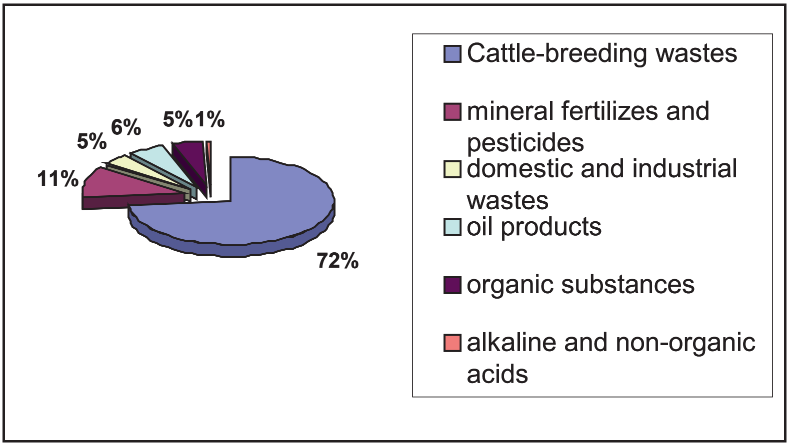

Figure 3. Potential sources of groundwater contamination in the Caucasus Mineral Water region.

3. METHODOLOGY OF GROUNDWATER VULNERABILITY ASSESSMENT AND ITS PRACTICAL APPLICATION

To assess groundwater vulnerability in the region, contaminant behaviour was studied in all parts of the protective zone. The widely known and used methodology DRASTIC was implemented as the foundational approach (Aller *et al.*, 1987). Parameters used for vulnerability assessment (such as the impact of the vadose zone, and soil layer and aquifer properties) were characterized quantitatively, and time of contaminant penetration towards water table used as a criterion for determining areas with different vulnerability.

3.1 Thickness of the Vadose Zone

This parameter characterizes depth to groundwater level and determines the thickness of rocks through which contaminants migrate (Figure 2). Additionally, it influences the different physical-chemical processes taking place in the vadose zone. Most notably, oxidation is a key process that may be enhanced in the uppermost layers through with pollutants migrate.

3.2 Infiltration Recharge

This parameter can be determined from the simple equation:

$$W = P - (E + R)$$

where P is atmospheric precipitation; E is evapotranspiration; and R is runoff (all expressed in similar units, e.g. mm/y).

On average, the value of atmospheric precipitation for the region is taken to be 600 mm/y. According to long-term observations, the surface evaporation rate is 500 mm/y. Hence, the calculated value of infiltration was equal to about 100 mm/y. It should be noted that surface runoff value was not taken into account because of the difficulties in its estimation.

3.3 Aquifer Structure

This parameter includes the lithological composition of the aquifer and the properties of those lithologies (Figure 2). The latter of these can be determined by laboratory methods or taken from the literature. The specific parameter included in this section is the distribution coefficient that determines sorption properties of the different rocks. However this parameter is quite difficult to determine in field experiments, so is best estimated using laboratory methods or taken directly from the literature.

3.4 Soil Layer

In the strict sense, soil is the upper layer of the terrestrial part of the earth crust, mainly consisting of weathered minerals and organic substances usually confined in depth to one or a few metres, on which plants are growing and organisms are living. Its filtering properties are very important in assessing groundwater quality because it influences a great number of physical-chemical processes (e.g. sorption, precipitation and microbial decay), the rate of contaminant penetration towards groundwater and, hence, the vulnerability of groundwater.

If discussing the migration of a non-sorbed conservative pollutant (tracer), its migration time can be determined by the simple equation:

$$t_s = t_{sor} = (m_s RFC)/W$$

where R is the retardation factor, determined as $R = 1 + \left(K_d^{\,s} \rho/\theta\right)$ where ρ is dry density and θ is moisture content; FC is field capacity; W is groundwater recharge; and $K_d^{\,s}$ is the soil distribution coefficient.

3.5 Topography

This parameter is characterized by slope elevation and is determined on the base of morphological zoning. Moreover it is closely connected with surface runoff, and, hence, with infiltration recharge.

3.6 Influence of Aeration Zone

As an influence on the soil layer, this factor is of great importance in assessing groundwater vulnerability. This is connected with the fact that organic soil matter can sorb some pollutants partly or totally. With infiltration flow, the residual part can penetrate the vadose zone and reach groundwater level. As volumetric moisture content is the main parameter determining contaminant penetration time through the vadose zone, then the time of contaminant migration towards groundwater level in the aquifer closest to the surface can be determined using the equation:

$$t_{vz} = \left[m_{vz}\left(1 + K_d \rho/\theta\right)\right]/W$$

where m_{vz} is thickness of the aeration zone; K_d is the distribution coefficient; ρ is bulk dry density; θ is volumetric moisture content, and W is infiltration recharge.

The volumetric moisture content complicates determination of contaminant migration time because it varies with both geological-hydrogeological conditions and time of investigation. Hence, porosity is used as a proxy. In this case, contaminant migration time can be determined as:

$$t_{vz} = \left[m_{vz}\left(1 + K_d \rho / n\right)\right]/W$$

where m_{vz} is thickness of the aeration zone; and n is porosity.

However, it is necessary to note that received values of contaminant migration time are slightly overestimated and characterize the maximum permissible time of contaminant penetration through the vadose zone towards the underlying water table. Hence they provide a more reliable forecast of possible contamination.

3.7 Conductivity

By using this parameter in conjunction with another (e.g. pollutant toxicity, concentration), the risk of a pollutant reaching the source of groundwater use (e.g. well field) can be determined. This parameter can be calculated using the equation:

$$t_h = \left(n\Delta l^2\right)/k\Delta H$$

where n is porosity; Δl is the distance from the source of contamination and well field; k is the filtration coefficient; and ΔH is the head difference.

3.8 Prediction of Groundwater Vulnerability

Lindane migration times for different rock types and thicknesses in the Caucasus Mineral Water region are tabulated in Table 1. The estimated times clearly demonstrate the influence of greater thicknesses, and low permeability more sorbing (high organic carbon, fine particle size) rock types causing increased migration times. For a given thickness, the rock type may cause over an order of magnitude difference in migration time.

Table 1. Lindane contaminant migration times for different rock types and thicknesses in the Caucasus Mineral Water region.

Rock type	Thickness (m)	Migration time (years)
Gravel, coarse gravel with sandy aggregate	0-5	0-13
	5-10	13-25
	>10	> 25
Layers of clay, loam, sand and fragmental material	0-5	0-28
	5-10	28-56
	>10	>56
Layers of loam, sandy loam and clay	0-5	0-103
	5-10	103-205
	>10	>205
Clay, occasionally heavy loam	0-5	0-119
	5-10	119-238
	>10	>238

Lindane migration times to the water table calculated spatially over the region have allowed the predicted groundwater vulnerability to be shown across the region (originally depicted in Figure 2) in Figure 4. Vulnerabilities are classified into three categories. The most vulnerable areas tend to occur along valley areas where depths to groundwater are low.

Close comparison of Figures 2 and 4 also confirms geological controls on groundwater vulnerability.

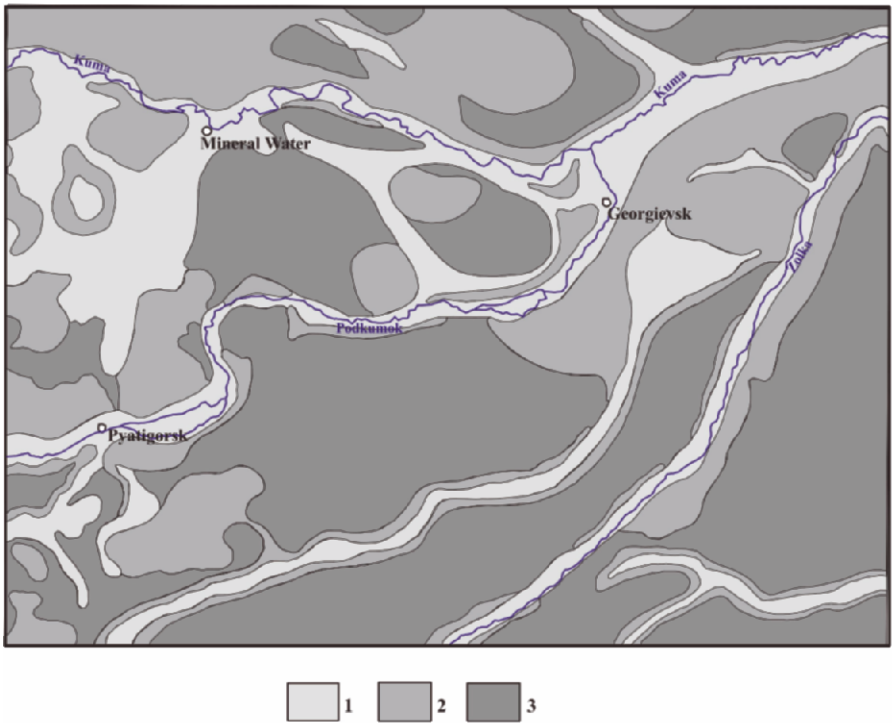

Figure 4. Schematic map of groundwater protection from pollution by lindane pesticide. The Categories of groundwater vulnerability are: 1 – vulnerable; 2 – partly vulnerable; 3 – non-vulnerable (very low vulnerability).

4. CONCLUSIONS

The combined methodology suggested is suitable for both regional and local investigations because it combines the advantages of DRASTIC with a method of quantitative assessment of contaminant penetration through the aeration zone towards the groundwater. Moreover, it reflects contaminant behaviour in all parts of the protective zone (soil layer and aeration zone) and also shows the possibility of a pollutant reaching the source of water withdrawal.

The results of the quantitative assessment and mapping of the vulnerability of groundwater to contamination have an important practical meaning. They can be used:

- for developing strategies of groundwater use and protection in regions with different intrinsic vulnerability;
- for designing and planning large industrial and agricultural centres containing hazardous waste sites;
- for determining schemes of groundwater use for potable water supply and irrigation and for well-field location;
- for forecasting groundwater quality in terms of anthropogenic pollution;
- for determining different protective measures hydrogeologically;
- for determining suitable sites of waste accumulation and storage.

The main practical result of such assessment and mapping of groundwater vulnerability is the possibility of comparing different areas based on groundwater protection. Consequently, it would be possible to identify which territory is better-protected from contamination, which areas are unsafe as well locations for the supply of groundwater for drinking water, and where protective measures are more necessary.

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ENVIRONMENTAL PROBLEMS ASSOCIATED WITH UTILIZATION OF MINERAL WATERS IN URBANIZED AREAS OF AZERBAIJAN

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Abstract: Azerbaijan is rich in oil and gas, deposits of various minerals, and mineral waters. Thermomineral waters, present in 5 regions, are very important economically: their occurrence, properties, and environmental impacts are reviewed. Overall, it is concluded that Azerbaijan has a very considerable potential for further development of its thermal waters; although this development will necessitate care if environmental problems are to be avoided, its use could save about 1.5 million tonnes of conventional fuel every year.

Key words: thermomineral waters; geothermal waters; iodine; bromine; arsenic; salinity; Azerbaijan; Caucasus; Absheron Peninsula; Nakhchivan

1. INTRODUCTION

The Republic of Azerbaijan is very rich in natural resources. As in any country, a sustainable economy and increasing population is directly dependent on such resources. Within Azerbaijan, more than 200 areas of mineral waters have been investigated with a potential discharge of over 100 million l/day. These waters have various ionic and gas compositions, many with therapeutic qualities, and display a diverse range of temperatures. The exploitation of these mineral resources has proved economically viable, and many industrial plants, producing iodine, bromine, boron, arsenic, liquid carbon dioxide, various salts, and other products, have been constructed. The heat content of the waters could also be used in various spheres of the national economy. In Azerbaijan more than 50 million m^3 of thermal waters are potentially available, e.g. at Istisu, Alasha, Arkevan, and Shikhovo, with maximum temperatures of up to ~95^o C. The heat generated is equivalent to that obtained by burning about 20

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thousand tonnes of coal. The potential of these waters is therefore huge, as they contain an inexhaustible amount of heat energy which could be successfully used for heating sanatorium-resort complexes, populated areas, medical and communal buildings, and sports venues, and for the supply of hot water and electricity.

This article summarizes data from all the thermal water fields of Azerbaijan, including their geological and hydrogeological settings, water chemistry, temperatures, and estimated resource potential. Using this information, the potential for utilization can be assessed.

2. COUNTRY-WIDE OVERVIEW

Azerbaijan, occupies an area of 86,600 km^2 on the west coast of the Caspian Sea between the mountain chains of the Greater Caucasus and the Lesser Caucasus / Talish mountains. To the south it is bordered by the River Araz and Iran and Turkey. Azerbaijan is rich in oil and gas, deposits of various minerals, and mineral waters.

Internationally, the use of alternative sources of energy has become popular because of the perceived advantages from an environmental / ecological point of view and because sources of cheap hydrocarbons are being exhausted. Most of the advanced European countries have been using wind and heat, and later geothermal energy, since the Middle Ages. Utilization of wind energy is developing in Azerbaijan now. Today alternative energy sources (including water energy) forms 10-12% of the total energy balance of the country, and the proportion is increasing every year. Azerbaijan is rich in alternative sources of energy, especially geothermal resources.

The Republic is intersected by many major faults and many thermomineral waters are associated with these features. The characteristics of the tectonic structure of Azerbaijan allow us to consider the hydromineral resources on a large scale, covering the Greater and Lesser Caucasus and the Talish region.

Within Azerbaijan there are about 200 mineral water fields (Figure 1) with a total operational reserve estimated to be more than 100 million l/day. These fields contain waters which possess diverse ionic and gas contents, are reputed to have therapeutic properties, and have various temperature regimes.

The geological history of the region has been the controlling factor in the development of the thermomineral water fields, and in the resulting thermal and chemical character of the waters of these fields. Using this dependence, prospective thermal areas can be delineated, and these are shown in Figure 2.

Figure 1. Schematic map of the hydrothermal areas of Azerbaijan.

Figure 2. A schematic map of zones in the Republic of Azerbaijan where thermal waters might be utilized: 1. Prospective zones for exploration for thermal waters. 2. Local, unexplored areas. 3. Non-prospective zones.

The thermal waters of Azerbaijan are divided into 3 categories: subthermal – 20 to 37°C; thermal – 37 to 42°C; and hot – 42 to >100°C (the latter being rarely met). The total estimated thermal water resource is 245,000 m^3/day.

Five thermal water-bearing complexes are found in Azerbaijan: Absheron, Productive layer, Maikop, Eocene and Cretaceous.

3. REGIONAL DESCRIPTIONS

3.1 Introduction

For the current purposes, Azerbaijan can be divided into 7 regions: the Greater Caucasus; the Lesser Caucasus; the Samur-Devechi piedmont; the Absheron peninsula; the Cura depression; and the Nakhichevan and Talish mountains.

3.2 The Greater Caucasus

The Azeri part of the Greater Caucasus is rich in mineral waters. 25 of the 100 studied mineral waters fields are thermal, including those of Haltan, Jini, Ilisu, Qumerukh, and Bin. The thermal waters of the Greater Caucasus in the main are found in deposits of Jurassic and Cretaceous age, and in the "Productive Layer". The temperature of the waters fluctuates between 22 and 50°C, the salinity of the water is not high, and the estimated discharge is about 2000 m^3/day. The waters are sodium-bicarbonate in dominant composition. Methane is present, produced by biochemical processes.

3.3 The Samur-Devechi Region

The Samur-Devechi piedmont (Figure 3) is especially rich in its thermomineral waters. The mineral waters of the area in the main are associated with the Jurassic and Cretaceous deposits, and with the Productive Layer. There are more than 50 mineral water springs even in Kuba alone, including Ansar, Amirkhanli (81°C), and Talabi Kine (59°C). In the Devechi area there are also more than 50 springs, more than 25 of them being thermal. The total discharge of the thermal waters of the Samur-Devechi piedmont is about 21,700 m^3 /day. Depending on depth, the temperature of the thermal waters reaches 30-97°C (e.g. Sovetabad 125°C, Khudat 140°C). The chemical composition of the thermal waters is very interesting. In the main they are sodium-bicarbonate in type, but they contain high concentrations of iodine and bromine. In addition to the natural springs, thermal waters have been encountered in the wells drilled to extract oil: at different depths, temperatures ranged from 30°C to 140°C. Discharge of these waters reaches 425 m^3/day at most (Tagiyev, 2001).

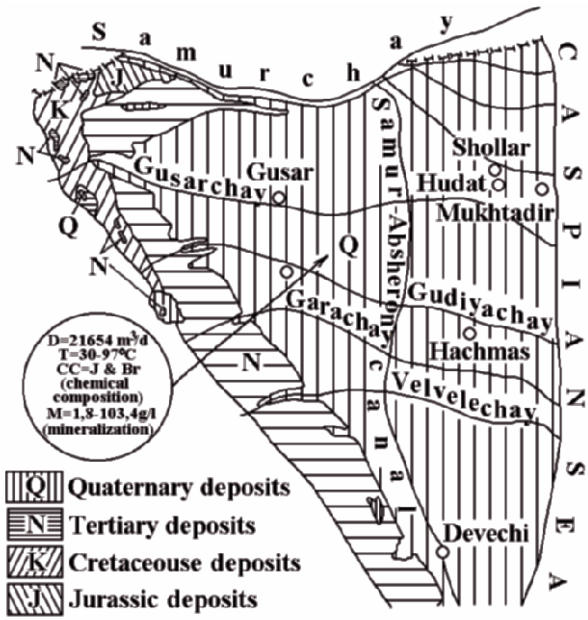

Figure 3. Geological map of Samur-Devechi piedmont.

3.4 The Absheron Peninsula

The Absheron peninsula (Figure 4) is closely connected with the Greater Caucasus structurally, and occupies the southeastern periphery of it. Mineral waters of the Absheron peninsula are associated with various stratigraphic horizons. Three main groups of mineral waters are found in the area according to the classification used by the mineral water clinics: waters without specific dominant components; sulphide-rich waters; and iodine/bromine-rich waters. The thermal water found in Shikhov is methane-rich. Discharge is about 500, 000 l/day, and the temperature is 68°C. In one well (No. 15) in Jiloy island, water rich in methane, iodine-bromine, chloride, hydrocarbons, boron, silica, sodium, and calcium was found, with a total mineralization of 12-13 g/l. The temperature of this water reaches 40°C, and the (pumped) discharge is 60 m^3/day. The total discharge of the thermal waters of Absheron peninsula is 20,000 m^3/day, at temperatures from 20° to 90°C: these waters contain 1-10 g/l. The waters are localized (Aliev, 2000).

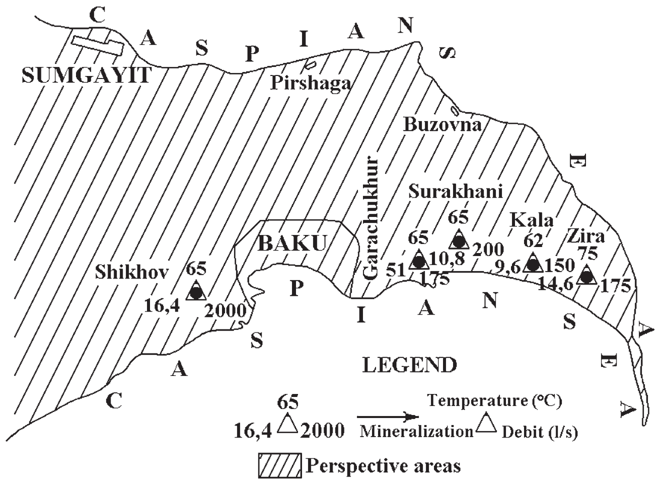

Figure 4. Thermal waters of the Absheron peninsula (mineralization in g/l; debit = discharge).

3.5 The Lesser Caucasus

The Azeri part of Lesser Caucasus is situated between two rivers, the Cura in the northeast and the Araz in the southeast. Most of the mineral water fields are associated with tectonic structures. Mineral water springs are very common in the central part of the Lesser Caucasus. Carbonate rocks control the chemical composition of the waters. The thermal waters are associated with the magmatic centres of Paleogene-Neogenic age and active volcanic manifestations of Pliocene and Quaternary. The waters with the highest temperatures occur adjacent to the Celbajar zone. They are connected with the magmatic activity of the most recent volcanism. About 80 fields have been found in this area; their temperature ranging between 32 and 74°C. Mineralization of the waters is not high. The fields are localized, and the total estimated discharge of the area is 4170 m^3/day.

3.6 The Nakhchivan Fold-Belt

The Nakhchivan fold-belt occupies the southern part of the Lesser Caucasus. The mineral water fields are concentrated in the western and central parts of this zone, where Paleogene-Neogene and Upper Cretaceous igneous and sedimentary rocks are present. Two hydrochemical areas have been distinguished: Central Nakhchivan, and Ordubad-Araz. The water of this field has been characterized using samples from well No. 10. It is CO_2-, B-, and As- rich; temperature reaches 52°C, and the total discharge is 6825 m^3/day. Mineralization of waters varies from 2 to 35 g/l. The high concentrations of B and As are connected with the contemporary volcanism (Alikuliyev *et al.*, 2003).

3.7 The Cura Depression

The thermal waters of the Cura depression are found in the northwestern and central parts of the area. There is a great quantity of thermal water, which is present nearly everywhere in the area at a depth of about 300 m. These are matrix and fracture systems of Quaternary and older ages from the Alazan-Avtoran intermontane zone and the Cura-Araz lowland. The temperature of the waters varies between 20 and 100° C. They possess high concentrations of iodine and bromine, and have salinities of between 5 and 192 g/l.

3.8 The Talish Region

The Talish region (Figure 5) occupies the extreme southeast part of the Republic of Azerbaijan. Topographically, the region is dominated by the Lenkoran lowland and the Talish fold-belt. Structural features include the Astara and Boorovar anticlinorium (which consists of Eocene volcanic complexes) and the Southern Mugan uplift in Talish area. Thermal waters are found across the whole area. About 200 mineral water fields have been discovered. High concentrations of mineral springs are observed along the lines of faults. Two groups of mineral waters have been identified. One, which is situated in the eastern part of the area, is represented by the Masalli-Lenkoran-Astara line of thermal springs: the other is represented by the Yardimli-Lerik group of cold-water springs. Temperature varies between 36° and 95°C. The discharges are large. The Masalli group of thermal springs in the north of the area are highly-mineralized (to 17 g/l), with temperatures reaching 66°C. Chemically, they are of chloride or sodium-chloride type, and contain methane and hydrogen sulphide. In the south, the Astara group of hot springs (38°-50°C) yield highly mineralized waters. Borehole have located the presence of waters from the Eocene sequence with a temperature of 39°C, a discharge of 81-518 m^3/day, and salinities of 9.9-38.2 g/l. Total discharge of the whole area is 22,300 m^3/day (Babayev, 2000).

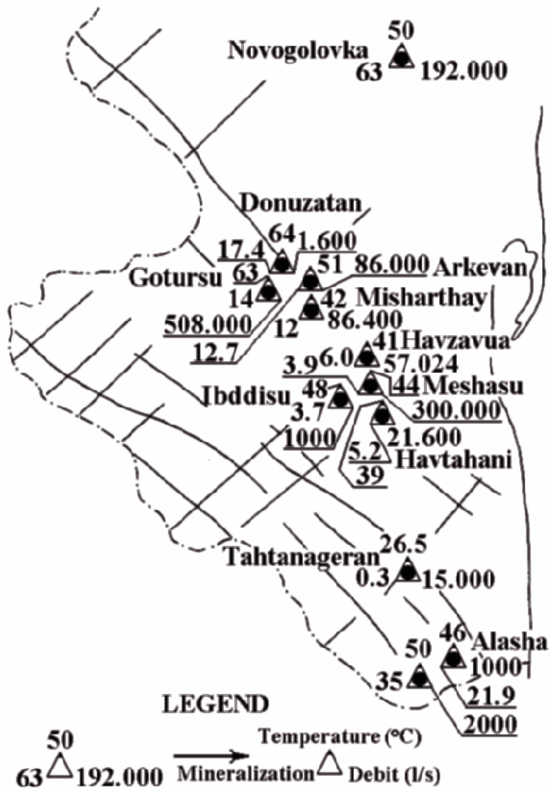

Figure 5. The thermal waters of the Talish fold zone (debit = discharge).

4. DISCUSSION

Compared with some other countries, the geothermal resources of Azerbaijan are relatively little studied; they are also relatively little used, and then only for thermal/mineral water resorts (for example, those of Istisu, Shikhovo, Meshasu, and Alasha). Investigating the thermomineral water fields has great practical importance, because besides their use in the sanatorium-resort industry, they can be used in the chemical industry and, most importantly, as a source of heat energy. About 50 million litres of thermal waters are discharged every day to the ground surface: assuming a maximum temperature of about 75°C, this discharge carries a heat load equivalent to that which would be got by burning about 200 thousand tonnes of coal. Thus the potential energy resource is very large and effectively inexhaustible. Such energy could be used for heating of sanatoria and resorts, populated areas, medical and sporting facilities, and municipal buildings, for hot water-supply, for flower and early vegetable growing, and for electricity generation.

The present limited knowledge of the geothermal systems and the economic state of the republic has hindered development of these resources and serious investment in hydrogeological investigation is required in order to realize the potential. Example issues which need to be resolved include:

- methods for re-injecting highly-mineralized waters after they have been used into deeper systems in order to avoid attendant environmental/ ecological problems;
- predicting and reducing the effects of fall in temperature and change in chemical composition during transport of the waters;
- managing the systems so that premature exhaustion is avoided.

Thus a concrete programme of research should be devised.

5. CONCLUSIONS

Reviewing what is known of the thermomineral waters of Azerbaijan permits the following inferences to be made:

1. Efficient utilization of the thermomineral resources to obtain electrical energy, and to supply heating will require further hydrogeological investigation.
2. Localized areas within the Greater Caucasus could be utilized independently.
3. The thermal waters of the Absheron peninsula are currently used only in sanatoria, but they are suitable for other uses as well.
4. New technology for removing chemical elements from these waters should be devised, so that the environment can be protected, as most thermal waters discharge in urbanized areas.
5. Special attention should be paid to the highly mineralized thermal waters, as they are the more aggressive and after their utilization they must be pumped into deeper horizons to avoid environmental / ecological impacts.
6. It is recommended that foreign experience in thermomineral water development also be studied.

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CONCLUSIONS FROM A NEGATIVE TRACER TEST IN THE URBAN THERMAL KARST AREA, BUDAPEST, HUNGARY

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Abstract: To prove the hydraulic connection between the urban Rózsadomb recharge area (Buda Thermal Karst System, Budapest, Hungary) – through its hydrothermal inactive caves – and the springs at the foothills has been an important question since the 1980s. These cold and lukewarm springs have been utilized as thermal baths since Roman times, and in modern times, occasionally, chemical and bacterial contaminants of human origin have been detected in them. It is hence of considerable importance to know whether these contaminants originated from the Rózsadomb recharge area or from close to the discharge points. According to the results of a previous test in a cave passage high upon the hill, it was tempting to suppose direct connection between the Rózsadomb area and the Boltív Spring in the foothills. The expected breakthrough-time predicted from this previous study was between 10 hours and 42 days. The tracer test documented in this present work showed that no breakthrough occurred. It is proposed therefore that the risk of the Boltív Spring being contaminated by infiltration in the Rózsadomb recharge area is low as compared with the possible contamination from sources near the discharge area. We suggest that in this case the negative tracer test clearly helped to prove and to understand better the role of "natural attenuation" in the Rózsadomb area. The efficient dilution observed in the tracer test facilitated the prediction of a hitherto unknown, large, phreatic cave-system as well.

Key words: thermal springs; geothermal waters; karst; tracer test; Tinopal CBS-X; Budapest; Hungary; bacterial contamination; springs

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1. INTRODUCTION

The internationally appreciated thermal bath culture of Budapest has developed because of the particular hydrogeological setting of the town, i.e. the cold, lukewarm, and hot karst springs arising near to each other along the River Danube. Nevertheless our knowledge of the operation of the Buda Thermal Karst System is far from complete. It is based on conceptual models of Schafarzik (1928), Vendel and Kisházy (1964), Alföldi (1978, 1981), and Kovács and Müller (1980). Geographically defined recharge areas cannot be properly related to the springs and hence qualitative and quantitative determination of protection zones for the springs is difficult. There is good reason to think that the cold-water of the karst system is being partly recharged within the urban area of Budapest, including the Rózsadomb, one of the hilly "villa" quarters of the town with an extent of 4 km^2 (Hazslinszky *et al*., 1993). During the 20th century several inactive hydrothermal caves of total length 35 km were discovered beneath the villas of Rózsadomb at a depth of only 10 to 30 m below ground surface (Takács-Bolner and Kraus, 1989; Leél-Őssy, 1995). The Molnár János Cave, the entrance of which is situated in the foothills, is the only known active hydrothermal cave within the area of Rózsadomb. Exquisite mineralization is associated with these caves and their situation at shallow depth beneath a densely populated urban area strongly suggests that the possibility of their protection against contamination should be investigated.

The aim of the present project was to acquire direct information with the help of a tracer test on how the Buda Thermal Karst System works between the Rózsadomb and its discharge area. The results of the experiment can be considered as a first step towards setting up a protection policy for these urban caves and springs, specifically based on the 3D hydraulic response model of the area.

2. HYDROGEOLOGICAL BACKGROUND AND LAND USE

Hydrologically, the Rózsadomb area, located above the central part of the Buda Thermal Karst System, belongs to the groundwater system of the Transdanubian Central Range (TCR) (Alföldi, 1979, 1982a, 1982b) (Figure 1). The aquifer is a several thousand metres thick Mesozoic carbonate sequence. Though karstic recharge plays a role in about 15% of the whole area of TCR only (Lorberer, 1986), it feeds a complex geothermal flow system with perennial discharge areas. Infiltrating cold water is heated up because of thermal convection of upward groundwater flow, which is characteristic for the TCR (Alföldi, 1982a, 1982b).

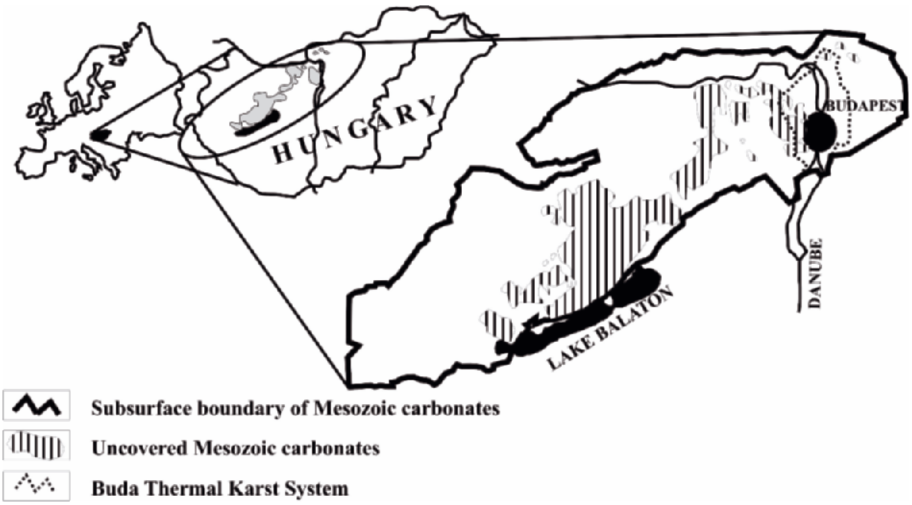

Figure 1. Location of the Buda Thermal Karst system in the Transdanubian Central Range.

After following a deep, regional-scale flow path, water returns to the surface as thermal water. ^{14}C measurements show that the age of this water is about 5,000–16,000 years (Deák, 1978). The rest of the infiltrated water discharges as cold or lukewarm springs or seepage via local or intermediate flow-systems.

The Buda Thermal Karst System is situated in the NE part of the TCR. Along with Mesozoic carbonates, Eocene limestones also serve as aquifers in this system. Thermal water discharge is localized here by the step-faulted boundary between the subsided basin to the east and the uplifted hilly range in the west. The course of the Danube follows this boundary and represents the base level of erosion, which is also the discharge site of the cold and lukewarm karst waters (Figure 2).

One of the hills near the Danube is Rózsadomb, which has relatively steep slopes towards the river. Its highest point is at 195 m asl. The base level is 104 m asl at the riverside, where springs of widely different temperatures arise. Surface karst phenomena are rare because of the geological and speleological characteristics of the thermal karst area. Since the 1920s human activities have completely changed this part of Budapest obscuring the "natural conditions". The original vegetation has been destroyed completely, and extensive construction works and limestone quarrying have modified the relief (Hazslinszky *et al*., 1993).

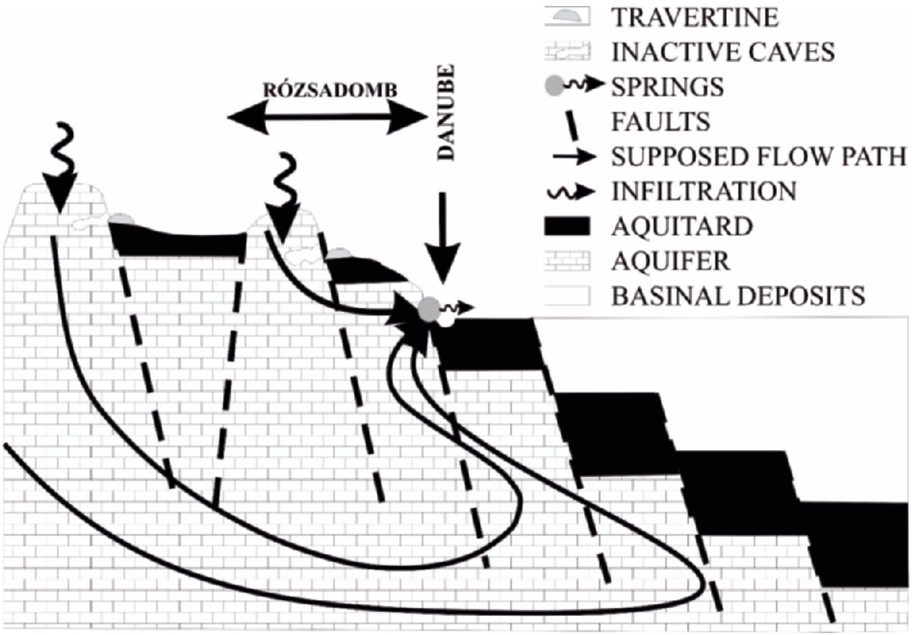

Figure 2. Hypothetical cross section of the Buda Thermal Karst (modified after Kovács and Müller, 1980).

Accelerated urbanization of the capital reached this area in the 1970s, and now more than 80% of the whole 4 km^2 area has been built over.

The infiltration is about 170 mm/y on the uncovered Triassic and Eocene limestone areas, whereas over the marly terrains this value is considerably less, around 35 mm/y (Hazslinszky *et al.*, 1993).

The original yield of the springs at the foothills of Rózsadomb is estimated as 34,000 m^3/d: by 1985 this had decreased to 12,000 m^3/d (Hazslinszky *et al.*, 1993). The main reason for the observed drop was groundwater abstraction in the underground mines of NE Transdanubia.

3. MEASUREMENTS

3.1 Introduction

The discharge volume of springs represents the overall response of a karstic reservoir to the input precipitation events. Measurements of this discharge can help to indicate how the karst system works. Also, these measurements may be utilized during the planning of tracer tests (injected concentrations, predicted dilution etc.).

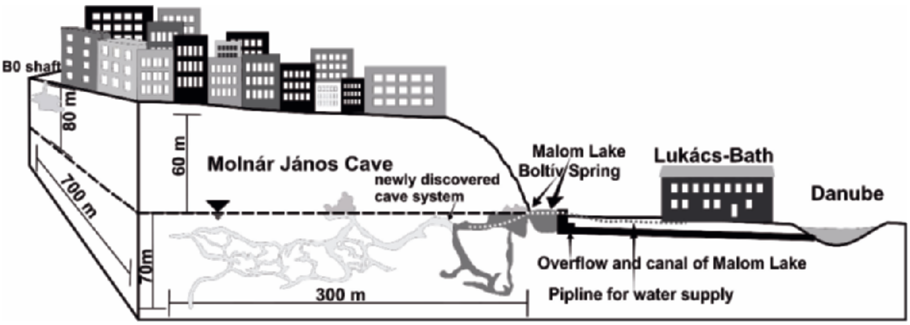

Figure 3. Sketch of the Rózsadomb area (modified after Wolk, 2003).

3.2 Discharge Measurements

On the basis of scattered measurements, Alföldi *et al.* (1968) estimated the flow rates for the Boltív Spring at the Malom Lake as 3000–6500 m^3/d. These values were used for planning our tracer test.

We have carried out continuous spring discharge measurements in the foothills of Rózsadomb, on a three-month time-scale. Daily precipitation was recorded during the same period. There are difficulties with measuring the discharge directly because of the urbanized environment of the once natural spring-outlets. Accordingly, measurements had to be focused on the Boltív Spring, the only natural, free outlet feeding the artificial Malom Lake. The discharge volume was estimated from two components. Firstly, with the help of a water-meter it was possible to measure how much water was actually used by the bath associated with this spring (Lukács Bath). The overflow of the Malom Lake was considered as the second component of the total discharge and it was measured by an impeller-type current meter in the canal (Figure 3).

3.3 Tracer Test

According to the results of a previous test in a cave passage high upon the hill (Figure 4), it was tempting to suppose direct connection between the Rózsadomb area and the Boltív Spring (Sárváry *et al.*, 1992). The expected breakthrough-time predicted from this previous study was between 10 hours and 42 days.

Tinopal CBS-X, an optical brightener in a solution concentration of about 15 g/l (10 kg dissolved in 600 litres water) was used as a (fluorescent) tracer for the experiment.

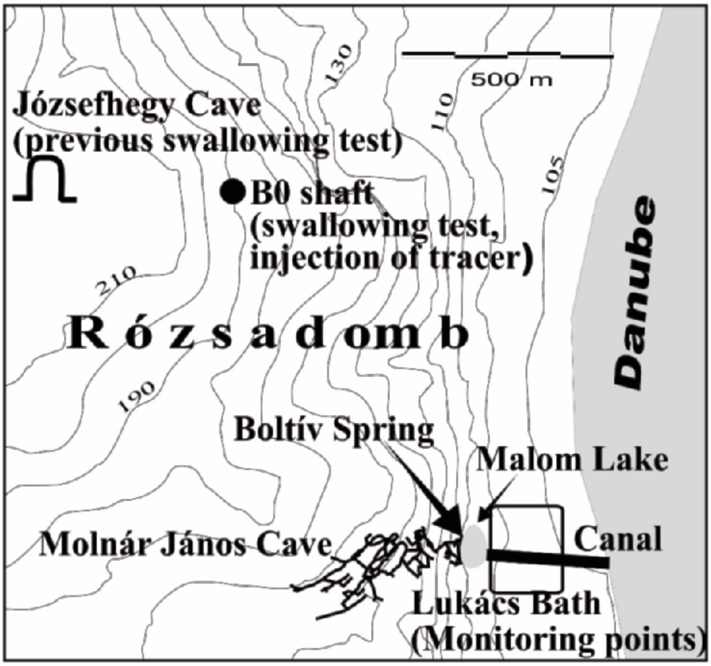

Figure 4. Location map of the tracer test.

Although Tinopal was not the best choice as a tracer due to its relatively high affinity for sorption (Schudel *et al.*, 2003), this was the only allowed tracer which could be used for water tracing around the bath. Shaft B0 in the Józsefhegy Cave sector of Rózsadomb was selected to be the most appropriate injection place (Figure 4).

Prior to the tracer injection, 500 m^3 of water was injected into the shaft to make sure that its 'swallowing' capacity would not limit the operation and that the underground channelways were properly saturated, to avoid obstruction of the descending dye by dry sections.

For this 'swallowing' test, the sampling period after the tracer injection was two months, with a gradually decreasing sampling frequency. In total, 1100 samples were collected at ten different sites (wells, springs) in the area of the Lukács Bath (Figure 4). Additionally, at the Molnár János underwater cave the tracer was continuously monitored with a flow-through field fluorometer (Schnegg and Doerflieger, 1997) (detection limit for

Tinopal: 10^{-9} g/ml), connected to the water supplying pipeline that was deeply immersed in the water of the adjoining phreatic Molnár János cave.

4. RESULTS

Discharge measurements in the canal between the Malom Lake and the Danube showed a rather voluminous discharge (~13,700–15,700 m^3/d), much greater than that previously estimated by Alföldi *et al.* (1968) (3,000–6,500 m^3/d). Based on the simultaneous observation of precipitation and discharge, it is also concluded that there is no significant correlation between the variation of discharge and precipitation (Figure 5).

Despite the carefully executed measurements, no tracer breakthrough could be detected during the two-month-long observation period.

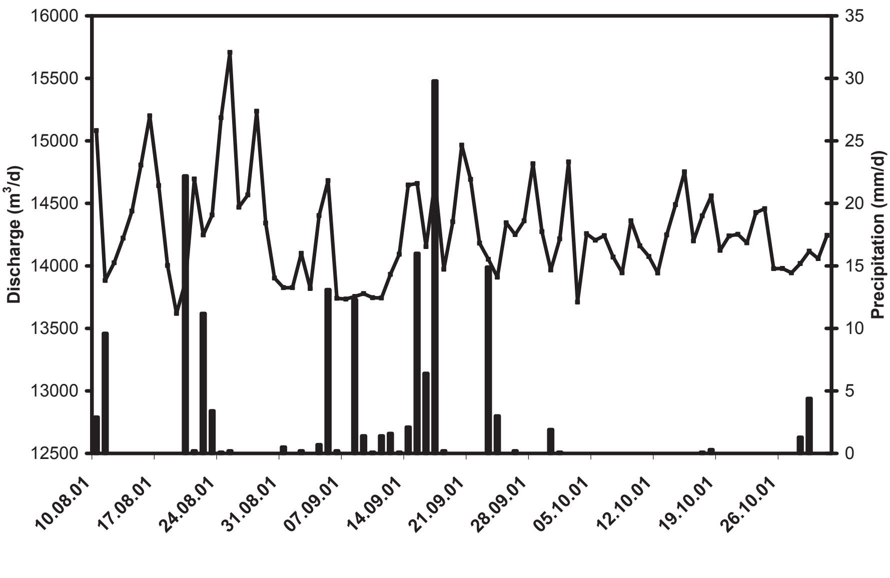

Figure 5. Total discharge of the Boltív Spring (line) and the recorded precipitation (columns) during the tracer test.

5. CONCLUSIONS

It turned out that previous authors have clearly underestimated the discharge of the Boltív Spring. No significant correlation was found between the variation of discharge and the recorded precipitation events.

The most reasonable explanation for the above facts is that the direct recharge from Rózsadomb is several orders of magnitude less than the discharge at the Boltív Spring.

No tracer breakthrough could be observed. There are three possibilities to explain this negative result as follows:

The injected water-cum-tracer was retained in an unknown, vadose-zone cavity somewhere under the injection point at shaft B0, and therefore it could not reach the karst water table. Since shaft B0 is supposed to have originated from the collapse of a vault of a larger cave underneath (Leél-Őssy, 1995), this possibility cannot be excluded.

The tracer reached the karst water table; however from there it followed an unknown flow path, different from the one leading directly to the Boltív Spring. It may have reached the Danube that serves as a "line sink" for all free-draining groundwaters. In this latter case the tracer might have easily been discharged, unnoticed, at some unknown, maybe underwater springs debouching directly in the river.

The third possibility considered here is that the tracer reached the karst water table and descended towards the Lukács Bath, however, behind the Boltív Spring in the Molnár János cave, it met such a huge volume of water that it became diluted to such an extent that its detection was rendered impossible.

In addition, it cannot be excluded that a combination of the above outlined mechanisms was responsible for the result.

6. SUMMARY

The negative tracer test did not provide any direct answer to the original question, namely, how the Buda Thermal Karst System works between Rózsadomb and the discharge area at the Boltív Spring. However, the results of the experiment greatly improved our understanding of the underground flow system of the area. Now we have good reasons to hypothesize that contaminants originating from the Rózsadomb urban recharge area cannot reach the discharge area: 1) because of the efficient and extensive marl cover (acting as aquitard), 2) because of the adsorptive capacity of an almost 80 m thick unsaturated zone, and 3) due to dilution, because of the large volume of deep groundwater added to the system. More risks for the water quality arise from the fact that contaminants may infiltrate into the aquifer in the close proximity of the discharge area where the water table is as close as 1 m to the level of the tram-rails and the basements of the houses.

7. EPILOGUE

Two years after the tracer test, scuba-divers have discovered a succession of enormous cavities connected to the Molnár János Cave (Figure 3) (Kalinovits pers. comm.). The length of the "newly discovered" largest passage is *ca*. 80 m; the width is 16 to 24 m, the depth 5 to 24 m (62 m below the water table), and the total length of the cave is *ca*. 2.6 km. The estimated water volume in this cave is therefore about 2–3 x 10^4 m^3. This confirms the idea that mechanism No. 3 outlined above may very well be valid.

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SECTION V:

BIOLOGICAL WATER QUALITY

MONITORING AND MANAGING THE EXTENT OF MICROBIOLOGICAL POLLUTION IN URBAN GROUNDWATER SYSTEMS IN DEVELOPED AND DEVELOPING COUNTRIES

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Abstract: Monitoring of urban aquifers has highlighted faecal contamination in both developed (UK) and developing (Mozambique) country settings. This has underlined gaps in our knowledge of not only the flux of contaminants through the complex urban water system but also the fate and transport of pathogens once in the subsurface. Research aiming to achieve a better understanding of these issues is described here along with potential management strategies looking at water reuse in urban areas.

Key words: faecal contamination; bacteria; viruses; microbial pollution; water quality; Mozambique; UK; transport of pathogens; pathogens; management strategies; Developing Countries.

1. INTRODUCTION

The storage and transmission of sewage in the shallow subsurface via septic tanks, pit latrines, soakaways, and sewers presents clear risks of faecal contamination of groundwater. This risk is exacerbated when these sanitation systems are poorly constructed or maintained. This is true both for developed and developing country settings. This directly threatens groundwater quality and can result in the sanitation system failing to meet its primary role of breaking the faecal-oral transmission route of waterborne pathogens. The risk of failure of any of these systems increases with population density, as does the potential severity of the public health consequences. Hence, the large urban populations dependent on

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groundwater for domestic use are often at greatest risk of exposure to waterborne pathogens. This is of particular concern as, by late 2007, for the first time in human history, the majority of the world's population will be urban dwellers (UN-HABITAT, 2003).

The transmission of disease through contaminated groundwater resources is well documented (e.g. Grabow and Prozesky 1978; Hurst 1997; Macler and Merckle 2000). There are now known to be over 100 viral, bacterial and protozoan pathogens that can contaminate groundwater from faecal and other sources (Bennett *et al.,* 1987; Herwaldt *et al.,* 1992; Kramer *et al.,* 1996). Inadequate treatment of groundwater continues to be an important factor contributing to outbreaks of waterborne diseases (CDSC, 2000; Misstear *et al.,* 1996).

Ideally, the occurrence and levels of all pathogens in drinking water should be monitored to restrict the transmission of water-borne diseases. This ideal is, however, far from attainable. Many of the pathogens that are found in groundwater are highly infectious and represent a potential risk to health even when present in very low numbers. Most enteric viruses and protozoa require only 10 or fewer infectious particles or cysts to cause infection. Bacteria, however, do not usually cause infection unless more than 10^3 infectious cells are ingested (USEPA, 1992).

Pathogens and indicators in groundwater are present either as organisms in suspension or bound to colloids and so do not exhibit the uniform concentrations typical of solutes. Hence, sampling regimes over short time periods lead to distinct changes in the levels of a particular organism. In these cases quarterly, weekly or even daily sampling may be insufficient to describe contamination events (Macler and Merckle, 2000). Sampling groundwater for the common bacterial indicators of faecal contamination normally requires the collection and analysis of a 100mL sample. The major difficulty in monitoring for viral and protozoan pathogens has been the need to use complex and specialised sampling methods to detect low concentrations of infectious particles in large volumes of water. The detection of potentially low numbers of virus particles therefore involves a preliminary concentration stage. Directed adsorption has become the preferred option for recovering viruses from large volumes of water (Hurst, 1997), and forms the basis of the standard method for enteric virus detection in water (APHA, 1998). Some recent studies (Vilaginès *et al.,* 1993; Powell *et al.,* 2000) have successfully applied a glass wool trap for the concentration of enteroviruses from large volume groundwater samples whilst in the field. These are the sampling methods used in the case studies discussed below.

The management of microbial water quality in urban aquifers is thus a complex task with multiple potential pollution sources (e.g. sewers/latrines, cemeteries, landfills). Compounding this are variable dynamic distributions in hydraulic head due to complex abstraction patterns and often a multitude of abandoned boreholes in urban areas (Cronin *et al.,* 2003a). This paper documents efforts to monitor and understand the extent of microbiological pollution in urban groundwater systems in developed and developing country cities. Case studies are presented from the UK and Mozambique to illustrate the results of these efforts. Further complicating factors, such as the large variability in microbial contaminant source loading and concentration variations during transport within the complex urban water cycle, are also examined. In addition, the present paucity of reliable information needed to assess the survival and transport of pathogens in groundwater systems is highlighted. Selected strategies to best protect groundwater resources from increasing urbanization are also discussed.

2. THE DEVELOPED WORLD SITUATION

Groundwater is a very important resource for urban areas in the EU, North America and other industrially developed countries because of the large populations these urban areas sustain and the fact that many rely on underlying aquifers (Table 1). Approximately 75% of Europeans, for example, live in an urban area (Table 1) and over 40% of the water supply of Western and Eastern Europe and the Mediterranean region comes from urban aquifers (Eiswirth *et al.,* 2003).

Table 1. Breakdown of the projected number of urban dwellers on each continent by the year 2010 (UN-HABITAT, 2003).

	Urban population (millions)	% of region living in an urban area
Asia	1784	43
Europe	536	75.1
South America/Caribbean	470	79
Africa	426	42.7
North America	273	79.8
Oceania	26	75.7

A key facet of urban groundwater that remains poorly understood is the depth and rate at which contaminants penetrate urban aquifers. Previous urban groundwater studies in the UK have used shallow monitoring piezometers and/or pre-existing boreholes, the depths and construction details of which are often uncertain. Sampling results can, therefore, be masked by the mixing of waters from several horizons (Parker *et al.,* 1982).

To monitor the depth-specific variation in concentrations of both solute and microbial contaminants in a typical urban aquifer, bundled multilevel piezometers were installed in the Triassic Sherwood Sandstone Group of Nottingham (Taylor *et al.,* 2003). The results showed that the temporal and depth-specific characteristics of faecal indicator microorganisms [thermotolerant coliforms (TTC), faecal streptococci (FS), sulphite reducing clostridia (SRC), enteroviruses, and coliphage (i.e. viruses that infect *E. coli*)] and inorganic (*e.g.* nitrate, chloride, sulphate) contamination over the investigation period differ significantly and reflect the contrasting transport characteristics of surface-loaded solutes and particulate microbial species (bacteria and viruses) within the Triassic sandstone (Cronin *et al.,* 2003a).

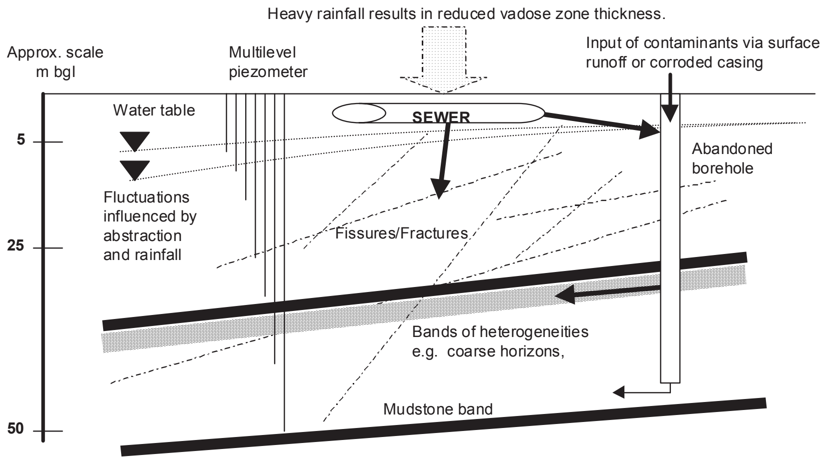

Figure 1. Conceptual diagram outlining possible routes for the transport of solute and microbiological contamination in urban aquifers (modified from Cronin *et al.*, 2003a).

Differences result from a variety of factors, including microbial die-off, dilution, and the contaminant-source characteristics. Observations made during this study show that low levels of microbial contamination should be expected at depth in fissured sandstone, though the magnitude of this contamination will vary over time (Figure 1). This work has highlighted problems of viable microbes reaching significant depths (>40m) in slow-moving groundwater systems. This is of major concern because it implies that urban groundwater-protection measures based on solute-transport estimates may not be applicable to microbial contamination. As Taylor *et al.* (2004a) point out, such groundwater-protection measures are frequently based on the invalid assumptions that pathogens travel to wells at average linear groundwater velocity and survive in groundwater for known lengths of time (*e.g.* 30 to 50 days). The Nottingham results prove the flawed nature of at least one of these assumptions. Further investigations in the Sherwood Sandstone at Doncaster (UK) reinforced these findings (Cronin *et al.,* 2005; Rueedi *et al.,* 2005). Improving our understanding of such transport and survival issues relating to pathogens in order to better manage the urban groundwater resource is discussed further below.

3. THE DEVELOPING WORLD SITUATION

The greatest increases in global urbanization are occurring in developing countries, particularly within small- to medium-sized towns (UN HABITAT, 2003, Taylor *et al.*, 2004b). Africa, for example, will have over 400 million urban dwellers by 2010 (Table 1), two thirds of whom will live in towns with fewer than 500,000 inhabitants. Such towns often have fewer resources and less expertise available to them than the larger cities. The effects of these increases in urbanization in the developing world are noticeable; the number of hours of piped water supply has dropped, and queuing time at community wells in East Africa has grown over the last 30 years (Thompson *et al.,* 2001). This study showed that the reliability of piped water supplies has declined at most sites over the period 1967-1997, in part because of the inability of municipal authorities to provide adequate services and because rising populations, particularly in urban areas, impose extra stresses on supplies and create conflicts within the communities. Average daily consumption figures for such users of piped urban supplies have dropped over this 30-year period from 128 to 66 litres/person/day (Thompson *et al.,* 2001). Furthermore, consumers of unpiped urban supplies still use more water than their rural counterparts (24 as opposed to 18 l/person/day) though Thompson *et al*. (2001) make no mention of contrasting water quality.

Ongoing monitoring work in the town of Lichinga (Mozambique) has highlighted severe problems of faecal contamination of groundwater in urban Africa (Cronin *et al.,* 2004; Cronin *et al.,* in press). Lichinga (13° 18‘S, 35° 15‘E) has a population of ~100,000 people and is reliant on on-site sanitation and groundwater resources. This single case study is discussed below but work carried out by the Robens Centre in Mali, Uganda, and Kenya has led to similar findings. Water quality monitoring was conducted using the portable *Delagua* water testing kit (~US$ 1600); this is a robust, relatively low-cost method of assessing sewage contamination by analysing the concentration of thermotolerant coliform bacteria (TTC) in water samples. Sanitary risk inspections (as recommended by the World Health Organization and the American Water Works Association) were also carried out and proved a useful tool. These inspections entail the systematic logging of observable faults in the wellhead vicinity that may lead to the degradation of water quality (Lloyd and Bartram, 1991). Each fault is considered as one point on the sanitary risk inspection score. Coupling water quality monitoring with sanitary risk inspections was carried out so as to

1. identify possible causes of sewage contamination;
2. identify potential risks to groundwater quality;
3. raise awareness among stakeholders as to the impacts of unsanitary conditions or practices on groundwater quality.

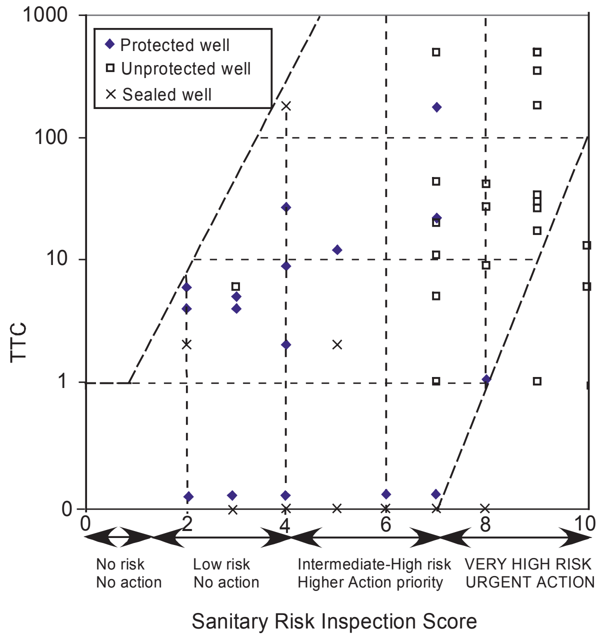

Figure 2. TTC (colony forming units/100ml) vs. sanitary risk score for a subset of representative wells in Lichinga, Mozambique (n=54). (from Cronin *et al.,* 2004).

A representative subset of water quality monitoring and sanitary risk inspections from groundwater-fed water sources in Lichinga are presented in Figure 2. Protected or sealed wells generally demonstrate both lower TTC counts and sanitary risk scores than unprotected wells. A broad relationship between sanitary risk scores and concentrations of TTC (log scale) is observable. Reduction in sanitary risks would therefore be expected to lead to a decrease in the magnitude of contamination. The relative importance of each risk factor can also be statistically tested (Howard *et al*., 2004) to identify the main causes of contamination. This approach is also useful in investigating the common assumption that microbiological contamination of groundwater derives from poorly sited or constructed sanitation facilities (Melian *et al.,* 1999). Preliminary data from

Lichinga show a strong association between unsanitary wellhead completion measures (such as animal access and the unhygienic storage and use of the rope and bucket) and the magnitude of sewage contamination. This is consistent with studies in Uganda (Howard *et al.*, 2004) and Guinea (Gelinas *et al.,* 1996).

In conclusion, the groundwater quality of protected (but unsealed) wells in Lichinga employing local, low-cost (sustainable) technologies (*e.g.,* windlass, plinth) is significantly better than that of unprotected wells. This is important as it demonstrates that local, sustainable abstraction technologies can begin to deliver significant improvements in quality. Surveillance has highlighted the complex interaction of a variety of factors (particularly sanitary completion measures and population density) that affect groundwater quality. Population density is not an easy issue to tackle but improving sanitary completion measures is being addressed by training in sanitary well construction and hygiene awareness programmes.

4. UNDERSTANDING THE ISSUES

The case studies presented above demonstrate that the proper management of urban groundwater requires a detailed understanding of:

- Concentrations of micro-organisms in the source loading – this must be identified both at the source and then the variations mapped through the urban water cycle
- Improved understanding of the fate and transport characteristics of the microbes once in the groundwater

The first issue has been addressed via an EU 5FD initiative termed AISUWRS (Assessing and Improving Sustainability of Urban Water Resources and Systems). This analysed a range of existing urban water supply and disposal scenarios by demonstrating how each scenario differs in its handling of contaminants (including pathogens) within different urban water systems, and structured around these a Decision Support System for urban water management strategies (Wolf *et al.*, 2005). In order to evaluate both existing urban water systems and alternative strategies, it is required that the sources of contaminants, their flow paths and volumes (*e.g.* unaccounted-for water, recharge from pipe leakage) and the sinks must be identified for different urban areas, and that quantification of the contaminant loads (e.g. sewage exfiltration, water losses) is carried out. This was undertaken by the Robens Centre and the British Geological Survey via an intensive field sampling campaign of groundwater, sewers, and stormwater in the UK city of Doncaster. With a population of around

300,000 people, Doncaster was chosen for the study because its supply system is relatively simple and depends heavily on groundwater. Modelling work, using a mass balance model (Figure 3) for the urban cycle [Urban water Volume and Quality model (UVQ)], was applied to track (conservatively) the parameters P, N, Zn and *E.coli*. for a suburb of Doncaster in order to assess the current water supply system and to compare it with potential new scenarios of water management (Rueedi *et al.*, 2005).

The scenarios tested were: (i) combined versus separate storm/foul water systems; (ii) introduction of permeable roads and pavements, (iii) greywater reuse at household level and (iv) rain water diversion from roof runoff for irrigation. The biggest impact on water quality and quantity leaving a typical household through sewer, storm water and infiltration system was obtained by re-using greywater from kitchen, bathroom and laundry for irrigation and toilet flush. Using all strategies outlined above, mains water use could be reduced by up to 20% (Rueedi *et al.*, 2005). This example demonstrates the potential power of assessing how chemical and microbial contaminants can be modified through the urban water cycle using different management strategies.

The second constraint to improved knowledge of urban groundwater protection from microbial contamination, as outlined above, is that we still do not fully understand how pathogens survive and move in various subsurface media. A range of substances have been used in the past to trace groundwater flow. Most of these were dissolved in the water, for example fluorescein and rhodamine dyes or conservative anions such as bromide, though suspensions of particles such as polystyrene beads have also been used. However, none of these can provide a comprehensive model for pathogenic viral transport as they do not have comparative chemical composition, charge, surface properties or physical size (Harvey, 1997).

Bacteriophage (a group of viruses that infect bacterial cells but not humans, plants or animals) are being investigated at laboratory and field-scale by the Robens Centre and the University of Birmingham as the tracers needed to develop more robust models of microbial transport. The survival of poliovirus and MS2 in several water types have both shown very low inactivation rates (Collins *et al.*, 2004). Four types of bacteriophage (MS2, PRD1, H40/1, ϕX174) were chosen for further research due to their sharing similarities in key characteristics with common waterborne viral pathogens (Table 2). These bacteriophage were introduced directly into an urban aquifer (Triassic Sherwood Sandstone, Birmingham, UK) in a series of tracer tests (with fluorescein acting as a conservative comparative tracer) so that the fate and transport responses could be monitored over specific time periods and under field conditions.

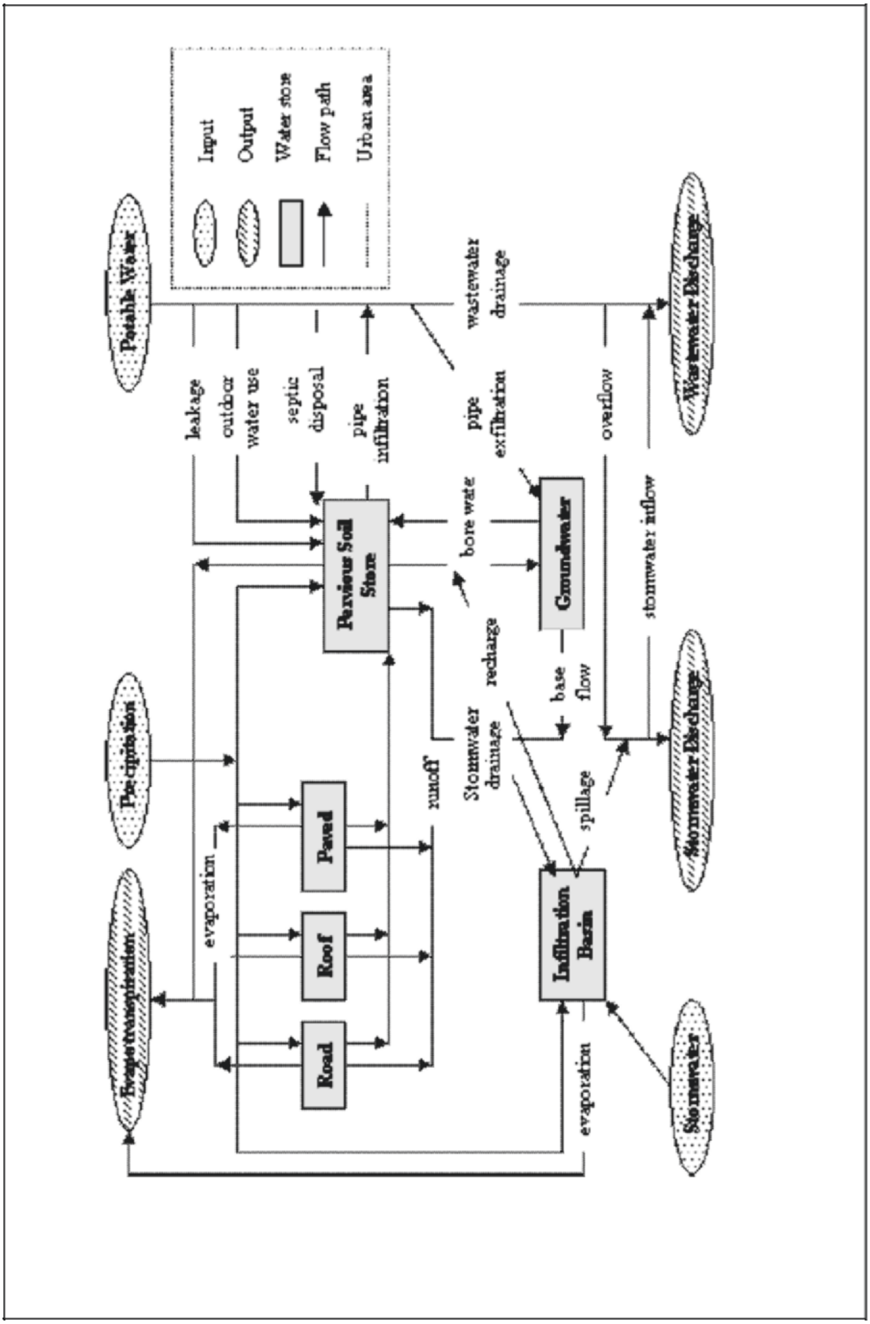

Figure 3. Outline of the modelling approach taken by UVQ (modified from Rueedi et al., 2005).

The resulting information was then used to create a phage reference tool to aid environmental and public health management of groundwater by allowing quick hypothesis testing of the properties and behaviour of an appropriate bacteriophage in the subsurface. This then enables inferences to be drawn with the pathogenic virus of interest (Joyce *et al.,* 2006).

Table 2. Characteristics of possible phage surrogates (MS2, PRD1 ϕX174, H40/1) and pathogenic viruses; N/A = Not available. (Apthovirus = Foot and Mouth DV).

Virus/Phage	*Size*	*Symmetry*	*Isoelectric Point*	*Genetic material*
MS2	26	Icosahedral	3.5-3.9	ss-RNA
PRD1	62	Icosahedral	4.2	ds-DNA
ϕX174	25-27	Icosahedral	6.6	ss-DNA
H40/1	82-85*39-43	Icosahedral and Tail	<4.5	ds-DNA
Adenovirus	80-110	Icosahedral	N/A	ds-DNA
Astrovirus	27-30	Polyhedral	N/A	ss-RNA
Coxsackievirus	28-30	Icosahedral	4.8	ss-RNA
ECHOvirus	28-30	Icosahedral	5.0-6.4	ss-RNA
Apthovirus*	27-30	Icosahedral	N/A	ss-RNA
Norovirus	35-39	Icosahedral	5	ss-RNA
Poliovirus	28-30	Icosahedral	4.5-6.5	ss-RNA
Rotavirus	80	Icosahedral	3.9	ds-DNA

5. TACKLING THE ISSUES

The urban water cycle mass balance modelling described above shows the importance of an integrated approach, both in terms of technically understanding pollutant fluxes but also in regard to developing management strategies. This is becoming increasingly important as water management is moving away from the development of new water sources towards reuse strategies. Water management policy can be divided historically into three distinct time periods. The first stage, *supply management*, involved utilizing previously undeveloped sources to satisfy growing demand requirements. As such sources became increasingly difficult to find, or transportation costs became too expensive, the water sector entered an era of *demand management*, using various measures to influence the demand for water (Lundqvist and Gleick, 1997). This has incorporated such initiatives as metered billing, repair of leaking mains, low-water use toilets, and the treatment and reuse of water by industries in their processes. The success of industry in treating and reusing its process water has led some to re-think management strategies for household water and wastewater. By separating streams of discharges from households, it becomes easier to treat or clean each stream to a sufficiently high quality that the end product can be reused. This water management strategy is termed *reuse management*, the main objective being to divide waste into appropriate streams which can be more easily transformed into useful products.

Reuse management has potential applications that are important to consider both for developing and developed country settings. Urine and faeces are the major nutrient sources in domestic wastewater with approximately, 75%, 80% and 90% of P, K and N respectively in the total domestic wastewater load coming from human waste (Henze and Ledin, 2001). There is great potential to directly recycle these nutrients back into the soil to maintain fertility, increase crop yields and avoid the use of chemical fertilizers (Jönsson, 2002). This has potential also to save on water mains supply volumes use, as the flows of urine (~1.5l/pers/day) and faeces (~0.2 l/pers/day, excluding flushwater) are small compared to total present day wastewater flows (~150 l/pers/day). Proper storage and/or treatment of the faecal waste is necessary to eliminate pathogenic risks, though further research is needed to test this in different country settings. If wastewater is to be reused then the other contaminant loadings must also be considered. Heavy metal content of household wastewater is generally small enough to be manageable, though more research is needed on the potential harmful effects of pharmaceutical residues in reuse of these waste streams.

Such alternative systems need to be actively considered for developing country cities (especially their peri-urban areas) as rapid urbanization, increasing volumes withdrawn from the underlying urban aquifers and the high capital and maintenance costs associated with conventional piped systems mean that such conventional piped systems are not viable options in the near future (Drangert and Cronin, 2004). 700 million urban dwellers are still without adequate sanitation, a situation that must be remedied as improvements in sanitation has impacts on diarrhoeal incidences at all levels of water supply (VanDerslice and Briscoe, 1995). Hence, careful and informed choices are vital.

Greywater and stormwater are other urban water streams that can be reused. In fact, these contribute most of the volume to urban water flow volumes. However they also contain organic contaminants (both BOD and toxic substances) and heavy metals (Vinnerås, 2002). The contaminant concentrations in stormwater are mainly from corrosion of roof material (*e.g.* copper roofing, galvanised steel) and surface runoff (Eriksson, 2001) and are generally low compared to those in greywater and conventional wastewater, making it easier to reuse. Sustainable urban drainage systems (SUDS) re-infiltrate stormwater into the aquifer, though such systems also need careful evaluation to ensure excessive groundwater quality degradation is not occurring. Most of the greywater contaminants (e.g. BOD, COD and toxic organic pollutants) originate from the household chemicals used, such as shampoos and detergents (Eriksson *et al.,* 2003). Thus, choices made by consumers, industry, and the health sector, among others, have major impacts on the contamination levels entering the urban water cycle. Hence, reuse management relies not only on technical solutions but also on consultation and education of the community; again relevant in all cities regardless of the stage of development.

Further monitoring of urban aquifers is needed, especially in developing countries. It is important that Non-Governmental Organisations (NGOs) and donor agencies working in the water and sanitation sector are able to address contamination issues *and* to demonstrate the effectiveness of the projects they fund. Initiatives such as the Sphere Standards (Sphere, 2004), a set of minimum expectations for humanitarian relief programmes, have formalized the need for the donor community to measure, analyze, document, and report their findings in the WATSAN sector, among others. This approach needs to be applied to alternative technical solutions and management systems.

Records of water-quality monitoring in developed countries are collected primarily in order to comply with monitoring requirements and are used occasionally to identify contaminated wells. Cronin *et al.* (2003b) have shown rising solute concentrations on a regional basis in the Nottingham urban area but also that monitoring programmes have been reduced in scale over successive years. Both of these trends are worrying and require proactive action to alter them. In addition, recent efforts toward depth-specific monitoring can usefully inform regional assessments of groundwater quality, as such data not only constrain the rate at which contaminants loaded at the surface or near-surface penetrate urban aquifers but also determine the opportunity for remediation.

6. CONCLUSIONS

Case studies in both developed country urban areas (Nottingham and Doncaster, UK) and a developing country urban area (Lichinga, Mozambique) have highlighted the vulnerability of urban aquifer systems to faecal contamination. Current sanitation systems can all contribute to this contamination, and current protection methods, such as setback distances that are based on solute transport times, will not adequately mitigate against continued degradation (and the associated public health threats) of such resources. New technical solutions and management strategies (including reuse systems) are needed but many such solutions need further testing. However, the complexity of urban groundwater systems means that this research is needed at a number of different levels. These include on-going monitoring to understand the vulnerability of urban aquifers, using surrogates to better understand the transport and survival characteristics of pathogens themselves once they enter into the subsurface, and more modelling to better understand the hydrogeology of urban aquifers and the flux of water and contaminants through the many complex facets of the urban water cycle.

ACKNOWLEDGEMENTS

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MICROBIAL POLLUTION OF GROUNDWATER IN THE TOWN OF WALKERTON, CANADA

Implications for the Development of Appropriate Aquifer Protection Strategies

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Abstract: In May 2000, the town of Walkerton, Ontario suffered one of Canada's worst outbreaks of waterborne disease. In the final toll, 2,300 people became ill and seven people died. The principal pathogens were *Escherichia coli* O157:H7 and *Campylobacter jejuni*, both found in the town's municipal well-water supply. The outbreak sparked intensive hydrogeological investigations and a nine-month long independent commission of inquiry. Two reports were issued. Part 1 focused on Walkerton and identified a cattle farm adjacent to the primary pumping well as the most probable cause of the outbreak. Part 2 delved deeply into all aspects of water supply in the Province and documented 93 commission recommendations concerning the Province-wide delivery of safe drinking water. The very first recommendation highlighted the need for adequate groundwater management and protection. To its credit, the government responded well to many of the issues raised, mostly as they relate to infrastructure, the operation of municipal water sources, the training of operators and data collection. However, from a hydrogeological perspective, the government's response has been disappointing, particularly with regard to its prescribed methodologies for groundwater protection. A program for mapping groundwater vulnerability and wellhead protection areas was hastily assembled following the Walkerton outbreak but is seriously under-funded, piece-meal in approach, and in several cases scientifically ill-conceived. Ultimately, there are no simple solutions and no easy short cuts, and the successful implementation of groundwater protection measures in Ontario will demand a serious commitment of funds and resources to advance significantly our basic understanding of the Province's groundwater resources and provide key input data.

Key words: bacteria; pollution; E. coli; aquifer protection; fracture flow; abandoned wells; Walkerton, Canada.

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1. INTRODUCTION

In Ontario, Canada, over 80% of the population is supplied with drinking water from centralized municipal sources. In the larger towns and cities, Great Lakes water is normally the preferred source option; groundwater is an important, if not the major source of domestic water supply for millions of rural users residing in small towns and villages.

Walkerton, Ontario is a typical rural town (population 4850), entirely dependent on groundwater for its drinking water supply. Located in cattle country, about 200 km north-west of Toronto (Figure 1), the town once had little to distinguish it from dozens of other small Ontario towns. This changed dramatically, and no doubt permanently, in May 2000 when Walkerton suffered one of Canada's worst outbreaks of waterborne disease. In the final toll, 2,300 people became ill and seven people died. The principal pathogens were *Escherichia coli* O157:H7 and *Campylobacter jejuni*, both present in the town’s municipal well-water supply. The outbreak sparked intensive hydrogeological investigations and a commission of inquiry that lasted nearly a year.

Two reports were issued (O’Connor, 2002a, b). Part 1 focused on the events at Walkerton and concluded that manure spread on a field at a cattle farm adjacent to the primary pumping well was the most probable, if not only contaminant source. Part 2 examined broader water supply issues and documented 93 commission recommendations concerning the Province-wide delivery of safe drinking water. The very first recommendation highlighted the need for adequate groundwater management and protection. The response of the government has been generally positive, particularly on issues relating to infrastructure, the operation of municipal water sources, the training of operators and data collection. Its response to hydrogeological issues has been less satisfactory, notably with respect to its formulation of meaningful groundwater management and protection methodologies. In this paper I review the hydrogeological issues raised at Walkerton and the progress, if any, that has been made in the four or more years that have elapsed since the tragedy unfolded.

2. THE EVENTS OF MAY 2000

In the town of Walkerton, groundwater is pumped from semi-karstic limestones and dolostones of the Devonian-Silurian Bass Island and Bois Blanc formations (Hewitt and Freeman, 1972). Throughout most of the area, the Paleozoic carbonate aquifer is draped, and locally confined, by the

Elma and Dunkeld Tills, relatively thin, stony, sandy silt to silt tills of Late Wisconsin age (13,000 - 16,000 years BP).

The original Walkerton wells (wells 1 and 2, Figure 1) were drilled in 1949 and 1952 respectively to depths of over 70 m. The quality of groundwater from these wells was relatively poor due to natural mineralization but the supply was considered adequate until 1962 when an improved supply was obtained from well #3. Wells 1 and 2 were eventually retired, but were never properly decommissioned in accordance with Ontario Ministry of the Environment (MOE) guidelines.

In due course, well #3, a shallow well, was superseded by wells 5, 6 and 7. These wells were drilled in 1978, 1982 and 1986 to depths of 15m, 72.2m and 76.2m respectively. From the beginning, well #5 was recognized as being vulnerable to surface contamination but at the time it was not standard practice to post explicit operating and monitoring conditions on the well's Certificate of Approval.

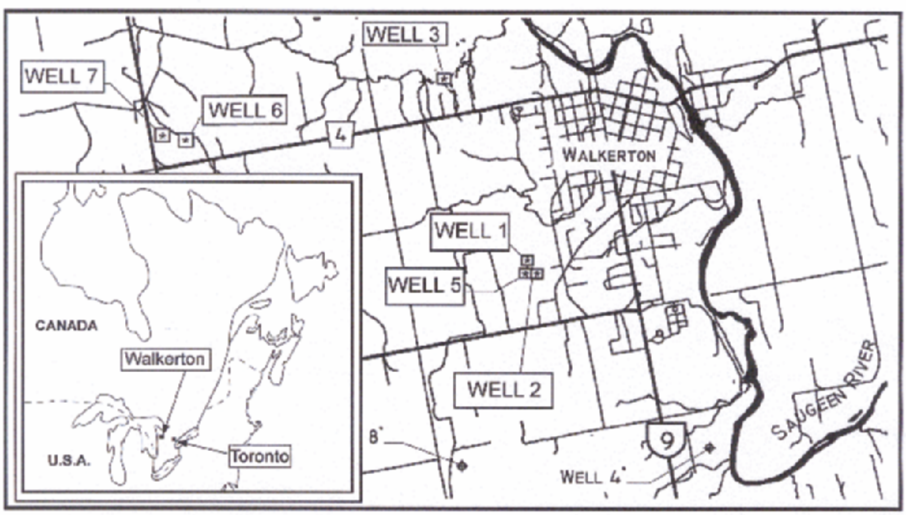

Figure 1. Location map showing wellsites around the town of Walkerton.

During the days of May 2000 when large doses of bacteria entered the delivery system, well #5 was the main source of the town's water and well #6 cycled on and off. Well #7, the primary, most productive well was, officially at least, out of service. As a general practice, all three wells would be fitted with chlorinators and chlorine residuals would be measured daily.

While the first reports of illness occurred May 14th, evidence of a widespread public outbreak did not materialize until May 18th. Initially, contaminated food was suspected, since inquiries made on May 19th to the general manger of the Walkerton Public Utilities Commission (PUC) regarding bacteriological reports of well water samples collected on May 15th, were met with false reassurances. It was not until May 21st that the likely root cause of the problem – contaminated well water - was recognized and a boil-water alert was issued. By Tuesday May 23rd, when the results of new tests confirmed the presence of *E. coli* in the town's water supply, 160 people had sought hospital treatment and another 500 had called area hospitals complaining of symptoms. One, an elderly citizen, had died. Even at that stage, the magnitude of the problem had not been appreciated. Within days, four more deaths were recorded and reports of sickness became widespread. By July 26, Walkerton's water had been associated with 2300 cases of illness and six deaths. Later studies set the death toll at seven. Some children who narrowly survived the tragedy face life-long kidney problems.

On 12th June, 2000, the Government of Ontario established an independent commission of inquiry under the Public Inquiries Act. The mandate of the commission, led by Justice Dennis O'Connor, was to examine the circumstances that caused the tragic events at Walkerton (Part 1) and to make recommendations to ensure the continuing safety of water supplies in Ontario (Part 2). Starting in July 2000, and lasting 9 months, the hearing heard from 114 witnesses including two former ministers of the Environment, and the Premier. Part 1 of the report was published in 2002 (O'Connor, 2002a).

3. THE PUBLIC INQUIRY PART 1

3.1 Source of the Contamination

Early in the hearing, evidence was given that well #7, the town's main well but supposedly out of service had, in fact, been operated without a chlorinator during the first week of May, 2000 and between May 15th and May 19th. The general manager was aware that samples collected from the distribution system on May 15th had indicated serious problems with the town's water but withheld this information since he did not want health officials to know he had operated well #7 without a chlorinator. Ironically, later evidence demonstrated that operation of well #7 was not the primary cause of the outbreak and that microbiological pathogens – namely, *E. coli* O157:H7 and *Campylobacter jejuni* bacteria – entered Walkerton's water system via well #5 on or shortly after May 12, at a time when well #7 was not being used. Well #5 was serviced by a functioning chlorination unit at the time but the available chlorine was being completely overwhelmed by bacteria entering the well. This fact went unnoticed since plant operators followed their usual routine and did not measure the chlorine residual, a practice which should have been followed daily.

Genetic fingerprinting using pulsed field gel electrophoresis (PFGE) (Danon-Schaffer, 2001) demonstrated that cattle manure applied to a neighbouring field during late April, 2000 (Figure 2) was the primary, if not only, source of the bacteria. Exceptional rainfall between May 8 and May 12, 2000 (Auld *et al*., 2004) (Figure 3) would have greatly assisted the transport of bacteria to the well. The forensic DNA study considered potential livestock sources on 13 farms within a 4-km radius of wells 5, 6 and 7 and revealed a strong association between molecular subtypes of *E. coli* O157:H7 and *Campylobacter* spp. isolates in cattle manure from the farm adjacent to well #5 with those found in the majority of human cases. Significantly, manure at the farm was applied to within 80m of well #5 using industry-standard, best management practices.

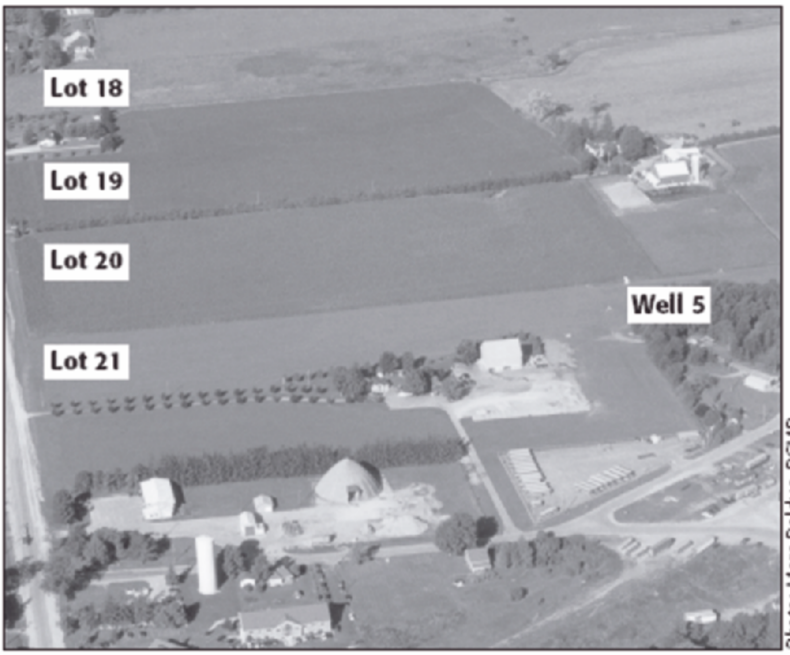

Figure 2. Cattle manure was applied to Lot 20 just 80m from well #5 (O'Connor 2002a).

Justice O'Connor never conclusively identified the path followed by the contaminant but observed that the till overburden was thin (2 to 4.5m thick), contained sand and gravel lenses and was partially penetrated by fence post holes. He also noted that unplugged abandoned wells (wells 1 and 2) and a spring adjacent to well #5 which stopped flowing when the well was pumping, were also possible routes for rapid vertical transport of contaminants.

3.2 Allocation of Responsibility

Justice O'Connor concluded that the outbreak could have been averted if monitoring procedures had been strictly followed. Evidence presented at the hearing showed that PUC operators had engaged in numerous improper practices, including failing to use appropriate doses of chlorine, failing to monitor chlorine residuals daily, making false entries about residuals in daily operating records, and misstating the locations at which microbiological samples were taken. If PUC operators had manually monitored the chlorine residuals at well #5 during the critical period, the problem would have been identified quickly and the extent of the outbreak would have been significantly reduced.

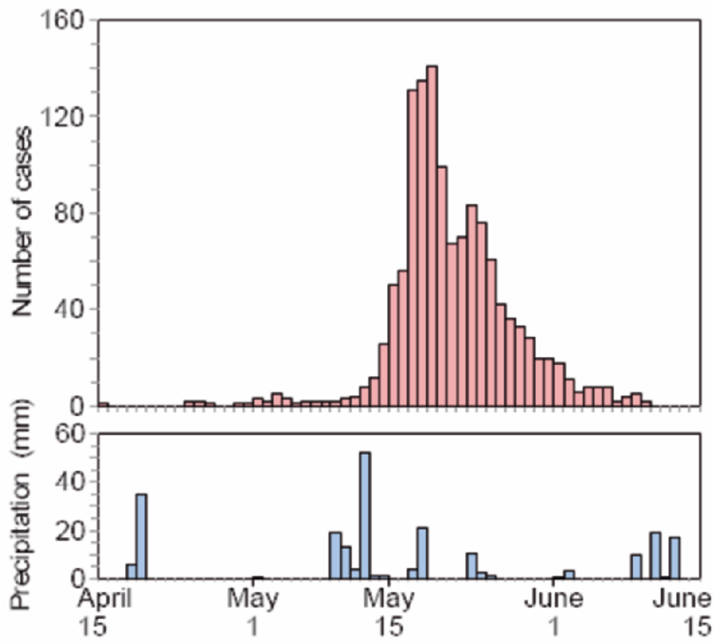

Figure 3. Comparison of illness onset dates and precipitation (after Bruce-Grey-Owen Sound Health Unit, 2000).

PUC operators did not burden all the blame. The outbreak would have been prevented entirely if continuous chlorine residual and turbidity monitors had been installed at well #5. It transpired that MOE inspectors were aware that well #5 was vulnerable to contamination by surface water, a condition referred to as "GUDI" (groundwater under the direct influence of surface water) and should have known that the Ontario Drinking Water Objectives (MOE, 1994) require such sources to be monitored continuously for residual chlorine and at least four times each day for turbidity. Also, the MOE inspection program should have detected the improper practices of PUC personnel. Provincial budget reductions were directly implicated in the failings of the MOE.

4. THE PUBLIC INQUIRY PART 2

Part 2 of the Inquiry Report, published in May, 2002 (O'Connor, 2002b), was much larger in scope and considered all issues that could affect the safety of drinking water across the Province. It was entitled "A Strategy for Safe Drinking Water" and included 93 recommendations for improvements to Ontario's water delivery system. They included:

- Source Protection (17 recommendations)
- Drinking Water Quality Standards (12)
- Water Treatment (4)
- Water Distribution (2)
- Monitoring (5)
- Laboratories (3)
- Role of Municipal Government (7)
- Quality Management (8)
- Training of Individual Operators (6)
- Provincial Government (16)
- Small Systems (7)
- First Nations (6)

Significantly, the very first recommendation dealt with hydrogeology, aquifer management and aquifer protection. It delivered a message **(Recommendation 1)** to the government that had been heard before, but never acted upon, namely that drinking water sources should be protected by developing watershed-based source protection plans, and that such plans should be introduced for all watersheds in Ontario.

Importantly, Justice O'Connor clearly defined what he meant by "watershed-based source protection plans". He said that, at a minimum, they should include:

- a water budget for the watershed, or a plan for developing a water budget where sufficient data are not yet available;
- the identification of all significant water withdrawals, including municipal intakes;
- land use maps for the watershed;

- the identification of wellhead areas;
- maps of areas of groundwater vulnerability that include characteristics such as depth to bedrock, depth to water table, the extent of aquifers, and recharge rates;
- the identification of all major point and non–point sources of contaminants in the watershed;
- a model that describes the fate of pollutants in the watershed;
- a program for identifying and properly decommissioning abandoned wells, excavations, quarries, and other shortcuts that can introduce contaminants into aquifers;
- the identification of areas where a significant direct threat exists to the safety of drinking water;
- the identification of significant knowledge gaps and or research needs to help target monitoring efforts.

In fact, it was the perfect "blueprint" for the management and protection of Ontario's groundwater.

5. THE GOVERNMENT RESPONSE

To date, the government's response has been disappointing. It certainly started very quickly and began well, bringing on-line some initiatives even before the Inquiry reports were published. For example, it:

- gave $15 million to the Municipality of Brockton to restore a safe water supply for Walkerton;
- launched Operation Clean Water which established strict protocols for operating large water works;
- committed $240 million under the "SuperBuild" program to upgrade health and safety infrastructure including municipal waterworks.

However, its subsequent activities concerning groundwater management and protection were seriously under-funded and in some cases ill-conceived. Of special concern are MOE programs prescribing approaches to groundwater vulnerability and the mapping of wellhead protection areas.

5.1 Groundwater Vulnerability

In November, 2001, just 18 months following the Walkerton outbreak, the Ontario Ministry of the Environment (MOE) introduced a very simple indexing approach to vulnerability mapping that can be described as antiquated at best and, at worst, a potentially serious liability. The approach was first described in a MOE technical guide - Technical Terms of Reference (MOE, 2001b) and has since been used to generate Groundwater Intrinsic Susceptibility (GwIS) maps for counties throughout the Province.

The prescribed method begins with the calculation of an intrinsic susceptibility index (ISI) for each available well. ISI is calculated by summing the product of the thickness of each geological unit in the well log and a corresponding “K-Factor”, from ground surface to the top of the first significant aquifer.

i.e.

$$ISI = \sum_{1}^{i} b_i K_{F_i}$$

where:

i = the number of geologic units identified in the water well record

b = the recorded thickness of each geologic unit

K_F = the representative K-Factor (see Table 1) (a dimensionless number that is very loosely related to the exponent of the vertical hydraulic conductivity in m/s).

Table 1. K-factors for calculation of ISI-values (after, MOE, 2001b).

Geological material	Representative K-Factor (dimensionless)
gravel; weathered dolomite/limestone; karst; permeable basalt	1
sand	2
peat (organics); silty sand; weathered clay (<5m below surface); shrinking/fractured & aggregated clay; fractured igneous and metamorphic rock; weathered shale	3
silt; loess; limestone/dolomite	4
weathered/fractured till; diamicton (sandy, silty); diamicton (silty, clayey); sandstone	5
clay till; clay (unweathered marine)	8
unfractured igneous and metamorphic rock	9

The final GwIS map is produced using an algorithm such as kriging to interpolate between the data points. Areas of low, medium and high susceptibility correspond to ISI values of >80, 30 to 80, and <30, respectively.

The approach has been roundly criticized, primarily for its almost total reliance on the K-Factor, a single, crudely developed surrogate for downward travel time to the aquifer. In its hasty efforts to provide a quick, convenient, low-cost method that could utilize the Province’s existing (though hardly reliable) water well database of almost half a million records, the MOE spawned scores of vulnerability maps that are little more than a coarse screening tool with little practical value for site specific planning decisions (Kell, 2004). The problems are clearly evident from a brief examination of Table 1. Two aquifers, one overlain by 12m of clean sand (K_F = 2) and the other overlain by 3m of marine clay (K_F = 8) will have identical ISI values (ISI = 24) (presumably indicating equivalent degrees of susceptibility). This is absurd. Similarly, an unconfined sandstone aquifer (no overburden) with a depth to water table of 6m, an unconfined gravel aquifer (no overburden) with a depth to water table of 30m, and a limestone aquifer (non-karstified) (no overburden) with a depth to water table of 8m will all be classified as having medium susceptibility (30<ISI<80), while a flowing-artesian aquifer confined by 3.5m of unfractured clay till or marine clay would be classified as high vulnerability (ISI <30). It is perhaps no surprise that at least one jurisdiction near Toronto has rejected the vulnerability maps produced according to the MOE technical guide, as scientifically unsound and totally inappropriate for planning purposes.

Golder Associates Ltd. (Golder, 2003; Warman *et al.*, 2004), has suggested that an AVI (Aquifer Vulnerability Index) approach developed in-house (Golder, 2001) offers certain advantages over the MOE ISI method and is considerably more conservative. To its merit, the AVI method certainly relies less on individual point data and more on a regional understanding of the subsurface geology. However, its highly simplistic three-component scoring system offers little, if any, improvement over the ISI approach.

If matters could be made any worse, the MOE provides various degrees of latitude in its Technical Terms of Reference to consultants charged with performing the vulnerability mapping exercise. Examples range from allowing consultants to modify the K-factors to better reflect field data, to providing consultants with full discretion over the selection of mapping procedures. For example, in its hydrogeological evaluation of Grey and Bruce Counties, Waterloo Hydrogeologic Inc. (WHI, 2003) assigned ISI values of 20 (high vulnerability) to all known areas of karst and all areas where the overburden thickness was understood to be <6 m, presumably to provide an additional margin of safety. While MOE deserves some credit for acknowledging the limitations of its prescribed approach and allowing consultants to incorporate their experience and local hydrogeological knowledge into the vulnerability mapping exercise, this does mean that vulnerability classifications produced by a consultant working in one area, may differ significantly from classifications developed by another consultant working in a neighbouring area.

5.2 Well Head Protection

In October, 2001, the Ontario Ministry of the Environment published a protocol for the delineation of wellhead protection areas (WHPAs) around all municipal groundwater supply wells known to be under the direct influence of groundwater (GUDI) (MOE, 2001a). A month later, the protocol was extended under Ontario's Clean Water initiative to include all municipal wells, and the protocol was published, in a slightly revised form, alongside the newly developed GwIS mapping approach in MOE's Technical Terms of Reference (MOE, 2001b).

MOE's technical guidelines require wellhead protection areas to be established separately for each municipal well and well field, and represented on maps as the surface projection of the entire 3-dimensional capture area. In turn, each WHPA must be sub-divided into a minimum of 3 well capture zones, classified according to saturated travel time - TOT (time of travel). These zones would include, as defined in the protocol:

1) **Zone 1** : 0 to 2 year TOT. Land uses in this zone need to be managed to avoid all possible risks, including those from bacteria and viruses.

2) **Zone 2** : 2 to 10 year TOT. The main focus of the land use management in this zone should be to minimize risks from all chemical contaminants; however, the bacterial and viral risks may still be a concern.

3) **Zone 3** : 10 to 25 year TOT. The land use management in this zone needs to address risks from persistent and hazardous contaminants.

Within Zone 1, an area demarked 50-day TOT is also required, presumably, though not explicitly, to identify land area requiring greatest protection from bacterial sources of contamination.

The technical guidelines also describe the various techniques that can be used to define WHPAs. These range from relatively simple analytical and semi-analytical methods to sophisticated 3-D modeling approaches. The Ministry states, 'In the majority of cases, three-dimensional, steady-state computer models should be used to delineate capture zones. When properly set-up and calibrated, these models produce the most realistic time of travel boundaries'. MOE goes on to suggest that 3-D modeling 'is the only method for accurately delineating capture zones where there is a significant presence of: (1) discrete fractures, (2) anisotropy, (3) spatial variations in hydrogeological parameters, (4) vertical movement of water and variation in total hydraulic head with depth, and/or (5) changes in water levels seasonally or through time'.

Predictably, the rapidly conceived MOE protocol for municipal well head protection attracted significant criticism, some, but by no means all, beyond the responsibility of the MOE. From a generic standpoint, the method hosts a series of problems that often go unrecognized by end users. For example, well head protection zones:

- are dynamic and change according to pumping conditions, recharge conditions and the effects of pumping at adjacent wells;
- can be geometrically complex, especially in karst and glacial aquifer systems, and can only be determined reliably where hydrogeological conditions are relatively simple and at steady state;
- can provide an indication of travel time, but take no account of contaminants already in the system and say nothing about the water quality of water that discharges from the well;
- provide no explicit guidance on the land use controls necessary for adequate groundwater protection;
- can protect wells and perhaps springs but do not provide for protection of the aquifer as a whole (i.e. contaminants released to the subsurface must have an end destination somewhere within the aquifer).

The MOE erred in its suggestion that 3-D models can accurately delineate wellhead protection zones where there is a significant presence of discrete fractures. This was amply illustrated through the work of Worthington *et al.* (2001a, 2001b, 2002) who compared wellhead protection zones developed using MODFLOW with the results of field tracer tests. The results are shown in Figure 4. 30-day capture zones developed with MODFLOW, using an equivalent porous medium approach to represent the semi-karstic carbonate aquifer, extended less than 300 m from the municipal well. Convergent flow tracer testing showed true groundwater velocities to be some 80 times faster than indicated by the model simulation. Calculations based on the tracing results and downhole video data indicated that the rapid transmission took place via solution channels with apertures of between 5 and 20 mm. Quite clearly, the capture zone analysis is meaningless in dual-porosity media.

The Ministry's program for establishing WHPAs has also been criticized for a) requiring the entire 3-dimensional zone to be projected to surface, and b) failing to provide adequate guidance for land use controls within specific well capture zones. With respect to a), many deep aquifers are very remotely connected to the active recharge system and are well-protected from contaminated infiltration in the short- and medium-term by overlying aquitards. The projection of deep capture areas to the surface creates a false impression of the threat imposed by land use change at the surface within such zones. With respect to b), well capture zones, even when reliably defined, have little practical value unless there is adequate understanding of the threat imposed by various types of potential contaminants. Time of travel can be an important criterion for protection from bacteria, or from chemicals that biodegrade during transport.

However, it can be a meaningless consideration for chemicals such as chloride from road de-icing salt that rely on dilution within the capture zone to achieve acceptable levels. It is essential that sensible, scientifically defensible guidelines be established for each of the well capture zones defined.

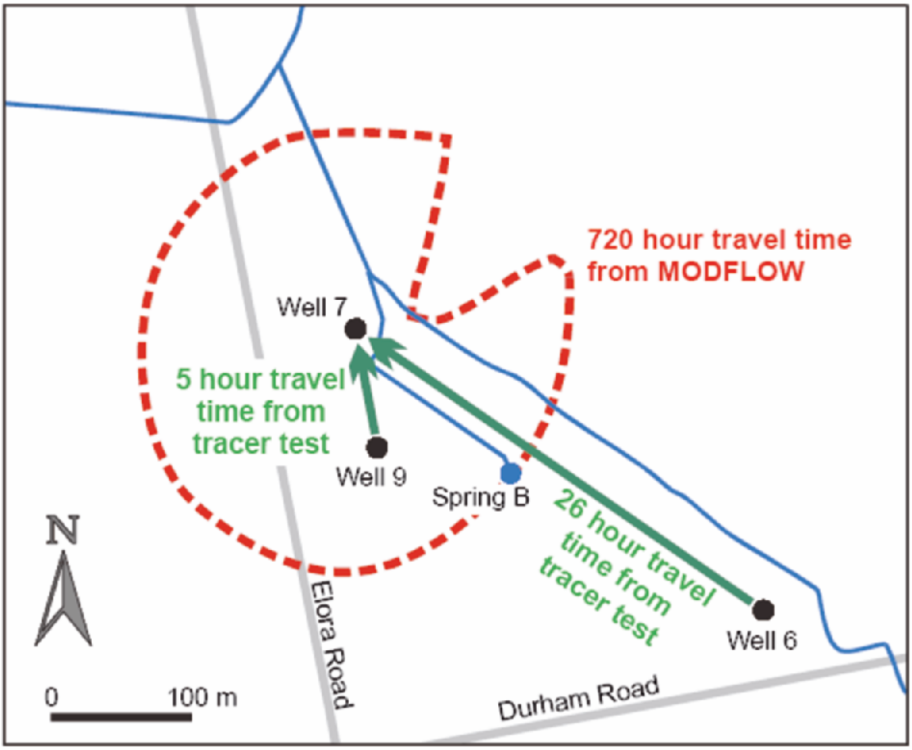

Figure 4. Trajectories and travel times for the tracer tests started on October 29th 2001, showing velocities >300 m/day, compared with a 30 day capture zone predicted by Worthington *et al*. (2001b) using MODFLOW.

6. CONCLUDING DISCUSSION

In the spring of 2000, Walkerton, Ontario endured one of Canada's worst outbreaks of waterborne disease. 2,300 people became ill and seven people died due to *Escherichia coli O157:H7* and *Campylobacter jejuni* in the town’s municipal well-water supply. Authorities responded quickly and a lengthy independent commission of inquiry uncovered the source of the problem and released a report documenting 93 recommendations that would ensure the safe delivery of drinking water across the Province. The very first recommendation concerned groundwater management and protection.

In the 4 years since the outbreak the Ontario Ministry of the Environment has responded well to a number of the issues raised, mostly as they relate to infrastructure, the operation of municipal water sources, the training of operators and data collection. From a hydrogeological perspective, however, the government's response has been disappointing, notably with respect to its prescribed methodologies for groundwater protection. It is not that the problem has been ignored but that the MOE's hastily assembled initiatives concerning groundwater vulnerability and the mapping of wellhead protection areas have been seriously under-funded, piece-meal, and in some cases scientifically ill-conceived.

Ontario is in urgent need of effective and reliable strategies for the protection and management of its ground and surface water resources. The task is considerable but the goal is achievable. There are no simple solutions and no easy shortcuts. To be effective, it is essential that groundwater protection be included as an integral part of an overall groundwater management plan that addresses both quality and quantity issues.

Wellhead protection methodologies can provide a viable approach for protection of Ontario's major municipal wells but require the development of three-dimensional models that are carefully calibrated and fine-tuned using both potentiometric heads and flows. This, in turn, demands an extensive and reliable hydrogeological database including a good quantitative knowledge of aquifer recharge, discharge and well production, together with a sound understanding of aquifer geometries, aquifer/aquitard properties and aquifer system behaviour. In addition, considerable work is required to develop appropriate and realistic land use controls for the protected areas, and the wellhead protection methodology requires further development to consider water quality impacts (i.e. contaminant concentrations) and not simply travel times to the well.

A serious limitation of the wellhead protection zone approach is that it will not protect groundwater in the aquifer beyond the well's zone of contribution. In this regard, aquifer vulnerability/sensitivity/susceptibility mapping can provide a useful supplementary approach, but the definitions of these terms need careful re-examination in light of the Province's needs, and the approach needs to be made more quantitative and scientifically defensible through a more rigorous appreciation of soil/rock properties and the incorporation of vertical hydraulic gradients. Travel times are not the only consideration when evaluating the damage contaminants released at the surface can cause to underlying aquifers. Thus, serious consideration should also be given to the incorporation of attenuating processes such as dilution and degradation, in the susceptibility analysis.

Ultimately, it must be recognized that the successful implementation of groundwater protection measures in Ontario will demand a serious commitment of funds and resources to advance significantly our basic understanding of the Province's groundwater resources and provide key input data.

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EFFECTS OF ARTIFICIAL STORMWATER INFILTRATION ON URBAN GROUNDWATER ECOSYSTEMS

Ecological Issues in Urban Groundwater

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Abstract: Urban groundwater is commonly recharged artificially by stormwater runoff. This paper focuses on the effects of artificial stormwater infiltration on groundwater ecosystems using data from a specific study site. It is demonstrated that at this site artificial stormwater infiltration increased local fluxes of organic matter into the groundwater ecosystems. An insufficient residence time of stormwater in the vadose zone resulted in organic matter reaching the water table and subsequent oxygen consumption. The invertebrate assemblages in the underlying urban groundwater system were enhanced, probably because organic matter enrichment stimulated microbial production. Thus, urban groundwater biodiversity was higher under stormwater infiltration basins for shallow water-table conditions. Furthermore, groundwater biodiversity peaked at the water table, and decreased with depth. Invertebrate assemblage composition showed a vertical shift with depth below the water table, thereby suggesting that competition processes occur among taxa. This work highlights interesting ecological issues in groundwater and provides several recommendations for urban stormwater management.

Key words: stormwater; infiltration basins; artificial recharge; wastewater; grey water; ecosystems; biodiversity; dissolved organic carbon; dissolved oxygen; invertebrates; microbial populations; vadose zone; organic matter

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1. INTRODUCTION

1.1 Groundwater as a Living System

Despite their intensive use for drinking water, agriculture and industry, groundwaters are among the most poorly described ecosystems (Danielopol, 1989; Marmonier *et al*., 1993). Indeed, groundwater ecology and biodiversity have only recently received attention. Nevertheless, groundwaters, present in karstic (e.g. limestone), fissured (e.g. granitic), and porous (e.g. alluvial) aquifers, are inhabited by microbes (Gounot, 1991; Chapelle, 1993; Claret *et al*., 2003), protozoa (Golemansky and Bonnet, 1994; Strauss and Dodds, 1997), and invertebrates (Ginet and Decu, 1977; Culver, 1982; Danielopol, 1980; Danielopol *et al*., 1989, 2003; Camacho, 1992; Juberthie and Decu, 1994; Marmonier *et al*., 1993; Gibert *et al*., 1994, 1997; Malard and Hervant, 1999; Datry, 2003). Depending on the nature of their occurrence in the subsurface, invertebrates found in groundwater are classified into three ecological categories (Thienemann, 1926; Gibert *et al*., 1994). Stygoxenes are surface water species accidentally found in groundwater; stygophiles spend their entire lives in either surface or subsurface waters; and stygobites are true groundwater species, completing their entire life cycle in subsurface waters. Stygobite species exhibit strong morphological, metabolic, and behavioural adaptations relative to surface invertebrates (Gibert *et al*., 1994). Figure 1 summarizes the typical diversity of stygobite species found in porous aquifers.

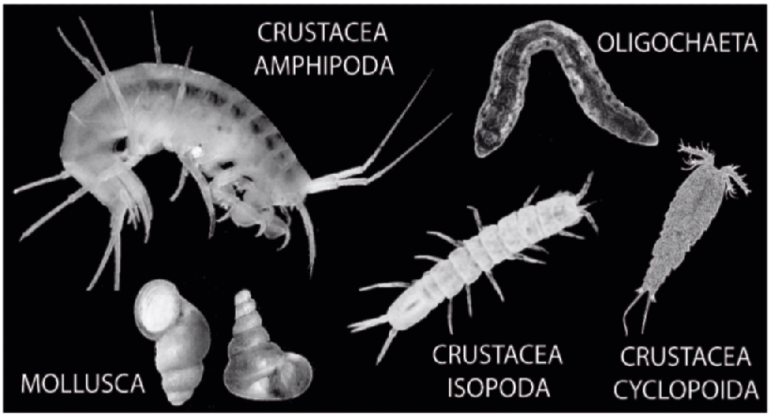

Figure 1. Stygobite invertebrates typically present in porous aquifers. Animal size varies from hundreds of μm (e.g. Cyclopoïda) to a few cm (e.g. Amphipoda).

1.2 Conceptual Framework for Studying Groundwater Ecosystem Responses to Urban Stormwater Infiltration Practices

In groundwater systems, permanent darkness prevents photosynthesis. Such heterotrophic systems are heavily dependent on organic matter and oxygen provided by recharge (Ginet and Decu, 1977; Gibert *et al*., 1994; Malard and Hervant, 1999; Baker *et al*., 2000). Organic matter fluxes in groundwater ecosystems are regulated partly by the soil/vadose zone system and partly by the geochemical processes occurring in the upper layers of groundwater (Ronen *et al*., 1987; Starr and Gillham, 1993; Malard and Hervant, 1999; Pabich *et al*., 2001). Upon reaching the water table, organic matter is consumed along a predictable sequence of oxidation-reduction reactions (Champ *et al*., 1979; Malard and Hervant, 1999). Aerobic respiration occurs first, supplied by dissolved oxygen (DO) as a terminal electron acceptor. Once anoxic conditions develop (i.e. after complete consumption of DO), nitrates, Mn^{2+}, Fe^{3+} and sulphates are alternatively used as terminal electron acceptors, depending on the prevailing oxidation-reduction potential (redox). In the case of highly reducing conditions, methanogenesis may eventually occur. These reactions are typically driven by a consortium of microbes interacting intimately with invertebrates. The invertebrates graze the microbial biofilm and modify its physical habitat by bioturbation (sedimentary reworking), causing its permanent regeneration and stimulation (Cummins and Klug, 1979; Danielopol, 1989; Hakenkamp, 1991; Boulton, 2000; Mermillod-Blondin *et al*., 2005). Several studies have shown that organic matter, expressed as dissolved organic carbon (DOC), is consumed rapidly within the first few metres below the water table, resulting in sharply decreasing vertical gradients of DOC and DO concentrations (Ronen *et al*., 1987; Malard and Hervant, 1999; Pabich *et al*., 2001). Hence, by regulating DOC and DO fluxes, biogeochemical processes occurring in the upper layers of groundwater systems affect the physicochemical and biological composition of deeper groundwater.

Urbanization results in the widespread development of impervious surfaces. This profoundly changes the recharge regime of aquifers, but does not necessarily decrease the total amount of aquifer recharge (Yang *et al*., 1999; Lerner, 2002). Diffuse recharge from rainfall is limited by impervious urban surfaces. In contrast, the artificial infiltration of stormwater runoff into the ground by means of infiltration basins is being used increasingly as an alternative to the direct disposal of stormwater into streams (Chocat, 1997; Pitt *et al*., 1999). Hence, in urban areas, the diffuse natural infiltration of rainwater into the ground is progressively replaced by the highly localized infiltration of large volumes of stormwater generated by urban runoff. This results in a clear spatial concentration of solute, contaminant, and organic matter fluxes in groundwater. Conceptually, three scenarios may describe the responses of groundwater ecosystems to increasing inputs of DOC (Figure 2). DOC reaching the water table depends on stormwater DOC concentrations, infiltration rate and vadose zone filtering capacity. The latter relies strongly on the residence time of infiltrating stormwater in the vadose zone system. In various settings with similar soil types, land use, and rainfall conditions, this residence time and subsequent DOC inputs at the water table are determined primarily by the vadose zone thickness (Ronen *et al*., 1987; Starr and Gillham, 1993; Datry, 2003).

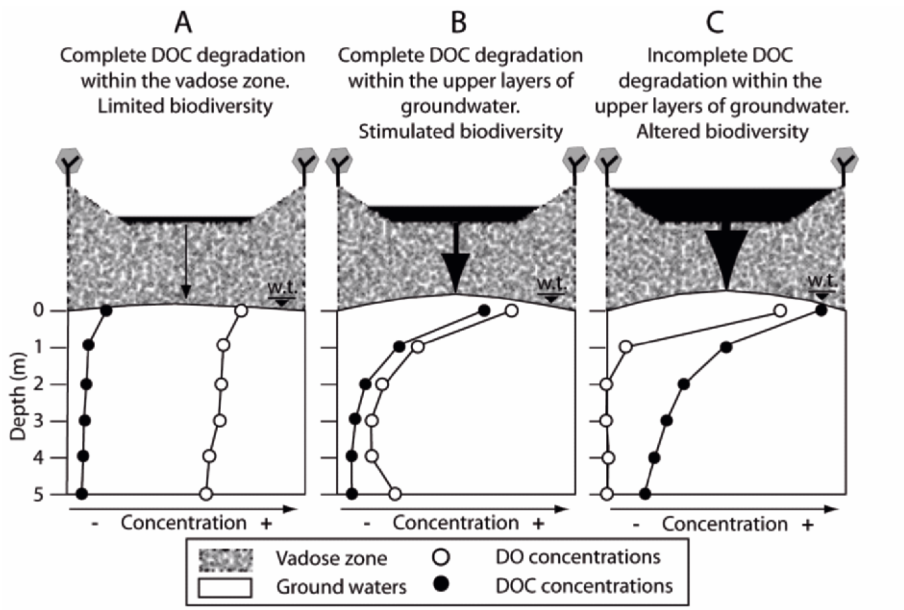

Figure 2. Possible scenarios for the response of groundwater ecosystems to artificial recharge by urban stormwater runoff. Responses are illustrated by the vertical distributions of dissolved oxygen (DO) and dissolved organic carbon (DOC), and by indications of biodiversity in the upper layers of the groundwater.

The first scenario (A) corresponds to weak refractory-DOC supply at the water table, following a complete degradation of the biodegradable fraction in the vadose zone system. There is strong evidence that biogeochemical processes and microbial communities are DOC-limited in such groundwater systems (Jones *et al*., 1995; Malard and Hervant, 1999; Baker *et al*., 2000). Hence, in this scenario, urban stormwater infiltration does not affect groundwater ecosystem functioning or biodiversity. In the second scenario (B), higher DOC fluxes reaching the water stimulate microbial activity through aerobic respiration. Biodiversity is promoted because invertebrates take advantage of the increase in microbial biomass. Strong vertical gradients of DOC and DO exemplify the intense biological activity in the first few metres below the water table (Ronen *et al*., 1987; Pabich *et al*., 2001). The third scenario (C) simulates massive DOC inputs at the water table that exceed the ecosystem assimilation capacity. Microbial activity is strongly stimulated and DO completely disappears in the water table region (Ronen *et al*., 1987; Starr and Gillham, 1993). Excessive DOC, as well as anaerobic respiration end-products (i.e., NH_4^+, Mn^{2+}, Fe^{2+}, HS^-, CH_4), accumulate and migrate to deep groundwater. Density of invertebrates does not peak despite an increase in microbial production, because groundwater organisms cannot live permanently in anoxic groundwater (Malard and Hervant, 1999).

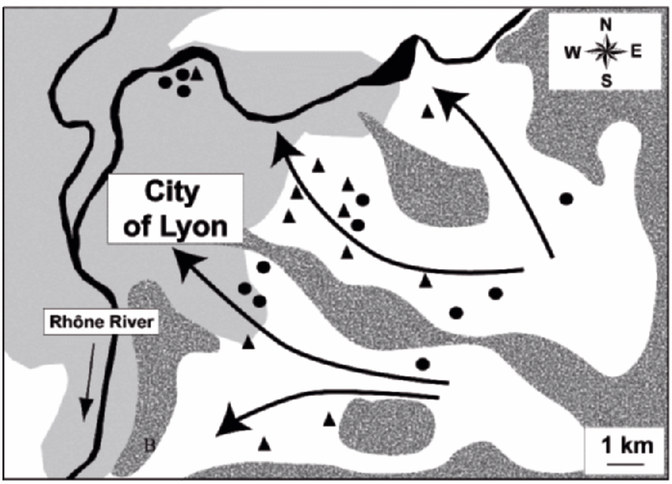

Figure 3. Location of the selected sites over the eastern aquifer of Lyon. Triangles and circles represent, respectively, stormwater infiltration basins and reference sites. Light and dark grey patterns correspond, respectively, to densely urbanized areas and to moraine hills. Arrows indicate regional groundwater flows in the 4 glaciofluvial corridors.

This theoretical model was used as a holistic framework to investigate the responses of groundwater ecosystems to artificial stormwater infiltration practices in a set of studies by Malard *et al*. (1999) and Datry (2003). The methodological approach comprised two complementary aspects. First, a detailed study was performed at a test site with a 2 m-thick vadose zone selected to test the hypotheses of scenario B of the model. Particular attention was paid to the vertical distribution of solutes and invertebrates in the upper layers of the stormwater-recharged groundwater by means of an original sampling design. Described elsewhere (Datry *et al*., 2004; Malard *et al*., 2004), this sampling design fully integrated the spatiotemporal dynamic of stormwater plume in groundwater. Second, a multi-site study was carried out over a glaciofluvial aquifer in the eastern part of Lyon, France (Figure 3). The 24 selected sites, 13 of which were stormwater infiltration basins and 11 of which were considered as reference sites, covered a large range of vadose zone thickness.

The present paper provides a synthesis of the major results of this work. Additional results can be found in Datry (2003) and Datry *et al*. (2003a, b, c; 2004). The first part of the paper focuses on groundwater organic matter enrichment resulting from urban stormwater infiltration. The second part emphasizes the responses of groundwater invertebrate assemblages below stormwater infiltration basins.

2. ARTIFICIAL STORMWATER INFILTRATION: A POTENTIAL CAUSE OF URBAN GROUNDWATER ANOXIFICATION

Urban stormwaters are highly contaminated, as a result of atmospheric wash out and runoff from urban surfaces (Chebbo *et al*., 1995; Pitt *et al*., 1999). Most contaminants, primarily metals and hydrocarbons, are adsorbed on stormwater-derived sediments (Chebbo *et al*., 1995; Masson *et al*., 1999). Highly contaminated stormwater-derived sediments are stored in the infiltration beds of basins, generally composed of coarse gravel, natural soil or subsurface deposits (e.g. fluvial or glaciofluvial deposits), or geotextiles. Several authors have investigated groundwater composition below infiltration basins (e.g. Appleyard, 1993; Barraud *et al*., 1999; Pitt *et al*., 1999; Fisher *et al*., 2003; Datry *et al*., 2004) but none found major contamination of groundwater by metals or hydrocarbons, thereby indicating that infiltration beds are efficient sinks for such contaminants. In contrast, as predicted by Mikkelsen *et al*. (1994), it is demonstrated that infiltration beds of basins act also as a source of groundwater contamination by other contaminants. Indeed, within infiltration beds, the mineralization of organic matter in stormwater-derived sediments produced DOC, phosphates, and ammonia, and caused the complete anoxification of infiltrating stormwater.

Hence, the spatial concentration of organic matter fluxes into groundwater caused by urbanization was largely amplified by geochemical processes occurring in the infiltration beds of basins. As a result, DOC fluxes towards groundwater were 340-fold higher below the selected test site than at a nearby reference site where natural recharge prevailed (i.e., 68 *vs*. 0.2 g DOC /y/m^2; Datry, 2003). Consequently, significant groundwater enrichment by DOC and phosphates was measured below the stormwater basin (Figure 4; Datry *et al*., 2004). In the first 2 metres below the water table, DOC enrichment was more pronounced during rainfall events (Figure 4).

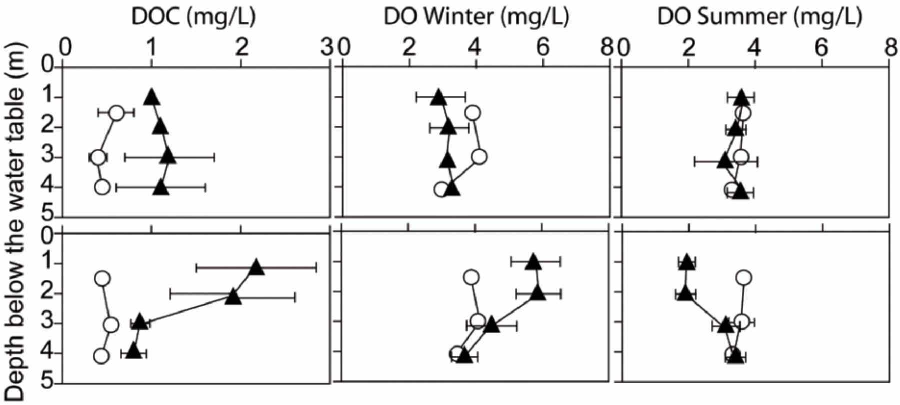

Figure 4. Vertical distribution of dissolved organic carbon (DOC) and oxygen (DO) in groundwater beneath a reference site (white circles) and a stormwater infiltration basin (black triangles) for dry (upper panel, n=3) and rainy (lower panel, n=3) weather.

Such a configuration illustrated typically scenario B of the predictive model. Nevertheless, DO gradients were more complex than those expected and displayed seasonal variation (Figure 4). Within the first two metres below the water table, winter rains caused weak oxygenation of groundwater, whereas summer rains induced DO depletion. A two-ended mixing model was used to predict DO concentrations expected in groundwater, with regard to DO concentrations in infiltrating stormwater (Datry *et al*., 2004). Comparisons with measured DO concentrations indicated a permanent DO deficit at the water table. This DO deficit was ascribed to: 1) anoxic stormwater inputs at the water table resulting from microbial respiration occurring in the infiltration bed; and 2) *in situ* aerobic respiration following DOC inputs. Seasonal variations of DO in groundwater corresponded to differences in the respiration rate in the infiltration bed, hot summer rains causing a strong stimulation of microbial activity and subsequent inputs of anoxic water at the water table (Figure 4).

DOC enrichment of groundwater resulting from artificial stormwater infiltration was also detected by Barraud *et al*. (1999). DOC concentrations in impacted groundwater were 21.5 mg/l, as opposed to 0.9 mg/l at a nearby reference site. The authors concluded a weak impact on groundwater, however, because no significant increases in metal and hydrocarbon concentrations were detected. Other evidence of severe groundwater anoxification caused by artificial stormwater infiltration exists in the literature. For instance, severe DO depletion (i.e., DO < 0.5 mg/l) was detected by Fisher *et al*. (2003) below several infiltration basins in southern New Jersey, USA. DO depletion was associated typically with high NH_4^+ and Fe^{2+} concentrations, strongly suggesting that anaerobic respiration occurred. These results show that DOC inputs below stormwater basins may, in some cases, exceed the groundwater ecosystem assimilation capacity, as illustrated by scenario C of the conceptual model of Figure 2.

Groundwater anoxification caused by artificial stormwater infiltration is controlled by the residence time of infiltrating stormwater in the vadose zone. The multi-site study indicates that DOC and DO groundwater concentrations below infiltration basins are correlated significantly and linearly to the vadose zone thickness (DOC negatively correlated with r^2=0.38; DO positively correlated with r^2=0.30). Moreover, groundwater composition, particularly DOC and DO concentrations, was dissimilar at stormwater basins and reference sites for vadose zones thicker than 10 m (p.value > 0.05; Figure 5). In contrast, groundwater was significantly enriched in DOC (p.value < 0.01) and depleted in DO (p.value < 0.05) for shallow water-table configurations (Figure 5). Similarly, Starr and Gillham (1993) showed that differences in groundwater DOC and DO between two sites in an unconfined aquifer in southern Ontario, Canada, were primarily explained by differences in the thickness of a sandy vadose zone.

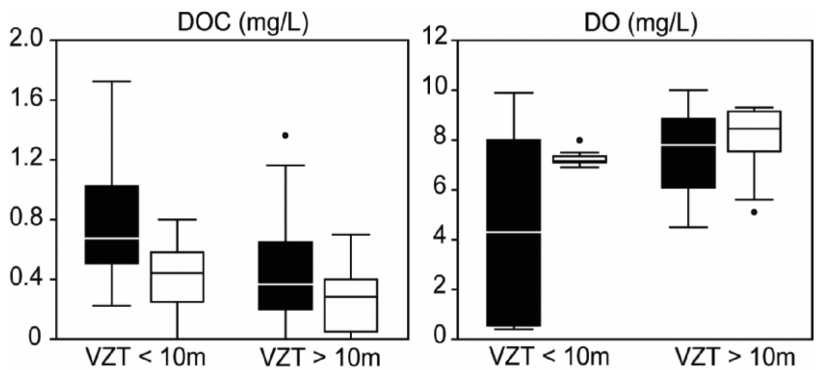

Figure 5. DOC and DO concentrations measured in groundwater for two categories of vadose zone thickness (VZT, < and > 10 metres) under reference sites (white boxes, n=5 and 6) and infiltration basins (black boxes, n=5 and 8). Top of box is the 75th percentile, bottom 25th percentile, and horizontal line is median; T-lines indicate total range; points represent outliers.

3. ARTIFICIAL STORMWATER INFILTRATION PROMOTES URBAN GROUNDWATER BIODIVERSITY

The theoretical framework of this work hypothesizes that invertebrate assemblages in groundwater are promoted by DOC supply as long as DO concentrations remain sufficient (i.e. > 0.3 mg/l, Malard and Hervant 1999). This implies that biodiversity distribution patterns are predictable in urban groundwater impacted by stormwater infiltration. Firstly, invertebrates might preferably colonize the recharge areas of aquifers, as these areas receive higher DOC fluxes. For shallow water-table conditions, densities should thus be higher under infiltration basins. Secondly, in recharge areas, invertebrates might preferably colonize the water-table region, where DOC concentrations peak. Under infiltration basins, densities should be higher in the first metre below the water table and then decrease with depth.

No differences were measured in groundwater invertebrate assemblage densities and diversity between recharge and non-recharge sites in the case of thick vadose zones (p value > 0.05; Figure 6). In contrast, it is demonstrated that shallow water-table urban groundwater artificially recharged by stormwater harboured invertebrate densities 7 times higher than in reference sites (p value < 0.01; Figure 6; Datry, 2003). These results support largely the hypothesis that life in groundwater systems is limited by DOC supply. However, groundwater artificially recharged with stormwater did not harbour a statistically higher number of species, whatever the vadose zone thickness being considered (p value > 0.05; Figure 6). Earlier studies conducted by Husmann (1975) in phreatic groundwater along the Fulda River, Germany, and by Sinton (1984) in the Templeton gravel aquifer, New Zealand, showed also that the density of groundwater invertebrate assemblages increased in response to organic matter enrichment caused by the infiltration of sewage effluent.

At the test site, invertebrate densities were 10-fold higher in groundwater under the infiltration basin, regardless of the hydrological conditions (p value <0.01; Figure 7). Hence, invertebrates integrated the long term changes occurring in their environment. Moreover, under the basin, invertebrate densities were 12-fold higher 1 m below the water table than 4 metres below, with 411 $\pm$ 378 invertebrates per 50L at 1 m depth compared with 33 $\pm$ 16 at 4 m depth. Densities exhibited a sharp decreasing gradient with depth below the water table whatever the meteorological conditions (Figure 7; Datry, 2003). This gradient could not be ascribed solely to artificial stormwater infiltration, because it also existed at a nearby reference site (Figure 7). Decreasing gradients of groundwater invertebrate densities with depth below the water table may be a common pattern in groundwater systems, as long as the water-table region remains oxic. Such vertical distribution gradients were previously detected in a non-urbanized area of the alluvial aquifer of the Rhône River (France) by Mauclaire and Gibert (2001), and in the Lobau aquifer (Austria) by Pospisil (1994).

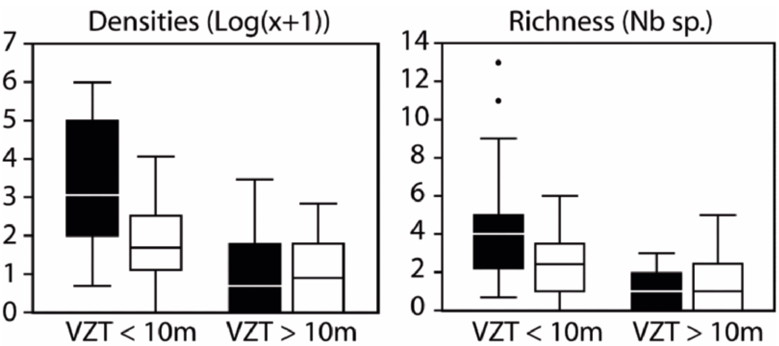

Figure 6. Densities and richness of invertebrate assemblages in groundwater for two categories of vadose zone thickness (VZT, < and > 10 metres) under reference sites (white boxes, n=5 and 6) and infiltration basins (black boxes, n=5 and 8). Animal densities log(x+1) transformed; Nb sp. = number of species.

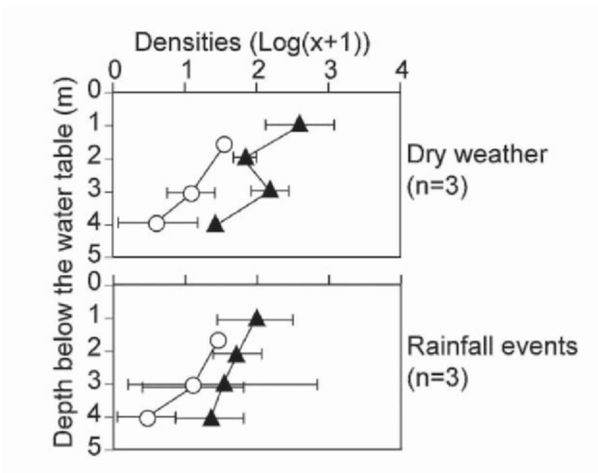

Figure 7. Vertical distribution of invertebrate densities in groundwater under a stormwater infiltration basin with a thin vadose zone (i.e. 2 m, black triangles) and at a nearby reference site (white circles).

At the test site, we detected a vertical shift in the composition of groundwater invertebrate assemblages with increasing depth below the water table, whatever the hydrological conditions (Figure 8; Datry, 2003). Some species, of which a majority were stygophiles, occurred preferentially in the vicinity of the water table, whereas others, the majority of which were stygobites, colonized deeper layers of groundwater (Figure 8). Such a vertical shift in the composition of invertebrate assemblages has been described previously in the hyporheic zones (i.e. water-saturated interstices below and at the margins of rivers containing a certain proportion of surface water) of many rivers (e.g. Danielopol, 1980; Meštrov *et al*., 1983; Dole-Oliver *et al*., 1993; Brunke and Gonser, 1999). However, the results outlined here are the first to provide evidence of a shift in the composition of groundwater invertebrate assemblages with increasing depth below the water table. As proposed by Brunke and Gonser (1999), based on observations made in the hyporheic zone of the Töss River, this compositional shift in invertebrate assemblages could reflect interactive processes between species (i.e. competition). Some species, including several stygobites, might be limited by food supply (i.e. DOC inputs at the water table) but they were out-competed in the uppermost layers of groundwater by other species, including several stygophiles (Figure 8). More data are needed to test definitively this model of community organization in groundwater.

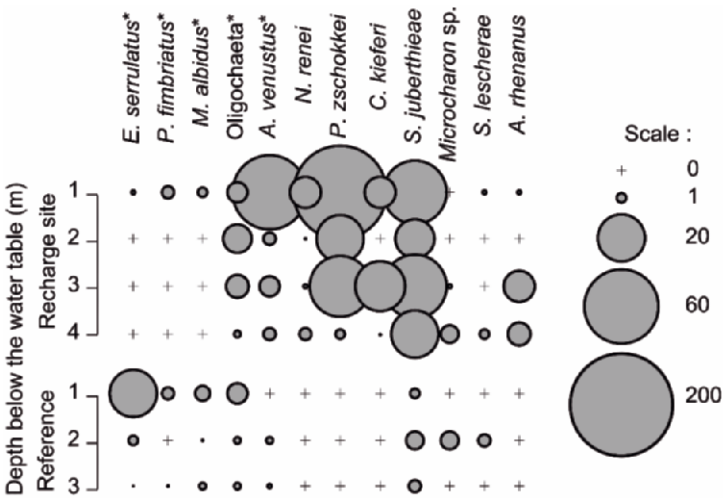

Figure 8. Vertical distribution of selected species below the water table at the recharge site (upper panel) and reference site (lower panel). Asterisks indicate stygophile species. Diameters of circles are proportional to the mean number of individuals collected per 50L of pumped water (n = 6 dates). Crosses correspond to sites where no individuals were collected.

4. CONCLUSIONS AND PERSPECTIVES

Within a conceptual framework and using an original methodological approach, it is demonstrated here that artificial stormwater infiltration practices could lead to urban groundwater anoxification. In thin vadose zones (i.e. < 10 m), the concentration of organic matter fluxes towards groundwater ecosystems caused strong DOC inputs at the water table and subsequent DO consumption. However, groundwater biodiversity was promoted as long as groundwater remained oxic. This demonstrates that food supply limits the development of invertebrate assemblages in groundwater systems. Hence, we suggest that invertebrates are likely to be concentrated in recharge areas and at the water-table region of groundwater systems.

These results provide interesting perspectives for further study of groundwater ecosystems. Because invertebrates are known to interact strongly with microbial biofilms, they act as indicators and also as instigators of the functioning of groundwater ecosystems. Thus, we suggest further research should focus on the potential role of groundwater invertebrates in regulating microbial respiration and DOC and DO fluxes in the water-table region of aquifers. Laboratory microcosms are being used to study the influence of invertebrates on organic matter processing in the beds of stormwater infiltration basins (Mermillod-Blondin *et al.*, 2005). Furthermore, a discrepancy is likely to exist between the metre scale of our sampling design and the centimetre scale at which geochemical processes occur. Indeed, it is suspected that the vertical gradients of DOC, DO and invertebrates are sharper, especially in the first metre below the water table. Hence, new sampling design and devices are being developed to increase the spatial resolution of investigations.

These results also highlight several points of great concern for urban groundwater management. First, managers should ban stormwater infiltration practices in locations with shallow water-tables. Moreover, when designing stormwater infiltration basins, they should avoid artificially reducing the vadose zone thickness. Indeed, in this study, 10 m of glaciofluvial deposits were needed to prevent the influence of infiltration basins on groundwater. Second, sampling protocols and schedules for monitoring the effects of existing stormwater basins must clearly: 1) use a systematic groundwater control point; and 2) take into account the spatio-temporal dynamics of the stormwater plume flowing into the groundwater. Moreover, pertinent integrative parameters, such as DO, constitute powerful and cheap tools for examining whether groundwater ecosystem assimilation capacity is exceeded under infiltration basins. Finally, even if further research is needed, invertebrate assemblages represent potential biological indicators of urban groundwater quality, because they integrate long-term transformations of their environment. Hence, further studies of the impact of stormwater infiltration, or other urban-derived practices affecting urban groundwater systems, should include a biological characterization of groundwater.

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SECTION VI:

REMEDIATION

ORGANIC CONTAMINANT REMEDIATION IN URBAN GROUNDWATER

A Review of Groundwater Remediation Technology Development

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Abstract: The historical development and current status of groundwater organic contaminant remediation is reviewed. Such development awareness underpins effective implementation of current and future remediation strategies. Significant remediation has been attempted since the 1980s with the US initially pre-eminent following a widespread programme of pump-and-treat that later developed to more innovative *in situ* technologies aimed at accelerating source mass removal or control. The emergence of the NAPL (non-aqueous phase liquid) paradigm in the 1980s was critical to both understanding the failings of pump-and-treat and later remediation strategies that recognized the importance of separate remedial strategies for source and dissolved plume zones. A plethora of *in situ* technologies have emerged over the past two decades or so including vapour-based methods, enhanced mass removal chemical-based methods, monitored and enhanced natural attenuation (bioremediation) and more passive technologies such as permeable reactive barriers. A summary of remediation issues is presented that requires on-going consideration for remediation efforts to remain central to the sustainable management of urban groundwater resources.

Key words: remediation; organic contaminants; contaminated land; NAPL.

1. INTRODUCTION

National estimates of contaminated land and groundwater present in Britain at the year 2000 indicated that 100,000+ contaminated sites could exist. Britain's contaminated-land liabilities were put at ~£15 billion with some 360,000 hectares of land potentially thought to pose risk of significant

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harm (Anon, 2000). The occurrence of anthropogenic organic chemicals in urban groundwater is of particular concern and came to prominence in the late 1970s and early 1980s in industrially developed nations of Europe and North America (Zoeteman *et al.*, 1981). Awareness of the problem developed in parallel with advances in chemical analysis methods (Bellar and Lichtenberg, 1974) with earliest awareness occurring in the Netherlands and the US (United States).

Discovery of contamination inevitably leads to groundwater remediation consideration and activity. In the US, national/state drinking water surveys prompted national/state groundwater surveys (~1976-79) that in turn prompted much site-specific investigations. By the late 1970s it was clear that VOCs (volatile organic compounds) such as trichloroethene (TCE) were a significant problem (Pankow *et al.*, 1996). The enacting of CERCLA (Comprehensive Environmental Response, Compensation and Liability Act), commonly known as "Superfund", lead to remedial action at many contaminated sites in the US from the 1980s onwards.

This paper overviews the development of groundwater remediation technologies relevant to organic contaminants and summarizes current remediation concerns relevant to the sustainable management of urban groundwater systems.

2. PUMP-AND-TREAT

The primary response to the discovery of extensive groundwater plumes in the US was pump-and-treat (P&T) remediation of groundwater. P&T, comprising the extraction contaminated groundwater via extraction wells and treatment at surface via sorption to granular activated carbon or air stripping, was widely practiced (US EPA (Environmental Protection Agency) 1989, 1998). Expectations of P&T were high in the US. Remediations were expected to complete within a few years and goals set in the hope of achieving restoration, i.e. cleanup to pristine conditions, or else low drinking water standards. However, many P&T operations displayed slowly declining concentrations (tailing), even “plateau” quasi-stable concentrations in some cases orders of magnitude above drinking water standards. The potential for some P&T schemes to operate for perhaps decades was gradually recognized (Mackay and Cherry, 1989; Zhang and Brusseau, 1999).

Emergence of NAPL (non-aqueous phase liquid) paradigms occurred in the 1980s in parallel with this P&T activity (Schwille, 1988). NAPL implications for P&T performance were conceptualized by Mackay and Cherry (1989). The NAPL paradigm presented caused a fundamental shift in the approach to sites. Key recognitions were: (i) source zones were often present as subsurface NAPL; (ii) NAPL sources contained significant mass able to generate dissolved plumes for extremely long time periods (decades); and, (iii) many common groundwater contaminants, e.g. chlorinated solvents, were derived from dense NAPLs, i.e. DNAPLs, that could reside below the water table. Importantly, it was recognized that P&T would have negligible impact on the depletion of mass in NAPL source zones of even just moderate size. Near source P&T wells would, at best, serve as source-containment wells. Such knowledge lead to revised remediation strategies for contaminated-groundwater sites. Source zones and dissolved plumes that developed down-gradient of the source were considered as separate entities within an overall site remediation plan. Plume removal without on-going source management, or source removal would just lead to re-growth of the dissolved-phase plume from that source (Pankow and Cherry, 1996).

Not all developed countries undertook significant P&T. During the 1980s, the UK expended most effort on diagnosing the extent of the groundwater problem rather than active groundwater remediation (Rivett *et al*., 1990). The negative P&T experience in the US caused the UK to veer away from the technology (Rivett *et al*., 2002). More positively, the US experience has lead to a generally more informed use of P&T, so-called "smart P&T" (Hoffman, 1993).

Key realizations were that aquifer restoration should not be anticipated and attainment of drinking water standards in source areas is not feasible within realistic time and financial constraints. Setting of pragmatic remediation goals became a key outcome of the early P&T experiences and has lead to the now widely adopted risk-based clean-up criteria approach in many countries (EA, 1999). Due to attenuation of contaminants, with biodegradation often the most significant process, risk-based approaches allow higher concentrations to remain in site with drinking water standards still being met at defined compliance points or receptors at distance.

3. VAPOUR-BASED TECHNOLOGIES

Innovative technologies to accelerate source mass removal or destroy that mass *in situ* were hence developed. Soil vapour extraction (SVE) methods were widely adopted in the mid to late 1980s, particularly in the US. Its ready applicability to the mitigation of hydrocarbon vapour risks was a key driver for its widespread application to fuel spill sites. Closely related vapour-based techniques subsequently developed to enhance unsaturated zone source mass and floating (light) LNAPL. Combined groundwater and vapour extraction methods and high vacuum extraction methods have allowed far more effective removal of floating hydrocarbons and contamination in lower permeability settings (Oostrom, 2005).

Air sparging (Johnson *et al*., 1993) was developed in the 1990s due to SVE's inability to remediate contamination present much below the water table. Air sparging involves injection of high-pressure air in well points located less than ~2 m below the water table. Air then rises in the formation into which VOCs partition ultimately removed by a SVE installed in the unsaturated zone above. A key concern with the technology has been the preferential movement of injected air phase through the more permeable strata causing contaminant mass to persist in lower permeability strata (Tomlinson *et al*., 2003).

4. ENHANCED MASS-REMOVAL TECHNOLOGIES

Key enhanced mass-removal technologies aimed at organic source zone removal include *in situ* chemical oxidation (ISCO) and surfactant-enhanced aquifer remediation (SEAR). ISCO involves the use of strong oxidants, e.g. the hydroxyl radical formed from Fenton's reagent (hydrogen peroxide and ferrous iron (Fe^{2+})) has been perhaps the most used method to date (Chapelle *et al*., 2005). Less severe oxidants such as permanganate are also used (Schnarr *et al*., 1998) on for example chlorinated ethenes (notably PCE and TCE). Reaction occurs on the dissolved-phase emitted from the NAPL surface. This causes a steepened concentration diffusion gradient and over 1-2 orders of magnitude increase over natural dissolution of solvent DNAPLs. There is potential via diffusion of permanganate oxidant to treat contaminant present in say low permeability clays or a matrix adjacent to a fracture. The consumption of oxidant due to the unwanted oxidation of aquifer natural organic matter requires consideration.

Surfactants most often used for SEAR are typically used within food products, e.g. Tween 80. They may enhance solubilities by up to a factor of ~20 leading to accelerated source dissolution (Abriola *et al*., 2005; Fountain *et al*., 1996). A concern for DNAPL sites is the avoidance of inadvertent DNAPL re-mobilization downward, usually minimized by careful surfactant choice. Other solubility enhancers include cyclodextrins (Tick *et al*., 2003), alcohol floods (Brooks *et al*., 2004). Significantly accelerated mass removals relative to simple P&T flushing are achieved.

5. NATURAL ATTENUATION-BASED TECHNOLOGIES

5.1 Monitored Natural Attenuation

Awareness of the importance of biodegradation of organic chemicals to cause significant natural attenuation (NA) of contaminants increasingly developed in the 1980s and 1990s (Wiedemeier *et al*., 1999; Barker *et al*. 1987). Much research effort has focused upon determining the controls imparted by specific electron acceptor conditions and the potential for biodegradation of more recalcitrant chemicals, e.g. chlorinated organics and MTBE (Wilson *et al*., 2002). Rates of biodegradation vary substantially, e.g. a specific organic such as PCE may biodegrade under very reducing conditions, e.g. sulphate reducing or methanogenic, but not degrade under aerobic or nitrate-reducing conditions. Specific bacteria strains may be required, e.g. the biodegradation of cis-DCE that often accumulates at TCE and PCE sites may be biodegraded by specific dehalorespiring bacteria (Yang *et al*., 2005) that may not always be present at sites.

The use and acceptance of monitored natural attenuation (MNA) as a remediation option has been gradually forthcoming. MNA aims to demonstrate that a dissolved-plume emanating from a source does not pose significant risks to receptors and that active remedial action (via technologies discussed above and others) is not required. The viability of MNA as a remedial approach has been shown for relatively biodegradable hydrocarbons by many, e.g. Wiedemeier *et al*. (1999), Barker *et al*., (1987), King and Barker (1999). Greatest debate on MNA relevance has occurred for the chlorinated solvents. Although chlorinated solvent biodegradation is frequently shown at sites (Wiedemeier *et al*., 1999), plumes can still reach greater than 10-km lengths (Jackson, 1998) suggesting NA is of limited influence in such cases. On-going research will remain targeted to: (i) establishing the application of MNA to the more recalcitrant compounds such as MTBE; (ii) understanding of where maximum rates of biodegradation occur in plumes where mixed redox conditions exist (Thornton *et al*., 2001).

5.2 Enhanced Natural Attenuation (Bioremediation)

Conditions for biodegradation may not always be optimal, e.g. specific bacteria, nutrients and, or electron acceptors are limiting factors. Enhanced NA (*in situ* bioremediation) may then become necessary. NA has typically aimed to enhance the electron-acceptor conditions, particularly the supply of oxygen (Wilson *et al*., 2002). Oxygen may be supplied through a variety of means including the direct injection of air via technologies such as air-sparging and bio-venting. A widely employed method, however, has been the employment of the commercially available oxygen-release compound ORC® from which oxygen is slowly released over time from a Magnesium oxide-based solid phase (Koenigsberg and Norris, 1999).

For contaminants that are not aerobically degraded, e.g. many highly chlorinated organics, the alternative enhanced NA approach has been to promote anaerobic conditions (Murray *et al*., 2001). This may be achieved via relatively fast-release electron donor compounds that are generally low-cost organic waste products such as molasses, or else slow-release electron donors (SREDs) such as hydrogen-release compound HRC®, a polylactate ester that gradually releases a lactic acid that is fermented by anaerobic bacteria that gain carbon and energy whist releasing hydrogen that is used in the enhancement of bacterial reductive dechlorination of the chlorinated organics. Fast-release compounds may induce too rapid fermentation inducing unwanted methanogenesis.

6. PERMEABLE REACTIVE BARRIERS

The permeable reactive barrier (PRB) concept was introduced in the early 1990s (Gillham and O'Hannesin, 1994; O'Hannesin and Gillham, 1998). It involves the emplacement of a porous reactive media across the path of a dissolved-phase plume (Boshoff and Bone, 2005). Plume contaminants are removed via a biological/chemical reaction or physical process during their contact with the reactive media within the barrier. The passive, sustainable nature of this source containment technology over other more active technologies is significant. Abiotic PRBs include: zero-valent iron (ZVI) for chlorinated hydrocarbons and trace metals; zeolites for organics, metals and radionuclides; activated carbon for organics and metals, and "E-Clays" for organics (Boshoff and Bone, 2005). Biologically-based PRBs have also been developed for both aerobic and reducing anaerobic conditions.

The technology has continued to develop, for example: the ease and depths of installation has increased with recent developments in the use of degradable biopolymers; the use of catalysts, e.g. palladium, to enhance have been used to enhance ZVI reactivity; and, the use of colloidal (nano) iron particles to allow injections directly into geologic media. To date, the PRB technology has enjoyed significant success with many PRBs having now operated for towards a decade (US EPA, 2003).

7. FUTURE ISSUES

The future role of organic contaminant remediation in the sustainable management of urban groundwater resources is both a multi-faceted and challenging one. A summary of remediation issues are listed below that require on-going consideration for remediation efforts to remain central to the sustainable management of urban groundwater resources. Such issues, although common to many countries, will assume differing national importance due to the varying local- and country-specific urban scenarios.

- In many countries awareness and documentation of organic chemical contamination within urban groundwaters still remains poor. Anthropogenic organic chemical use may have been significant, awareness of health concerns potentially present, however, affordable and widely available trace organic chemical analysis is often lacking.
- A plethora of remediation technologies have now been developed. The challenge for *in situ* groundwater technologies is that the chosen technology will achieve the remediation and risk reductions sought in the specific site scenario that is always chemically and geologically heterogeneous and unique.
- Remediation options for deep, consolidated and fractured aquifer systems are limited, even should the contaminant distribution be confidently delineated. In many cases P&T or MNA may be the only option; further technology development is necessary.
- It is difficult to foresee extensive remediation of groundwaters occurring in low-income, or developing countries without substantial foreign financial assistance. The result would probably be the remediation of just high profile cases. Remediation needs may, nevertheless, be huge, e.g. Albu *et al*. (2002).
- In many countries use of urban groundwaters is in decline due to industrial closures. As such, there may then be little incentive to undertake expensive groundwater remediations with activities perhaps restricted to the land surface and, or shallow groundwater.
- NA of organic contaminants is highly variable depending upon the nature of the contaminant release, contaminant type and the specific subsurface environment. Although generalizations can be made, e.g. many hydrocarbons readily attenuate, our ability to monitor NA more efficiently at sites needs to increase.
- Water treatment at the point of use, e.g. supply well, is in many circumstances a cheaper alternative to source, or plume remediation. Although not ideal, it may represent the only pragmatic and cost-effective solution in some cases.

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RECENT APPROACHES FOR URBAN GROUNDWATER POLLUTION PREVENTION AND REMEDIATION

Analysis and Recommendations

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Abstract: Groundwater protection and pollution prevention must be based on an integrated and conclusive protection and safety philosophy. Out of this philosophy a uniquely integrated concept for protection standards can be developed and implemented into technical and legal standards. State-of-the-art technical standards and regulations for the storage and handling of hazardous material, as well as state-of-the-art groundwater protection, have to refer to the complete spectrum of physical, chemical and biological properties of potential contaminants. Hydrogeologists have to integrate the latest research results and developments of material science, chemistry, physics, biology and the life sciences into their conceptual models and must even include the "green technologies" which start to produce new underestimated hazards. Remediation of soil and groundwater contamination can only be successful, if all the aspects of science and technology are professionally integrated in the planning of the remediation process. In most pollution cases only a well-balanced combination of various methods can guarantee a successful remediation, depending on the geological conditions, the specific properties of the aquifer itself and the extent of the contamination. Finally transboundary aquifers and rivers and the development of international public utility companies urge hydrogeologists to work together on an international basis for the solution of common problems.

Key words: groundwater protection; remediation; pollution prevention; safety standards; contaminants; limonene; halogenated hydrocarbons; international public utility companies; transboundary issues; Germany.

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1. INTRODUCTION

Urban environments suffer largely from groundwater pollution by industry, agriculture and their own infrastructure of water mains and sewers. As well overexploitation and cut-off of urban groundwater by subsurface infrastructure can, by lowering and rising water levels, cause severe structural problems to buildings due to changing mechanical properties of the soil. In the worst case landslides can occur.

Thus state-of-the-art protection concepts for groundwater protection and remediation have to be developed and applied to save this essential urban resource for future generations. This paper will focus on the protection of urban groundwater from contamination by hazardous substances and on the basic principles of urban groundwater remediation.

2. POLLUTION PREVENTION

Existing philosophies and standards for groundwater protection and remediation vary from country to country. When seen from a neighbouring country's point of view, they sometimes seem to be completely strange and even contradictory. Historical developments, local customs, or regional economic structures have largely influenced the development of legal and technical standards. Early clean water acts and safety regulations date back to medieval times and had to undergo significant changes during the 19th and 20th century.

Some of the earliest modern environmental safety standards came into existence, when large fires were caused by flammable liquids leaking from tanks, releasing poisonous fumes and liquids, causing large numbers of casualties by breathing or ingestion. Thus these standards focussed on the inflammability and toxicity of materials but did not yet fully recognise the problem of groundwater pollution or else simply ignored it. Some other protection standards focused on liquid hydrocarbons only and ignored the existence of other dangerous liquids as well as gaseous or solid contaminants.

2.1 Halogenated Solvents – A Series of Errors

The most costly experience with respect to groundwater pollution was probably made with the use of halogenated hydrocarbons, which were developed in the late 19th century and tested for various purposes. The development started with substances like carbon tetrachloride and tetrachloroethene (perchloroethylene, PCE) and continued with trichloroethene (TCE), 1,1,1-trichloroethane (TCA) and other chlorinated agents; later bromide and fluorine were added as halogens. While the bromide-containing halogenated hydrocarbons, only widely used as fire-extinguishing agents, were banned at an early stage due to their high toxicity, the fluorine-containing halones (chlorofluorocarbons CFSs) encountered a boom in use as cleaning agents, refrigerants, fire-extinguishing-agents and narcotics in anaesthesia. Due to their high ozone-depleting capacity they were internationally banned in the early 1990's.

The "classical" chlorinated hydrocarbons were regarded as all-round chemicals suitable for nearly all purposes. Even medical treatment seemed possible with this group of substances and PCE was for example tested as an anthelmintic for foxes and dogs, which successfully killed hookworms and ascarids but caused liver lesions in some cases as well (Figure 1) (Hanson, 1927). Over a prolonged period, PCE was used for extraction processes of all kinds, mainly in the production of vegetable oils and aromas in food industry and in various chemical processes.

As they are not inflammable, halogenated hydrocarbons were widely introduced in the 1950's in order to replace highly inflammable hydrocarbons like benzine or benzene (benzol) in cleaning and extraction processes. The prevention service of the fire brigades almost forced dry-cleaners and chemical manufacturers to change their processes to the new solvents and fire underwriters happily reduced the insurance premiums. The first disadvantage turning up was the toxicity of some halogenated hydrocarbons which caused nervous diseases and (liver) cirrhosis, when intake by breath or ingestion occurred for a significant time or in a significant concentration. Beyond that, halogenated hydrocarbons could be significantly accumulated in all fats and lipophilic matters, causing long term adverse health defects.

Later on it was found that this group of chemicals, which was highly esteemed in degreasing and coating processes for its extremely low surface tension, was able to permeate walls and concrete floors in liquid and gaseous phase. No wonder that chlorinated hydrocarbons were found ubiquitously in almost every aquifer close to urban agglomerates (Toussaint, 1986). Cleanup of water resources contaminated with chlorinated hydrocarbons tuned out to be one of the most costly procedures, requiring billions of Euros worldwide to date and remaining a significant environmental legacy (Rijnaarts, 2000).

CRITICAL TESTS OF TETRACHLORETHYLENE AS AN ANTHELMINTIC FOR FOXES[1]

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INTRODUCTION

In a recent paper Hall and Shillinger (*2*)[3] called attention to the promising results obtained by them in critical tests of tetrachlorethylene as an anthelmintic for dogs. They found the drug very effective against hookworms and ascarids. In a later paper these authors (*3*) published protocols of tests on about 60 dogs. Their experiments indicated that the effective dose was 0.2 c. c. per kilo. At this dose rate tetrachlorethylene removed 195 of 196 hookworms in 9 infested dogs and all the ascarids—32—in 4 infested dogs. Dogs survived doses of 3, 5, 10, and 15 c. c. per kilo, which indicates that the margin of safety is high in the absence of contraindications. Of 64 dogs treated with doses ranging from 0.2 to 0.6 c. c. per kilo, 3 died from 1 to 3 days after treatment. These three animals were showing clinical symptoms of distemper at the time of treatment. The authors concluded that the contraindications for tetrachlorethylene probably are identical with those for the related compound, carbon tetrachloride. Liver lesions were noticed after the administration of tetrachlorethylene.

Figure 1. Tetrachloroethene for the medical treatment of foxes and dogs (Hanson, 1927).

2.2 Alternative Solvents – Even More Errors

Even replacing chlorinated hydrocarbons by new substances, supposed to be less dangerous, turned out to be a mistake. One of the liquids with nearly the same cleaning properties as the chlorinated hydrocarbons is limonene, a terpene extracted from orange skins, when producing orange-juice. It is highly flammable and has to be handled with more care, but it has a pleasant orange smell, which makes limonene a chemical acceptable to those handling it. It has been used in traces for decades as a perfume additive for cleaning agents and washing powders (Ullmann, 1981). In Germany limonene was introduced by the green movement as a natural, healthy and biodegradable cleaning agent for persistent stains like tar and grease. In the beginning it was sold in simple glass bottles without any specific description. A few years later it attracted the symbols for inflammable and irritating substances and shortly after disappeared from the displays in the green warehouses.

Within industry, limonene was introduced in paint-shops, precision mechanics and injection-moulding plastic industries, where chlorinated agents were too aggressive and petroleum-based hydrocarbons would interfere with the material or the coating. Liquid limonene has a very low surface tension and can easily leak through small fissures and pores into the underground and the aquifers. The low vapour-pressure causes it to be a mobile vapour which can penetrate the sub-surface in a similar manner to the chlorinated hydrocarbons. It is biodegradable in low concentrations in groundwater under aerobic conditions, consuming the available oxygen quickly and than remains in an anaerobic environment as a contaminant as persistent as PCE.

Barrett *et al*. (1996 and 1999) found in UK city aquifers ubiquitous, elevated concentrations of limonene. Due to its irritating and allergenic properties the use of limonene as a solvent in industrial processes is decreasing but it is still widely used as a perfume and aroma additive in various products.

2.3 Principles of Groundwater Protection

As a result of such experiences, groundwater protection and pollution prevention must be based on an integrated and conclusive protection and safety philosophy, starting at the potential source of the contamination. Out of this philosophy, a uniquely integrated concept for protection standards can be developed and implemented into technical and legal standards.

State-of-the-art technical standards and regulations for the storage and handling of hazardous material have to refer to the complete spectrum of physical, chemical and biological properties of potential contaminants. In various cases protection devices, emergency plans and cleanup strategies must be able to deal with solid, liquid and gaseous material, which are inflammable or explosive and have toxic and biohazard properties at the same time.

One of the simplest principles for protecting groundwater from pollution by hazardous chemicals is the double-barrier principle for all storage and handling facilities: tanks and pipelines must be double-walled and equipped with leak-control instruments. This can be vacuum sensors, testing liquids or electronic tape sensors which can be calibrated for the detection of various substances (Figures 2 and 3). For the storage of barrels and containers retention basins or trays can be used (Figure 4). Otherwise storage rooms must be fitted with floors, sealed by coatings with a proven resistivity to the stored material (Figure 5).A large number of groundwater contamination incidents are also caused by road or railroad traffic accidents. As these infrastructural features cannot be protected against spills, except when crossing groundwater catchment zones for short distances, safety features have to be applied to the vehicles and the payload. There are still too many deficiencies in the construction of transport systems and vehicles, and at least in Germany the restrictions on the storage and handling of dangerous goods are much stricter than those on transportation, where the double-barrier principle is not generally applied. While inside storage facilities chemical reactions have to be avoided by using separate storage rooms for e.g. acids and bases, inflammable and oxidising agents, all these dangerous good may be transported on the same truck according to European Union transport regulations. Thus, in case of an accident, unforeseen chemical reactions might produce contaminants, which are extremely difficult to be removed from soil and groundwater.

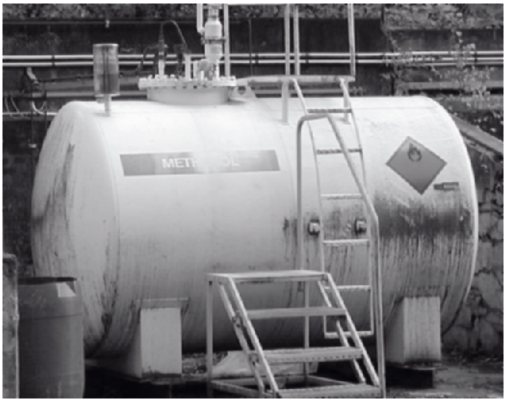

Figure 2. The double-barrier principle: a double-walled tank with testing liquid in the annulus between the inner and outer tank; a leakage will cause the testing liquid vanishing from the glass on top of the tank and initiate an acoustic alarm.

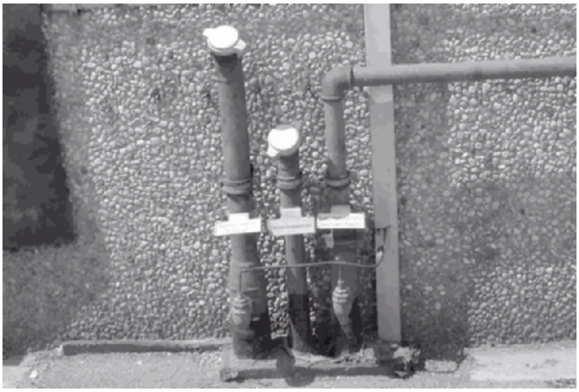

Figure 3. Alarming system: filling pipes for an underground tank equipped with vacuum sensors; the annulus between the inner product pipe and the outer protection pipe is evacuated and in case of a leak, the vacuum will cease and an alarm be started. The thin pipes vanishing in the grey cable duct, connect the vacuum sensor with the pipe annulus.

State-of-the-art groundwater protection must also consider all properties of the classic and new contaminants. Hydrogeologists have to integrate the latest research results and developments of material science, chemistry, physics, biology and the so-called live-sciences into their conceptual models and must even include the "green technologies" which start to produce new underestimated hazards.

Bio-diesel, Fatty-Acid Methyl Ester (FAME), is produced from vegetable oils and is a renewable, re-growing source of energy. Nevertheless, if released in larger quantities, it is a soil and groundwater pollutant producing the same problems as mineral-oil diesel. Beyond that FAME, as many other esters is able to destroy the most commonly used sealing materials. Every tank, pump and filling valve must be thoroughly fitted with new FAME-resistant seals, fittings and hoses (Arbeitsgemeinschaft Qualitätsmanagement Biodiesel e.V., 2004).

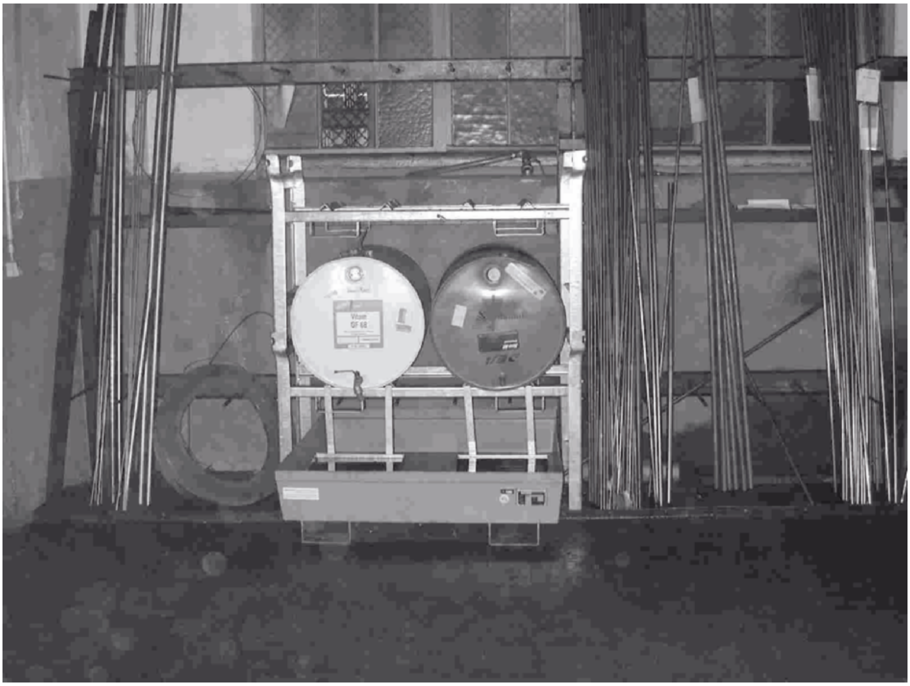

Figure 4. Retention tray with the possibility to secure barrels and safely transport them by a fork-lift.

2.4 Risk Assessment and Protection Strategies

Risk assessment for groundwater catchments and protection zones has to not only include flow patterns and the existing and planned land use with its living quarters and industry. It is also advisable to go back to the local and regional history, looking for ancient mines and their tailings, closed dump-sites or buried riverbeds, flow channels, sewers and large underground buildings.

Beyond that, the assessment must include airborne pollution e.g. by VOCs, pesticides or acid rain. Even a large part of the existing PCE and TCE contaminations have been caused by airborne pollution and in several cases during early groundwater remediations, contaminated air released from groundwater stripping units without activated carbon treatment re-infiltrated into the groundwater as gaseous phase a short distance away. No wonder that the remediation took much more time than initially supposed.

Hydrogeologists have to get actively involved in almost every regional planning process, if groundwater protection in general and pollution prevention specifically shall achieve real success. They have to go to the users of hazardous substances to analyse the risk for the groundwater and develop optimised protection strategies and emergency plans, well adapted to the local infrastructure and geological conditions.

Figure 5. Retention trays for containers with a volume of 1000 l. The tap at the left container is protected by an additional tray to retain small leakages from filling procedures. The whole storage room is constructed as a retention basin and coated with a chemical-proofed coating.

Education and training of personnel handling hazardous material must be treated as a continuous process in every company in order to acquire or maintain the state of the art in groundwater protection. Management systems for quality management (ISO 9000) and environmental management (ISO 14000 or the Eco-Management and Audit Scheme EMAS in the European Union) enable companies to control and optimise their workflow and material flow. When seriously implemented and generally accepted by the workforce, these management systems can also be used for controlling and optimising the management and handling of hazardous substances and so improve groundwater protection.

It is the extremely large and costly cases, which make the headlines, with tremendous health, social and economic problems, sometimes stretching out over hundreds of square kilometres in old industrialised areas or urban agglomerates, caused by decades of mismanagement, ignorance and inhuman attitude. Nevertheless the attention should also be drawn to the small and medium sized contaminations, which might also add up to a significant regional groundwater problem. Resulting from technical deficiencies, human errors and wrong decisions in emergency situations, many of them could well be avoided by simple technical equipment and a good deal of common sense.

One of the major causes for contaminations is poor decision making in emergency situations. After causing an oil spill polluters often try to improve the situation by using detergents and large amounts of water to clean stained surfaces. Mostly the contaminated water is not collected but washed straight into the sub-surface. Adequate binding agents might as well be applied, which should be available at every place where oils are stored.

In a case recently handled, about 100 l of heating oil, leaking from a central heating spread over a concrete slab of about 50 m² in front of a building. Instead of using binding material to soak the oil up, a mechanic took a high-pressure water cleaner and several litres of household detergents to clean the concrete and washed the whole liquid on to a lawn. The result was that about 200 m³ of soil had to be excavated, instead of just giving some hundred kg of contaminated binding material for incineration. In this case it meant instead of spending some 2,000 – 4,000 EUR for cleaning the concrete by a specialised company, more than 25,000 EUR had to be paid for excavation, refilling, constructing an observation well and renewing the lawn plus the renewal of 50 m² of inappropriately cleaned concrete.

This last example shows clearly, how much attention has to be paid to appropriate education and the rising of awareness on groundwater contamination problems, even in people who handle groundwater pollutants in their daily work.

3. GROUNDWATER REMEDIATION

Remediation of soil and groundwater contaminations can only be successful, if all aspects of science and technology, which are relevant for groundwater protection, are also professionally integrated into the planning of the remediation process (Table 1). In most pollution cases only a well-balanced combination of various methods can guarantee a successful remediation, depending on the geological conditions, the specific properties of the aquifer itself and the size of the contamination.

Nevertheless even the most modern cleanup methods have one important limiting factor: they are limited by costs or the funds available. Therefore it is necessary to evaluate the urgency of the measures to be taken, depending on the possible or probable impact the pollution might have on used groundwater and human health in the first place. If the urgency is high, immediate action has to be taken, in low urgency cases, one can also think about the concept of monitored natural attenuation as a means of "passive remediation".

Table 1. Basic information needed for risk assessment and successful remediation.

Properties of the contaminants	Material science
	Chemistry
	Physics
	Biology
	Medicine / toxicology
	Life-science (biotechnology)
	"Green technologies"
Geology	Hydrogeology
	Engineering geology
Local conditions	Groundwater use
	Buildings & use of buildings
	Subsurface structures (tunnels, service utilities)
Legal conditions	Limits of properties
	Terms & conditions of land use
	Federal, state & local acts, directives & regulations
Time available	Immediate action needed – risk of casualties
	Medium term
	Long term
Natural resources needed	Energy consumption
	CO_2-production
Costs	Detailed evaluation of costs

In most cases only a combination of various methods is able to guarantee a successful remediation (Table 2). This is usually performed in steps according to the case urgency and site accessibility as exemplified below.

Step 1.1: Removal of the primary source, e.g. the leaking tank, pipe or the hazardous waste in an old landfill;

Step 1.2: Removal (excavation) of contaminated soil as the secondary source of the groundwater pollution.

Step 1.1 and 1.2 can only be applied if the local conditions allow digging out material at all. Buildings on top of the sources or subsurface structures, like sewers or power transmission lines, might make it impossible to reach them. In these cases other procedures might become necessary and source treatment starts with step 2.

Step 2: Soil treatment by soil-vapour extraction of volatile contaminants or biological treatment of the soil. In many cases it is sufficient to seal the surface or to encapsulate the contaminated soil by cut-off walls or injection methods in order to prevent the contaminants from migrating into the groundwater.

Table 2. Remediation schedule and methods.

Step			
1.1	Source removal (if possible)	Primary sources	Old tanks Pipelines Disposed waste
1.2		Secondary sources	Highly contaminated soil Concrete etc.
2.1	Source treatment	Soil vapour extraction	
2.2		Biological treatment	Injection or infiltration of fertilisers for soil bacteria, aerobic and anaerobic
2.3		Encapsulating the source	Sheet pile walls Injections Infiltration-tight covers
3.1	Water treatment	Active interference into the groundwater flow	Diversion of plume by injection wells as first aid Drinking water treatment Pump and treat (various methods of treatment) with or without re-injection into the aquifer
3.2		In situ treatment	Biological treatment (injection or infiltration of fertilisers for soil and water bacteria, aerobic and anaerobic) Groundwater circulation wells Reactive walls Funnel and gate
3.3		"Passive remediation"	Natural attenuation Enhanced natural attenuation

Step 3: Groundwater remediation: various methods are available and can be combined as necessary for the individual case.

Step 3.1 Active remediation: pump and treat action (with or without re-infiltration) plus bioremediation in a further stage or the application of groundwater circulation wells (IEG, 2004).

Step 3.2 In-situ methods: depending on the local geological and infrastructure conditions it might be advisable to install reactive walls or funnel-and-gate constructions with either reactive material directly installed in the gate opening or with a subsurface reactor, which can be easily controlled, maintained or even exchanged (Finkel *et al.*, 1998; Gillham *et al.*, 1998; Stupp, 2000; Werner, 1998).

While the pump and treat systems need a continuous supply with energy and in some cases even a lot of maintenance due to the highly sensitive technology applied, reactive walls and funnel-and-gate-systems do not need additional energy as the natural groundwater flow is sufficient to operate the systems by driving the water through the treatment. On the other hand, the investment for the installation of both, reactive walls and funnel-and-gate systems is mostly significantly higher than for the installation of pump-and-treat systems, mainly due to the high prices for the installation of the enclosing walls.

Depending on the geological conditions, the specific properties of the aquifer itself and the size of the contamination, even large scale constructions might be installed, which could be sheet pile walls for depths down to 20m or soil-bentonite cut-of-walls down to 50 m. - In each and every case however, a detailed study has to evaluate the ecological and financial risks of the cleanup method to be recommended, as well as its potential legal problems with regard to public or private law.

Step 3.2: Passive remediation: natural attenuation.

If time pressures are absent, as a plume of contaminants is only moving slowly towards a protection zone, the natural attenuation approach might be considered as an alternative to the high-investment, energy-consuming active remediation strategies (Rijnaarts, 2000; Stupp and Paus, 1999; Teutsch and Rügner, 2000; Wienberg, 2000).

The U.S. Environmental Protection Agency (EPA) was among the first agencies to accept this remediation approach under the condition that all relevant processes are well monitored. This process re-named “monitored natural attenuation MNA” is defined by EPA (1999) as the "reliance on natural attenuation processes (within the context of a carefully controlled and monitored site cleanup approach) to achieve site-specific remediation objectives within a time frame that is reasonable compared to that offered by other more active methods. The 'natural attenuation processes' that are at work in such a remediation approach include a variety of physical, chemical, or biological processes that, under favourable conditions, act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in soil or groundwater. These in-situ processes include biodegradation; dispersion; dilution; sorption; volatilisation; radioactive decay; and chemical or biological stabilisation, transformation, or destruction of contaminants. When relying on natural attenuation processes for site remediation, EPA prefers those processes that degrade or destroy contaminants. Also, EPA generally expects that MNA will only be appropriate for sites that have a low potential for contaminant migration."

From the ecological point of view this approach combines in an almost ideal way the aims of CO_2-reduction by saving energy with the simple financial task of saving money (van Veen *et al.*, 2000). Nevertheless natural attenuation can only be applied if the land owners and the environmental agencies involved agree to such a long-time remediation process. If the contaminated land is to be sold and re-used for non-industrial purposes as office buildings or housing, it will be almost impossible to proceed with natural attenuation, as the value of the land is either significantly reduced or it cannot be sold at all until cleanup is definitely finalised.

4. PROTECTION AND REMEDIATION IN AN INTERNATIONAL ENVIRONMENT

Finally transboundary aquifers and rivers urge hydrogeologists to work together on an international basis for the solution of common problems; the EU water framework directive gives a good example of how multilateral contracts and laws can enforce international cooperation in hydrology and hydrogeology on a catchment-based approach.

North-west of Cologne in Germany large scale open pit lignite mining has been in operation for about 70 years. The mining company lowers the groundwater levels from an average 10 m below ground level (bgl) to a maximum of 500 m bgl for the time of mining operations.

The local effects of groundwater withdrawal are obvious: dry wells, dry springs, dry rivers, ponds/lakes, swamps and wetlands. Close to the mines all complaints have been taken seriously by the mining company, drilling new deeper wells or delivering water from other sources, filling up dry creeks and ponds with their pipe network. Long distance effects, however, have often to be handled by the courts of justice. The range of influence of the drawdown is supposed to be up to 40 km, even affecting waterworks and wetlands in the Netherlands, like the "Dutch-German Natural Reserve Maas-Schwalm-Nette" and the Dutch "Meinweg National Park", protected by the European Union as a "Natura 2000 Reserve", about 20 km from one of the actual pits (BUND, 2000). All these problems have now to be analysed and solved in common boards by Dutch and German authorities in order to secure the existence and availability of commonly used aquifers.

While these developments seem to improve the awareness of both, the water authorities and the general public, a new threat to our water reserves is now realised. In January 2005 the World Trade Organisation's (WTO) "General Agreement on Trade in Services (GATS)" becomes effective, stating that all public services must be opened for private investors.

On long term this could mean, that public water supply and wastewater systems can be completely privatised and that a worldwide trade with water resources can become possible. Depending on their financial power, city councils and public utility companies can decide to use the services of private companies only in a few fields as a Public-Private-Partnership (PPP) solution, or to go as far as to completely sell their water resources and infrastructure (Dümmer, 2004; Gramel and Urban, 2002).

Depending on the specific viewpoint of the observers, there are various possible results to be found in mega-cities with a privatised water supply infrastructure. In Manila, Johannesburg or La Paz, judgments range from "cheaper and more effective" over "more expensive and unsocial and ignorant towards the poor" to "new colonialism". The overall existing problems are that private companies always look for the Shareholder-Value and in many cases tend to run their business to the "Quick-Win-Principle". The latter might result in the following allegations levelled against several privatised utilities (Attac, 2003; Klas, 2002):

- Minimal investments in repair and maintenance, resulting in a deteriorating infrastructure.
- Postponement of measures for resource protection as long as other sources are available. No planning for the future.
- An essential resource of life comes under the complete control of economic interests, even in developing countries with highly sensitive economic and hydrological structures (e.g. Malawi; Dar-es-Salam, Tanzania).

In a few places citizens raised protest against the privatisation of their regional water supply systems and succeeded in stopping the process like in New Orleans (USA) or in Cochabamba (Bolivia). “GATS-Free-Zones” were declared in Canada, e.g. in the City of Vancouver and the Province of British Columbia.

In Cochabamba a violent citizen's protest against a 35 % increase of water prices in connection with the sale of the water utilities to Aguas del Tunari, a Bechtel Corporation affiliate, caused 6 deaths and 175 casualties in 2000. The "Cochabamba Declaration on the Right to Water" of December 8, 2000 (Table 3) can be regarded as the expression of rising public awareness towards the access to clean and affordable drinking water and the protection of its resources for future generations.

In the light of resource protection, all large scale water management projects should be watched and evaluated with due care and suspicion. Once water resources are sold to private companies, it is just a question of supply and demand whether a large project will be started or not. From time to time over the last 25 years there have been discussions about exploiting a groundwater reserve in the Lüneburger Heide, southeast of Hamburg

(Germany) and pipe the water over more than 2,000 km to central Spain for irrigation purposes. Presently such a project would be stopped by all local political boards in the earliest possible stage as an inappropriate use of a locally owned and locally protected resource. Under the condition, that the resources were privatised, only the enormous investment for a pipeline and the pumping equipment could stop the start of the complete exploitation of the groundwater reserves for international groundwater trade.

Table 3. The Cochabamba Declaration on the Right to Water.

"For the right to life, for the respect of nature and the uses and traditions of our ancestors and our peoples, for all time the following shall be declared as inviolable rights with regard to the uses of water given us by the earth:
1. Water belongs to the earth and all species and is sacred to life, therefore, the world's water must be conserved, reclaimed and protected for all future generations and its natural patterns respected.
2. Water is a fundamental human right and a public trust to be guarded by all levels of government, therefore, it should not be commodified, privatised or traded for commercial purposes. These rights must be enshrined at all levels of government. In particular, an international treaty must ensure these principles are non-controvertible.
3. Water is best protected by local communities and citizens who must be respected as equal partners with governments in the protection and regulation of water. Peoples of the earth are the only vehicle to promote earth democracy and save water."
December 8, 2000

5. CONCLUSIONS

Urban groundwater protection and pollution prevention have to start at the probable source. The transport, handling and storage of dangerous contaminants must be performed in a safe way, making spills unlikely to happen and preventing them from reaching soil and groundwater. All precautions must be based upon a profound knowledge of the contaminant's properties and include all state-of-the-art technical equipment available.

Groundwater remediation has additionally to include the latest information about contaminants and available clean-up technologies. Beyond that, ecological aspects like CO_2-production and energy consumption of the remediation should be considered, if various methods are in discussion.

For hydrogeologists, water-supply engineers and other technicians and scientists involved in groundwater protection and remediation the challenge of dealing with old and new contaminants will continue into the future. Nevertheless a sustainable success in the protection and conservation of our most essential resources can only be achieved, if all the professionals engage themselves for groundwater issues in public and get involved in all relevant planning processes and legislation. This should apply also to the question of possible effects of the privatisation of groundwater resources and supply systems and the field of international conflicts on the use of water.

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REDUCING THE GROUNDWATER POLLUTION RISK IN THE MINING AND INDUSTRIAL REGIONS OF CHIATURA AND KAZRETI, GEORGIA

Remediation of Mine-related Tailings and Wastes

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Abstract: Results are presented of an experimental study of bacterial leaching of tailings and wastes from mining industry enterprises. From the examples of Chiatura city and the industrial centre of Kazreti, it is shown that use of the suggested method for the leaching of ores can reduce significantly the pollution risk to water and soils in urban areas. The method also promotes both the improvement of the ecological system in general, and a more rational use of natural resources.

Key words: bacterial leaching; solution; metals; tailings; mine wastes; pollution; Chiatura, Georgia; Kazreti, Georgia.

1. INTRODUCTION

Large-scale contamination of groundwater and soils within Georgia, has resulted from the leaching of poorly stored mining-associated production tailings and waste. Remediation of such problems requires the application of remediation technologies that are both cost-effective and technologically sound.

This paper demonstrates the possibility of employing new biochemical methods to metalliferous mine-related waste tailings. The applied method achieved an 85% decrease of the toxic element content within waste from a copper-pyrite mine. Similar decreases were also observed in a manganese–ore district. Such reductions thus provide significant opportunity to reduce concentrations of these metals to safe, environmentally acceptable, levels.

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In light of the worldwide problem of protecting the environment from contamination, bacterial leaching techniques are generally much less harmful relative to other more aggressive methods (e.g. chemical) due to their limited impacts to the eco-environment. Furthermore, the processes described do not require large expenditure, either of energy or capital, and hence their wide applicability may be anticipated (Polkin *et al.*, 1982; Karavaiko *et al.*, 1989).

The research was undertaken using available data on copper- and manganese-ore production in Georgia. Mining facilities there may produce soil and groundwater contamination to exceedingly high levels. For example, waters leaching the spoil banks of Kazreti copper–ore quarry contain copper concentration is 100,000 times higher than the average content in the water globally. Cadmium, manganese and cobalt content in rivers and seawater can be 12,000 times higher. Leaching of such metals into the wider soil and groundwater environments may lead to ecological problems in what is for the most part rural localities.

A similar situation is observed in the Chiatura manganese–ore district, wherein water that has washed the tailings of dressing mills flows into the river Kvirila. Downstream of Chiatura the manganese content in the river is 600 mg/l. This is some 50,000 times greater than the average content for river water.

2. METHODS AND MATERIALS

This study did not follow the commonly practiced approaches that tend to involve specific selection and growing-on of specific micro-organisms that may facilitate a preferred degradation pathway. Rather, the approach adopted employed natural processes of weathering and leaching. The objective was to investigate the effect of the entire and complex microbial community rather than individual members. In particular, the possibility of the leaching of micro-organisms, and their participation in the gleying processes was assessed (Tsertsvadze and Zviadadze, 1999).

The purpose of these investigations was to show the possibility of bacterial purification of tailings and waste disposed of in mining and industrial regions thereby limiting the accumulation of ecologically dangerous quantities of pollutants. Various test sets were run to choose the optimal parameters for the leaching regime and to state the dynamics and main regularities for extracting certain metals.

The chemical composition of the experimental samples was analyzed by applying neutron-activation, X-ray fluorescence and atom-emission methods of the Joint Institute for Nuclear Research in Dubna (Ostrovnaya *et al.*, 1993) and the Moscow Geological Institute (Tsertsvadze *et al.*, 2001). Estimation of the element concentration in the solutions was conducted by employment of spectrophotometric and neutron-activation methods. Bacterial composition was elucidated by plating micro-organisms in selective media.

In accordance with the study aims, tests were conducted on the preparation of experimental samples and solutions for running bacterial leaching. For this purpose, peat was selected as the source of the micro-organisms, as its microbiological content is diverse. It includes many aerobic and anaerobic, both obligatory and facultative, forms.

Figure 1 illustrates the technological process of bacterial leaching. It is not always necessary to run the further stages of bacterial leaching for the remaining mass, and therefore leaching stages II and III are accomplished only. This is the case provided that a correspondence has been observed between the compositions of the solution and the remaining unleached mass, and the objective sought (Tsertsvadze *et al*., 2001).

The investigations have shown that the effect of leaching depends not so much on the quantity of micro-organisms in solution but on their activity and provision of the optimum conditions for their existence in new surroundings (i.e. the mineral-bearing waste). The energetic input of material increases markedly the intensity of development of micro-organisms in new surroundings.

The progression of the leaching effect is evaluated from the observed remaining elemental contents of the samples. In samples with a low concentration of Mn (<7%), better results of leaching were achieved. This correlates well with the microbiological composition of the solution inferring the participation of a greater quantity of sporiferous bacteria and reducing micro-organisms.

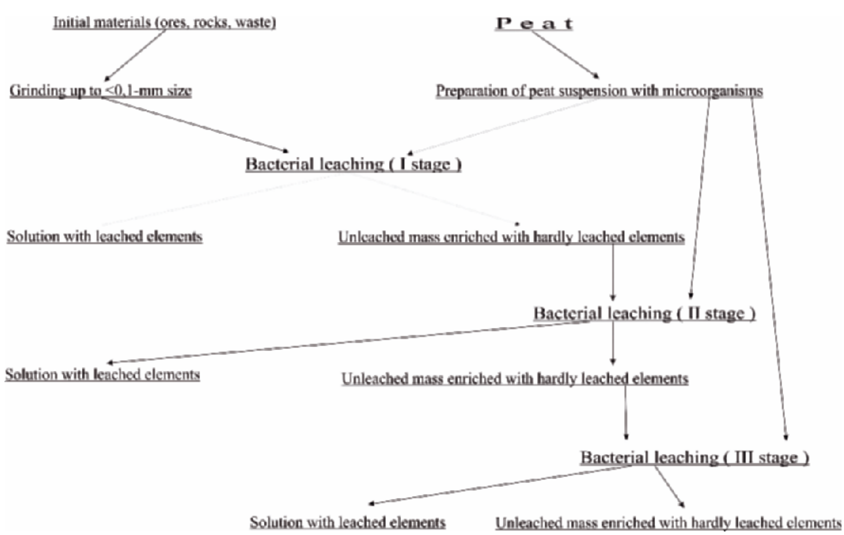

Figure 1. Schematic diagram of the technological process of bacterial leaching.

3. RESULTS AND DISCUSSION

The results obtained indicate that the proposed technique is promising for certain types of mineral waste product, depending on the specific goal. The latter, for example, may relate to decontamination of the waste, extraction of scarce metals, or else enriching the content of a specific metal.

The bacterial treatment technique developed enables leaching of a wide spectrum of elements, including radioactive and noble metals (gold in particular) from the mine tailings and wastes. Study of the performance of 60 elements showed that all elements studied were leached and extracted into the solution, albeit at varying efficiencies (Tsertsvadze *et al*., 2001). Elements with variable valence, whose concentrations in the leached evaporated solutions from manganese–ore production tailings reached, for example, over 2.61×10^5 g/t for Mn, and 107 g/t for Mo, are readily extracted as early as the first cycle of leaching. Thus, the extraction percentage for Mn was 55.5% and, for Mo, 54.8%. After the first cycle of bacterial treatment, Cu content in the copper-pyrite ore dressing tailings dropped by 57.5%, while the content of Zn in barite-complex ore tailings decreased by 83.3%. After further cycles of leaching the levels can decrease by over 90%.

Especially interesting is the prospect of leaching elements that are ecologically hazardous with high concentrations common in mine tailings and waste. This equally may apply to radioactive elements, as well as Cd, Hg, As, and other heavy metals.

The presence of these elements in the bacterially leached solutions indicates potential for their concentrations to be decreased in waste-tailings. This controlled accelerated leaching may in turn diminish their long-term, gradual natural leaching into the wider environment. Thus, Cd content in the evaporated leached solution from the copper–pyrite ore tailings is 6.4 g/t., that of Hg is 379 g/t., and that of As is 176 g/t. Uranium yield after only one bacterial leaching of the manganese–ore production waste reaches 30% (Tsertsvadze *et al*., 2001).

The diagrams below (Figures 2 to 4) indicate the concentrations of various metals in the initial and leached samples of copper– and manganese–ore production tailings. These data demonstrate explicitly the alterations in concentrations after only one cycle of bacterial leaching. Additional metals to those shown are extracted. This includes rare-earth elements, however, data were not obtained.

Figures 2 to 4 indicate concentrations in initial and leached samples of tailings from the copper-pyrite industry, situated in the area around the industrial centre of Kazreti and from spoil heaps produced by manganese ore treatment within the city of Chiatura. After one cycle of leaching, the content of toxic metals decreases significantly and already approaches the relevant quality criteria (Clark values). Repeated cycles of leaching lead to safer levels of toxic elements (such as uranium, cadmium, arsenic, molybdenum, cobalt, tungsten, vanadium and nickel). The data substantiate that such bioremediation methods may lead to significantly decreased risks of pollution to natural waters and soils in mining regions.

The other principal benefit of the experiments is that the leaching solutions may accumulate concentrations of some elements to a commercially viable degree.

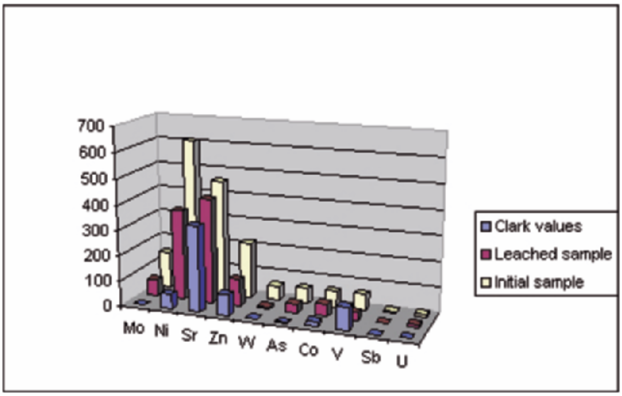

Figure 2. Variation of the concentrations of some metals (g/t) during the processes of leaching of tailings from manganese ore treatment.

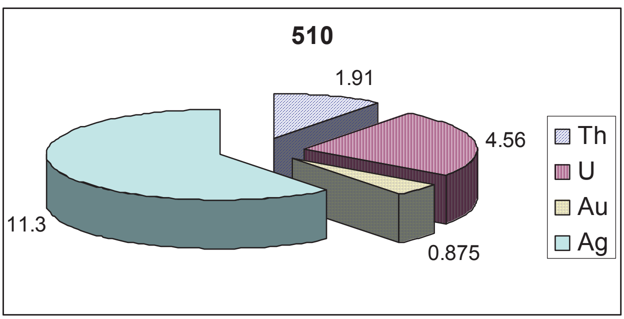

Figure 3. Comparison of the concentrations (g/t) of some elements in tailings of copper-pyrite ores, from initial sample 510.

It should be noted that, as bacterial leaching takes place, both subtraction of certain metals and a considerable enrichment of the remaining leached mass with some elements occur (Figures 3 and 4). The leached mass contains a concentration of metals that may exceed the quality criteria (Clark values) by many times. For example, the leaching of gold is a particularly interesting issue. According to the data obtained, gold is an element with an explicit trend towards enrichment, though at the same time metal subtraction into the solution also takes place.

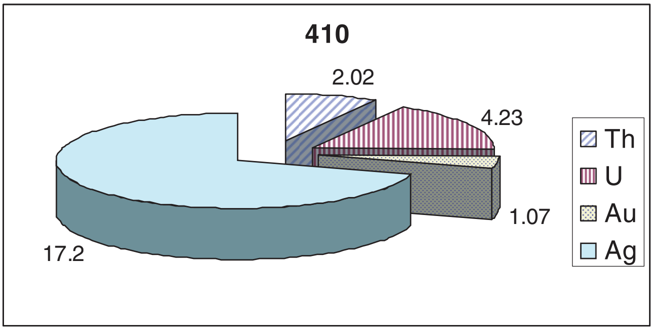

Figure 4. Comparison of the concentrations (g/t) of some elements in tailings of copper-pyrite ores, from leached sample 410.

The results demonstrate that employment of the developed method for enriching the waste and tailings of copper-pyrite ores with gold is likely to give an impressive economic profit. This is achieved via upgrading the tailings and waste to ore standard with regard to gold content, which would exceed many times the Clark value.

4. CONCLUSIONS

From this study it can be inferred that the bacterial treatment of waste and tailings from mining production diminishes both the risk of water and soil contamination by toxic metals, and increases the economic profitability through the use of the leached solutions for extracting certain scarce and valuable metals.

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THE REMOVAL OF NITRATE AND PESTICIDES FROM CONTAMINATED WATER

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Abstract: A water-saturated biological denitrification system was employed to remove selected pesticides and nitrate from drinking waters. In the study, approximately 99% nitrate removal was observed, but most of the samples included nitrite. Up to 95% removal of pesticides was also observed. The lowest removal efficiency of nitrate (63%) was observed when the temperature dropped sharply from 18 ^oC to 14 ^oC. Despite such oscillations, high removal efficiencies for trifluralin, fenitrothion and endosulfan (α+β) were observed and the highest nitrite concentration was 16 mg/l in the effluent of the reactor.

Key words: denitrification; pesticides; nitrate; water treatment; trifluralin; fenitrothion; endosulfan; bioreactors; agricultural pollutants.

1. INTRODUCTION

Water pollution by pesticides and nitrate from routine agricultural practices is a common and growing problem in the major agricultural areas of the world. Over recent decades, concern about the contamination of water sources has been raised due to increasing nitrate and pesticides concentrations. In regions where pesticide contamination is a problem, nitrate concentrations are often high (Hallberg, 1987). Although groundwater pollution by nitrate and pesticides is encountered mainly in rural areas, it is also a problem affecting urban groundwater (Wakida and Lerner, 2005).

Nitrate is found in moderate concentrations in most waters, but is often enriched to high levels in water supplies. This is predominantly through the increased use of nitrate fertilizers and discharge of raw and partially treated waste water. This has become an increasingly important problem and has

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limited the direct use of water supplies for human consumption in many countries. Increased nitrate concentrations in water supplies cause methemoglobinemia in infants and cancer of the alimentary canal (Wasik *et al*., 2001). There is no other group of carcinogens that can produce such a wide variety of tumours (Mirvish, 1991).

Drinking water regulations are necessary in order to limit both the risk to humans and environmental pollution. Pesticides and nitrates in drinking water are limited to 0.1 μg/l for a single pesticide, 0.5 μg/l for the sum of all pesticides and 45 mg/l and 50 mg/l for nitrates, by Turkish Drinking Water Standards (TDWS, 1988) and the World Health Organization (1984) respectively.

Biodenitrification is a promising technique for the simultaneous removal of nitrate and pesticides from drinking waters. The removal of nitrate in water by conversion to nitrogen gas can be accomplished biologically under anoxic conditions in a process known as denitrification. Denitrification in aquifers is in fact the main natural attenuation process that may serve to limit nitrate impacts in groundwater. However, in many sub-surface environments its occurrence is fairly limited unless confined or reducing groundwater conditions are prevalent (Buss *et al.*, 2005). Nitrate is reduced to nitrogen gas by several genera of bacteria namely, *Achromobacter, Aerobacter, Alcaligenes, Bacillus, Brevibacterium, Flavobacterium, Lactobacillus, Micrococcus, Proteus, Pseudomonas,* and *Spirillum*. These bacteria are heterotrophs capable of dissimilatory nitrate reduction, a two-step process. The first step is conversion of nitrate to nitrite. This stage is followed by production of nitric oxide, nitrous oxide, and nitrogen gas.

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

For reduction to occur, the dissolved oxygen (DO) level must be at or near zero, and a carbon supply must be available to the bacteria. The presence of DO will suppress the enzyme system needed for denitrification. Alkalinity is produced during the conversion of nitrate to nitrogen gas resulting in an increase in pH. The optimal pH lies between 7 and 8 with different optimums for different bacterial populations and the organisms are sensitive to changes in temperature (Volokita *et al*. 1996).

The majority of biodenitrification relies on heterotrophic bacteria that require an organic carbon source. Since drinking water has low carbon content and pesticides cannot be used as the sole carbon source, an additional carbon source is required. Therefore an external carbon has to be supplied for microbial growth. Several types of organic compounds have been used, such as methanol (Hoek and Klapwizk, 1987; Gomez *et al*., 2000; Wasik *et al*., 2001; Lee *et al*., 2001) and acetic acid (Dahab and

Kalagari, 1996; Bandpi *et al*., 1999). Though methanol assures the highest denitrification rate (Mansell and Schroder, 1999), it can constitute a certain risk if the treated water is used for drinking purposes (Adriaan, 1992). The main disadvantage of acetic acid as compared to other carbon sources is its high consumption ratio and high cost (Bandpi *et al*., 1999). Balszczyk *et al*. (1981) pointed out that using acetic acid could have a significant effect on the production of nitrite in the reactor. Consequently, the use of ethanol as an alternative is becoming more popular (Richard, 1989; Dahab and Sirigina, 1994; Delanghe *et al*., 1994; Green *et al*., 1994; Bandpi *et al*., 1999; Gomez *et al*., 2000; Fonseca *et al*., 2000; Gomez *et al*., 2000; Aslan, 2005).

The activated carbon/ion exchange process and reverse osmosis (Goodrich *et al*., 1991), combined membrane bioreactor/powdered activated carbon adsorption (Urbain *et al*., 1996), biofilm-electrode reactor (BER) (Sakakibara and Kuroda, 1993) and BER/adsorption process (Feleke and Sakakibara, 2001), fluidised biodenitrification reactor (Katz *et al*., 2000), and bioelectrochemical/adsorption process (Sakakibara *et al*., 1997) were developed in order to remove both pesticides and nitrate from drinking water. Removal of pesticides and nitrates using wheat straw as solid particles and carbon source with ethanol as a further carbon source were previously studied by the authors of this study (Aslan and Türkman, 2004; Aslan, 2005).

Pesticide consumption is not equally distributed over all the agricultural areas of Turkey. Dense application of pesticides can be seen in some areas, for example, while in other areas almost no application is observed. Uncontrolled use of agricultural chemicals in intensive agricultural areas of Turkey causes serious soil, surface, and groundwater pollution. If the pesticides are classified according to their volatilization, mobility, persistence characteristics and groundwater pollution potential, nearly 65% of the pesticides commonly used in Turkey have a high pollution potential (Aslan and Türkman, 2002).

Evaluation of the pesticides used in Turkey has led to a preliminary selection of widely used compounds – trifluralin, fenitrothion, and endosulfan (α+β or I+II) as appropriate substances for this study. This selection was based on both their widespread use in Turkey and their persistency. Their characteristics are summarized in Table 1.

The principal objective of this study was to research the simultaneous microbial removal of nitrate and endosulfan (α+β or I+II) ($C_9H_6Cl_6O_3S$), fenitrothion ($C_9H_{12}NO_5PS$), and trifluralin ($C_{13}H_{16}F_3N_3O_4$) in a biodenitrification reactor, using ethanol as carbon source.

Table 1. Characteristics of the selected pesticides. In the (EPA) toxicity score, 1 is high

Pesticides	Mol. Formula	Mol. Weight (g/mol)	Solubility in Water (mg/l)	Partition coeff. log K_{ow}	Toxicity Score	Half-Life (days)
Endosulfan	$C_9H_6Cl_6O_3S$	406.95	0.32-0.33 (22°C)	4.74-4.79	1	30-70
Fenitrothion	$C_9H_{12}NO_5PS$	277.2	21 (20°C)	3.43	2	4-20
Trifluralin	$C_{13}H_{16}F_3N_3O_4$	335.3	0.184 (pH 5)	5.27	3.5	57-126

2. MATERIALS AND METHODS

2.1 Concentrations of Pesticides

Each medium studied was spiked with 7 μg/l trifluralin, 7.6 μg/l fenitrothion and 10.9 μg/l endosulfan (α+β) solutions during the experiments.

2.2 Experimental Set-up of the Biological Denitrification Reactor

The experimental set-up consisted of a cylindrical stainless steel biological reactor of 15 cm inner diameter and 60 cm height, completely water-saturated and operating with an upward flow mode (Figure 1). The packed column was filled with 10 mm pieces of coiled plastic, which supported bacterial growth. The denitrification column had a liquid volume of 5.3 litres and the support particle surface area was $1m^2$, resulting in 190 m^2 surface area/m^3.

Denitrification micro-organisms were taken from the denitrification reactor used in the laboratory. The column packed with plastic materials was fed with medium solution prepared in distilled water. A medium solution spiked with selected pesticides was prepared daily.

2.3 Synthetic Medium Composition

The liquid medium used consisted of a mineral base supplemented with nitrate as sole electron acceptor and ethanol as donor. Other constituents were KNO_3 (100 mg NO_3/l), KH_2PO_4 (150 mg/l), and $NaHCO_3$ (325 mg/l). This basal medium was supplemented with 1% v/v of a solution containing $FeSO_4.7H_2O$, titriplex (0.565 mg/l), and with 0.1% v/v of a trace nutrient solution containing $ZnSO_4.7H_2O$ (0.1 g/l), $MnCl_2.4H_2O$ (0.03 g/l), H_3BO_3

(0.3 g/l), $CoCl_2.6H_2O$ (0.2 g/l), $CuCl_2.2H_2O$ (0.01 g/l), $NiCl_2.6H_2O$ (0.02 g/l), and $NaMoO_4.2H_2O$ (0.03 g/l). The final pH of the medium was adjusted to 7.5 using NaOH solution. The synthetic medium solution spiked with the selected pesticides for reactor study included mineral base media; 100 mg NO_3/l and CH_3CH_2OH, which were prepared daily. Nitrate and ethanol were added so that the C/N ratio was adjusted to 1.5.

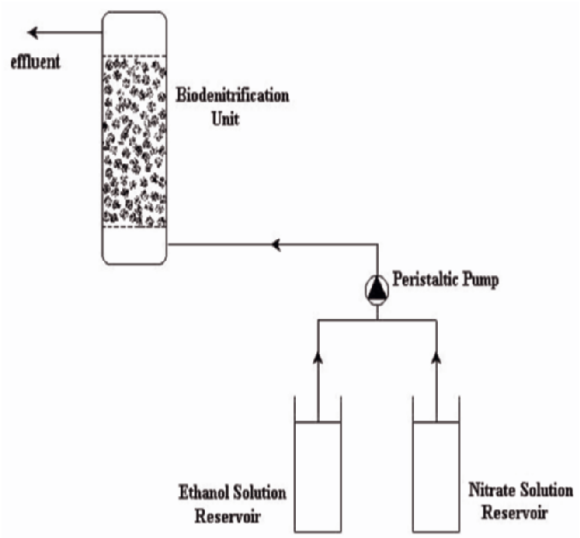

Figure 1. Experimental set-up of the biological denitrification reactor.

2.4 Analytical Techniques

2.4.1 Reagents

Pesticides used in this study have the following properties: endosulfan ($\alpha+\beta$), purity 99.3%, endosulfan-ether, purity 99.5%, endosulfan-sulphate, purity 99.4%, endosulfan-lacton, purity 99.5, fenitrothion, purity 98.5%, and trifluralin, purity 99.2% from Dr. Ehrenstorfer, and internal standard pentachloronitrobenzene from Aldrich, purity 99%. Acetone and n-hexane (Merck) were used as solvents. Octadecylsilica (C_{18}) was supplied by J.T. Baker.

2.4.2 SPE extraction for pesticides

Samples were extracted and pre-concentrated by solid phase extraction (SPE) prior to chromatographic analysis. Conditioning of the cartridges filled with 1g C_{18} was performed with 30 ml of acetone, 30 ml pure water containing %1 acetone and then washed with 30 ml pure water. Using the vacuum source, the liquid sample was passed through the C_{18} column at a flow rate of about 5 ml/min. The cartridge (C_{18}) was washed with 10 ml of distilled water. Pesticides collected on the C_{18} were eluted with 10 ml acetone by using the vacuum. The mixed solvent of acetone and pesticides was passed through anhydrous sodium sulphate and then evaporated to dryness by air heated to about 40 ^oC. The extract was then diluted with 500 µl acetone and the internal standard pentachloronitrobenzene was added for gas chromatographic analysis.

2.4.3 Gas chromatographic analysis

Chromatographic separation of the pesticides was performed at a fused silica capillary column (30m x 0.32 mm i.d.) coated with 5% phenylmethylpolysiloxane film thickness 0.25 µm (Supelco, USA) with a splitless injection volume of 1 µl. The pesticide analysis was performed with a Shimadzu GC-17A model gas chromatograph equipped with a Ni (63) electron capture detector and AOC-20i Autosampler.

Helium of highest quality (5.0) was used as the carrier gas and nitrogen was used as make-up gas at flow rates of 1.5 ml/min and 50 ml/min, respectively. The injector and detector temperatures were 250 ^oC and 300 ^oC, respectively. The initial oven temperature was kept at 45 ^oC for 1 min; then was programmed to 210 ^oC at a rate of 15 ^oC min/l, then was raised to 280 ^oC at a rate of 10 ^oC min/l, held for 3 min and finally at 300 ^oC at a rate of 30 min/l and held for 1 min. Pesticides concentration were calculated on the basis of peak area measurements.

2.4.4 Analytical methods

Samples withdrawn daily from effluent of the reactor were filtered using 0.45 µm, white 47 mm radius filters. Nitrate, nitrite and TOC analyses were then performed on the clear samples. Biomass concentrations in the effluent samples were determined by filtering 100 ml samples from the filter and drying the filter paper in an oven at 103 ^oC and then 550 ^oC to constant weight.

Nitrate determination was carried out according to the UV spectrophotometric screening method following Standard Methods (APHA, 1995). Nitrite and turbidity were determined with a Merck photometer SQ 300. A Merck Spectraquant analytical kit was used for nitrite (14776) analysis. Organic carbon was determined by means of a high-temperature

TOC analyser (Dohrmann DC-190). Dissolved oxygen (DO) measurements were carried out by using a WTW oxygen meter.

3. RESULTS AND DISCUSSION

In this study, the medium solution was prepared using tap water, the composition of which is given in Table 2.

The biodenitrification reactor packed with plastic materials was inoculated with micro-organisms acclimatized to ethanol taken from the denitrification reactor used in the laboratory. The inoculation lasted 3 days to enable microbial growth and attachment onto the plastic coils with daily replenishment of nitrate and ethanol in medium solution. The inoculation period was carried out in the batch mode of the reactor. Ethanol was used as a feed stock carbon source with C/N ratio of 1.5 when the NO_3-N was 22.6 mg/l (100 mg as NO_3^-/l). The nitrate was almost completely removed in 24 hours after start-up. After this period, the reactor was operated at continuous flow. The biodenitrification reactor was operated for about three months before studying pesticides elimination. Micro-organisms were observed to be covering the plastic filling materials.

Table 2. Composition of the tap water used in this study.

Components	Concentrations
Total hardness	240 mg as $CaCO_3$/l
Ca^{2+}	44 mg/l
Mg^{2+}	32 mg/l
Total alkalinity	116 mg as $CaCO_3$/l
Cl^-	40 mg/l
Electrical conductivity	335 μmhos
pH	7
AOX	150-190 μg/l
TOC	3-5 mg/l

Hydraulic retention times (hrt) were changed between approximately 4.5 and 8 hours into the experiment. In the study, about 99% of nitrate removal was observed, but most of the samples included nitrite (Figure 2). Using a biodenitrification unit, nitrate concentration in the effluent water was in the range of 0-5.5 mg/l. During the study, because of temperature oscillations, nitrate concentration increased to 37 mg/l (removal efficiency was 63%), but high nitrite washout occurred from the reactor. Temperature was an important parameter on the nitrate removal and nitrite formation.

Different mechanisms have been found to be responsible for nitrite accumulation, such as repression of the nitrite reductase synthesis in the presence of oxygen (Gomez *et al.*, 2000; 2002) and inhibition of the enzymatic activity by pH.

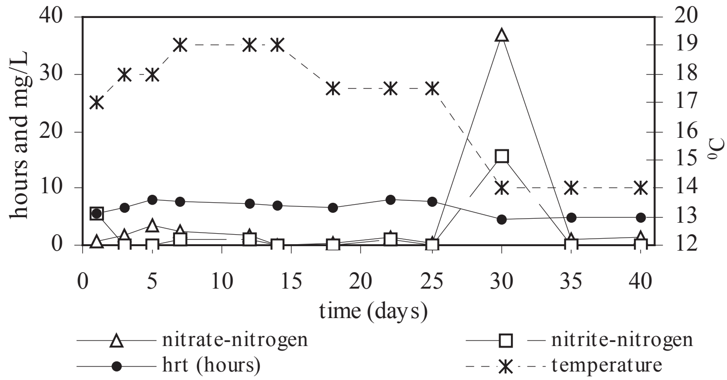

Figure 2. Effluent NO_3-N and NO_2-N data from the biological denitrification reactor.

The lowest removal efficiency of nitrate was 63%, seen when the temperature dropped sharply from 18 °C to 14 °C. Trifluralin, fenitrothion and endosulfan ($\alpha+\beta$) removal of 73%, 77%, and 80%, respectively was observed, and the highest nitrite concentration (16 mg/l) was determined in the effluent of the reactor. Up to 95% removal of pesticides was observed, however effluent water still could not meet the requirements of drinking water. Pesticide removal efficiencies of the biological denitrification reactor are given in Figure 3. Breakdown products of endosulfan were not detected in the effluent water; it is assumed that they were below the detection limits.

Although the reactor worked at 14°C, nitrate and pesticides elimination were quite high. When the temperature was stable at 14 °C, denitrification organisms were not negatively affected except for lower than 10 °C. Volokita *et al*. (1996) also noted the temporary breakthrough effects on the nitrate removal performance. In the newsprint packed columns, nitrate removal rate at 14 °C was around one-third of the rate observed at 32 °C.

The hydraulic retention time dependence of the biological reaction is very important in assessing the overall efficiency of nitrate removal. When the hydraulic residence time was decreased to under 2 hours, nitrate concentration increased and nitrite was observed in the effluent water. Increasing hydraulic residence time to higher than 2.4 hours had little effect on the effluent concentration of nitrate and organic carbon, but above this point no nitrite was observed (Aslan, 2005). Dissolved oxygen concentration did not exceed 0.3 mg/l in the biological denitrification reactor.

In the study, high nitrate elimination was observed, despite the negative effects of the temperature, enabling drinking water standards to be met. However, pesticide concentrations were not at an acceptable level, with nitrite the problem in effluent water.

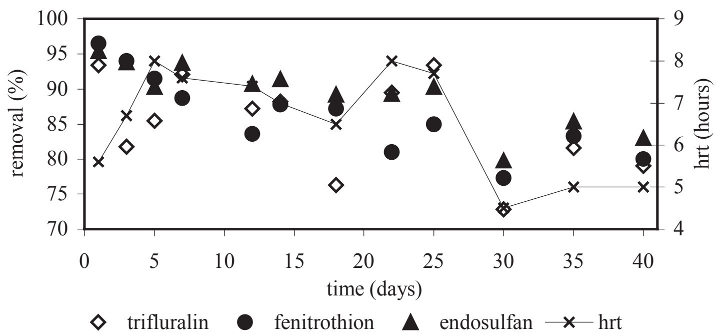

Figure 3. Pesticide removal in the effluent from the biological denitrification reactor.

4. CONCLUSIONS

Temperature is an important parameter in nitrate removal and nitrite formation. The diminished performance of the biological activity at low temperatures might be amended by introducing a longer contact time or by decreasing the rate of water flow. Although high nitrate removal was achieved, even with hydraulic residence time as short as 4 hours, higher hydraulic residence time has to be provided for significant pesticide removal. The hydraulic residence time dependence of the biological reaction is very important in assessing the overall efficiency of nitrate and pesticide removal. As a result of this study, it can be concluded that additional treatment is required to meet drinking water standards. Thus, in addition to biodenitrification, activated carbon adsorption is necessary. It should also be pointed out that the operating conditions of a water treatment plant would be different from lab-scale units. It is expected that better operating conditions could be achieved in drinking water treatment plants than in laboratory conditions.

ACKNOWLEDGEMENTS

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SECTION VII:

ENGINEERING

CHEMICAL AND GEOTECHNICAL PROBLEMS ASSOCIATED WITH THE TBILISI WATER STORAGE RESERVOIR, GEORGIA

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Abstract: The Tbilisi reservoir (Georgia) represents an important water supply for the city of Tbilisi, but has suffered considerable problems over the past 50 years. Initially the problems related to leakage of stored water via the schistose bedrock. It contains soluble gypsum and is highly sensitive to particulate removal by the process of suffusion. To some extent the leakage problems were ameliorated by the construction of vertical grout curtains but they have never been fully resolved. In recent times, an additional problem has appeared related to the inflow of saline groundwater from the bedrock host during times when reservoir levels are low. This groundwater locally contains as much as 16,000 mg/l total dissolved solids due primarily to gypsum dissolution. This release of groundwater to the reservoir has impaired reservoir water quality, increasing total dissolved solids from about 250 mg/l in 1988 to almost 500 mg/l in 2002. This has important implications for the long-term viability of the reservoir as a supply of potable water for the city.

Key words: Tbilisi, Georgia; reservoir; water supply; dissolution; sulphate; gypsum; suffusion

1. INTRODUCTION

The Tbilisi water reservoir (Figure 1) is located in the northeast of Tbilisi, the capital city of Georgia, and extends over a distance of about 12 km from northwest to southeast. It provides water for the city of Tbilisi and is also used to irrigate farmland (Buachidze and Chumburidze, 2002).

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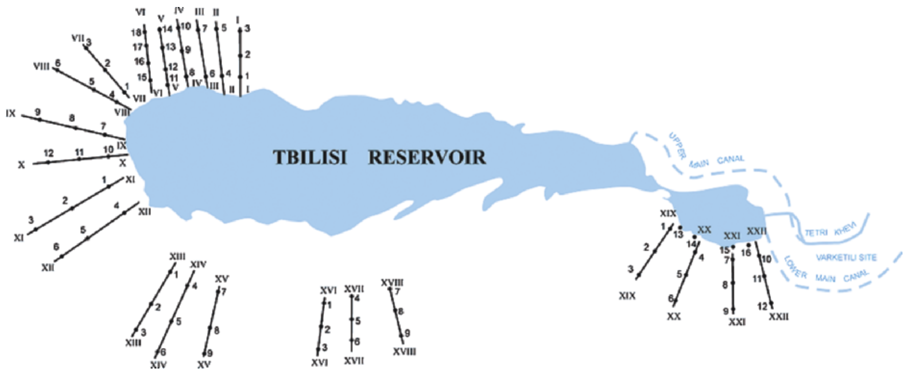

Figure 1. Tbilisi reservoir, showing cross section locations described in the text. The reservoir is about 8.9 km from its northwest tip (left) to its southeast tip (right).

The reservoir was created in the 1950s by directing water from the Rivers Iory and Aragvi into a series of natural depressions occupied by three small saline lakes (Kukia, Ilguniani and Avlabari). To maximize the reservoir storage, two concrete weirs (or “dams”) were constructed along the northwestern fringes of the reservoir to raise the land elevation to approximately 550 m above sea level:

- Weir No. 1 - 237 m in length, comprising 11 blocks with a maximum height of 7 m.
- Weir No. 2 - 290m in length, comprising 14 blocks with a maximum height of 14 m.

The weirs were built on a foundation of thinly bedded to massive schistose sandstones interbedded with mudstones and thin schistose clays. The rocks contain soluble sulphate salts, are easily weathered, and are characteristically unstable due to suffusion processes.

2. LEAKAGE PROBLEMS

The water reservoir was initially filled in 1953. As water levels approached the design capacity, significant losses of water occurred through the weir foundations. Water losses averaged 120 l/s and several concrete blocks subsided by as much as 80 mm. Subsequently, the water level in the reservoir decreased by between 10 and 16 m.

Between 1957 and 1959, a vertical grout screen was constructed in the vicinity of weir No. 1 and, during 1964 to 1965 when the reservoir was refilled, filtration losses were observed to have decreased by a factor of 13. However, no equivalent grout screen was constructed at weir No. 2, where a combination of sulphate salt dissolution and sediment removal due to suffusion increased water losses by a factor of 8 (from 1 l/s to 8 l/s).

In 1973, when the reservoir water level was 542 m above sea level, two major leaks ("gryphons") were observed along the southern flanks of the reservoir. When water levels in the reservoir were lowered to between 535 and 536 m, one of these leaks stopped flowing and the other was considerably diminished.

3. GEOTECHNICAL CONSIDERATIONS

In 2002 a new observation network was constructed at 9 sites, involving a series of boreholes along each of 27 sections radiating from the reservoir. 22 of these sections are shown on Figure 1. A typical section (V-V) is shown on Figure 2 and major ion hydrochemical data for the reservoir water and groundwater collected in boreholes along sections I to V [in the vicinity of the constructed dams (see example in Figure 2)], are provided in Table 1.

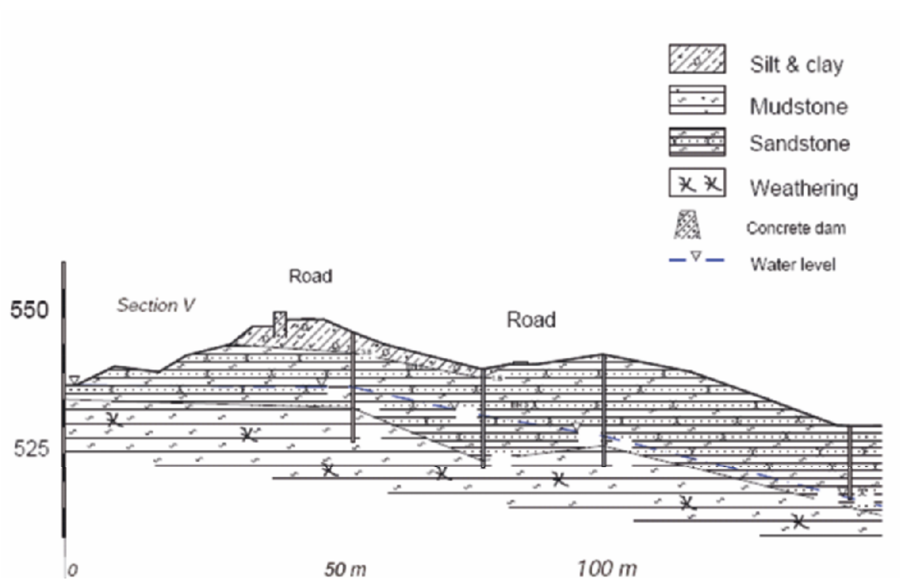

Figure 2. Typical section from reservoir across dam.

Table 1. Water chemistry of Tbilisi Reservoir water and adjacent groundwaters (for Section locations, see Fig. 1; BH = borehole; WL = water level; Dist = distance from reservoir; TDS = total dissolved solids).

Section	BH	WL (m)	Dist (m)	TDS (mg/l)	HCO_3 (meq/l)	Cl (meq/l)	SO_4 (meq/l)	Na+K (meq/l)	Ca (meq/l)	Mg (meq/l)
Res		537.5	0	484	0.8	0.48	3.6	4.8	1.75	1.3
I-I	1	536.6	30	3996	6.0	0.96	52.6	37.3	9.7	13.6
I-I	2	528.1	63	4136	1.8	0.96	60.2	29.5	16.0	17.5
I-I	3	527.7	82	7144	3.4	0.96	104.6	72.1	12.8	24.1
II-II	4	529.0	27	2356	5.8	0.96	31.9	12.7	12.4	13.6
II-II	5	526.0	64	2616	6.6	0.96	32.6	14.6	14.6	10.9
III-III	6	529.8	40	1397	3.6	0.96	16.2	3.6	11.5	3.6
III-III	7	526.5	72	4200	6.6	0.96	57.8	44.3	4.9	16.2
IV-IV	8	537.2	38	5478	5.0	0.96	78.0	40.1	11.1	32.8
IV-IV	9	536.7	70	9303	8.2	1.92	132.5	53.2	16.8	72.6
V-V	11	537.9	51	5007	15.4	1.44	59.8	55.7	11.7	19.2
V-V	12	532.1	74	12008	12.5	2.31	163.2	90.9	16.4	71.7
V-V	13	529.0	97	16040	16.6	1.44	11.4	20.3	1.7	7.4

In considering sections I to VI, three geotechnical elements (GE) can be identified (Tevzadze, 2000).

GE-1 is saline due to the presence of sulphate. It is a weak to moderately swollen loamy material with inclusions of gravel (27.6%). The plasticity index is 0.136 and rock consistency is hard. The rock is characterized by:

- a low compressibility (0.022 - 0.015 MPa^{-1});
- a module of elasticity equal to 27.0 MPa;
- a compressibility of 0.036-0.016 MPa^{-1} when saturated;
- a module of elasticity of ~ 22.2 MPa when saturated;
- a calculated angle of internal friction of 26.1°;
- a cohesion of 0.0492 MPa.

GE-2 is heavily weathered, softened and has turned into plastic loam. The mudstone eluvium includes dry, unweathered gravel-sized fragments of the same rock. The rock is characterized by:

- a mean volumetric moisture content of 26.9%;
- a bulk density (ρ_b) of 1.90 g/cm^3:
- a matrix density (ρ_s) of 1.58 g/cm^3;
- a water saturation of 77%;
- a plasticity index of 0.192;
- a “free swelled size” of 4.6%;
- a hard consistency;
- a gypsum content of 1.645%; (pore water SO_4 8.82 meq/l, Ca^{2+} 5.40 meq/l, and Mg^{2+} 3.62 meq/l);

- a cohesion of 0.06 to 0.027 MPa^{-1};
- a modulus of elasticity of 10.86 MPa;
- an angle of internal friction of 13.2°;
- an adhesion of 0.014 MPa;
- a hydraulic conductivity of 10^{-3} m/day, increasing to 0.1 m/day for less weathered mudstone.

GE-3 is represented by thin, friable schistose sandstones and moderately thick, hard schistose sandstones. The matrix density (ρ_s) of the former does not exceed 2.14 g/cm^3, and of the latter is 2.47 to 2.60 g/cm^3. The compressive strength (c) under uniaxial compression is between 22.4 and 26.5 MPa, the angle of internal friction (ϕ) is 34 to 35° and adhesion is 12.3 to 12.7 MPa. Under saturated conditions the compressive strength and angle of internal friction correspondingly decrease (ϕ = 14° – 15° and c = 5.0 to 5.6 MPa).

Based on the laboratory and field geotechnical analyses, the following main conclusions can be drawn:

1. The foundations of the concrete weirs of Tbilisi reservoir can be subdivided into three geotechnical elements: GE-1 (loam), GE-2 (heavily altered mudstone eluvium containing readily soluble sulphate salts), and GE-3 (sandstones). The zone of weathering extends to a depth of 20 m and is characterized by high volumetric moisture contents and relatively low densities. The less weathered sandstones are characterized by low volumetric moisture contents (1.66 to 3%) and relatively high densities (2.59 to 2.68 g/cm^3) and high compressive strengths under uniaxial compression (22.4 to 26.5 MPa).
2. The flow of water as leakage from the reservoir via the weir foundations significantly influences the geotechnical properties of the different rock types. This has implications for the long-term structural stability of the reservoir.

4. WATER QUALITY CONSIDERATIONS

In addition to the geotechnical investigations, comprehensive studies of the water table and water chemistry were undertaken using a network of installed piezometers. These investigations showed that groundwater flow directions radiate away from the reservoir during times when the reservoir is full and water levels in the reservoir exceed groundwater levels. However, part of the subsurface outflow returns to the reservoir when reservoir levels are lowered to below groundwater levels. This reverse flushing of water has an important influence on water quality in the reservoir and may enhance both gypsum solubility and suffusion processes.

Chemical analyses of water samples from the reservoir and from piezometers 1 to 9 and 11 to 13 at Site 1 (Sections I to V of Figure 1) are presented in Table 1. In all cases:

- Solute concentrations in the groundwater (total dissolved solids ('dry residue') values of 1.397 to 16.040 g/l) significantly exceed the concentration in the reservoir (0.484 g/l).
- There is a significant correlation between groundwater salinity and distance from the reservoir, with remote piezometers showing considerably higher mineralization levels. In section I–I for example, salinity increases from 3.996 to 7.144 g/l away from the reservoir; in section II–II it increases from 2.356 to 2.616 g/l; in section III–III it increases from 1.397 to 4.2 g/l; in section IV–IV it increases from 5.478 to 9.303 g/l; and in section V–V it increases from 5.007 g/l to 16.04 g/l.

In all cases, the dominant anion is sulphate derived from gypsum. Concentrations range from 3.6 meq/l in the reservoir to between 16.2 and 163 meq/l in the groundwater. Cations are more variable but Na^+ plus K^+ is usually dominant. In all cases concentrations of individual ions in the groundwater exceed corresponding concentrations in the reservoir.

Historical data demonstrate the important influence of elevated groundwater salinity on water quality in the reservoir:

08. 04. 1988 – 0.152 g/l – source water from R. Aragvi;
07. 07. 1988 – 0.214 g/l – water reservoir;
09. 08. 1988 – 0.248 g/l – water reservoir;
10. 08. 2003 – 0.484 g/l – water reservoir.

The first observation is that in 1988 the inflow of groundwater to the reservoir had the effect of increasing reservoir salinity from 0.152 g/l (the source water) to between 0.214 and 0.248 g/l. By 2003, 15 years later, inflow of groundwater to the reservoir during times of low reservoir water level had resulted in an increase of salinity by a factor of 2 to 0.484 g/l. This relatively rapid increase of salinity to concentrations that are only marginally lower than permitted by drinking water quality standards has important implications for the long-term viability of the reservoir as a supply of potable water for the city.

5. CONCLUSIONS

During the past 50 years, the Tbilisi reservoir has suffered many problems. Initially, the problems concerned leakage of stored water via a schistose bedrock that contains readily soluble gypsum and is also highly sensitive to particulate removal by the process of suffusion.

Observed geotechnical and subsurface flow impacts include:

- Local mudstone weathering resulting in its transformation to a saturated clay-rich sediment with inclusions of gravel
- A reduction in matrix density and increases in volumetric moisture content and water saturation
- An increase in rock compressibility and a reduction in the modulus of elasticity
- Reductions in adhesion and the angle of internal friction
- A decrease in hydraulic conductivity from 10^{-1} cm/s to between 10^{-5} and 10^{-6} cm/s
- Sub-surface water losses initially as high as 120 l/s, and associated subsidence of up to 80mm along the weir crests
- The appearance of major concentrated leaks ("gryphons") when water levels in the reservoir are at or above 542.5 m.

To some extent water loss problems were improved by the construction of vertical grout curtains but they have never been fully resolved.

In recent times an additional problem has appeared related to the inflow of saline groundwater from the bedrock host. Due to gypsum dissolution, this groundwater contains as much as 16,000 mg/l total dissolved solids, and the flow of this groundwater into the reservoir has raised lake reservoir salinity from around 250 mg/l in 1988 to almost 500 mg/l in 2002. This rapid rise in salinity has important implications for the long-term viability of the reservoir as a supply of potable water for the city.

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HYDROGEOLOGY AND ENGINEERING GEOLOGY OF THE 'SLEEPING DISTRICT' (VARKETILI) OF TBILISI, GEORGIA

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Abstract: Buildings in the Varkethili 'Sleeping District' of Tbilisi are often affected by subsidence. They are underlain by the extensive third left bank terrace of the River Mtkvari, formed by products of erosion of the flat-lying slopes above and cut by natural drainage channels. Geotechnical and chemical analyses of the sediments suggests that the main reasons for subsidence are chemical and pore water pressure changes in the reactive sediments brought about in part by changes in the recharge regime which in turn was the result of urbanization.

Key words: subsidence; buildings; Tbilisi, Georgia; terrace deposits; chemical reactivity; engineering; River Mtkvari

1. INTRODUCTION

The Varketili massif, around 82.5 hectares in area, is situated within the limits of Tbilisi, the capital city of Georgia, on the left bank of the River Mtkvari. The climate in Tbilisi is temperate continental, characterized by dry, hot summers and mild winters, whilst orographically,the study region lies at the eastern end of the Adjara-Trialeti fold system.

The study region is located on the left bank of the River Mtkvari in an area characterized by low relief. Plateaus and sloping valleys are developed, and as a result, the catchments are relatively flat-lying. It has been established that the main reason for the deformation of buildings found in the area is intensive ground shrinkage beneath the buildings, despite the

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wetting of the upper soil horizons by atmospheric precipitation and water losses from leaking water supply pipelines.

2. HYDROGEOLOGICAL AND GEOLOGICAL CONDITIONS

The hydrographic network of the region is associated with the River Mtkvari basin. The left-bank valleys of the River Mtkvari are weakly developed and inflows are small because of the low elevation of the catchments, the low intensity of the precipitation, and the high evaporation.

The most significant inflow from the left bank on the east of the study area is from the River Lochini, characterized by a comparatively developed flow system. The only other notable tributary near the study area, the River Orkhevi, is formed by the two smaller Rivers Pokaraant-Khevi and Tatris-Khevi, which start at the southern foot of the Kashveti-Tselubani plateau. Apart from these tributaries, there are many ravines and dry gorges, which only flow during periods of rain.

All the study area is underlain by Tertiary sediments (Buachidze, 1970) present within Adjara-Trialeti fold system. These sediments were formed in a geosynclinal regime, and are very thick. Quaternary alluvial and hill-wash (proluvium and deluvium) deposits overlie the bedrock. These sediments are characterized by very variable thicknesses (1 to 20 m). Generally, fill material is present immediately below ground surface. This artificial fill, represented by loams, is usually no more than 3 m thick.

The proluvium-deluvium sediments are widely spread over the research area. They were formed by the relatively easy erosion of the existing sediments, especially clays, and their creation is hence a product of geological distributions, geomorphological gradients, and climatic conditions. They are composed of sand, clay, and yellowish-rusty coloured loams, often mixed with detritus and sometimes larger clasts of bedrock. Frequently re-worked material from higher terraces and Miocene-Pliocene pebbles are found in these deposits. At their most developed, they are up to about 18 m in thickness.

Modern alluvial sediments are not wide-spread in the region. Their creation is associated with the erosion by the River Mtkvari and its tributaries. The main sediment types are small and large pebbles, gravel, and sand. Total thickness is no more than 3 m.

Ancient alluvial sediments are developed on all the flood terraces of the River Mtkvari and its tributaries, from the modern floodplains of the River Mtkvari gradually step-by-step through higher terraces as height climbs towards Mount Makhata and Mount Varketila, a change in elevation from 2-3 to 520 m.

The research area is located on terrace III. The alluvial deposits contain pebbles, clays with boulders, limestone blocks, and loams, with a total thickness of 3-6 m.

According to the hydrogeological classification of Georgia (Buachidze, 2000), the study area is located in the Tbilisi water 'pressure' system. Here, above the very large, water volcanogenic formations of middle Eocene age lie relatively low permeability clay-sand sediments of the upper Eocene. The upper Eocene sediments are permeable in the weathered zone, and fracture flow is dominant. This horizon is characterized by SO_4-HCO_3-Na and SO_4-Ca-Mg water types. The salinity is 2.5 to 4.5 g/l. Sources yield no more than 0.1 l/s but the water is drinkable.

Quaternary sediments overlie the bedrock all over the study area, and are represented by alluvial and deluvial-proluvial deposits of thickness 1 to 18 m. The most permeable are the alluvial sediments. Groundwaters are characterized by high permeabilities (15-20 m/d), with well yields of up to 0.5 l/s. The groundwaters are of SO_4-Ca-Mg type, with a salinity of 2.0-3 g/l.

The Varketili plateau is comparatively small in size, with smooth hills; to its southwest is Mount Makhata. The plateau is elongated almost east-west. This plateau is high above the east side of Tbilisi and decreases in elevation slowly towards the Tbilisi Sea hollow. The absolute elevation of the plateau on its western side is 650 m, and on its eastern side is 635 m. It is 270 m above the River Mtkvari. The plateau has a length of 4 km and a width of 1 km. On all sides, slopes are smooth.

The surface of the Makhata terrace is located at an elevation of 500-520 m. It is cut by parallel channels, described below. There are fragments of flattened surfaces with elevations of 466-470 m, the "Upper plateau", where new blocks of flats are situated.

Precipitation flows to the channel network. The largest channel with permanent flow is located at the western end of the area behind Djavakheti Street, and is marked as channel 1 on Figure 1. This channel flows into the River Mtkvari near Dimitrov`s factory. Its axis is aligned northwest/southeast, becoming north/south in the downflow (southerly) direction. Within the study area, the most developed tributaries to channel 1 are from the left-side (1.1, 1.2, 1.3). Channel 1.1 has third-order channels (1.1.1 and 1.1.2) which pass out of the study area. Channels 1.2 and 1.3 cut through the surface sediments. Channel 1.2 passes between blocks 7 and 4 of the Varketili housing estate. Channel 1.3 passes near Trialeti Street. Channels 2 and 3 are to the east of the study area. Finally, there is short channel, flowing adjacent to Tsnoristskali Street.

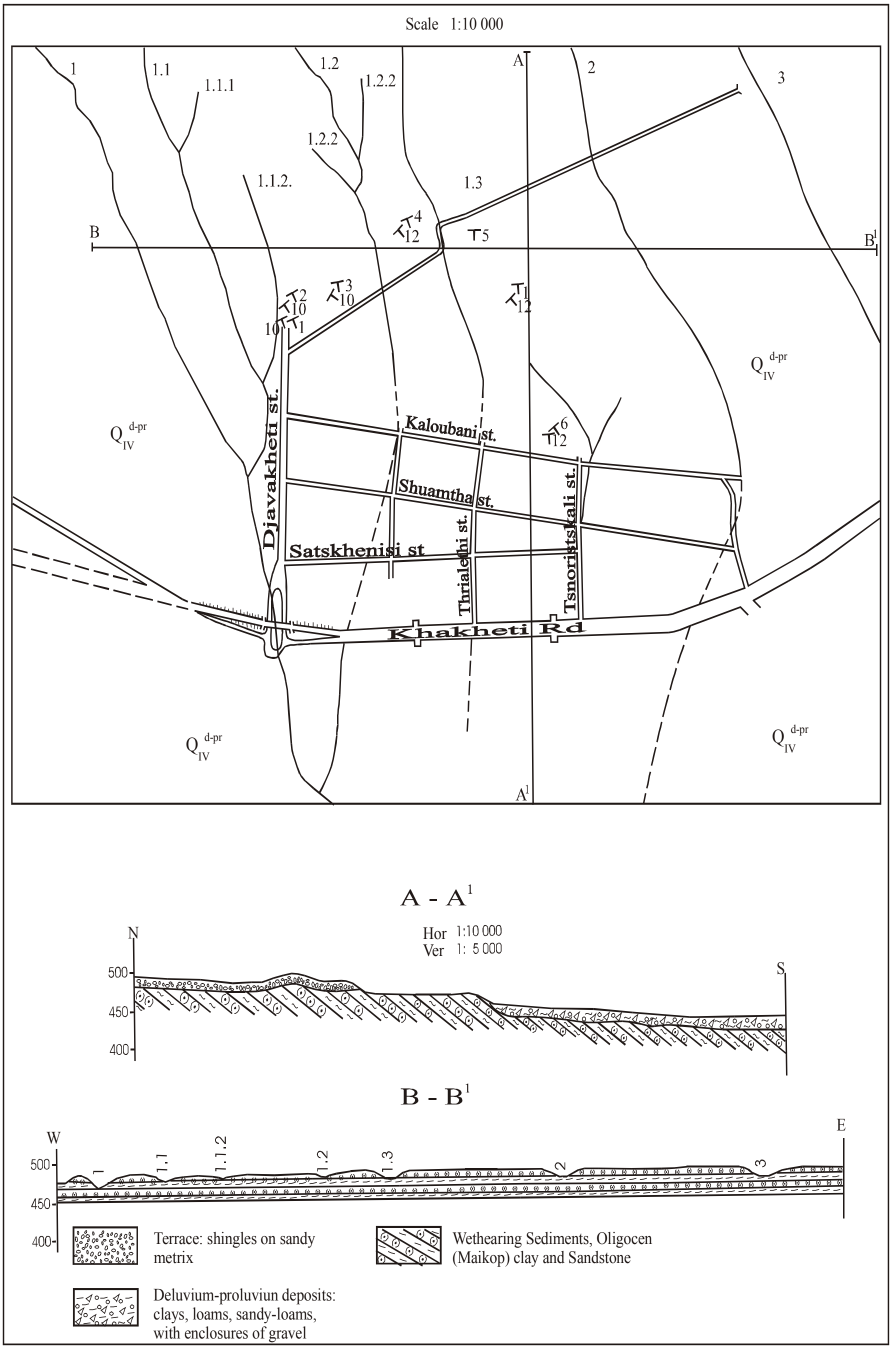

Figure 1. Schematic engineering geology map and cross-sections.

The plateau surface is underlain by loamy-sand with pebbles and clay. These sediments overlie sands and clays of upper Eocene and Oligocene age, dipping to northwest at an angle of 25^0. Quaternary sediments of different thicknesses and comprised of sands and finer-grained deposits are present all over the area.

The most widespread deposits are the deluvial-proluvial sediments, represented by different gypsiferous, carbonate-containing loams with pebbles and clay. The median thickness of such layers is 5-9 m; in the central part of the area, the loams are 2-5 m in thickness, but to the southwest they increase in thickness up to 9-12 m. (Figure 1).

3. HYDROGEOCHEMICAL AND PHYSICAL CONDITIONS

The clay mineral content of the deluvium-proluvium loams includes hydro-mica, hydro-mica/montmorillonite, and montmorillonite; gypsum content is 5-30% and sometimes more. Very saline loams are present up to 3 m depth, and sometimes to 6 m. Loams with gypsum are found mainly at a depth of around 6-6.5 m. Their colour is whitish, yellowish, brownish and dark brown.

The hydraulic and Atterberg limit properties of these loams are similar (Table 1). The plasticity indices are 10-16, and the liquidity and plasticity indices are 19-36% and 7-23% respectively. Bulk density ranges from 1,740 to 2000 kg/m^3, and porosity from 39 to 46%. The shrinkage limit is high, with values between often above 20%, and a range of 10 to 27%. Cohesion is quite low, being between 0.3 and 0.53 Pa, and the angle of internal friction ranges from 47 to 49^o.

The hydrogeology of the study area is controlled by the geomorphological and geological structure. A characteristic feature of the rocks, developed on the Varketili massif, is the relatively deep groundwater level and the permanent recharge by surface waters.

The level of the groundwater within the area is 2.3 (hole No. 342) to 12.0 m below ground surface (hole No. 312). Groundwater levels roughly correspond to the boundary of the 'bedrock' (i.e. pre-Quaternary sequence) and the Quaternary sediments, and the water level varies with bedrock relief. Where pebbles block the upper Eocene Sands, the water level is increased accordingly and filling this aquifer. On the northeast of the area, on Kaloubani Street the water level is at 4.0 m (hole No. 123, 156) to 6.0 m depth (hole Nos. 278, 279, 3): here alluvial pebbles block 'bedrock' sands and finer deposits at a depth of 5-7 m. Comparatively high groundwater levels occur on the southeast of the area [from 2.5 m depth (hole Nos. 283, 282) to 5.0 m depth (hole Nos. 267, 13, 321)]. In general, throughout the Varketili area, the groundwater level changes from 8 to 12 m depth, with the decrease in elevation from north to south.

Table 1. Geotechnical properties of the weathered deposits [B/H = borehole; depth = depth of sample θ = natural moisture content; LL = liquid limit; PL = plastic limit; PI = plasticity index; ρ = density (subscripts: w = water; g = grain; b = bulk; db = dry bulk); n = porosity; RS = relative shrinkage; ϕ = angle of internal friction (^{o}); c = cohesive strength.

B/ H	Depth m	θ %	LL %	PL %	PI %	ρ_g /ρ_w	ρ_b /ρ_w	ρ_{db} /ρ_w	n %	RS %	ϕ ^{o}	c Pa
Weathered Sandstones												
1	6.0	21	36	23	13	2.70	2.88	2.40	0.11	27.3		
2	6.0	17	34	20	14	2.74	2.12	1.81	0.33	23.6		
		13	28	16	12	2.70	2.09	1.84	0.32	14.3		
Weathered Mudstones												
10	7.0	18	36	17	19	2.74	1.78	1.52	0.44	14.3		
13	9.0	23	33	16	17	2.70	2.04	1.65	0.39	14.3		
14	10.5	17	40	22	18	2.70	1.91	1.63	0.40	14.3		
Weathered Clays												
2	6.0	23	33	16	17	2.69	1.86	1.51	0.44	27.3	49	0.53
2	3-3.2	18	32	16	16	2.71	1.95	1.65	0.30	23.6		
2	6.2	15	19	9	10	2.72	1.76	1.53	0.43	27.3		
Weathered Loams												
21	6-6.2	30	21	7	14	2.70	1.86	1.49	0.45	23.6		0.30
29	6-6.2	19	29	14	15	2.71	1.81	1.61	0.40	14.3		0.40
13	6.0	16	32	20	12	2.71	1.74	1.50	0.45	27.3		0.38
8	6.0	19	21	12	9	2.69	2.00	1.98	0.30	27.3		
9	3.0	23	19	13	6	2.67	1.89	1.50	0.39	27.3		
13	1.5	17	28	12	16	2.70	1.73	1.47	0.46	23.6		
14	3.0	14	25	9	16	2.69	1.79	1.58	0.42	10.3		
16	3.0	19	35	22	13	2.69	1.78	1.50	0.45	10.3		
22	6-6.2	16	32	18	14	2.71	1.92	1.63	0.40	27.3		
3	3-3.2	16	36	23	13	2.71	1.83	1.57	0.42	24.3	48	0.38

The groundwater of the Varketili massif can be classified according to chemical content into 2 types using total mineralization and the content of the following components (SO_4^{2-}, HCO_3^-, Cl^-, Ca^{2+}, Mg^{2+}, Na^+) (Table 2):

- Type I waters – these are SO_4-rich waters and are found over almost all the area at the contact of the Quaternary clays and loams with the pebbly alluvium where these block the upper Eocene sands and finer deposits. Groundwaters of the gypsum loams and bedrock sands, as a rule are SO_4 – Ca or SO_4-Ca-Mg in composition, with salinities of 2-3 g/l. In boreholes 1, 6, 22, 189, 113, 256, 362, and 358, groundwater salinity is 2.2-2.9 g/l, and the water type is SO_4-Ca-Mg.

- Type II waters – these are SO_4-rich waters with greater mineralization (3-5 g/l) (hole Nos. 187, 183, 128, 126, 127, 188). They are found where the Quaternary gypsum loams lie directly on bedrock sands and finer-grained deposits without pebbles being present. We think that this situation occurs between the channels, and that the increase of mineralization is connected with the fact that water circulation difficulties slow allowing greater time for solution of SO_4, Na, and Mg ions from the upper Eocene sequence.

Water samples, taken from the Tbilisi water collector, channels, and water mains are HCO_3-Ca-Na in type with salinities of 0.2-0.4 g/l. Identical waters are not seen in the boreholes, although at most borehole sites the upper levels of the profiles (1.5-2.5 m depth) are moist because of water leakages and household discharges.

Table 2. Chemical data for the groundwaters (mg/l unless otherwise stated; TDS = total dissolved solids).

BH No.	**Date day/m**	**pH (-)**	**TDS (g/l)**	**Na**	**K**	**Ca**	**Mg**	**Cl**	**SO_4**	**HCO_3**
T1*										
1	20/9	6.9	2.775	52	5	608	770	21	1686	306
6	31/9	7.2	2.584	68	2	608	0.1	18	1590	238
22	18/8	7.2	2.596	75	6	624	35	28	1516	311
113	20/9	7.1	2.693	81	1.6	616	30	36	1584	238
189	20/10	7.1	2.62	56	8	620	55	25	1562	294
256	15/10	7.2	2.556	51	4	656	57	28	1579	281
358	26/10	7.2	2.188	60	6	620	31	25	1420	24
T2*										
126	22/10	7.1	5.013	860	4	540	123	178	2880	469
127	22/10	6.9	4.909	750	2.3	528	102	170	2842	415
128	10/11	7	4.945	720	1.6	540	210	178	2869	427
183	10/10	7.2	3.25	171	78	600	105	36	1990	271
187	10/10	7.1	3.884	164	741	956	88	37	1987	272
188	10/10	7	3.215	155	6.2	600	118	32	2045	259

**T1 = 2-3 g/l TDS, sulphate type; T2 = 3-5 g/l TDS sulphate type*

4. CONCLUSIONS

From the data summarized above, it is concluded that the influence of groundwaters on building stability is in part due to the water/rock chemical reaction, which leads reduction in ground strength, porosity increases, and plasticity increases: this in turn leads to settlement. The main factors causing building deformation are therefore groundwaters and the composition of the rocks. Precipitation recharge and household discharge waters result in dissolution of the SO_4- and CO_3- rich underlying rocks, thus changing their structure.

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AN INTEGRATED EVALUATION PROGRAM FOR THE ASSESSMENT OF VARIATIONS IN URBAN GROUNDWATER LEVEL

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Abstract: Bridging the gap between a need for preserving unique historic cities and a requirement for modern infrastructure development is one of the major challenges for sustainability of urban areas. Many historic cities around the world have complicated ground conditions, commonly represented by weak soils and a high groundwater table. Natural, or historically formed, hydrogeological conditions need to be maintained carefully in such ground conditions, since even insignificant variations of groundwater level and gradient may affect valuable historic buildings. This paper discusses an integrated evaluation program for the assessment of causes and consequences of variations in hydrogeological parameters within urban historic environments. An analytical quantitative method based on hierarchy structures is used for the program. Some examples of the potential use of the assessment program in Hamburg and St.Petersburg are given.

Key words: biocorrosion; groundwater level; underground construction; wooden foundations; historic buildings; St Petersberg Russia; Hamberg Germany.

1. INTRODUCTION

Underground space plays an important role in the modern development of historic cities. In city downtown districts, where lack of available space is observed both above and below ground, new facilities are required in order to maintain an up-to-date infrastructure.

The integrated evaluation program for assessment of variations in urban groundwater level variation described herein pays particular attention to hydrogeological issues arising from environmental impact assessments of

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urban underground structures. However, the program can also be used separately to assess hydrogeological conditions from an environmental standpoint and assessment results can be used for decision making on the location and size of underground facilities yet to be installed.

Another aspect considered in the program is the safety of existing buildings and underground structures, since groundwater level plays a major role in the geotechnical stability of soils.

2. IMPACTS ON GROUNDWATER

Groundwater level plays an important role in the stability of natural and artificial structures within urban areas. There are a number of important artificial factors which affect groundwater level, and, in turn, groundwater level has a significant impact on environmental parameters (see Figure 1).

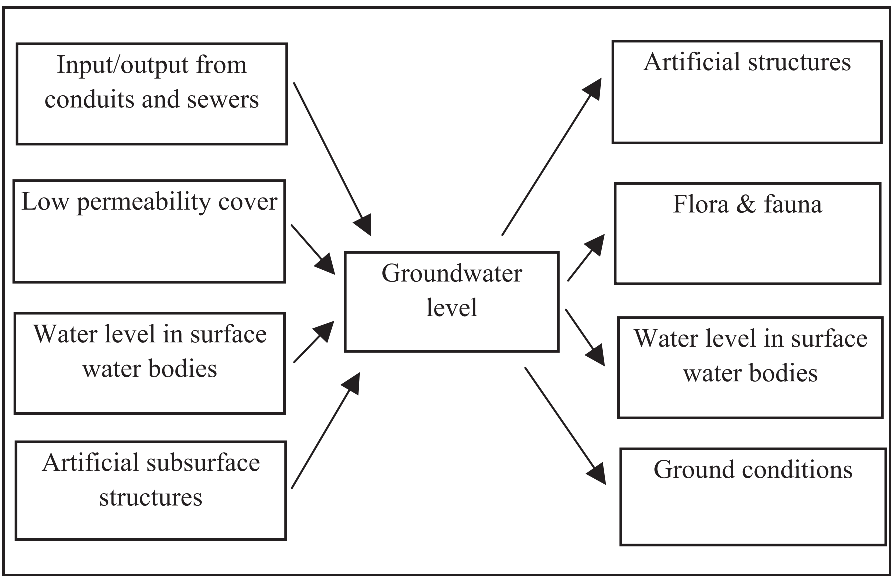

Figure 1. Artificial impacts of and on groundwater level within urban areas.

Uncertainties in characterizing ground conditions and the technical condition of underground structures can result in geotechnical failures. Underground space beneath old cities is saturated with caverns, conduits, archaeological and historical features, and recently developed structures (Bobylev et al., 1998). Hydrogeological conditions within these regions are primarily formed by the impact of artificial factors, and it is very important to study those factors before carrying out underground construction activities (Bobylev, 1999). A case study from the city of St. Petersburg serves as a good example of this. After the excavation of a new site, engineers discovered the hydrogeological situation to be totally different from that anticipated. The reason was the presence of an old canal, with a lining made of bricks. Filled with broken bricks, this canal was providing perfect drain, and gradients of groundwater ingress into this canal were 1.7 times higher than into the nearby river.

Amongst pressures on the groundwater systems of urban sites, one could mention input and output from pressured conduits and sewers, surface water input from floods and heavy showers, water ingress to caverns with low quality waterproofing, release of above- and subsurface contaminants situated outside the construction site, and leakage of liquid pollutants from the construction site.

Embankments, faced with stone slabs, represent a special problem for groundwater discharge. A damming effect is observed over most of the embankment, but at points with defects in facing, high velocity groundwater flow can occur, leading to possible suffusion.

3. ENVIRONMENTAL IMPACT ASSESSMENT

Integrated evaluation programs for the assessment of urban groundwater level variation were originally used as tools for understanding hydrogeological environmental impacts. The program described in this study uses an analytical quantitative method, based on a hierarchy structure, which is itself founded on a multi-criteria optimization approach. The analytical basis is the method of analytical hierarchies defined by Tomas Saati and the “Expert Choice” computer program (Saati, 1993). Quotient multi criteria values are combined according to an original analytical solution at the final stage of the present method. Results are represented by environmental criteria, which provide the basis for decision-making on environmental quality in the area considered. An example of a hierarchy structure for environmental parameters used for the method is presented in Figure 2.

Environmental impact assessment is an important tool for decision-making processes in civil engineering. A new development of underground infrastructure within a historic city requires the special attention of environmental experts and hydrogeologists, since groundwater issues are critical and must be considered. Groundwater impacts are normally most pronounced during the creation of an underground opening, and the main output of this study is the production of guidelines for the mitigation of such impacts.

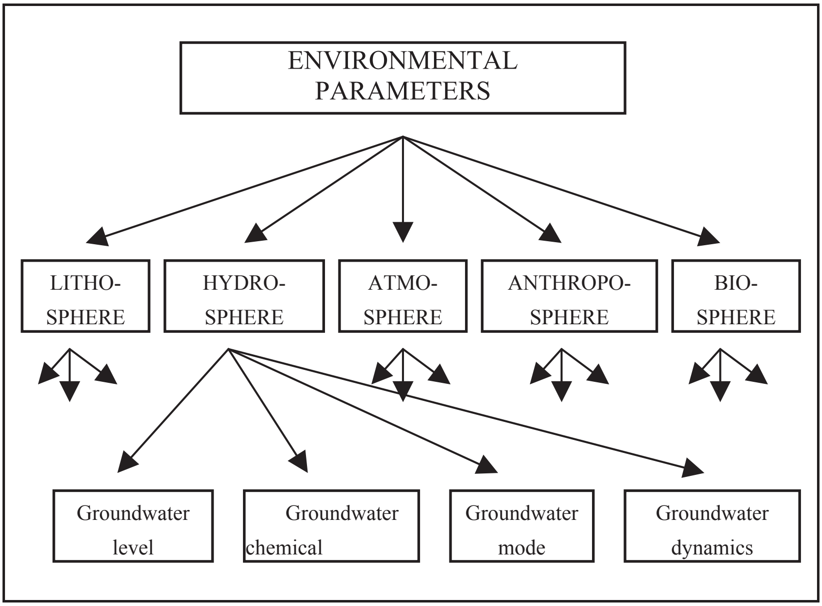

Figure 2. Example of a hierarchy structure for environmental parameters.

4. GROUNDWATER LEVELS AND FOUNDATIONS OF HISTORIC BUILDINGS

One of the most important factors to consider in the maintenance of groundwater at its natural level in old towns and cities are the wooden foundations of unique historic buildings. Typically, ancient building foundations built on weak soils consist of wooden piles or logs, with stones positioned directly beneath them. An example of excavated wooden piles can be seen in Figure 3.

The wooden logs and piles of such foundations are very well preserved in the anaerobic conditions formed by a high groundwater level provided the water is uncontaminated. Unfavorable impacts on those logs can occur via organic contaminants in the groundwater (e.g. leakages from sewers and infiltration of above-ground contaminants) and the lowering of groundwater level. Aeration of wooden logs leads to their biocorrosion and the destruction of the foundation, eventually causing an uneven settlement of the historic building above. A number of studies have been carried out describing and analysing the connection between wooden log properties, groundwater chemical composition, time and conditions of aeration, and chemical preservations that should be used for preventing biocorrosion

(Ulitsky *et al.*, 1999). Old conduits and underground structures can also be affected by biocorrosion.

Figure 3. Thousands of wooden piles are excavated while creating a new underground development in Hamburg, Germany.

5. CASE STUDY OF ST. PETERSBURG

5.1 The Structure and Environment

An underground multi-purpose structure at Truda Square in the city of St. Petersburg encompasses a pedestrian route and shopping area. The total area of the structure is 2509 m²; with plan dimensions of 66 x 78 m; the typical depth of the foundations is 5.8 m, with a maximum depth of 9.8 m; and the floor-to-ceiling height of this one-level structure is 3 m.

Construction started in 1993, but between 1994 and 1997 it was halted due to lack of financing. A structure surrounding the excavation proved to be insufficiently waterproofed, leading to a fall in groundwater level. As a consequence, House #6 on Konnogvardeisky Street was severely damaged (the foundations suffering differential settlement of up to 3.07 cm), whilst a number of other buildings suffered less serious damage. All affected buildings had wooden foundations. Figure 4 shows a cross-section of the underground structures, including the excavated opening and adjacent buildings that were affected by the lowering of groundwater level. The problem was finally remedied in 1998 by the creation of a waterproofing barrier via high-pressure injections of clay slurry.

Truda Square is located in an historic area of St. Petersburg, bordered by the Krukov canal on one side and Konnogvardeisky Street on the other. The groundwater level is situated quite close to the surface, being only two to three metres below it. Ground conditions in the area consist mostly of silt and silty sand, with layers of peat commonly found.

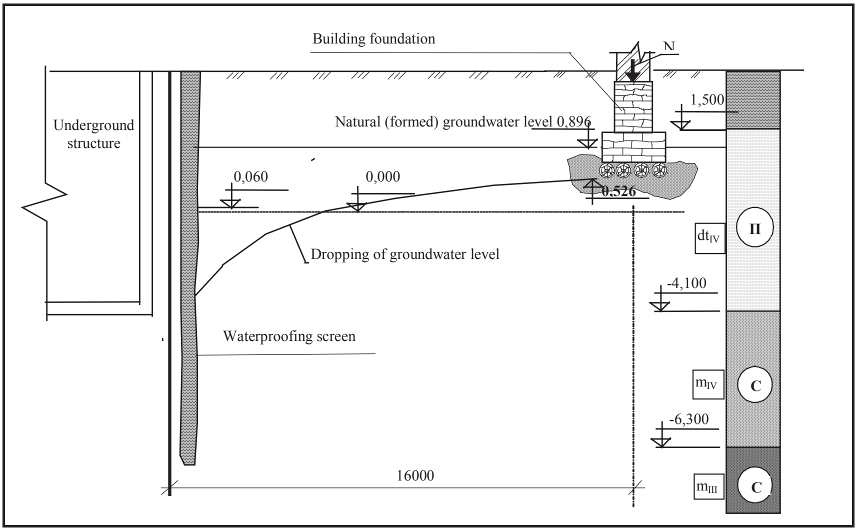

Figure 4. The settlement effects of aeration and the lowering of groundwater level on a building with wooden foundations. Truda Square development, St. Petersburg.

5.2 Groundwater Level Variation

Groundwater level dropped significantly during the earlier years of construction but, after the waterproofing barrier was created, a damming effect was observed. Additionally, shortly after the opening was created, the Krukov canal became a recharge source instead of providing discharge. This led to changes in flow gradient, with high groundwater velocities observed beneath the foundations of many buildings. This situation changed once the barrier was created: the near-stationary nature of groundwater in the affected area was restored, but the edges of the underground structure continued to experience high groundwater velocities. This was observed approximately two years after the barrier was created, with a maximum damming effect of +0.5 m. Currently the maximum damming effect constitutes not more than +0.2 m. The region immediately surrounding the new structure experienced significantly changed hydrogeological conditions during the six-year period. It was clearly evident that soil properties in a large number of locations were damaged during that period, and cement injections had to be made to protect building foundations and conduits. Figures 5 and 6 show the variation in groundwater level at the two points where it was most pronounced (both points being located at the border of the underground structure).

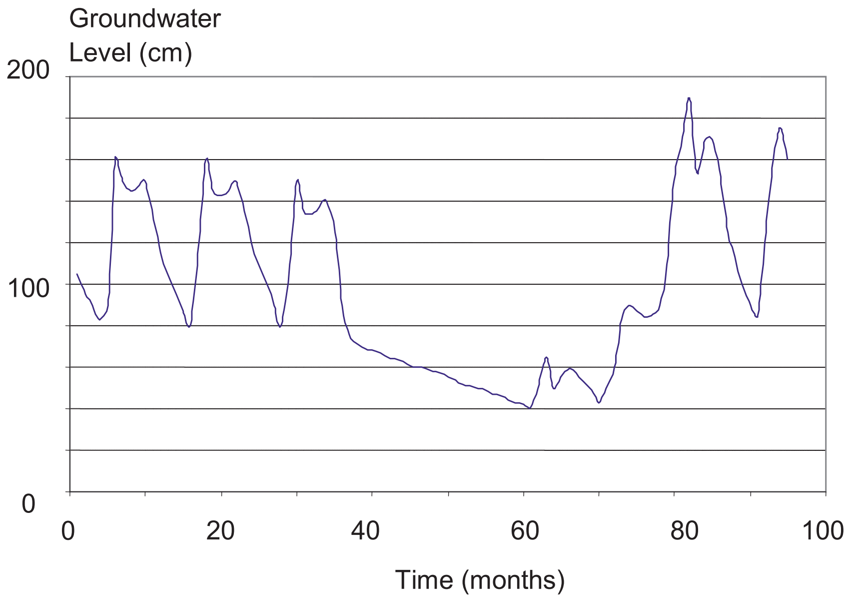

Figure 5. Groundwater level variation over the period of maximum lowering of water table.

6. CONCLUSIONS

Research into variation of urban groundwater level requires many issues to be taken into consideration. A multi-criteria approach provides an opportunity for integrating miscellaneous information originated from different spheres of knowledge. Natural and artificial factors both affect hydrogeological conditions within urban areas, and groundwater in its turn is one of the major factors affecting the stability of the built environment. Some of the environmental problems arising from activities of underground construction works have only recently been revealed. The true reasons for groundwater contamination, groundwater level variation, and suffusion while excavating openings are very often shown to be the poor condition of old conduits and foundations. Uncertainty over the condition of the artificial environment (both existing structures and those that have been abandoned or destroyed) and its characterization seems therefore to be the major concern while carrying out underground works in old, historic cities.

An integrated evaluation program for the assessment of urban groundwater level variation could be used for feasibility studies on both current and future underground construction activities. The analytical basis

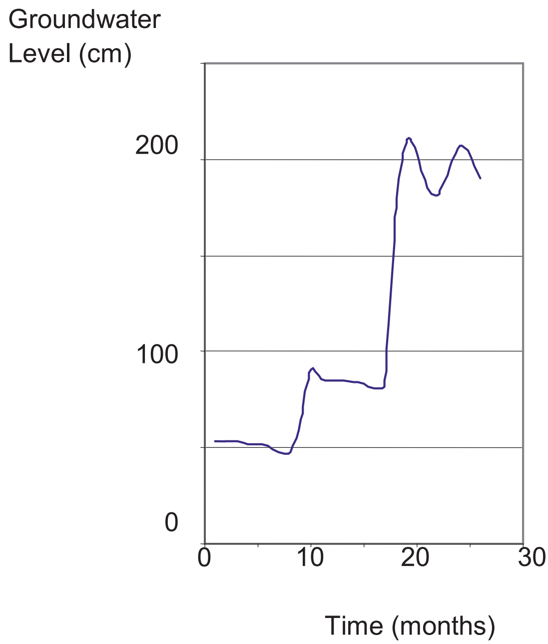

Figure 6. Groundwater level variation at the point of the maximum damming effect.

suggested is the method of analytical hierarchies defined by Saati (1993), and included in the "Expert Choice" computer software. This program will be helpful also in planning and zoning, decision-making on urban underground infrastructure rehabilitation, renovation, and installation.

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THE HYDROTHERMAL SYSTEM OF TBILISI, GEORGIA

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Abstract: Investigations were carried out on hydrothermal resource exploitation in the Tbilisi hydrothermal reservoir, Georgia, using a multidisciplinary approach involving geophysical, geological, and hydrogeological techniques. High-resolution monitoring of temperature and pressure in several boreholes was carried out over a long period of time using an innovative geophysical tool. The system was numerically modelled, being calibrated against both the hydraulic and thermal monitoring data. It was found that the oil and thermal fields are connected hydraulically, that the thermal resource is of the order of 2-4 x 10^4 m^3/day, and that the hydraulic system responds to the stresses associated with seismic events.

Key words: water levels; temperature measurement; earthquakes; modelling; hydrothermal; geothermal; Tbilisi; Georgia; oilfields.

1. INTRODUCTION

Tbilisi, the capital of Georgia, is situated in the Caucasus seismoactive zone at the contact of tectonic plates. The most powerful earthquake in the Tbilisi area in recent years was one of magnitude 4.5 which occurred on April 25, 2004. This earthquake caused great economical damage and human losses.

Both confined and unconfined aquifers are present within the area. Hydraulic continuity is present between them, and this complicates the flow systems. However, thermal water discharging from the Middle Eocene sequence has a dominant role.

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Thermal mineral water or “sulphur springs” have been of particular importance for its population throughout the 1500 year history of Tbilisi. The water from these springs is hot (40-50 °C), and a little sulphurous due to the presence of hydrogen sulphide. It is used for therapy and recreational purposes. Hot natural springs issue from the exposed sediments of Middle Eocene in the River Mtkvari gorge. The water-bearing volcanic complex of Middle Eocene age is an important unit within the Tbilisi thermal field. Several boreholes, with depths of 2000-2500 m and located in the Lisi district in the northwest of the city, encountered sulphurous thermal water at 60-70 C^{o}. This water is used for room heating.

From west to east, these Middle Eocene volcanic deposits are buried under younger rocks. 20-30 km from the thermal water deposit, oil was found in an anticlinal structure. Intensive exploitation of this oil deposit caused a reduction in flow rate at the thermal springs during the 1980s. The hydrodynamical interactions are not understood in detail yet. In addition, it is possible that seismic activity, which was occurring at the time, may have also been involved with changes in the flow systems.

Without detailed research, the sustainable and ecologically correct use of this urban thermal resource is impossible. This paper describes progress towards this understanding. The main aims include the quantification of the resource, the evaluation of the interconnection between the thermal field and the Tbilisi oil field, and the establishment of the effects that seismic activity may have on the groundwater system.

2. THE NATURAL SETTING OF THE TBILISI HYDROTHERMAL FIELD

2.1 Geological and Hydrogeological Setting

From the geological point of view the region belongs to the Sartichala sub-zone of the Ajara-Trialeti folded system of the Lesser Caucasus. It is composed of sedimentary rocks of upper Cretaceous, Palaeogene, and Neogene age.

The volcanic Middle Eocene deposits are the most important part of the sequence from the point of view of the geothermal activity. They are represented by tuffs, sandstones, conglomerates, and breccias, and are characterized by intensive jointing; the joins are usually open, which promotes free circulation of groundwaters. The total thickness of the Middle Eocene deposits is 500 – 800 m. Waters are of bicarbonate/sulphide–calcium/sodium, bicarbonate/chloride-sodium-calcium-magnesium types. The total salinity of the waters fluctuates from

0.4 g/l to 0.1 g/l. The water has a temperature of 40-50 °C. Water-bearing volcanic formations of Middle Eocene are present across the whole area of the Tbilisi Hydrothermal Field (THF), in boreholes in the Lisi and central districts of the city, and also in the oil production boreholes in the Eastern part of the study area (Fig. 1).

Within the Tbilisi district the strike of the main geological structures is oriented nearly east-west. The sequence comprises Upper Cretaceous, Palaeogene, and Neogene units. To the north, south, and east, the region is bounded by major faults, and another fault that is still in dispute crosses the centre of the city, following the valley of the River Mtkvari. The magnitude 4.5 earthquake of 25th April 2002 is very probably generated by this latter fault.

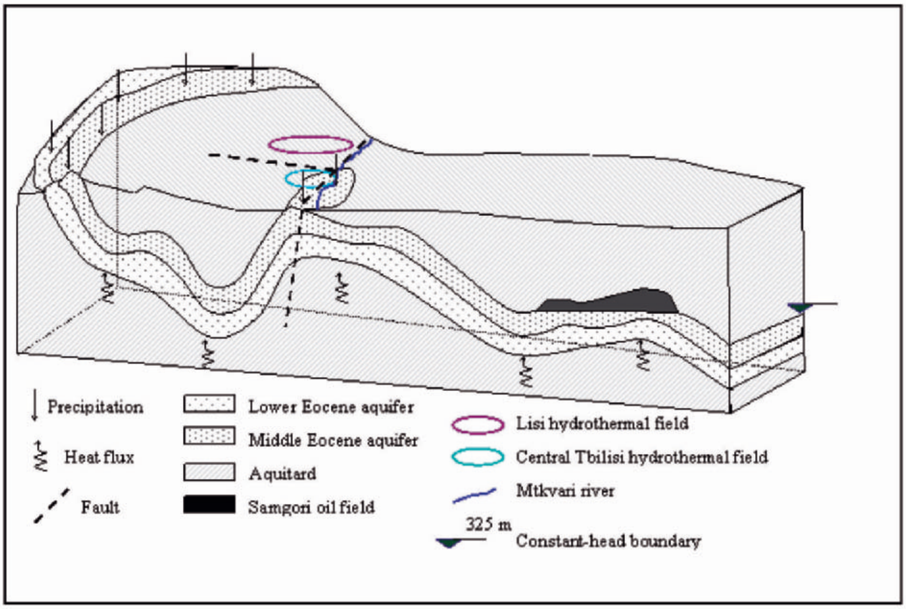

Figure 1. The geology of the Tbilisi area and the locations of boreholes.

2.2 The Geothermal Conditions

Tables 1 and 2 indicate the properties of the aquifers of the Middle Eocene sequence and their groundwaters as determined from investigations on boreholes whose location areas are shown in Figure 1.

Table 1. Characteristics of the Middle Eocene sequence aquifers and their groundwaters.

Borehole Number	Depth of Unit (m)	Yield (m^3 /day)	Temperature (°C)	Water Pressure (kg/m^2)	Salinity (g/l)
3-tech	3075-3286	86	33		0.44
4-tech	1050-2685	680	70	13.1	0.22
5-tech	1086-1878	3400	62	2.87	0.28
7-tech.	2118-3702	425	62	4.16	0.30
8-tech.	1740-2529	155	46	5.20	0.35
1-sab.	2140-2867	370	66	16.28	0.25

Table 2. Geothermal gradients.

	Geothermal Gradient (°C/100m)		
Age of sediments	Borehole No. 1, Lisi	Borehole No. 2 tech.	Borehole No. 3tech.
Oligocene	-	2.50	1.66
Upper Eocene	2.37	3.30	1.88
Middle Eocene	2.04	2.97	-
Lower Eocene	2.10	-	-
Average	2.17	3.86	1.71

Structural and hydrogeological data allow two main parts of THF to be singled out: the Saburtalo thermal water basin; and the Tbilisi Central thermal water basin. It should also be noted that the operations at the Sartichala-Teleti oil field affects the thermal water regime.

3. MONITORING SYSTEM

3.1 Equipment

In order to achieve the objectives of the present research, innovative equipment (Buntebarth, 1999) was tested and installed in boreholes in the THF. The devices are able to record very weak water flows in the subsurface either by high-precision temperature measurements or by combined temperature and water level monitoring. The equipment can operate in automatic mode with energy saving electronics, recording data for several weeks or months depending on the storage capacity and the reading frequency using three alkaline batteries of size D. The water level, pressure, and temperature data were collected at boreholes located in the Saburtalo (Lisi) and Tbilisi Central hydrothermal areas as well as in Sartichala-Teleti oil field.

Temperature and water level data were recorded in boreholes with an accuracy of 0.5 μK and 0.1cm respectively at a frequency of 10 or 20 minute intervals at the depths of order of 200 m (Buntebarth, 1999).

3.2 Observation Sites

The monitoring network comprised four boreholes. Three are in the thermal systems: Botanical Garden (No. 1); Lisi (No. 1T); Varketili (No. 46.). The fourth borehole, Lisi (No. 1), monitors the non-thermal groundwater system.

The natural regime of the thermal waters can be influenced by many factors: exogenous (precipitation, atmospheric pressure, tides) and endogenous (earthquakes, creep, tectonic strain impacts). Accordingly, a multidisciplinary database of the areas under investigation was compiled to provide all known data which are or which might be relevant to a contextual understanding of the thermal groundwater reservoirs. It contains topographic, geological, seismic/tectonic, hydrogeological, geophysical, borehole, meteorological and tidal data.

3.3 Hydraulic Tests

Knowledge of the hydraulic properties of well sites is necessary for a correct interpretation of monitoring data. These parameters were obtained by hydraulic testing of the wells. Slug-tests (i.e. recording of water level change with time in a well after injection (or withdrawal) of a known volume of water) were used to estimate the main hydraulic characteristics of aquifer: they were carried out between 1999 and 2002.

The interpretation of the field data was undertaken using the Cooper et al. (1967) and Hvorslev (1951) methods. The results are summarized in Table 3.

Table 3. Hydraulic properties as determined using slug-tests (L is test zone thickness).

Borehole	$T_{\text{Cooper et al}}$ (m^2/s)	$S_{\text{Cooper et al}}$	S_S (=S/L) (/m)	K_{Hvorslev} (m/s)	$K_{\text{Cooper et al}}$ (=T/L) (m/s)
Lisi No. 1	1.9×10^{-6}	1×10^{-6}	1.4×10^{-8}	2.4×10^{-8}	2.7×10^{-8}
Lisi No. 1T	1.4×10^{-5}	1×10^{-6}	5×10^{-8}	6.5×10^{-8}	7×10^{-8}
Varketily No. 46	20×10^{-3}	1×10^{-6}	1.5×10^{-8}	4.7×10^{-5}	2.5×10^{-7}

The table shows that hydraulic conductivity and transmissivity are relatively low in the Lisi area, but they are within the range of typical values for similar geological formations.

3.4 Numerical Modelling

One of the main purposes of the project was to quantify the available hydrothermal resources. Accordingly, a numerical model was constructed. Figure 2 shows the grid. A downward flux representing annual recharge with a mean temperature of 15ºC was imposed across the modelled area.

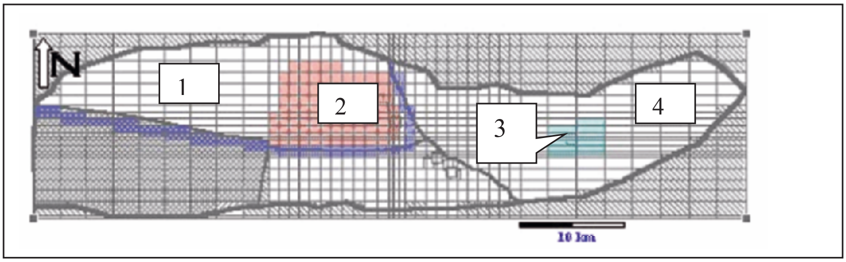

Figure 2. Regional numerical model grid with calibrated hydraulic conductivity distribution. 1- $K_x = K_y = K_z = 0.72$ m/d. 2- $K_x = K_y = 0.11$ m/d, $K_z = K_x/10$. 3- $K_x = K_y = K_z = 0.021$ m/d. 4- $K_x = K_y = K_z = 4x10^{-4}$ m/d. Shaded cells are inactive.

Calibration for the transient state was accomplished using the data collected when the influence of the exploitation of the oil fields was becoming a major concern for the exploitation of the Central thermal field. The information dates from 1974 to 1990, and clearly shows that the increase in volume of pumped oil caused considerable drawdown and consequent loss of yield in borehole 1BG (Botanical Garden), one of the main sources in the Central Thermal field (Figure 3).

Calibration of the model showed that the best fit was obtained with an infiltration equivalent to 5 mm/year. This represents resources of $2.27x10^4$ m^3/day (or 260 l/s) for a recharge area of 166 km^2. In some simulations a reasonable calibration is obtained using a value of infiltration of 10 mm/year in the northern boundary of the model. In this case the resources available are 3.79 x 10^4 m^3/day: this model provides the best fit with field data.

Table 4 shows the values found for the mean hydraulic and thermal parameters for the two hydrothermal fields. Notice that the hydraulic conductivity, and therefore the transmissivity, was the only parameter that must be ascribed different values in each thermal field. The model considered a constant thickness of 600 m for the Middle Eocene.

In the case of the storage coefficient, the model performs its computation using the values of porosity, rock and water compressibility, and fluid density. This last parameter varies with temperature and density, and therefore each cell of the model may have a different value of storage coefficient. The storage coefficient is only relevant for transient state behaviour, while in terms of resources management, the steady-state response to pumping is more interesting. The values presented in Table 4 are the ones that provided the best-fit calibration.

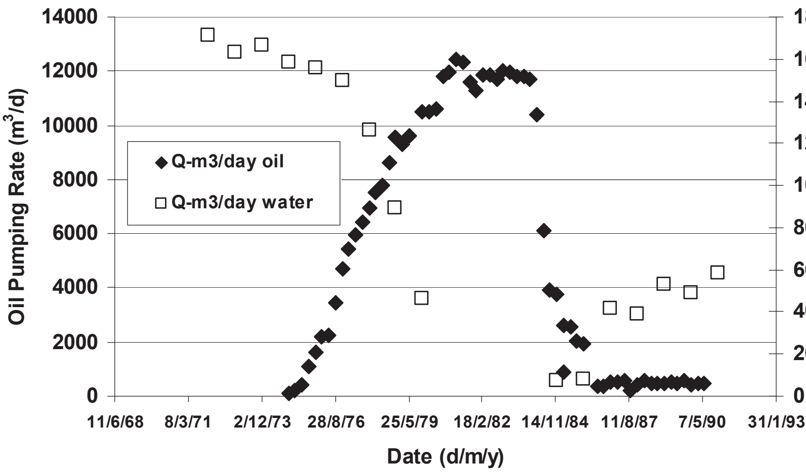

Figure 3. Influence of oil field pumping on drawdown, and hence yield, of the Central thermal water field.

The calibrated model can be used as a predictive tool to define strategies for aquifer exploitation, giving indications of which areas are more interesting to explore when considering new boreholes, determining what the influence of oil field exploitation may be, or identifying what areas may be more important to protect.

Table 4. Parameters adopted for the whole modelling domain (Middle Eocene).

Parameter	Value	Units
Porosity	0.012	-
Rock compressibility	7.5×10^{-11}	Pa^{-1}
Heat capacity	2.24×10^{-6}	J/kg K
Hydraulic conductivity	See Fig. 2	
Thermal conductivity	3	W/m K

4. EFFECTS OF SEISMIC ACTIVITY

Periodic observations of temperature and water levels were made to investigate seismically-induced effects. The data collected also allow the problem of the interconnection of three main thermal fields to be investigated.

In general, some weeks before the earthquakes, water level rises and the amplitude of tidal variations increases (Figure 4). Earthquakes occur either at the rising phase or during the falling phase of the water level fluctuations (Gavrilenko et al., 2000; Matcharashvili, 2001). We interpret the rise of water level to be due to the compressional strain: the Tbilisi area is in a zone of continental collision and focal mechanisms are mainly showing thrust faulting. After the earthquake, or just before it, the stresses (at least in the area surrounding the plane of the impending faulting) are released and the connections between pores increase; both these phenomena should promote water level drop. Due to the heterogeneity of the stresses, water level variation patterns are different in each well (Melikadze, 2000; Melikadze & Ghlonti, 2000; Buntebarth & Melikadze 2002).

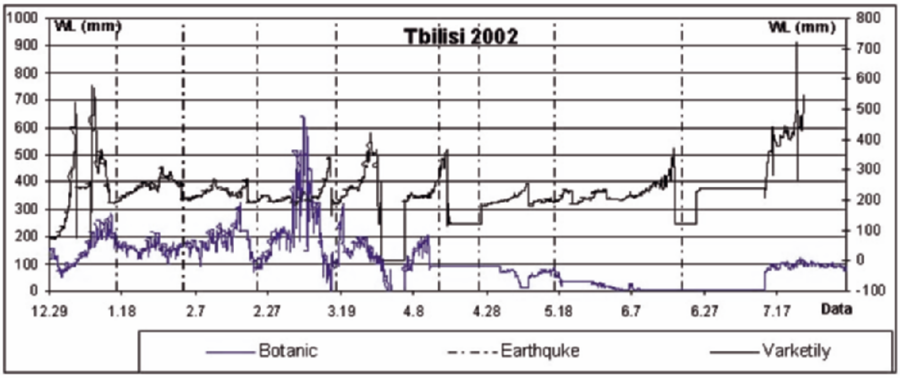

Figure 4. The relationship between borehole water level variations and seismic events (x-axis indicates month and day in 2002).

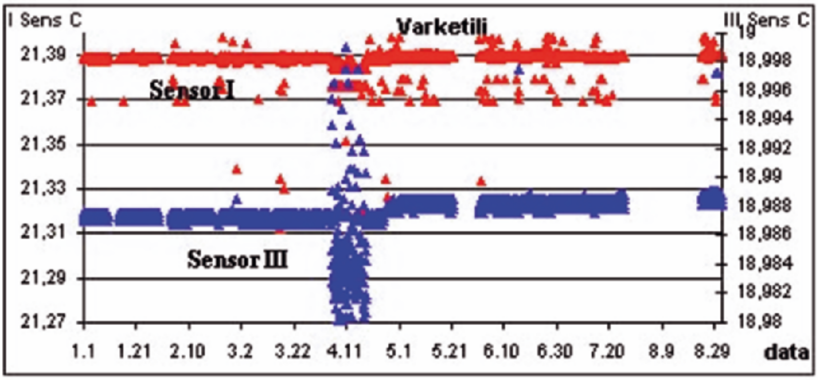

Figure 5. Temperature change in the Varketili borehole during the April Tbilisi earthquake of 25.04.02 (x-axis indicates month and day).

Several days before the large event in April 2002, anomalies were observed at the Lisi and Botanical Garden wells: namely, the rise of average water level and the increase of amplitude of water level variations of tidal origin. After the earthquake, water level fell, which is characteristic for a strain-relaxation process (Figure 4). Also, temperature variations were seen at the Varketili borehole (Figure 5).

5. CONCLUSION

Numerical modelling of the Tbilisi hydrothermal reservoir achieved an acceptable calibration with temperature and pressure data. It confirms the hydraulic connection of the thermal water and oil production areas. This means that uncontrolled exploitation of the oil deposit will strongly decrease the fluid pressure in the thermal wells. Earthquakes also affect the system, and hydrodynamical precursors of earthquakes have been recorded in all monitored wells.

The results obtained demonstrate the considerable potential of the monitoring network. It not only allows the recognition of earthquake-preceding phenomena, but also allows monitoring of environmental conditions in the Tbilisi region.

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ESTIMATING THE BASIC MATERIAL AND TECHNICAL RESOURCE NEEDS FOR THE OPERATION OF WELL DRAINAGE SYSTEMS IN URBAN AREAS WITH HIGH WATER TABLES

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Abstract: In a number of cities in Uzbekistan, infiltration from irrigation and surface drainage structures, leakage from water supply pipelines and sewers, and discharges of waste water all contribute to rising groundwater levels. To prevent damage to buildings, well fields have been installed to act as vertical drainage systems. These vertical drainage systems need to be managed properly if they are to achieve their purpose. This paper presents a calculation scheme for estimating the requirements for basic materials and other resources for urban areas where drainage by well systems is necessary. The scheme takes into account many different types of pumping failure, and is supported by empirical relationships developed from a large data set collected across the country. The principles of the approach could be adapted for other management issues concerned with urban water development.

Key words: Uzbekistan; drainage; management; modelling; infrastructure; prediction of performance; engineering issues; rising groundwater levels.

1. INTRODUCTION

Vertical drainage is an efficient but technically complex method of drainage that is associated with high costs, high level of maintenance, and careful implementation. Large vertical drainage systems (VDS) consisting of 200 to 800 wells are constructed and operated in some provinces of

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Uzbekistan. The total number of operating wells in the beginning of 2001 was more than 10,000 including 3,702 wells for vertical drainage, 4,621 wells for irrigation (data from the Ministry of Agriculture and Water Economy), and 1,886 wells in cities, towns and settlements (data from the Department of Communal Drainage). The vertical drainage operation in these area has resulted in the development of sustainable reclamation processes, i.e. soil desalinization and the maintenance of groundwater at optimal level with minimum costs. In urban areas, vertical drainage is used generally for the prevention of basement flooding of buildings though sometimes the water is used for supply. The wells extract water from the second or third layer of the aquifer beneath cities and developments. The management of the groundwater level (GWL) is achieved due to the head difference between the shallower and deeper aquifers. In Uzbekistan, vertical drainage is used in 205 cities, towns, and settlements, extracting more than 239 million m^3 of water per year (Table 1) (Saliev and Saliev, 2003).

Table 1. Operation characteristics of VDS in populated areas (PA) of Uzbekistan.

Province	No. of PA	No. of wells	No. of operating wells	Volume of pumped water (million m^3 /y)			Drained area (ha)
				Plan	Act.	%	
Republic of Karakalpakstan	24	149	91	11	11.0	100	2,585
Andijan P.	18	90	54	11	14.8	134	1,100
Bukhara P.	21	181	126	24	24	100	4,190
Djizzak P.	10	131	71	14	15	101	1,520
Kashkadarya P.	14	155	115	36	37	101	6,028
Navoi P.	1	56	36	4	4.3	102	685
Namangan P.	16	130	94	17	18	103	4,500
Samarkand P.	23	82	49	7	8	101	1,312
Sirdarya P.	7	148	84	32	32.7		4,761
Surkhandarya P.	11	36	19	2	3	133	658
Tashkent P.	34	215	164	13	13.4	101	2,022
Farghana P.	18	242	171	38	40.9	107	5,858
Khorezm P.	8	271	195	17	17.5	103	3,234
TOTAL	205	1,886	1,269	224	240	104	38,453

A lower level of operation occurring in the past few years in the majority of vertical drainage wells has led to a decrease in the efficiency of many systems. It is therefore necessary to change the management systems for the well fields, and to do this, methods are needed to assess what resources are required to achieve a given standard of efficiency. Accordingly, this paper presents a calculation scheme for assessing what basic materials and other infrastructure are required for maintaining vertical drainage systems at prescribed performance criteria.

2. PERFORMANCE CRITERIA

Three performance criteria will be defined here: (i) the average coefficient of readiness; (ii) the system efficiency; and (iii) the coefficient of system utilization.

The average coefficient of readiness (C_r) is the probability that a well will be operable on a randomly chosen basis. It is calculated by:

$$C_r = \frac{\sum t_p}{\sum t_p + \sum \tau} \tag{1}$$

where Σt_p is the duration of well operation over a certain period, and $\Sigma\tau$ is the duration of stoppages due to technical problems for the same period (stoppages due to power switch-outs and stoppage requests are not taken into account).

The system efficiency (SE) is calculated by:

$$SE = \frac{\sum_{i=1}^{n} \cdot \sum_{j=0}^{T} t_{i \cdot j}}{n \cdot T_k} \tag{2}$$

where t_{ij} is the actual duration of operation of i^{th} well during the j^{th} time interval, and T_k is the sum of all the time intervals (i.e. the total time period over which efficiency is being assessed).

The required duration of VDS operation is determined on the basis of calculations of the optimum GWL. So SE (taking into account well discharge and the number of wells) indirectly may characterize the draining capacity of a considered area:

$$SE = \frac{D_b \cdot F_m}{\sum_{i=1}^{n} q_i \cdot T_k} \tag{3}$$

where q_i is the i^{th} -well discharge, F_m is the area, and D_b is the total pumped water volume.

The VDS development designs envisage SE to be no less than 0.75-0.80, whereas the "Manual for designing of operational regimes in vertical drainage systems for conditions of Central Asia" (Anon, 1977) envisages SE values to be no less than 0.85. However, the actual average SE for many years even in the best VDS of Uzbekistan does not exceed 0.72.

If the drainage operation level is evaluated by the coefficient of system utilization (CSU), i.e. the ratio of actual pumped volume (D_a) to the planned pumped volume (D_{pl}) calculated in accordance with the well's operational regime:

$$CSU = \frac{D_a}{D_{pl}} \frac{\sum_{i=1}^{n}\sum_{j=0}^{T} t_{ij} \sum_{i=1}^{n} q_{if}}{nT_{пл}\sum_{i=1}^{n} q_{inn}} \tag{4}$$

it turns out that at present the vertical drainage in many towns does not provide the designed drainage capacity necessary for maintenance of the optimal GWL.

3. SYSTEM PERFORMANCE IN UZBEKISTAN

Projects envisage CSU equal to 0.85-0.90, but in practice it does not exceed 0.2-0.3. This fact can be explained not only by low SE but also by a gradual decrease of well discharges due to filter clogging, metal pipe corrosion, and sedimentation. The low level of VDS operation in Uzbekistan is caused by interruptions to pumping due to stoppages caused by frequent breakdowns of pumps and high power costs; these stoppages make up 28 to 60 % of total time. Analysis of the structure of VDS stoppages shows that most of the longer stoppages are those due to pumps failure (Table 2).

The pumping stoppages are caused mainly by breakdowns of pumps in gravel-pebble systems due to sand pumping, the aquifer here also containing fine- and medium- grained sand (Figure 1).

The analysis of the VDS operations shows that it is practically impossible to obtain SE equal to or greater than 0.85 in large VDS due to the following causes:

- poor technical design and quality of well construction, inappropriate selection of gravel filters leading to abundant sand pumping and frequent failure of pumps and power supplies;

- insufficient and untimely equipment repairs, which are the cause of delayed rehabilitation of pumps and power supplies;
- insufficient supply of spare pumps and power units, other spare parts, transport, and repair equipment.

Table 2. The reasons for pump stoppages in VDS wells in three provinces of Uzbekistan in 1990. Note: present day indices are lower.

Cause of stoppage	Percentage of stoppages in listed province		
	Sirdarya	Farghana	Bukhara
Repair of draining networks	6-10	5-18	14-23
Farm requests	12.5-18.0	25-86	3.8
Lack of power	22-36	2-23	23.9-41.8
Pump / power faults	55-56	2-31	23.0-53.6
Others	5-10	5-11	9.5

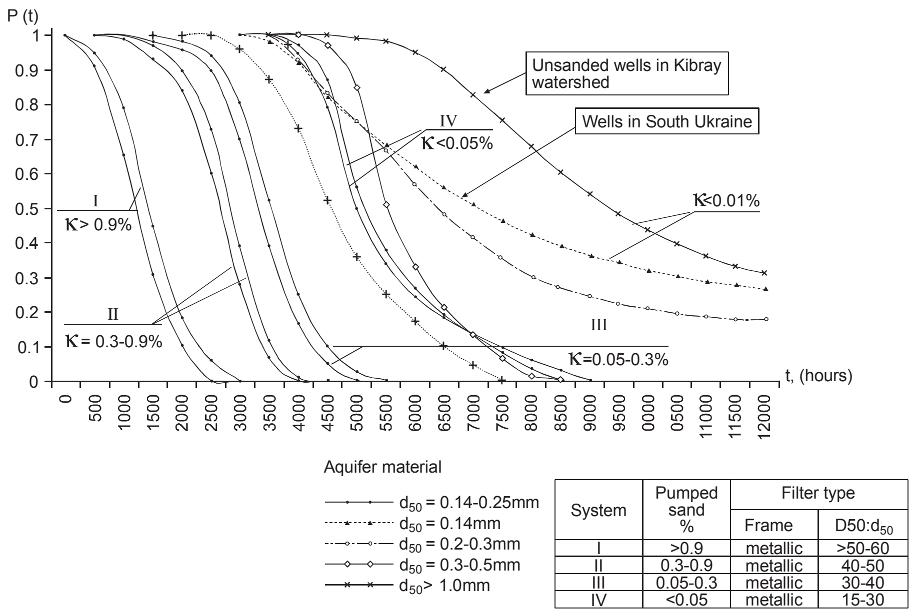

Figure 1. Failure of electric pumps.

The poor technical efficiency of VDS operations leads to a sharp rise of GWL in populated areas. This fact as well as the relatively low SE levels in other areas indicates that it is necessary to raise the efficiency of VDS operations by satisfying the demands of managing organizations for necessary material and technical resources.

At present, both designs and performance measures in operating VDS revolve around the availability of material and technical expertise. In part, the provision of amortization allocations is for low discharge water wells, though these wells by their purpose, operational conditions, and parameters differ considerably from VDS wells.

Meanwhile, the planning of material and technical resources must take account of the planned complete pumping regime of VDS wells over the full year. SE is the main index characterizing the complete pumping regime of VDS. On the one hand, it reflects the technical condition of the system (reliability of wells and their elements, provision of material and technical resources, spare parts, repair technician teams, power, etc.). On the other hand, it characterizes the groundwater drainage capacity which, in its turn, is closely associated with formation of the optimal GWL.

The VDS well pumping regimes proceed from:

- provision of the optimal water and salt regime of soils in the unsaturated zone over a year;
- establishment (as far as possible) of steady intensity pumping (i.e. a more even schedule of SE) over a year; and
- achieving of a higher SE both on a monthly and yearly basis which in turn will permit a reduction in the number of wells.

A constant SE over a year facilitates:

- the determination of the number of wells necessary for the operation of the system;
- the setting up of goals for current operations and capital expenditure for the repair of wells and other facilities;
- the planning of the provision of material and technical resources; and
- control over the operational processes and the introduction of progressive bonus systems for wages.

However, stochastic faults inherent in the constructed elements of the VDS result in forced stoppages of the VDS. The provision of material and technical resources has to be planned taking into account the prompt elimination of the causes of these stoppages. It is necessary to test the potential of the complete pumping regime with the available material and technical resources by comparing the actual SE with the SE value calculated on the basis of values of the system reliability and repair capacity.

4. ESTIMATING SYSTEM EFFICIENCY

For evaluation of the SE, which can be achieved in a considered VDS with the available material and technical resources (new equipment acquired at the expense of amortization allocations, repaired units, repair teams, lifting and transport means) and actual SE as well as selection of means to increase it, we propose to use the following expression taking into account the reliability indices:

$$SE = \frac{1}{1 + \lambda_p \tau_p + \lambda_{os} \tau_{os} + \lambda_{dn} \tau_{dn} + k_1 + k_2 + k_3} \tag{5}$$

where λ_p, λ_{os}, and λ_{dn} are the frequencies of failure ($[T^{-1}]$) of the electric pumps, the control stations, and the drainage network, respectively; τ_p τ_{os}, and τ_{dn} are average times required to mend the pumps, control stations, and drainage network, respectively; and k is the fractional stoppage time due to switch-outs (k_1), other technical causes (k_2), or requests / changes in pumping regime (k_3).

Using data collected by SANIIRI, the values of the k_1, k_2, and k_3 coefficients are 0.09, 0.03, and 0.05, respectively, in Sirdarya province; 0.18, 0.09, and 0.67, respectively, in Farghana province; and 0.53, 0.15, and 0.06, respectively, in Bukhara province (Yakubov et al., 1987).

Using data collected over a long period by SANIIRI scientists covering a variety of conditions in Central Asia and South Kazakhstan, we have derived some empirical relationships for evaluating the intensity of failures of well elements and the time needed to repair them. The intensity of electric pump faults (γ_p) is defined as follows. Based on the available data on grain size distributions for gravel pack and aquifer materials, an 'interlayer coefficient' is calculated from:

$$IC = \frac{D_{50}}{d_{50}} \tag{6}$$

where D_{50} and d_{50} are the median diameters of particles in the pack and aquifer materials, respectively. If there are no data on the composition of the gravel pack materials, analysis of a selection of samples from 1.5-2.0 m depth needs to be carried out at wells where subsidence has not occurred. Using the IC, the sand concentration (SC) in pumped water and the frequencies of the electric pump failures (λ_p) are evaluated using the relationships presented in Figures 2 and 3. The relationship given in Figure 3 is derived from the results obtained in a selection of wells where 15% had new pumps and 85% had pumps which had undergone capital repairs.

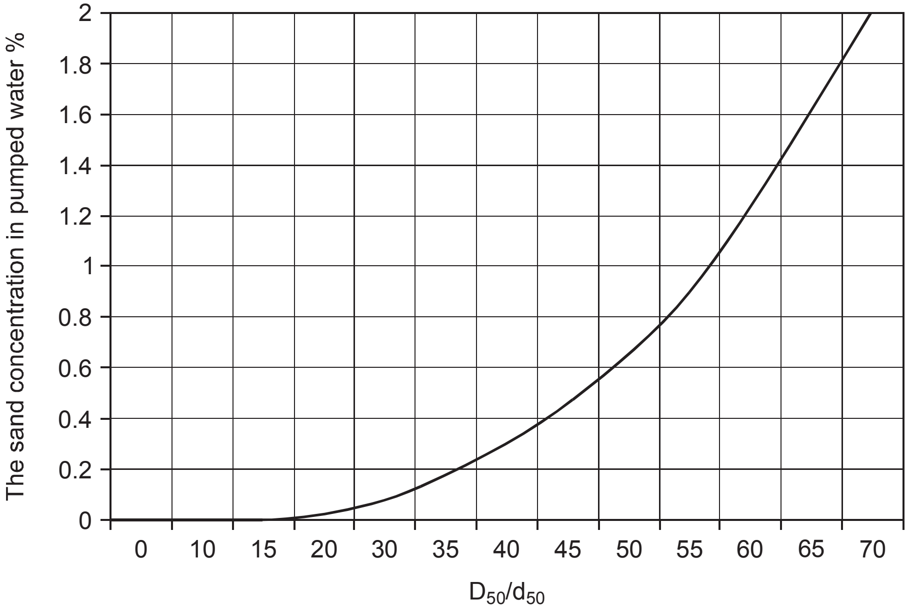

Figure 2. The sand concentration in pumped water (SC) as a function of interlayer coefficient (IC).

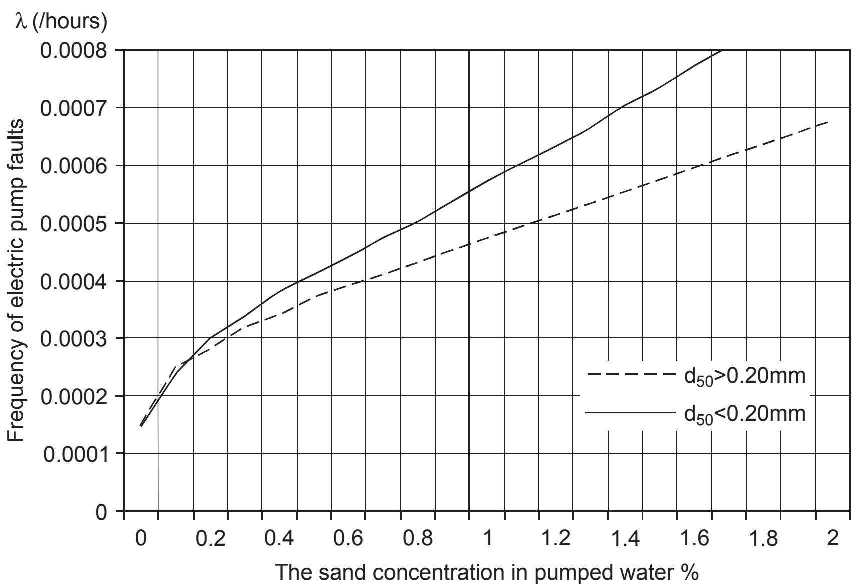

Figure 3. Frequency of electric pump failures (λ_p) as a function of sand concentration in pumped water (SC).

The frequency of electric pump failure has to be determined individually depending on composition of the gravel pack and the aquifer materials. For VDS, the weighted average value is given by:

$$\lambda_p = \frac{\lambda_н^1 n_1 + \lambda_н^2 n_2 + ... + \lambda_н^i n_i}{n_i} \tag{7}$$

where n_1, n_2, ... n_i are the numbers of wells with the same interlayer coefficients.

Equation (5) requires the evaluation of τ_p which is the average time required to mend the pump in a well. Generalizing to any well component k, τ_k is the average time required to repair this component, and can be estimated from:

$$\tau_k = \frac{a_k^u \tau_k^u + a_k^p \tau_k^p + a_k^D t_{kj}^k}{a_k^u + a_k^p + a_k^D} \tag{8}$$

where a_k^u is the number of stoppages dealt with by replacing faulty parts; a_k^p is the number of stoppages dealt with by repairing faulty parts; and a_k^D is the number of stoppages which are not dealt with within a certain time interval t_{kj} due to the absence of a working 'element' (insufficient repair capacity, absence of parts, inadequate maintenance of wells, etc.). τ_k is the time spent on dealing with stoppages using either spare parts (τ^u) or repairs (τ^p):

$$\tau_k^u = \tau_f + \tau_o + \tau_d + \tau_i \tag{9}$$

$$\tau_k^p = \tau_f + \tau_o + \tau_d + \tau_j + \tau_i \tag{10}$$

where τ_f, τ_o, τ_d, τ_j, and τ_i are the times spent for: finding the problem; preparation of an order for the repair work; dismantling the faulty system; completing repairs; and re-installation.

The occurrence of well stoppages due to repairs and lack of failed elements must be taken into account when the actual SE is analyzed for elimination of faults. The demand for material and technical resources and capacity of the repairs base in VDS operation projects must be defined so that the faults of elements in any time interval can be eliminated at the expense of available spare parts.

The frequency of control station failures according to data collected by SANIIRI is 0.00016 per hour (Nasirov, 1976).

The time for repair of a control station failure (τ_{os}) is determined in a similar way as for repair of an electric pump (i.e. $\tau_k = \tau_{os}$ instead of τ_p in equations (8) to (10)) except that the time spent for unit installation and dismantling is set at 30 minutes.

Skrilnikov (1983) proposed a method for evaluation of frequency of well failure in a case when the drainage network consists of parabolic flumes (typical type for the region). The general calculation scheme is also acceptable for other constructed drainage networks.

The transport capacity of stormwater is defined by the formula:

$$\frac{\rho}{\gamma_w} = \left(\frac{2JF_r^{0.3} - 0.000414}{3.5} \right)^{1.147} \tag{11}$$

where ρ is turbidity [M/L^3]; γ_w is the density of water [M/L^3]; J is canal water level; and F_r is the Froude number. The volume of sedimentation in a flume network depends on the number of turn-ons and the term of operation after which flumes must be cleared of sedimentations is calculated. Suppose that sedimentation is occurring in a flume L metres long (assuming the form of a pyramid with a volume 1/3 wL). Then sedimentation volume (m^3) G per turn-on is:

$$G = \frac{Q}{\gamma_s} \sum_0^t t_i (\rho_i - \rho_o) \tag{12}$$

where: Q is the flow rate [L^3/T]; γ_s is the density of the sedimented volume, [M/L^3]; ρ_i is the flood turbidity [M/L^3] in the time interval t_i; and ρ_o is the initial flood turbidity [M/L^3]. The depth of sedimentation (H_s) in the first part of flume is calculated as follows:

$$H_s = \left(\frac{9GN}{4L\sqrt{2g}} \right)^{1/1.5} \tag{13}$$

where: G is in m^3; N is the number of well turn ons for a certain period; g is the acceleration due to gravity.

The number of turn-ons until the critical sedimentation depth h is reached is therefore given by:

$$N_{cr} = \frac{4L\sqrt{2pH}^{1.5}}{9G} \tag{14}$$

The intensity of the drainage network failure is defined as follows:

$$\lambda_{dn} = \frac{N_{cr}}{N_y} \tag{15}$$

where N_y is the number of wells starts per year. According to data collected by SANIIRI, N_y varies from 40 to 400.

The well stoppages due to faults in the drainage network (τ_{dn}) are caused by its silting by sandy-loamy sediments contained in the pumped water. The value of τ_{dn} depends on the quantity of alluvium, the hydraulic elements of the drainage network (ferro-concrete flumes, closed pipelines, or open canals), and the transport capacity of the network.

The failure repair time is calculated by:

$$\tau_{dn} = \tau_f + \tau_j + \tau_{fc} \tag{16}$$

τ_{fc} is the time necessary for flume clearing:

$$\tau_{fc} = \frac{GN_{cr} * R_h}{n_s} \tag{17}$$

where R_h is the time needed for extraction of 1 m^3 of the soil by a team of workers (e.g. for topsoil to 1 m depth, R_h = 0.85 hours); and n_s is the number of teams involved.

5. ESTIMATING THE REQUIRED RESOURCES FOR A GIVEN SYSTEM EFFICIENCY

As mentioned above, the VDS designs have to include a calculated estimate of the provision of material and technical resources needed for the SE value chosen. In particular it is necessary to determine: (i) the need for working reserve units; (ii) the number of maintenance teams required for installation and dismantling of the well equipment, clearing and repairs of the drainage network, hydraulic structure maintenance etc.; and (iii) the types and amounts of equipment needed. These material and technical resources are known as the exchange reserve. The exchange reserve (Q_{er}) can be split into two parts: (i) the resources needed to repair the faults expected to occur during the period of interest (Q_k); and (ii) the insurance reserve (I_k) required to even-out the supply of working equipment, making sure that components are always available for undertaking repairs/replacements:

$$Q_{er} = Q_{\kappa} + I_{\kappa} \tag{18}$$

The number of working units necessary (i.e. groups of items required for undertaking the repair) (U_k) is determined by:

$$a_k = U_k = \lambda_k T_k * SE * n \tag{19}$$

The number of working units necessary will be supplied by the new units obtained at the expense of amortization allocations (n_{kn}) and repaired units (n_{kr}):

$$U_k = n_{kn} + n_{kr} \tag{20}$$

$$n_{kn} = k_s n \tag{21}$$

where k_s is an empirical coefficient.

Comparing the number of necessary working units for a given period with the number of new ones obtained at the expense of amortization allocations, we determine the number of faults which must be rectified by undertaking repairs, i.e. the required repair capacity:

$$C_k = U_{\kappa} - n_{kn} \tag{22}$$

The insurance reserve I_r is calculated as follows. The starting data for calculations are: (i) the number of wells in the VDS; (ii) the SE allocation in the course of the year according to the plan for the VDS pumping regime; (iii) the frequency of occurrence of the faults (λ_k); (iv) the delivery of new elements at the expense of amortization allocations (n_{kn}); and (v) the annual capacity to undertake repairs (C_k). The delivery of new and repaired elements is accepted to be evenly distributed throughout the course of year or is taken from the operational practice for the element concerned. The expected monthly number of element faults is determined by equation (9) taking into account the monthly SE in accordance with the planned VDS pumping regime.

By comparing the expected number of failures during each month with the number of new and repaired working elements delivered during the same period, we can calculate the necessary insurance reserve for every month:

$$I_{kj} = (n_{knj} + n_{krj}) - a_{kj} \tag{23}$$

where, when $I_{kj} \geq 0$, $I_{kj}=0$.

After determination of the insurance reserve, the required repair capacity (C_k) and the number of new units delivered at the expense of the amortization allocations (n'_{kn}) can be calculated:

$$C_k = U_k + I_k - n'_{kn} \tag{24}$$

where $n'_{kn} = (n + I_k)C_s$

The number of maintenance teams necessary is calculated as follows. The annual amount of work, W_k, needed for repairing breakdowns is calculated by:

$$W_k = \lambda_k * SE * T_\kappa * n * S_t \tag{25}$$

where S_t is the standard time for installation and dismantling of the well element concerned [man-hours].

The yearly working hours per man for one-shift working days and six-day working weeks is calculated using the equation:

$$t_w^y = 8.2T_w \tag{26}$$

where T_w is the number of working days in a year taking into account vacation, temporary inability to work, etc.(= 4.4 % of T_k). Thus, the number of workers N_{wk} needed for installation and dismantling is calculated by:

$$N_{wk} = \frac{W_k}{t_w^y} \tag{27}$$

and the number of maintenance teams (M_t) is given by:

$$M_t = \frac{N_{wk}}{n_{mt}} \tag{28}$$

where n_{mt} is the number of workers in a team.

6. CONCLUSION

The method presented here for the evaluation and provision of materials and technical resources at the assigned SE level may be used also in other regions with vertical drainage systems. The approach could also be adapted for a range of other issues within the context of urban aquifer management.

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HYDROGEOLOGICAL CONDITIONS IN URBAN AREAS IN THE GEORGIAN BLACK SEA COASTAL ZONE

Case Studies of the Towns of Poti and Batumi

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Abstract: This study describes the hydrogeological conditions of the coastal urban areas of Poti and Batumi, Georgia. A clear hydrogeological zonation can be mapped out. The hydrological influence of the rivers Rioni and Chorokhi on the environment of Poti and Batumi is evaluated. Infilling of the coastal zone with inert material has proven to be, and will remain, the most effective method of rehabilitation of the areas where coastal erosion is occurring.

Key words: coastal aquifers; coastal erosion; Poti, Georgia; Batumi, Georgia; Rioni River, Georgia; Chorokhi River, Georgia; fill material.

1. INTRODUCTION

The aim of this paper is to present and discuss hydrogeological conditions in the urban areas of the Georgian Black Sea coastal zone. One issue concerns coastal erosion. The urgency of these problems is undoubted, as the Georgian section of the "Asia-Europe" transport corridor has already begun to function. This corridor includes, for example, railway ferries from Batumi and Poti, the Supsa oil terminal, the Baku-Jeikhan oil pipeline, the Middle Asian gas pipeline, and the substantial expansion of Black Sea ports. It should also be recognized that the whole Black Sea coastline of the region is a recreational area. All these factors bring about a dramatic increase of anthropogenic pressure upon the natural habitat. In this paper, case studies of the towns of Poti and Batumi are presented.

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2. HYDROGEOLOGICAL AND GEO-ECOLOGICAL CONDITIONS

The hydrogeology and hydrology fundamentally influence the geoenvironment of urban areas. Groundwaters may change the composition and properties of rocks that may in turn promote the activation of modern geological process.

Within the Kolkhida plain, the upper aquifer is developed in marine, alluvial, and marsh sediments (Figure 1). Among the water-bearing rocks, the main lithologies are loams and clays interbedded with beds of sand, sandy-loams and peat. Aquifers are fed mainly by atmospheric precipitation (mean annual precipitation in Poti is 1660 mm) and are classified according to the predominant regime-forming factors:

- Climatic: e.g. that of the mountainous area of west Georgia, and that of the upper part of the piedmont plain;
- Hydrological: e.g. coastal areas and river flood plains.

In order to reveal the character of interaction between the sea and groundwaters, observations were carried out in the delta of the River Inguri (v. Pichora) and the swampy lowlands of the River Rioni (v. Grigoleti). The groundwater was studied at distances of 50, 200, 550, 1000 and 1500 m from the sea, with each study site consisting of 3 boreholes of 5, 10 and 20 m depth.

Groundwater discharge into the sea occurs freely when the water level is high. With low water levels the discharge is reduced, particularly over the first 200 m of the shoreline. According to the data collected, groundwater discharges into the sea are also hindered by storms and groundwater flows into the marshes.

Analysis of the groundwater flows demonstrates that with high water levels the groundwater discharges freely at 5 m depth while at 10 and particularly 20 m depth the discharge becomes restricted (Figure 2). The seawater feeds the aquifer over the first 400 m from the shore when the groundwater level is low.

Recent marine and alluvial sediments are common in the coastal area, the width of the latter varying from few hundreds of metres to several kilometres the widest part reaching the River Rioni mouth. Groundwaters are present in sand and gravel at 1 to 5 m depth. Chemical analysis of the fresh-water lens revealed its calcium-potassium bicarbonate and potassium-calcium bicarbonate composition. Total salinity typically ranges from 0.3 to 0.5 g/l. However, in a dry period the salinity may increase to 3 g/l with corresponding changes in the water chemical composition (increased chloride, potassium, and magnesium).

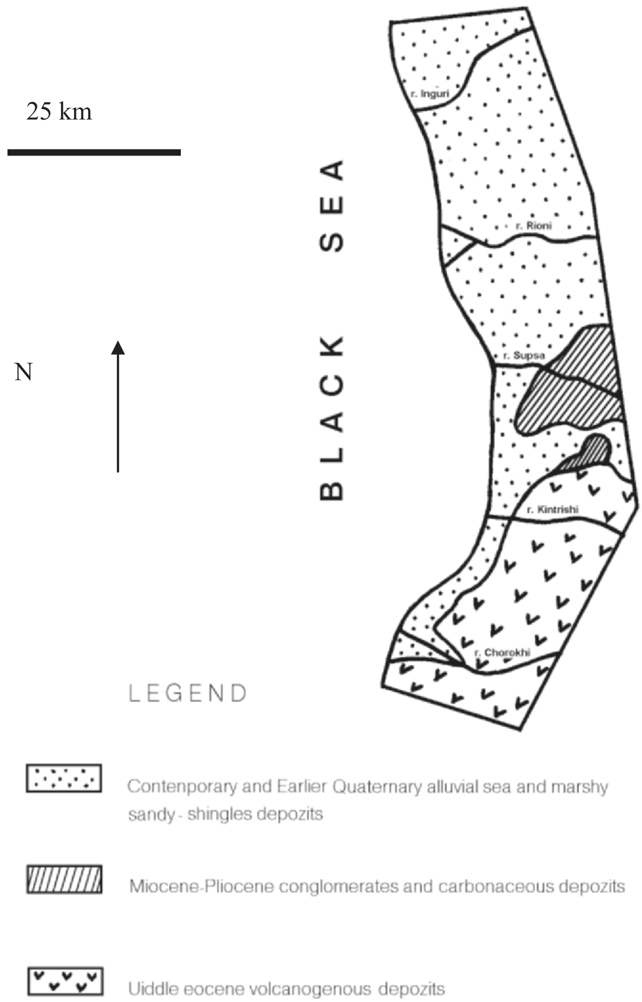

Figure 1. Geological distribution.

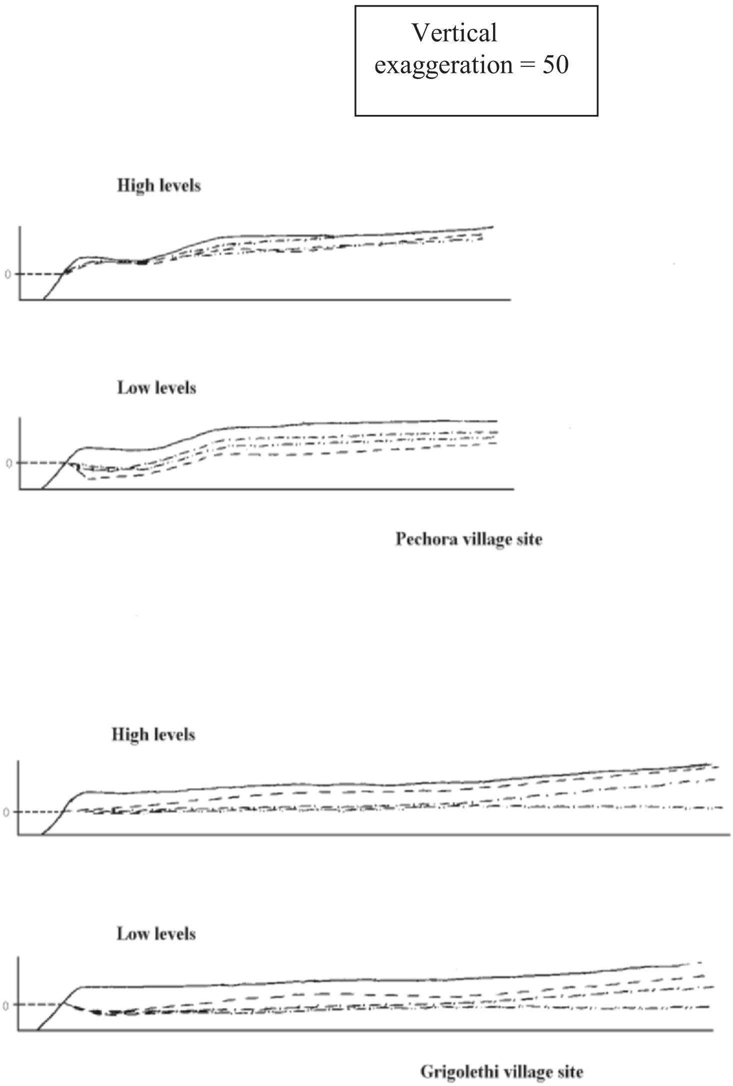

Figure 2. Relationships between sea level and (fluctuating) groundwater levels.

Groundwaters in the Recent marine and alluvial sediments are easily contaminated. Nevertheless, being the only source of water supply in the area, they are frequently used by the local population.

Groundwater circulates freely in the sand and sandy loam. Peat, loam and clay are, however, less permeable thus ensuring that high heads typically occur in the alluvial-marine and marsh sediments. The permeability of the argillaceous sand and sandy loam partings and lenses ranges from 0.1 to 1.0 m/d, reducing to 0.05 – 0.1 m/d in the marsh loam, silt and sandy clay.

The aquifer is fed by atmospheric precipitation, river filtration losses, groundwater flows from adjacent areas, and up-flow from deeper confined aquifers. Groundwater losses occur through evaporation from marsh surfaces and groundwater discharge.

Despite the drainage of near-surface groundwater (interflow) and run-off through the thoroughly investigated network of open canals in the area, heavy argillaceous areas (clay, loam) preserve high moisture contents for long time periods owing to capillary forces and capillary rise (Mikadze *et al.*, 2003).

The main river arteries of the Kolkhida lowland dominate the relief and strongly influence the water level regime in the coastal area. In the area of v. Mukhuri, boreholes were drilled at distances of 0.8 km and 3.3 km from the River Rioni. In the first case, groundwater levels were in direct continuity with river level fluctuations. The annual amplitude of these fluctuations is 0.3 m. In the second case the influence of the river declines, the annual amplitude reducing to 0.25 m.

Rivers have other influences additional to groundwater. In the district of Poti, the River Rioni is the chief water artery. Diversion of the river mouth to north of the port of Poti in 1939 caused erosion of the south shores of the port. Later a strip, nearly 800 m wide, of the town area was washed away (Iashvili and Iashvili, 2003).

In Batumi area jointing rocks and half rocks of the Middle Eocene tuffaceous series descend to the Black Sea brink (Figure 1). They possess high filtration and discharge properties.

The water in the tuffaceous series is of a calcium-bicarbonate type. Thaw and rainwater freely penetrate into the jointing zone of the mountain rocks.

Aquifers may hold significant resources of fresh water and noticeably influence the geoenvironmental / geo-ecological situation in the area. This water has a high physical and chemical quality and is an important source of industrial water supply in the district. Recent river and marine alluvial sediments frequently are permeable, with well discharge rates of 1-5 l/s, increasing to 20 l/s in the coastal strip. The depth to groundwater table in the alluvium varies from 1.5 to 3.0 m (Khachapuridze, 1990).

Of other factors affecting the geoenvironmental conditions, hydrology is worth mentioning. River arteries are of particular importance both in Batumi and Poti. After the building of the Port of Batumi in 1878 the amount of suspended sediments from the River Chorokhi diminished. This resulted in intensive wash-out on the Makhinjauri-Chakvi reach, an intensively populated zone.

Construction of a high dam has recently started in the head of the River Chorokhi in Turkey. Regulation of the river flow is anticipated to result in a great reduction of sediments supplying the seashore zone in Batumi area. Again, the consequence is expected to be washing-out of sediments in populated districts.

3. RECOMMENDATIONS

Filling of the coastal zone with inert material has proven to be, and will remain the most effective way of rehabilitating the eroding areas. Any practical recommendations for protection of the geoenvironmental systems should aim primarily to attain natural equilibrium and remove any negative effect of natural and anthropogenic influences on the urban area.

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SECTION VIII:

SOCIO-ECONOMICS

GROUNDWATER INSTITUTIONS AND MANAGEMENT PROBLEMS IN THE DEVELOPING WORLD

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Abstract: Based on a literature review, the paper analyses groundwater management problems in the urban and the agricultural setting in the developing world. It is argued that in both settings groundwater rights can neither be established nor enforced. Groundwater rights are based on land rights and the economic capacity of the user to follow the falling groundwater table. The connection of groundwater access rights to land rights and economic capacity may cause social instability and could lead to further depletion of the resource.

Key words: management; water supply; agricultural water use; water rights; developing world; socio-economic issues; land rights.

1. INTRODUCTION

A ‘groundwater revolution’ began in the middle of the last century (Scott and Shah, 2004). Since then, groundwater utilization and the number of mechanized pumps have risen tremendously. Initially, groundwater utilization has often been perceived as a purely technical matter, but it also poses social and institutional questions, such that researchers discussing groundwater in the developing world speak of its ‘potential for social instability’ (Moench, 2002). As with surface water, groundwater access can face problems if the water rights are not defined or defined rights are not enforceable. Depending on the rate of recharge, open groundwater access can lead to conflicts among users, the marginalization of users and to its depletion as a natural resource.

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This paper provides a literature-based overview of institutional and organizational problems of groundwater management in rural and urban settings in the developing world, with special emphasis on South Asia. The second section looks at institutional and management aspects of groundwater, and the problem of enforceability of groundwater rights. This is followed by a section on the dynamics of groundwater utilization in the agricultural sector and its social consequences. The fourth section looks at the management problems in the expanding cities of South Asia. The presented data show that, both in rural and urban settings, groundwater utilization raises equity questions. In both cases a strong state is necessary to manage groundwater utilization but, though such management appears to be possible in urban settings, the situation in rural settings is much more complex, with very little hope of implementing water rights or controlling groundwater pumping.

2. INSTITUTIONS, POLICIES AND ENFORCEMENT

The groundwater revolution has caught hold in South Asia, the Middle East/West Asia, and North America and, to a lesser extent, in Africa and South America (Scott and Shah, 2004: p.150; see Table 1). Although both developed and developing countries have experienced a dramatic increase in groundwater utilization, their circumstances are rather different. The number of users, groundwater structures, the organizations controlling its utilization, and the enforcing capacities of the state vary tremendously between the countries of the developed world and those of the developing world.

Shah *et al*. (2001) argued that groundwater rights in the developed world are clearly defined and easily enforced, either because of the small number of users (e.g. the American West), or because of strong states (e.g. Western Europe). In the western United States for example, groundwater is subject to the doctrine of prior appropriation, with the strict distinction of senior and junior groundwater rights (Kropf, 2003). Under this system, a junior right-holder cannot interfere with the established rights of a senior appropriator. To receive a share, new users must bear the costs of upgrading the senior's appropriation.

In countries of the developing world which have a strong state and a small number of groundwater users, it is possible to monitor individual groundwater utilization. An example is Jordan, which has only 2,449 wells, used for agricultural, municipal and industrial supply. The government has enforced the metering of withdrawals and introduced charges for domestic and industrial users (Salameh, 2002). However, an ability to monitor individual withdrawals does not necessarily mean that groundwater rights are defined.

Table 1. Extent and magnitude of the global groundwater revolution, based on data from Shah et al. (2001).

Country/region	Annual ground-water use (km^3)	No. of groundwater structures (million)	Extraction/ structure (m^3/year)	Population dependent on groundwater (%)
India	150	19	7900	55-60
Pakistan-Punjab	45	0.5	90,000	60-65
China	75	3.5	21,500	22-25
Iran	45	0.5	58,000	12-18
Mexico	29	0.1	300,000	5-6
USA	100	0.2	500,000	<1-2

However, studies in India and Pakistan show that, in some countries, groundwater laws are either undefined, not enforced or not enforceable. In the case of India, Narain (1998) argued that the current regulations on groundwater have sought only to regulate the establishment of groundwater structures, rather than the quantity of water extracted. There are no *de jure* rights on groundwater but all landowners have the *de facto* right to groundwater underlying their land, with no limit on how much water they withdraw (Narain, 1998; Prakash and Ballabh, 2004). There are however some traditional community management systems: Van Steenbergen and Oliemans (2002) described the example of karezes in Pakistan, an ancient network of subterranean canals. Community management of the karezes gives all users the right to a certain amount of water and to decide who can utilize the system, as well as rights to the infrastructure (Beccar *et al.*, 2002), although it should be noted that this does not ensure equitable utilization. Mosse (1997) observed that traditional water management systems in India often reflected local hierarchies. Karezes were protected under customary law (the *harim* rule of Islamic origin) that determines the distance between new and old wells and which is comparable with the senior and junior rights system of the western USA. However, changes in technology, the introduction of deep tubewells, and decreasing groundwater tables have led to the collapse of many traditional groundwater management systems.

India has been drafting a groundwater bill for 20 years, but has been unable to make it law due to doubts about its enforcement (Shah *et al.*, 2001). Kumar (2000) listed the problems facing India in introducing a top-down legal and regulatory framework: (1) national legislation would not reflect local problems; (2) poor infrastructure and inaccessibility would make enforcement difficult, and (3) general political pressure against groundwater regulations from a strong farming lobby would limit the government's ability to enforce the law (Kumar, 2000: p. 425). The problem of a weak state with an inability to enforce regulations makes it questionable whether any top-down approach to the restriction of groundwater withdrawals will be successful. While Kumar (2000) focused on the availability of groundwater, Molle (2004: p. 209) pointed out that 'rights defined bureaucratically are likely to conflict with local formulation of rights and equity'. Hence, they could contradict and potentially destabilize existing local management systems.

Because of the limited effectiveness of top-down approaches, Kumar (2000) advocated the creation of local community organizations, which could be scaled up for aquifer management [see also Narain (2000) and Molle (2004)]. Citing the example of Mehsana district in Gujarat, India where, with the help of an NGO (non-governmental organization), a local community constructed local recharge structures and implemented water savings through optimization, Kumar (2000) reasoned that local groundwater management organizations can be successful. The success of the project resulted from politically easy solutions, which did not force utilization limits onto users. However, such projects maintain the status quo of groundwater use based on land rights and the advantage of deep tube wells in comparison with shallow wells. Depending on the level of additional recharge and water savings, it is possible that powerful users could benefit more than marginalized users.

While the above reasoning was focused on direct laws, rules and regulations, it may be possible to regulate groundwater withdrawals through indirect means. The supply of electric power to rural areas has been a primary driving force in the utilization of groundwater for irrigation; regulating electricity supply could be an effective tool for managing groundwater utilization. One possibility would be to increase the costs of electricity. However, even though electricity for the agricultural sector in India and Pakistan is subsidized (Kumar and Singh, 2001; van Steenbergen and Oliemans, 2002), the recovery of electricity charges is low. In Pakistan they do not exceed 50 per cent (van Steenbergen and Oliemans, 2002). This makes it questionable whether changing electricity prices will have an impact on groundwater utilization. Scott and Shah (2004: p. 162) made an argument for regulating electricity supply, giving three options ('In order of least to most difficult or acceptable socially, politically and technically in the Indian and Mexican context'): '(a) restrictions on new connections; (b) caps on capacity or amperage; and (c) reductions in hours of power supply'.

However, in many places in the developing world, the political reality is that powerful farmers can utilize their political clout to avoid regulations, obtain credits or electricity connections, or switch to diesel pumps (Kumar, 2000; Prakash and Ballabh, 2004). Hence, as with direct regulation, indirect regulations might be difficult to enforce.

3. GROUNDWATER UTILIZATION AND SOCIAL CONSEQUENCES IN RURAL AREAS

Though large-scale groundwater pumping was anticipated initially as being used only in land reclamation, the benefit of augmenting surface irrigation supplies was soon recognized (van Steenbergen and Oliemans, 2002). Today, groundwater development is perceived in Asia and Africa as more amenable to targeting poverty than surface irrigation, and therefore it has become the central element of livelihood creation programmes for the poor. Shah *et al*. (2001: p. 11) stated that 'groundwater irrigation likely creates more wealth per m^3, alleviates more poverty per m^3 targeted to the poor and spreads benefits more widely than surface irrigation generally does'. In India and Pakistan, support policies were implemented to encourage groundwater usage in the agricultural sector. These policies included subsidies for the costs of pumps and wells, credit support and subsidies on power supply to agriculture (Narain, 1998; Shah *et al*., 2001; van Steenbergen and Oliemans, 2002; Prakash and Ballabh, 2004). As a consequence, the utilization of groundwater has increased rapidly over the last 50 years.

Though high groundwater tables were, in certain areas, a threat to agriculture thirty years ago, today that situation has changed. Groundwater can be regarded no longer as a limitless resource: Shah *et al*. (2001) reported that the average depth of groundwater in the Indus basin has declined from 3.6 metres in 1988 to over 7 metres in 1996, and that in some areas groundwater decreased by 1.5 metres per annum. Narain (1998) stated that in a number of Indian states the exploitation rates had reached unsustainable levels, such as Punjab (94 per cent), Haryana (84 per cent), Tamil Nadu (60 per cent) and Rajastan (51 per cent). The higher administrative levels, such as state level, do not give clear evidence on the reality of groundwater mining, but at lower administrative levels (e.g. the district level) the situation appears to be more serious. Narain (1998) presented data for individual districts in Punjab and Haryana showing that the utilization rate is between 140 and 259 per cent respectively. In certain districts of Tamil Nadu and Gujarat the aquifers have already become permanently depleted (Narain, 1998: p. 358), whilst Prakash and Ballabh (2004) stated that in Northern Gujarat the water table had declined from a depth of approximately 3–5 metres in the 1960s to 150 metres by the year 2003.

The social implications of groundwater being an open access resource and the increase of groundwater pumping have to be placed in a time dynamic perspective. While the distribution of groundwater was determined initially by a few ‘water lords’ who could afford pumps and therefore establish a ‘seller market’, the low cost of pumps and the pooling of small farmers increased the number of pumps per hectare and therefore changed the market to a ‘buyer market’. Van Steenbergen and Oliemans (2002: p. 331) argued that in Pakistan ‘the fast development of tubewells was a democratization of groundwater access’ but with the rapid decline of the groundwater table the situation has changed again. Today, resource-poor farmers do not have the ability to chase the water table. Narain (1998: p. 358) stated ‘the groundwater depletion has serious equity implications, as falling water tables make the resource out of reach of small and marginal farmers as the costs of extraction increase’. The old pattern of dependencies seems to have reappeared: Prakash and Ballabh (2004: p. 33) argued that new ‘water lords’ have emerged and that water buyers are almost totally dependent on the sellers. Deep tubewell owners cannot only extract rents from buyers, by charging more than the pumping costs, but they have primary rights also: they are the first to irrigate. Water buyers have subordinate rights, being served only once the water need of the owner is satisfied. Hence, the unregulated private ownership of groundwater pumping structures in south Asia puts into doubt the potential use of groundwater in reducing poverty, and therefore undermines the non-critical promotion of groundwater irrigation during the last century.

4. UTILIZATION AND CONSEQUENCES IN URBAN SETTINGS

As Puri and Romanenko (this volume), Wegerich (this volume), and Maria (this volume) indicate, primary, secondary and tertiary cities in the developing world all have problems in providing sufficient water supply. Urban water authorities rely either on surface water or groundwater to supply the population. Showers (2002) analyzed data from 38 urban areas in Africa and stated (Showers, 2002: p. 624) that, between the early 1970s and the 1990s, ‘the number of locations reliant upon groundwater decreased from 58% to 47% while the number using river water increased from 55% to 68%’. However, the data did not provide any information on the shortcomings of the urban supply infrastructure, such as whether it was well-maintained or whether it kept up with the expansion of urban boundaries. Showers (2002) examined mainly the natural resource scarcity and the coping strategies of the municipalities. Helwig (2000) argued that there are three main reasons for water supply shortages: natural resource scarcity, lack of capital to expand infrastructure, and lack of capital to increase the efficiency of the current infrastructure. The lack of financial capital has different consequences for developing countries and for countries in transition. While in the context of the developing countries, it is mainly argued that the urban water supply infrastructure is under-constructed (see below), Davis and Whittington (2004) made the case for Odessa in Ukraine that the urban infrastructure has been over-built and is therefore unsustainable. Hence there are different inefficiencies in public water supply, each of which requires individual strategies. One possible strategy is groundwater utilization.

To analyze the situation in urban areas it is important to examine the different groundwater users and the different technology employed in groundwater pumping. It is argued that, in South Asia, large parts of the urban population rely on private pumping for domestic water supply (Khan and Siddique, 2000; Basu and Main, 2001; Foster, 2001; van Steenbergen and Oliemans, 2002). In their case study of West Java, Indonesia, for example, Braadbaart and Braadbaart (1997) argued that domestic users mainly utilized shallow wells. However, to obtain a detailed picture, it appears necessary to distinguish between individual houses, with a small number of users, and new apartment blocks. In Calcutta for example, Basu and Main (2001) pointed out that deep tubewells are utilized by high-rise apartment blocks and housing estates. Following the arguments of Maria (this volume), it appears that these new buildings are especially attractive to higher income urban residents. Hence, one might need to distinguish between a poorer population utilizing shallow wells, and a wealthier population utilizing deep tubewells. In this case, as in the agricultural sector, falling water tables have the potential to increase the socio-economic division.

While Braadbaart and Braadbaart's (1997) study of West Java and Das Gupta and Babel's (2005) study of Bangkok distinguished only between domestic and industrial users, Basu and Main (2001) made further distinctions, arguing that hotels, restaurants, and other businesses also utilize shallow wells. However, they also argued that, in Calcutta, deep tubewells are utilized by factories and businesses. Hence it is unclear which businesses utilize which forms of technology for groundwater pumping. Basu and Main (2001) argued that falling water tables in Calcutta are due to domestic use, whereas Braadbaart and Braadbaart (1997) argued for West

Java that a manufacturing-led economic expansion is mainly to blame for the groundwater overdraft problem. According to their data, industrial abstractors were responsible for 60% of the increase in groundwater withdrawals in the period from 1950 to 1990. Meanwhile, Das Gupta and Babel (2005: p. 460) stated for Bangkok that 'sixty per cent of the total groundwater pumping is by the industrial sector; the remaining 40 per cent is by individual households and water works authorities'.

While regulatory frameworks exist in urban contexts, the lack of enforceability makes groundwater in such settings often an open access resource in which, as in rural settings, land rights are connected to water rights. Das Gupta and Babel (2005: p. 460) stated that most of the groundwater wells in Bangkok are not metered which, given the large number of shallow wells for domestic use, is perhaps to be expected. However, more concerning is the observation of Braadbaart and Braadbaart (1997) that, in West Java, most private firms with deep tube wells did not have mandatory abstraction licenses, did not have metered usage, or utilized illegal boreholes. Similarly, Basu and Main (2001) noted that, though the municipal authority of Calcutta was instructed to seal all illegal deep tubewells in nine high-risk wards, the lack of records and sheer number of illegal wells rendered this task impossible. These cases suggest that the municipal authorities have neither the logistical nor the human resources to monitor and control groundwater utilization, and therefore cannot prevent overdraft. However, Braadbaart and Braadbaart (1997) argued also that the sustainability of groundwater management is dependent on the interests of the management agency, whether they are primarily the collection of revenues or the conservation of resources. They stated (Braadbaart and Braadbaart, 1997: p. 205) that 'perversely, it was in the interest of the Revenue Agency that groundwater users consumed as much as possible, since more groundwater consumption implied more provincial revenues'.

Das Gupta and Babel (2005) showed how a change of interest can lead to a reduction in groundwater exploitation. They demonstrated (Das Gupta and Babel 2005: p. 459) that the municipality of Bangkok was able to reduce groundwater exploitation between 1983 and 1987, accomplished through the introduction of groundwater charges at a rate equivalent to that of the public water supply and by the restriction of new installations. However, because surface water supply could not meet the public water demand and because the supply system could not be expanded to newly developed suburbs, annual pumping rates increased again.

Foster (2001) argued that a top-down approach which restricts private pumping will not be successful; only community facilitation will allow the development of sustainable groundwater management in urban settings. Galofre *et al*. (2002) presented a similar argument with an example of a groundwater user association from a developed country. According to their study, the water administration of Barcelona initiated a groundwater user association and delegated some water resource management tasks to the association. They noted (Galofre *et al*., 2002: p. 6) that the association became 'the hands and the eye of the administration', collecting data on extraction and changes in water use. Das Gupta and Babel (2005) described the contemplation of a different strategy in Bangkok, where the authorities are considering the involvement of the private sector in a monitoring and charging system. In the case of groundwater user associations, one could learn from the experiences of the associations in surface irrigation, where it is argued that high levels of heterogeneity among the users might prevent collective action. Hence, it might be difficult to bring industrial and domestic user groups together. Whether examined with the involvement of the public or the private sector, the main issue to be resolved is whether a small number of deep tubewells or a large number of domestic shallow wells is the primary cause of groundwater over-extraction. Arguably, the first would be easier to control then the second.

As noted above, the demand from growing populations and the decreasing levels of water resources has led cities to claim water from surrounding areas as well as from distant regions (Showers, 2002; Knapp *et al*., 2003). While individual water rights in more developed countries are established, enforceable and transferable, in less developed countries this is not necessarily the case. A major concern therefore with water transfers from agricultural to urban areas is the equity effects on the regions supplying the water. In their case study on the conflicts between local communities in the Ixtlahuaca valley and Mexico City, Dyrnes and Vatn (2005) pointed to the unequal power relationships that led to a restriction of the water rights of local communities, the creation of dependencies and irregular compensation for lost harvests. Rising demand for water seems to be a normal process of increasing urbanization and therefore a reallocation of water resources might become a strategic option. However, as the case of Ixtlahuaca valley and Mexico City shows, the interests and the rights of the senior users have to be addressed, and compensation mechanisms have to be mutually agreed upon.

5. CONCLUSIONS

The discussion has shown that groundwater rights in rural and urban settings in the developing countries are based mainly on land rights and that therefore groundwater is an open access resource. In the more developed countries, the strength of the state authorities or the limited number of users have made it possible to define and enforce groundwater rights and to regulate utilization.

The review presented here suggests that rather than focusing on the definition and enforcement of rights, the focus in urban areas is on the control of infrastructure, such as restrictions on setting up and utilizing new tubewells, as well as metering and charging for groundwater utilization. In this sense, groundwater is a public resource and, at least on paper, a domain of the municipalities. However, the problems of expanding the public supply system, the unreliable and limited water supply in existing systems, and the limited logistical capacity to enforce restrictions as well as metering have hindered the implementation of these policies. The case of Bangkok suggests that the predictions of Maria (this volume) might be too pessimistic and not correct, but they depend on the strength of the municipalities in enforcing metering and groundwater charges.

The situation in rural settings appears to be worse, with very little prospect of implementing water rights or controlling groundwater pumping. A focus on increasing recharge and the efficiency of utilization might change the situation, though it is questionable whether this policy can be effective in different regions and whether it would be possible to finance an increase in irrigation efficiency. In addition, because the policy does not address the issue of groundwater rights, it might still leave poor and marginal groups worse off.

Overall, irrespective of whether top-down legislation or the use of individual allocations in local settings is implemented, the state would have to take responsibility by providing a general legal framework plus the responsibility to make the right enforceable: in the case of allocations, it would also have to give guidance and empowerment to the local community. This would enable the community to manage its water resources on an acceptable basis that would not create new groups of marginalized users, nor would create economic, social, and political dependencies on local ‘water lords’.

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THE ROLE OF GROUNDWATER IN DELHI'S WATER SUPPLY

Interaction Between Formal and Informal Development of the Water System, and Possible Scenarios of Evolution

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Abstract: This paper presents an economic approach towards urban groundwater in the context of a fast growing metropolis. It attempts to understand the current role of groundwater in the urban water supply for the city of Delhi, India, and proposes possible scenarios of future evolution. The main insight of this paper is that in spite of the public neglect of groundwater as a resource for urban water supply, groundwater plays a central role in meeting urban needs through a variety of private and uncontrolled systems. The potential role of these systems in the future and their impact on urban sustainability is questioned.

Key words: Delhi, India; urbanization; sustainability; technology; private supplies; uncontrolled systems; water supply; socio-economic issues

1. INTRODUCTION

The case of Delhi is representative of the situation of many urban water systems in developing metropolises. The low level of service provided by the public utility companies has led the upper and middle classes to develop coping strategies based on private groundwater abstraction. These include direct use of groundwater from a private tubewell, development of a private small supply network fed with untreated groundwater, or supply by tanker, groundwater remaining the primary source of raw water in most of the private supply chains. As a consequence, indiscriminate abstraction has led to a rapid fall in water tables across the city, threatening the short-term sustainability of the system.

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The response of public authorities to the crisis is not clear. It combines propositions of management reform of the piped supply, projects of raw water supply augmentation through building large reservoirs in the Himalayas, but also plans for dual piped supply, encouragement of roof-top rainwater harvesting, and some projects of local water harvesting at a larger scale. The largest unknown factor underlying most of those policies is the future role of local groundwater resources. Although falling water tables used to make the headlines in the capital's newspapers, in technical debates groundwater supply is primarily discussed only in terms of its contribution to the sources of public supply, which is minimal compared to that of private abstraction.

It appears that the main reason for this lack of acknowledgement is the current inability of the public authorities to regulate groundwater abstraction. In the absence of an enforceable formal property regime for urban groundwater, the resource is considered as a buffer, forcing the undeserved user to wait for an improvement of the network-bound public service. Thus, it is not properly included in the long-term planning process.

I will present an overview of urban development in the region of Delhi and its impact on groundwater resources. I will then discuss the different dynamics of development of the piped supply system, and of emergence of alternative systems giving a central role to groundwater resources. Based on this discussion, I will present possible scenarios of co-evolution of the formal system based on the water supply and sewerage system, and the informal system based on decentralized groundwater use.

2. THE CURRENT SITUATION

2.1 Urban Development and the Formal and Informal Role of Groundwater in the City's Water Supply

Delhi, as the capital of India, has a specific status in the Indian political federalism. The city is located in the National Capital Territory of Delhi (NCT Delhi). This territory has a pseudo-state status and is under the mixed control of the central government, and of a local government similar to that of other Indian states. The National Capital Territory spreads over a total area of 1483 km^2, of which more than 60% is now urban.

This political situation has a direct impact on Delhi's access to water resources since, in the Indian constitution, the management of water resources falls primarily under state responsibility. The National Capital Territory of Delhi, with a population around 15 million people, has therefore very few resources under its direct control.

Since the beginning of the twentieth century, the city of Delhi has experienced an exponential population growth (Fig. 1).

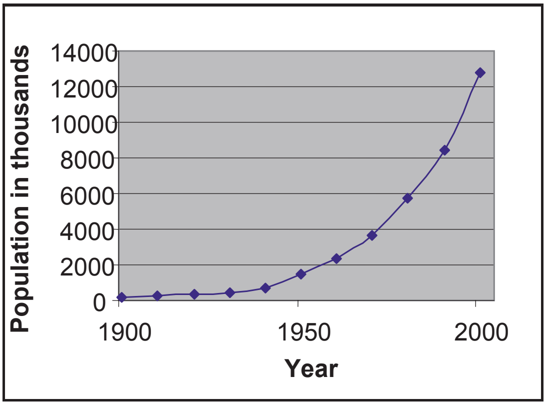

Figure 1. Population growth in Delhi (Census of India data).

The public authority in charge of housing development, the Delhi Development Authority, has not been able to cater for the consequent demand. According to various estimates, 50% of Delhi's population lives in some kind of informal, unplanned and/or precarious settlement. Among them, more than two million people, representing more than 15% of the total population, live in illegal squatter settlements and face a permanent risk of eviction. The accountability of the municipal utility in charge of water supply and sanitation towards populations living in informal settlements is loosely defined, and most of the latter do not have a proper connection to the network. According to the Water and Sanitation Program (WSP), more than 40% of the urban poor in India rely only on groundwater resources through handpumps and shallow wells for their water supply.

As shown in Table 1, around 75% of the households in Delhi are reported to have access to piped municipal supply either through a private connection or a common stand point, with around 20% of the population relying on hand pumps tapping the shallow aquifer for their water supply.

Table 1. Access to piped water and reliance on hand pumps in NCT Delhi.

Year	1991	2001
Households with access to piped water	75.3 %	75.6 %
Households relying on hand pumps	20.06 %	18.68 %

Source: Census of India 1991 & 2001.

The contribution of groundwater to the municipal supply is around 11% in terms of volume, which represents around 370 million litres a day (MLD). This figure is often used as an indicator of the relative importance of groundwater for the city's water supply. However, it does not accurately reflect the true role of groundwater in the city's water supply. In order to understand the hidden role of groundwater in Delhi's water supply, one has to understand the current state of failure of the municipal supply system. Water supply from the network is intermittent and unreliable. Private connections and public standpipes receive water for only one to four hours a day, resulting in great uncertainty over availability and demand. Zerah (2000) investigated the development of private strategies for coping with the unreliability of the public supply, and found that the total annual cost of these strategies represented twice the annual public expenditure in water in Delhi. A large percentage of these strategies involve the installation of private tubewells to complement or substitute for the public supply. Although the number of private tubewells legally registered to the authority in charge is around 100,000, different sources estimate the actual number of private tubewells to be between 200 000 (Planning Department, 2004) and 360 000 (Central Ground Water Board, 1996). Precise information on private groundwater use is difficult to obtain since this use often does not comply with the regulations. Therefore, owners of private tubewells are rather reluctant to provide information on their practices and hence no precise figure for the total volume of water abstracted through private tubewells is available. However, approximate estimates for the year 1996 give a total abstraction of around 1300 MLD (Central Ground Water Board, 1996).

One can compare these estimates with the volume of municipal water actually reaching the final user. The total capacity of the municipal system is 2950 MLD, with recent estimates giving a level of technical losses of 40% (Planning Department, 2004). The volume of municipal water reaching the final user is therefore around 1770 MLD. Assuming there to have been a growth in the private abstraction of groundwater between 1996 and 2004, it can be estimated that, in terms of volume, private abstraction of groundwater represents around 50% of the total supply to the final users.

2.2 Hydrogeological Characteristics of Delhi

The National Capital Territory of Delhi is part of the Indo-Gangetic alluvial plain. The river Yamuna, a tributary of the Ganga, flows through the eastern part of the territory, and a quartzitic ridge, rising up to 91 m above the surrounding plains acts as a groundwater divide between the western and eastern parts of Delhi (Fig. 2). The alluvial formations overlying the quartzitic bedrock have a different nature on either side of the ridge. The Chattarpur alluvial basin, which covers an area of about 48 km² and is almost entirely enclosed, is filled by alluvium derived from the adjacent quartzite ridge. The Yamuna flood plain meanwhile, contains distinct fluvial deposits from upstream sources. The alluvial plains on both the eastern and western sides of the ridge are characterized by the occurrence of older alluviums.

The average annual rainfall in the region is 611 mm, most of which falls between July and August, during the monsoon season. The total renewable resources have been estimated by the Central Ground Water Board (1996) at around 290 million m^3/yr. Datta *et al*. (1996) carried out isotopic investigations to assess the natural recharge in the NCT Delhi and found levels of recharge lower than 5% of precipitation in most of the area. In the urban centre, recharge was lower than 3%.

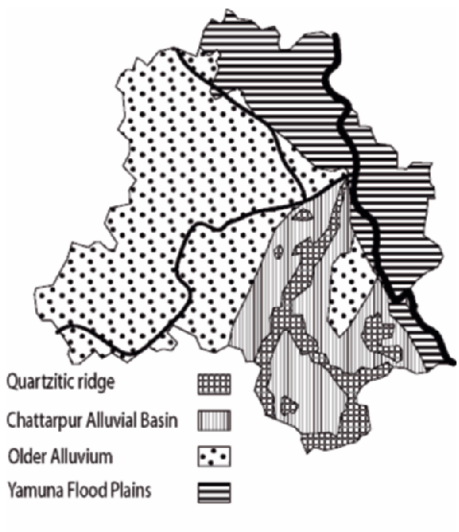

Figure 2. The main geological formations of the NCT Delhi region. Dark lines indicate surface water courses, the heaviest the Yamuna.

Although Delhi benefits from a large quantity of available groundwater, mainly in the alluvial aquifers, the uses of this water are compromised by the occurrence of saline water in the deeper layers. The repartition of the static groundwater resources between fresh, brackish, and saline water is given in Table 2.

The map in Figure 3 illustrates the distribution of salinity in the different regions of NCT Delhi.

Along with salinity, fluoride concentration is a major constraint to safe groundwater use for water supply. Fluoride concentration exceeds the WHO norms of 1.5 mg/l in 30% of the NCT area.

Table 2. Repartition of static resources according to salinity (TDS is total dissolved solids).

Water Quality	Comments	Static resources	% of total
Fresh Water	TDS < 1000 ppm	1866 Mm^3	30 %
Brackish Water	1000 < TDS < 5000 ppm	659 Mm^3	10%
Saline Water	TDS > 5000 ppm	3635 Mm^3	60%

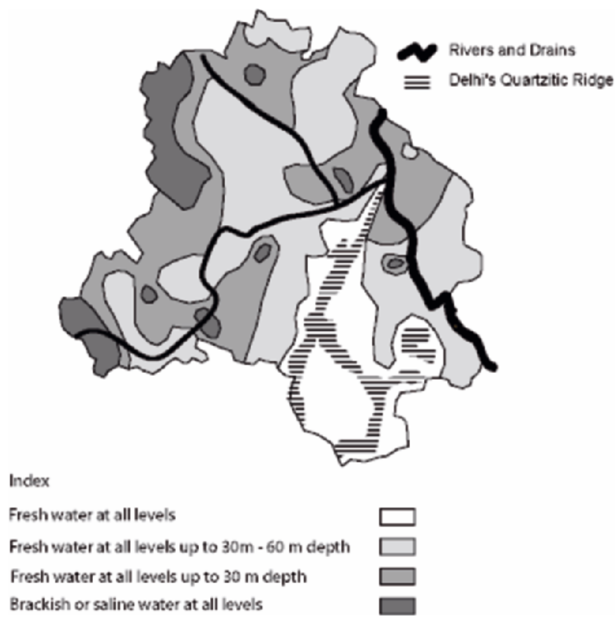

Figure 3. Patterns of groundwater salinity in the NCT Delhi region (modified after http://www.rainwaterharvesting.org/index_files/ground_water_quality.htm).

2.3 Depletion and Pollution of Water Resources

Although groundwater abstraction is in theory regulated in Delhi, the authorities in charge of the enforcement of regulations do not have the practical means of controlling private abstraction. In practice, abstraction is tolerated as a compensation for the failure of the public supply system. As a consequence, water tables are falling at an increasing pace in several areas of NCT Delhi (Fig. 4). The southern and western regions of Delhi, where piped supply does not match the growing residential demand, are the most severely affected areas. In the Chattarpur alluvial basin, where water tables are falling at the highest rate, the availability of fresh water will cease within a few years if abstraction continues at the current rate. A similar scenario is also likely in the western part of the state, where urbanization is occurring in areas affected by the occurrence of saline water at shallow depths.

In addition to this, Delhi’s groundwater resources are subject to various forms of pollution. Nitrates and pesticides are generated by agricultural activity in the rural areas in and around the NCT Delhi, heavy metals are accumulating due to the infiltration of urban run-off, and bacteriological contamination affects most shallow aquifers.

The many problems affecting Delhi’s groundwater resources could explain the attitude of the local authorities in regard to the process of long term planning for water supply. However, as will be shown in the next section, a number of systems are being developed that can give groundwater a central role in the city’s water supply.

3. DIFFERENT MEASURES AND THEIR INTERACTION

3.1 Introduction

Although the situation in Delhi can be summarized as “dry taps and falling water tables”, different approaches to solving the problem can be found among the stakeholders, each giving rise to a distinct set of dynamics in the evolution of the water system. Here, I make a basic distinction between measures that keep the network-bound water distribution and waste-water collection system as the core technical element of a strategy, and alternative approaches based on different flows of water and waste-water.

3.2 Network-bound Measures

3.2.1 The bureaucrat-engineer approach: more pipes, more dams

Thus far, the traditional approach of the administration to the problem of water supply in Delhi has been based on quantity issues. A demand-supply gap has been identified by comparing the supply capacity with a projected “demand” derived from population estimation and supply norms. Following the norms applied, the “demand-supply” gap is estimated at between 900 and 2000 MLD. This allows the problem to be expressed in terms of overall availability and to formulate solutions in terms of augmentation of raw water availability. With the existing pressure on local groundwater resources and the lack of administrative control on these resources by the municipal utility, priority has been given to the development of large scale projects based on the construction of dams and long distance transfers of water. Two ongoing projects are expected to deliver 360 MLD and 640 MLD respectively, the first consisting of the building of a channel parallel to an existing canal in order to reduce seepage and evaporation, the second consisting of a diversion of water from the recently built and highly controversial Tehri Dam on the Ganga river. The overall approach, however, seems to have reached its limits. Indeed, the ongoing projects of raw water augmentation have been delayed by several years, whilst proposed damming of tributaries of the Yamuna have been blocked due to a lack of assessment of their environmental and social impact.

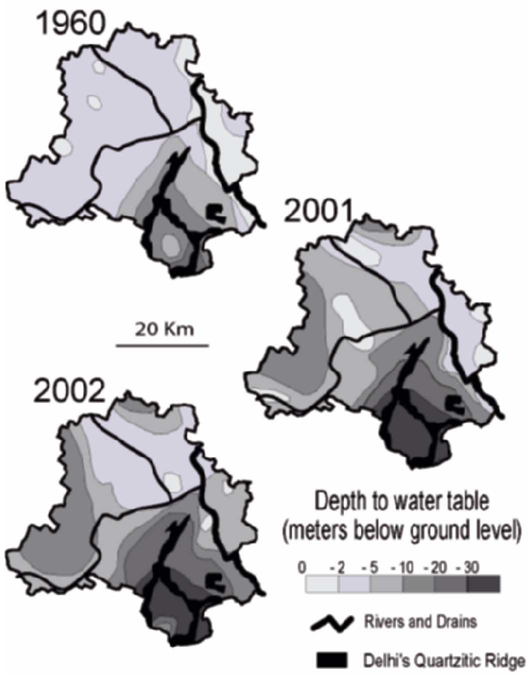

Figure 4. Changes in water table levels in NCT Delhi between 1960 and 2002 (modified after http://www.rainwaterharvesting.org/index_files/water_level_fluct.htm).

However, it is now argued that with the quantity of water currently available, a satisfactory level of service could be achieved, sustainable over the middle term as a consequence of the completion of the ongoing projects. This can be termed the 'economist approach'.

3.2.2 The economist logic: reform of the public utility

The economist approach considers management rather than raw water availability to be the central problem. In the current system, there is a very large disparity in the level of service offered to different users. Water tariffs are either fixed or, in the case of a metered connection, very low (0.35 Rs./m^3, equivalent to less than 0.01 US$/$m^3$). In both cases, there is no incentive for users benefiting from a consistent supply to restrain their consumption. Therefore, allocation has to be predetermined through restriction, by supplying water only during a limited period of the day. This mode of allocation, which is widespread in South Asian cities, leads to unreliable supplies, coupled with a high risk of contamination through waste-water infiltration into supply pipes. Based on the experiences of industrialized countries, the economic superiority of a single network supplying water treated to potable standards is assumed. As a consequence, this leads to reform of the municipal utility being set as the priority, with the goal of supplying water on a 24-hour basis and controlling demand through tariffs. This reform, although possible, requires considerable effort in terms of capacity building and public debate, and one can assume that it would take at least a decade to achieve the transition of the whole city to a new mode of management.

As outlined above, although it is proposed to improve the level of service through a centralized network-based system, the implementation of solutions is bound to take time. In this scenario, groundwater is supposed to act as a buffer, consumers being expected to cope with the inefficiency of the network-bound system in the short-term whilst waiting for its overall improvement in the long-term. It is worth considering, however, whether groundwater could play a different role in the process of evolution of the water system, be it through its inclusion in the planning process and coordinated development, or through the spontaneous emergence of independent self-sustaining systems.

3.3 Alternative Technical Systems and Their Mode of Emergence

3.3.1 Proposed alternative systems

The systematic development of alternative systems has enabled several propositions to be considered in regard to augmenting the availability of water in Delhi. Among them, rain-water harvesting (RWH) has been the most intensively promoted, first by the civil society and then by the public authorities. Existing systems are based on the collection of monsoon run-off from roof tops and paved areas for injection through specifically designed pits. Building regulation has been modified to make RWH mandatory on new buildings and, in severely affected areas, users of tubewells are instructed to install such systems to compensate for their abstraction. However the adoption of this system has thus far been quite limited and the overall potential of RWH is relatively small. It has been calculated that if the entire urban area of Delhi was equipped with RWH systems, the volume of water collected would provide only 10% of the current assumed consumption (Soni, 2003).

In addition to RWH, propositions for local harvesting and storage of regional monsoon run-off at a larger scale have been made. A local NGO, INTACH, has formulated propositions combining the development of drainage channels and floodplains as well as rehabilitation of urban lakes and traditional reservoirs in order to increase the storage potential of regional monsoon run-off in the city (INTACH, 1998).

Waste-water reclamation is also being considered. The only use currently considered by public planners for reclaimed waste-water is urban horticulture and irrigation in peri-urban areas. Although a number of benefits could be expected from the substitution of groundwater by reclaimed wastewater, the implementation of large scale waste-water reuse does not appear to be currently prioritized by the NCT Delhi authorities. This may be a result of the economic costing of the technical systems needed to dispatch treated groundwater from the existing large scale waste-water treatment plant to the different points of use. The use of treated waste-water for aquifer recharge is not considered to be technically feasible at the current level of understanding of the impact of such recharge.

Dual supply, which has already been experimented with in several housing complexes through private initiative, is also proposed as a logical development of the distribution system. This combines the piped potable supply with the decentralized supply of groundwater for non-potable uses.

The coordinated development of alternative systems based on combining the techniques discussed above with the existing network-based system, does not seem to be a priority of the present authorities. This might be explained by the fragmentation of responsibility, authority, and information between the different agencies in charge of urban development, water supply and sanitation, drainage, groundwater exploration, and groundwater control. The current status of groundwater is a central element. Although groundwater exploitation is regulated in theory, the failure of the public supply has led the authorities to tolerate a *de facto* private appropriation of groundwater (Ruet, 2002). In this context, groundwater appears not to be valued adequately by the public authorities.

As outlined above, the upper and middle class have invested massively in private tubewells, especially in the southern and western part of the city in which water supply is least reliable. According to Llorente & Zérah (2003) there are around 50 private companies in Delhi involved in the provision of water, either in bottles or jars, or from tankers. Most of it is groundwater, either treated when provided in bottles and jars, or untreated in the case of tankers. As revealed by the study of a recently urbanized area, the upper and middle classes settling in multi-storey apartments with limited municipal water availability have invested in systems of increasing complexity. Systems have emerged that deal with the occurrence of brackish water in the shallow aquifers,, combining treatment through reverse osmosis, ion exchange, conjunctive use of groundwater and municipal water, rainwater harvesting, and in certain cases, sullage recycling.

These systems are currently unsustainable and, in all areas where groundwater is used as a complementary source of supply, water tables are falling at an increasing rate. However, the private market for such equipment and solutions is evolving very fast. The market for individual reverse osmosis purifiers is attracting more and more companies. Indian companies are developing small-scale waste-water treatment plants able to provide tertiary level treatment of domestic waste-water at an affordable cost, and a large number of engineering consultants are now proposing customized solutions to residents eager to ensure themselves a reliable and sustainable supply of water. The cost of these systems is still prohibitive and their development has arisen only due to the current failure of the municipal utilities. However one might anticipate a rapid improvement in their efficiency. It is also possible that the hydrogeological knowledge of solution providers, which is currently limited to well discharge, will develop into a competent understanding of aquifer recharge and recovery.

4. CONCLUSION

In order to understand the interaction between the co-existing approaches to the problem of water supply described in this study, one has to take into account the time frame of the different dynamics involved. Once this is done, it is possible to differentiate three basic scenarios.

The first scenario is the development and reform of the centralized network-based system. This scenario could arise if privately developed systems lead to a depletion of groundwater resources, prior to the development of techniques (e.g. a combination of rain water harvesting and decentralized waste-water reclamation) that could make them sustainable.

The second scenario is the development of sustainable independent systems reducing the long-term reliance of the upper and middle classes on the centralized public system.

The third scenario is an approach coordinating the development of alternative systems with a reform of the public system. This coordination would require the management of groundwater as a collective resource, and the integration of both service management and local resource development.

Several factors will affect the evolution of the current system towards one of the scenarios described here. Among them are the following:

- the political process of interstate surface water allocation, which will determine the ability of Delhi to increase the volume of raw water available for its centralized system.
- the ability of the municipal utility to improve rapidly its efficiency and ability to manage demand
- the institutional integration between service management and local resources development
- the evolution of technologies for aquifer recharge and recovery, and decentralized waste-water reclamation
- the involvement of communities in the creation and management of alternative systems.

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WATER SUPPLY AND SANITATION SECTOR ANALYSIS OF THE SECONDARY TOWNS OF AZERBAIJAN

Does Groundwater Play a Role?

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Abstract: This paper concerns the bringing together of the science of hydrology and the socio-economics of water supply in order to deliver a high degree of human security to those people who have diminished, or complete loss of access to their water supply. Human security is measured in a number of ways, but one underlying factor for all humanity is access to adequate water (quality and quantity) at an affordable cost. In Azerbaijan there are over 50 secondary towns with populations in the range 10,000 - 100,000. While the larger towns had a well developed industrial and infrastructure base during the former Soviet Union, with its collapse and shut down of industry in the early 1990s, the town infrastructure has significantly deteriorated and water supply and sanitation systems are anticipated to collapse by the middle of this century. The situation in the smaller towns, where the economic base was more fragile, having been founded on agriculture and processing of agricultural products, the situation is no less dire. For a combined population of 1.45 million in secondary towns, the systems originally designed to a capacity of 179.6 Mm^3/y, actually delivers only 48.2 Mm^3/y. As a consequence, the population resident in these towns suffers considerable hardships incurring high personal costs, thus further impacting their incomes, and reinforcing widespread poverty. The water related health and environmental problems add to the burden, trapping some people into a vicious cycle of poverty.

Key words: Azerbaijan; secondary towns; poverty; water supply; infrastructure; socio-economic issues.

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1. INTRODUCTION

In the former Soviet Union (FSU) time, the vast majority of the towns in Azerbaijan were connected to a central water supply and sanitation (WS&S) system, and the people received unlimited quantities of free, potable water and sanitation services. Following independence, and the introduction of the free market economy, the cost of municipal services has been passed on to the citizens, who are being required to fund, what until recently were free services. Significant difficulties in transforming the old municipal utilities into modern operators that provide services, for which they might charge their customers, is ringing alarm bells. Their lack of capacity, access to finance and unsuitable management skills are leading to an inevitable end. Many water supply sources are threatened by poor source water (high sediment loaded rivers) or by collapsing intake systems (damaged river infiltration galleries). Although there may be adequate water resources, the means to abstract do not operate, as borehole pumps have burnt out from fluctuating electric power. Trunk and distribution pipelines are old, poorly completed and thus leak considerably. High sediment load in the supply means that consumers leave taps running adding up to a huge amount of unaccounted for water. Waste-water management was always a lower priority, consequently in some towns this is a direct threat to residents, who suffer from flooding of sewage. The situation is exacerbated in time of heavy rainfall, since storm drainage systems also fail to operate.

Most urban conurbations have to rely on some form of a centralised supply system; while many single family houses with small land plots may have wells or nearby streams. The multi storey apartment dwellers cannot access such sources, and thus cannot hope to have any relief from this daily irritant. Azerbaijan is not the only country in the Central Asian-Caucasus region with this problem (World Bank, 2001). Secondary towns in Kazakhstan, Kyrgyzstan, Uzbekistan, Georgia and Armenia (Messrs JINJ, 2003) all have similar, near-crisis conditions. Efforts are being made by most governments at improvements, though examples of success are still few.

2. PROBLEM IDENTIFICATION & STRATEGIES

2.1 Sector Analysis and a Financing Strategy

In an effort to address the situation at its basic levels and to find some sustainable technical-financial alternatives, a ‘sector analysis’ has been carried out for the secondary towns of Azerbaijan, which is the subject of this paper (Figure 1). There are more than 50 secondary towns in Azerbaijan defined by populations in the range 10,000 - 100,000. The sector analysis undertaken sought to find some strategic solutions to assist the government in planning the rehabilitation and improvement of the sector at affordable levels. Through an analysis of the 'business as usual' scenario it was possible to demonstrate that action is required at the earliest possible time – a point that has been made by analysts in the past several years (e.g. World Bank, 2000; OECD & DANCEE, 2003). In the event, despite all the problems, water is still being delivered to consumers somehow or the other, in a very limited and unreliable way – however, increasingly consumers are making their own arrangements, as they cannot rely on some future promises. Drawing from wells, streams, truck deliveries, sharing of sources are common; in smaller conurbations this alternative works for some of the time, for some of the consumers. In larger conurbations this is difficult. The question is how to provide reliable water and sanitation to all consumers?

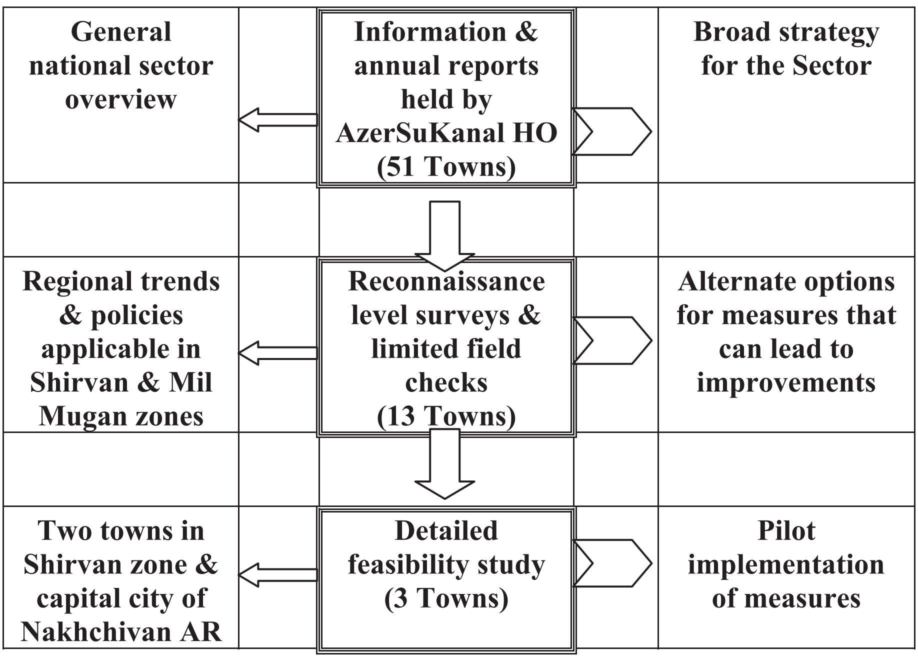

Figure 1. Steps towards the sector analysis.

The basic problem can be encapsulated by an appreciation of the fact that the level of service demanded is high and affordability is low. If the WS&S is to be rehabilitated to the former standards (i.e. 24 hour per day potable water delivered at the appropriate pressure at the household connection) the costs exceed affordability levels considerably (EU TACIS, 2003). The past WS&S infrastructure was oversized (following the FSU norms of between 250 lcpd & 300 lpcd); the actual need, as ascertained in this study could be as low as 80 lpcd). The rehabilitation and repair of existing oversized WS&S installations would leave the systems highly inefficient, making the O&M costs unrecoverable. Capital costs for a completely new system might reach $ 300 per capita (Figure 2), again a figure that is unaffordable at commercial and even concessional borrowing rates, though the O&M costs of a new, highly efficient system would be well within the affordability level.

The sector analysis conducted in the study took a wide ranging approach. After demonstrating that 'business as usual' was not an option (Figure 3), it analysed an optimum level of financing to restore and rehabilitate the old systems, because wholesale renewal in the 51 towns was also not affordable nationally, despite the annual 10% increase in national GDP. Taking the capital value of the WS&S assets in the year 2002 and estimating the annual repair and maintenance costs as a starting point, a series of performance simulations were carried out. In these, capital investment and the associated improvements were calculated. The results from several financing strategies were optimised to select the alternative that provided the highest output of treated water and the lowest losses from the delivery systems. The analysis suggested that capital sums equivalent to 60% of the current WS&S asset value, split 20% on the production and treatment side and 40% on the transmission and distribution side, disbursed over an 9 year horizon from 2003/2004, could make rapid and effective improvements in the WS&S systems.

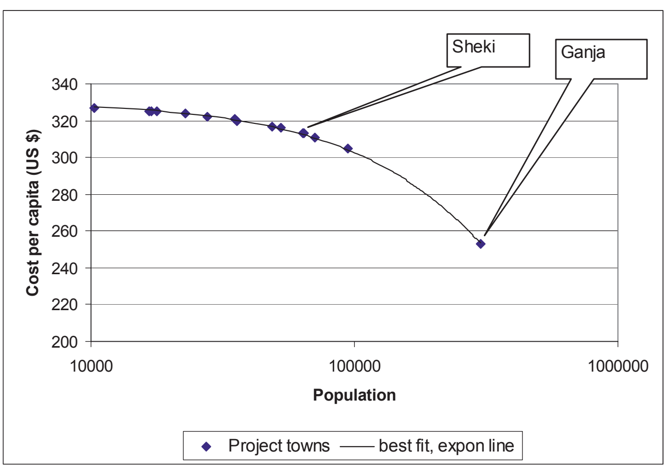

Figure 2. Indicative capital costs for rehabilitation of WS&S facilities.

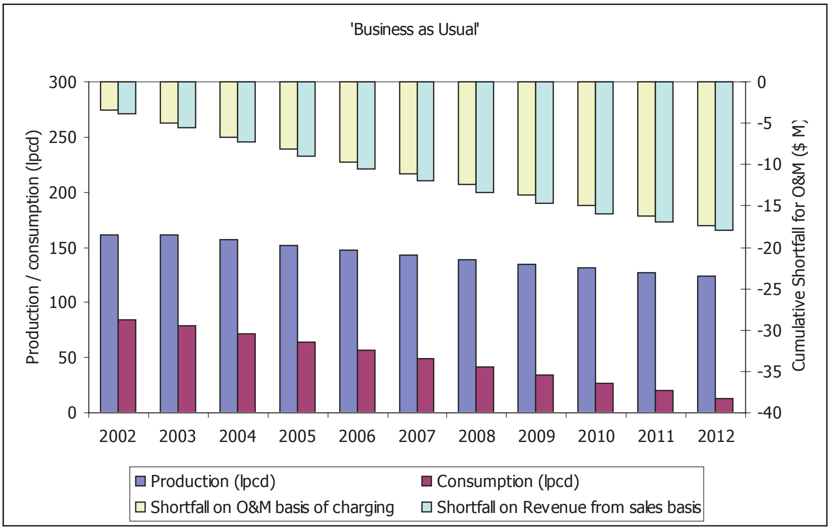

Figure 3. Outcome of 'business as usual'.

2.2 Pilot Project Approach to Test the Long Term Strategy

The analysis was based on data and information provided by the current water utilities. In the course of field work checking the information in the regions of several utilities, it was found that information was unreliable and often incorrect, because the means to collecting the data are simply not available to the utilities. No measurement equipment is held, records are poorly kept and data is badly managed. Despite all efforts to improve the data used in the analysis it was concluded that the main recommendations of the analysis should be tested in a series of pilot projects. Three towns were chosen for the pilot implementation of the strategy. These are Goychay (population 35000), Agdash (population 25000) and Nakhchivan (population 94000). A detailed feasibility study has been completed, running parallel with the sector analysis. A project involving a capital investment of $53 million has been designed. The water source for all the three towns is groundwater based, two projects will involve pumping from wellfields, one will involve gravity delivery. Co-financing between Asian Development Bank, the Islamic Development Bank and the Government of Azerbaijan has been proposed.

In all three towns the current WS sources are river gravel aquifers drawing water through infiltration galleries, fed to the towns' centres by trunk mains between 12 and 19 km. In two towns, Goychay & Agdash, the galleries have been completely destroyed from devastating floods in 2003;

an erosion process has started, creating a deep rill in the river bed, making the infiltration gallery unviable (Figure 4). The Nakchivcan gallery is safe, as it is protected by a major dam structure (Vaykhyr dam) a few kms upstream. Eight alternatives water supply systems were investigated and discussed with stakeholders. Least cost analysis suggested that wellfields would be the most viable option. As these would have to be pumped, losses in the distribution and the trunk mains cannot be tolerated to achieve a reasonable level of O&M cost. The capital cost distribution as demonstrated in the sector analysis was applied. In the case of Nakhchivan, given that gravity delivery can continue, distribution loss reduction can be somewhat postponed, while reliable water supplies are delivered to the town. Staged reduction of network losses were proposed for Nakhchivan.

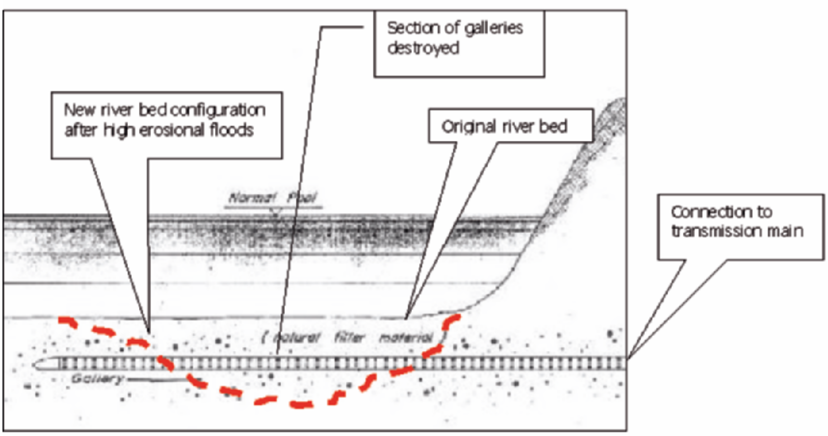

Figure 4. Damage to infiltration galleries from intense floods.

2.3 Local Water Resources - Local Management

One of the main conclusions of the study is that local water resources need to be developed for local management. Usually this is a local aquifer system. Many of the towns located along the lower flanks of the Caucasus mountains have small, reasonably yielding aquifers in alluvial cones. However, the rapid transition from fresh water to brackish and saline, at the interface with the clay-alluvial deposits of the Kura basin, makes them susceptible to quality deterioration if not managed with a good hydrogeological analysis. While data are available for most of these cones, they are somewhat unreliable and out of date. One of the challenges to the sustainable development and implementation of the sector strategy will be a good appreciation of the local aquifers around most of the secondary towns. The exceptions are those towns that overlie the thick Kura basin alluvium consisting of marine and lacustrine clays.

2.4 Institutional Development & Strengthening

A new WS&S system also needs modern efficient management. The sector analysis made a review and an assessment of the most viable institutional structure for secondary towns. The WS&S sector in Azerbaijan is in transition from central government administration to local management. The government has considered private sector participation, with input of equity as the main approach to improvements. In view of the developing legislation, regulations and formation of supervisory bodies, any institutional proposals have to be necessarily considered transitional. The Sector review has suggested a major revamping of the current structure, with strong local participation and a transitional strengthening of the new institution through a management contract (Figure 5).

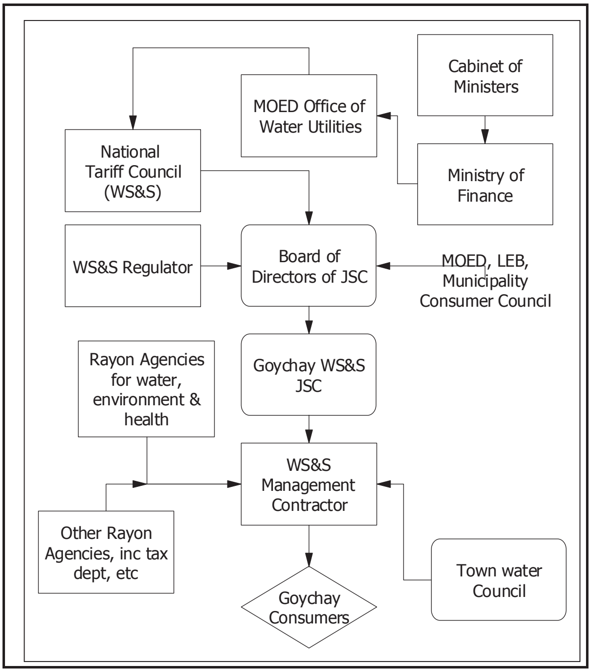

Figure 5. Proposed institutional structure for WS&S in pilot towns.

3. CONCLUSIONS

Urban water supply and sanitation is facing significant challenges in the Caucasus region. The challenges in the FSU are stark because the service level demanded and anticipated is far in excess of its affordability. Reliance on market forces as a means to adjust demand and supply may not provide a solution in the short term. Capital investment by governments is essential, while full O&M costs can be passed over to consumers; full cost recovery, inclusive of the cost of rehabilitation, from consumers is not possible. The most cost effective solutions, relying on local water sources, generally aquifers, will provide the optimum solutions. Waste water treatment based on simple technologies such as waste stabilisation ponds must be included if the local sources are not to be threatened. The suggestions made in this paper cannot be considered the final answers – they should be tested in pilot projects; the successful solutions should be systematically applied in all secondary towns. A well planned strategy should be adopted as soon as possible to avoid the current *ad hoc* repair and damage limitation resulting from hazards such as increasingly intense floods.

ACKNOWLEDGEMENTS

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GROUNDWATER UTILIZATION AS ADAPTIVE CAPACITY TO PUBLIC WATER SUPPLY SHORTAGES

Case Study of Two Cities in Khorezm, Uzbekistan

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Abstract: While the public water supply organization Suvokava claims that there are no problems of public water supply to the urban households in the Khorezm province of Uzbekistan, water shortages occur, particularly during the summer. During this period, urban households utilize groundwater as an alternative source to the public water supply. The data suggest that there are two common pool resource problems: first, between the agricultural and the urban sector; and secondly between upstream and downstream cities.

Key words: Khorezm Province, Uzbekistan; water shortages; common pool resource; adaptive capacity; allocative efficiency; public water supply; private water supply; socio-economic issues.

1. INTRODUCTION

While living and working in Khorezm (Uzbekistan) from 2003 to 2004, it became evident that the water supply to the urban areas is not always without problems. Questioning of the water management organization, Khorezm Suvokava, which is responsible for supplying urban areas with drinking water, prompted the reply that there were neither problems with the sufficient supply of water or with the infrastructure of water delivery. Their recent infrastructural work focuses on the installation of a sewage system in the urban areas and the installation of water meters. Water meters are installed to change from a water supply to a water demand management

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and to change the pricing system from fixed prices per household to flexible prices according to water utilization. Current international projects (such as the German Agro Action) focus only on the drinking water supply to rural areas, but not to urban areas. Other work is conducted on the quality of the public water supply (Froebrich and Kayumov, 2002). During the winter season it was observed that there were only a few shortages of water delivery, and in spring and summer the shortages increased. Hence, it seemed that the problem was not based on a failure of the infrastructure but on water shortages.

The research presented here has been conducted in Khanka and Khiva city in Khorezm Province, Uzbekistan in May to June 2004. Fourteen to sixteen households have been questioned in four different mahallas (urban units) in each city. A total of 120 households have been questioned in structured interviews about their current public water supply utilization, the water supply provided by Suvokava, the costs for water, and their strategies for coping with deficiencies. There are two problems with the survey. First, conducting the survey during the summer months might have had a negative influence on the responses of the interviewees. Second, because only 120 interviews were conducted in eight different mahallas in two cities, the results of the survey are not representative. Nevertheless, the results will give an indication about the current situation.

The paper is divided into four sections. The first section gives a brief introduction to the concept of adaptive capacity and the concepts of water supply and water demand management. The paper continues with a brief geographical background to the region, the water allocation to the different sectors as well as the current claims and future plans of the water management organization, Khorezm Suvokava. The third section presents and discusses the findings of the survey. The final section concludes that the urban population adapted to the considerable shortages of public water supply with an increase in the utilization of alternative water sources, mainly groundwater. Second, it is suggested that the shortages in public water supply can be prevented by implementing two strategies, namely end-use efficiency and allocative efficiency.

2. THE CONCEPT OF ADAPTIVE CAPACITY

The concepts of ingenuity (Homer-Dixon, 1995) and adaptive capacity (Ohlsson, 1999) can be used interchangeably. Both concepts are based on the same understanding; namely that a high level of ingenuity/adaptive capacity enables change and reduces transaction costs, improves the capacity to deal with changes, uncertainty, and crises, whether the crisis is institutional, political, economic, or environmental.

In terms of water management, the concept is used to explain how a group or society utilizes adaptive capacity to deal with natural resource scarcity. A distinction can be made between supply and demand management of water resources. While the emphasis of supply management is to meet the current demand or even future demands, the emphasis of demand management is to reduce consumption by increasing efficiency in use (compare Morris 1997, p. 230). Ohlsson and Turton (undated) argue similarly. They state that the objective of supply management is to "get more water" and the objective of demand management is to increase the "use per drop" (Ohlsson & Turton undated, p.2). While supply management increases the available amount of water resources to the end-user, demand management applies water saving strategies and technologies to reduce the amount required. Supply management does not affect the current amount of water utilization of the end-user; therefore it is often more feasible to implement than demand management of water resources.

In terms of water management, to increase the supply as a reaction to uncertainty is not always environmentally sound. Possible decreases in demand are mainly dependent on technological and economic feasibility. Allan (1998) points out that instead of focusing mainly on end-user efficiency, it is important to focus on allocative efficiency. Allocative efficiency allocates water away from one sector and towards another. An example of allocative efficiency is the decrease of the water allocation to the agricultural sector and the reallocation to the urban or industrial sector. Allocative efficiency is technically simple but socially and politically extremely complex.

3. BACKGROUND

The Khorezm Province is located in the northwest of Uzbekistan and is in the lower part of the Amu Darya Basin. Khorezm (Uzbekistan), Dashkhovuz (Turkmenistan), and Karakalpakstan (Uzbekistan) all receive water from the Tuyamuyun reservoir. The reservoir provides drinking and irrigation water for the lower Amu Darya region. The reservoir is located within Turkmenistan. It is upstream of the Amu Darya delta, 450 km south of the former Aral Sea shore line. According to a representative of the Amu Darya Basin Organization (BVO), 0.7 km^3 of the annual flow of the Amu Darya is reserved for the urban population and industries of that region (3 million inhabitants). The bulk of the river flow is allocated for irrigation purposes in the agricultural sector, and no water is allocated to the environment (Aral Sea) (Wegerich, in press). Figure 1 indicates the sector water use in Uzbekistan.

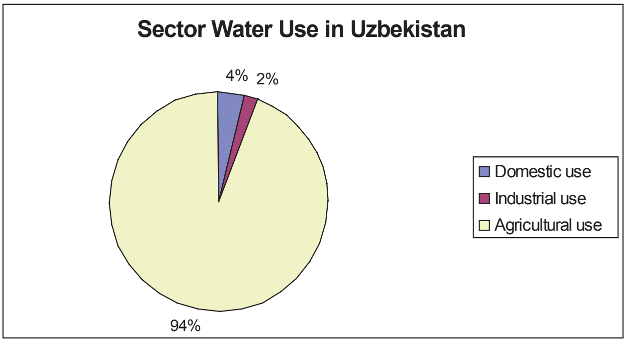

Figure 1. Sector water use in Uzbekistan (in percent) (after DFID 2001, p.17).

One organization, the Khorezm Suvokava, is responsible for the public water supply to the urban centres in Khorezm Province. According to the corporate development plan (CDP) of Khorezm Suvokava, the total water supply coverage of the urban areas was 96 % in 2003, and it was anticipated that the coverage would increase to 98 % in 2004. The CDP states that there were only 5 water supply interruptions in 2003, and it was anticipated that the number of interruptions would be reduced to 3 by 2004. In addition, the plan states that in 2003 the reaction time following a pipe burst was shorter than 4 hours, and that it was anticipated that this would be decreased to below 2 hours in 2004. The CDP was developed in cooperation with a World Bank project; after the termination of the project, the CDP has been continuously updated.

From the data in the CDP, Khorezm Suvokava suggest that the organization can provide sufficient water supply to the cities in Khorezm Province, and that interruptions of the public water supply are negligible because of their small number. Also the reported short times to respond to interruptions implies that there are no shortages of public water supply.

4. PUBLIC WATER SUPPLY SHORTAGES

The CDP of the Khorezm Suvokava portrays a picture of an organization which is able to supply the urban population in Khorezm Province sufficiently with water. Nevertheless, the findings of the survey carried out in the project reported here suggest that the data in the CDP do not reflect the real situation.

Of the eight mahallas considered during the survey, one to four are within Khanka city, and five to eight are located within Khiva city. The survey data on supply shortages, as summarized in Table 1, suggest that there is a seasonal variation in public water supply. According to the responses of the interviewees, there are few shortages of public water supply during the spring but many during the summer months. None of the interviewees stated that there were shortages during the autumn and winter season. Repeatedly it was stated in Khiva city that during the summer season water is not supplied for three to four days at a time.

Table 1. Seasonal variations in public water supply.

Mahalla	Access to public water supply (season)					
	Spring worst	Summer worst	Autumn worst	Winter worst	No difference between seasons	Total answers
	Khanka					
1	0	12	0	0	2	14
2	4	8	0	0	6	14
3	0	5	0	0	11	16
4	1	10	0	0	6	16
	Khiva					
5	2	7	0	0	8	15
6	5	11	0	0	4	15
7	0	14	0	0	1	15
8	4	13	0	0	2	15

The main irrigation phase in Uzbekistan is during the spring and summer season. The data of Table 1 suggest that the urban sector is in competition with the agricultural sector for water supply. The Tuyamuyun reservoir stores water for both sectors. It seems that there is a strong emphasis on supporting the agricultural sector with water, instead of supplying the urban population sufficiently.

The interviewees were also questioned on how many hours per day the individual households have access to public water supply. The responses, given in Table 2, indicate that there is a high level of water shortage and shortcoming in the service provision of Khorezm Suvokava. While in the CDP it was stated that there were only 5 interruptions per annum, the data sets of Table 1 and 2 suggest that the interruptions can be on a daily basis, especially during the summer season. It seems that the respondents answering to the question of hourly supply were referring mainly to the summer season, rather than to a yearly average.

As the data of Table 2 indicate, there are vast differences between the two cities. Khanka, which is closer to the Amu Darya and also closer to the Tuyamuyun reservoir, has on a daily basis more access to public water supply than Khiva, which is closer to the desert and more distant from Tuyamuyun. However, there is no other confirming evidence that the distance from Tuyamuyun is a cause for the variation in performance of the Khorezm Suvokava and that the closer to the reservoir the higher the level of public water supply. In addition, there seems to be significant differences in the responses from different mahallas within Khiva. These differences cannot be explained by the seasonal variation, but would have to be explained by management shortcomings in sustaining the infrastructure and responding to infrastructure breakdowns. However, there could be also a second explanation: water is distributed according to the first-come-first-served principle. Hence, over-utilization ‘upstream’ leads to water shortages ‘downstream’ in the public water supply system. This would indicate that the public water supply faces the problem of a common pool resource and is in need of stronger regulations in terms of share allocation and share utilization.

Table 2. Access to public water supply (in hours per day) (probably in summer only).

	Counts on amount of hours per day of public water supply					
Mahalla	1-2 hours	2-4 hours	4-6 hours	>6 hours	No supply	Total answers
	Khanka					
1	0	0	1	13	0	14
2	0	0	4	9	1	14
3	0	0	2	14	0	16
4	0	0	2	11	3	16
	Khiva					
5	1	1	7	6	0	15
6	1	4	2	8	0	15
7	7	2	3	2	1	15
8	9	2	1	3	0	15

5. COPING STRATEGIES FOR DEALING WITH THE PUBLIC WATER SUPPLY SHORTAGES

Does the urban population rely on the public water supply as the only supplier of fresh water? The data of Table 3 indicate that the respondents in the Khanka mahallas rely more on the public water supply than the respondents in the Khiva mahallas. Sixteen out of sixty respondents in Khanka city stated that they rely only on the public water supply (i.e. in

Table 3, tap in house, tap in yard, and shared tap). In Khiva, only 3 households relied completely on the services of Khorezm Suvokava. However, in some locations, there may be no useable water source other than the public supplies. For example, one of the respondents in Khiva city pointed out that the groundwater in the yard is saline, and therefore cannot be utilized as drinking water. The water quality of the private hand pumps and open wells is not controlled. Hand pumps and open wells are not utilized for drinking water when the users feel that the water is too salty.

Table 3. Utilization of only one water source for drinking water in households (public water supply includes: tap in house; tap in the yard; and shared tap).

	Utilization of only one source for drinking water in households				
Mahalla	Tap in house	Tap in the yard	Shared tap	Open well	Hand pump
	Khanka				
1	5	0	0	0	0
2	4	0	0	0	1
3	2	0	3	0	0
4	1	1	0	1	2
	Khiva				
5	0	0	1	0	0
6	1	0	1	0	0
7	0	0	0	1	0
8	0	0	0	0	0

Tables 4 and 5 show the different coping strategies for the households questioned. The main strategy is to rely on one public water supply source (either private tap or shared tap) and on hand pumps in case of public supply deficits. Only six households relied on two different services supplied by the Khorezm Suvokava, namely private tap and shared tap. Remarkably, four out of these six households are located in Khiva city, which is worse off in terms of public water supply.

Table 5 indicates that in Khiva city the respondents are relying more often on three sources for drinking water supply. This attitude seems to be especially important in Mahalla 8. Comparing the data of Table 5 with the data of Table 1 and 2, it appears that Mahalla 8 is the most severely affected by the public water supply shortages. This could be an explanation for the reliance on more than two sources. Altogether, the data confirm on the one hand that there are regional differences in public water supply (see Table 1 and 2), but also that there are different responses to the degree of uncertainty of the public water supply.

Table 4. Utilization of two water sources for drinking water in household.

Mahalla	Utilization of two water sources for drinking water in households								
	Tap in house & hand pump	Tap in house & shared tap	Tap in the house & open well	Tap in the house & other	Tap in yard & hand pump	Shared tap & hand pump	Shared tap & open well	Cistern & hand pump	Open well & other
	Khanka								
1	8	0	0	0	1	0	0	0	0
2	4	1	0	2	1	1	0	0	0
3	0	1	0	0	0	8	0	0	0
4	11	0	0	0	0	0	0	0	0
	Khiva								
5	10	0	0	0	1	0	0	0	0
6	1	2	0	0	0	7	0	0	0
7	6	2	2	2	0	0	0	0	0
8	1	0	2	0	0	1	2	1	1

Table 5. Utilization of three water sources for drinking water in households.

Mahalla	Utilization of three water sources for drinking water in households						
	Tap in house, shared tap & hand pump	Tap in the house, open well & hand pump	Tap in house, hand pump & electric pump	Tap in the house, open well & other	Tap in house, shared tap & open well	Tap in yard, shared tap & hand pump	Shared tap, open well & other
	Khanka						
1	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0
3	2	0	0	0	0	0	0
4	0	0	0	0	0	0	0
	Khiva						
5	1	1	1	0	0	0	0
6	1	0	1	0	0	1	0
7	1	1	0	0	0	0	0
8	0	3	0	1	2	0	1

So far the data presented give no indication of the amount of water which is utilized from alternative sources. These kinds of data would be important to see, in particular whether the demand for water changes with the degree of water shortage. Answers of the interviewees on how much water they utilize from alternative sources are summarized in Table 6.

Given the data presented above, it is not surprising that the daily amount utilized in Khiva city is greater than the amount utilized in Khanka city. However, it is surprising that even though Khanka city has a better public water supply, eleven households stated that they utilize more than 50 litres per day from alternative sources. In Khiva, only thirteen respondents claimed to have such high uses.

Going back to the approach of supply and demand management, and focussing only on the last column of Table 6, it seems that the households in Khiva adapted more to the water scarcity by utilizing less water from the alternative water source than the households in Khanka. This would indicate that the generally smaller supply of water leads to lower water use. However, because there are no data available concerning the total water use per household, this is only an assumption, which needs to be validated. Even the CDP of the Khorezm Suvokava gives no indication on the average utilization of households. The current billing system charges per household instead of actual water use, and therefore there are no figures available.

Table 6. Amount of water obtained from alternative sources in litres/day (?/NA = do not know / not applicable).

	Amount of water obtained from alternative sources						
Mahalla	< 10 l/d	< 20 l/d	< 30 l/d	< 40 l/d	< 50 l/d	> 50 l/d	? / NA
	Khanka						
1	1	2	3	0	1	3	4
2	0	2	3	1	0	1	7
3	0	0	1	3	1	5	6
4	0	0	3	1	2	2	8
	Khiva						
5	0	3	4	2	2	2	2
6	0	2	4	0	1	2	6
7	0	2	1	2	3	6	1
8	1	3	1	2	2	3	3

At the current time, the majority of the households are charged a fixed rate. According to the respondents, the amount charged is approximately US$ 1.2 per household per month. Khorezm Suvokava's CDP claims that in 2003 40 % of the households had water meters; however, in the study area, water meters were the exception. When metered, the current price charged to households is US$ 0.05 for 1 m^3. Taking only into consideration the supply shortages of the current system, water meters seem to be the more cost effective option for the customers.

The current prices for water do not represent the real costs of water delivery. However, it is likely that any increase in water rates to reflect the real price would increase groundwater utilization. This assumption is confirmed by Rogers *et al.* (2002), who argue that “proper water rates imposed by the public water supplier alone are insufficient to manage the resource. If the suppliers charged the full cost of water and there were no regulations to protect groundwater sources consumers will gradually exploit the groundwater sources” (Rogers *et al.* 2002, p.12).

In the CDP of Khorezm Suvokava it is claimed that the response time to complaints is less than two hours and that there are only 5 complaints per month per 10,000 customers. While in the mahallas in Khanka nearly 50 % of the respondents confirmed that Khorezm Suvokava responded to complaints within two to three hours, in Khiva the majority of the respondents stated that Khorezm Suvokava does not respond at all to the complaints of customers. The higher level of service in Khanka could explain why more customers rely only on Khorezm Suvokava for their drinking water supply. The greater service provision by the organization results in less initiative by the customers to find alternative water sources.

6. PERCEPTION OF THE CAUSES FOR THE WATER SHORTAGES

When the customers were asked about the reasons for the water supply shortages, the majority of the interviewees in Khiva stated that the water shortages are related to the usage of water for the agricultural sector. Only a few blamed the shortages on the conditions of the infrastructure of the water supply system. On the other hand, in Khanka the shortages were blamed equally on the usage of water in the agricultural sector, problems with the infrastructure, and on households using the public drinking water supply for watering their gardens. The answers of the interviewees in Khiva confirm the data presented in Table 1 on the seasonal variation of the water supply. As shown above about one third of the respondents in Khanka stated that there are no seasonal variations in terms of water supply (compare Table 1). The responses indicate that the water supply failure does not arise purely from the competition between water supply to agricultural and urban sectors. In addition, the collected evidence shows that the households in Khanka see their own mismanagement as a reason for the shortages. The technique used when watering gardens is the same as that applied in the agricultural sector, namely flood and furrow irrigation. This watering method has low water efficiencies.

It is surprising that groundwater is hardly used for watering gardens, tap water being the usual source (Table 7). The explanation could be that using hand pumps for flood and furrow irrigation would be very labour intensive. As stated above, currently water supply is not metered and the current charges for tap water do not reflect the real costs. The continuation of installing water meters and a rise in water prices, could have the affect that households either utilize tap water more efficiently in their gardens or switch to groundwater for irrigation. On the other hand, agricultural water should have a lower salinity than drinking water; therefore it is questionable if the groundwater could be used for watering gardens. Table 6 shows that there are hardly any differences between Khanka and Khiva in terms of using tap water for watering gardens. However, in Khanka, irrigation channels are utilized as often as tap water for watering the gardens, but in Khiva irrigation channels are not utilized at all. While in Khanka it was acknowledged that watering the gardens with tap-water leads to public water supply shortages, this was not acknowledged in Khiva, where the uncertainty of water delivery is much higher. Assuming that no raising of public awareness has taken place, it can be suggested that having a feasible alternative (irrigation channels) leads to this recognition.

Table 7. Drinking water used for gardens (? / NA = no reply / not applicable).

	Water source used in watering gardens				
Mahalla	Tap water	Ground-water	Cistern	Irrigation channel	? / NA
	Khanka				
1	7	0	0	7	0
2	6	0	0	6	4
3	1	0	0	0	15
4	6	0	0	8	2
	Khiva				
5	9	1	0	0	5
6	8	3	0	0	4
7	2	0	0	0	13
8	2	2	1	0	10

7. CONCLUSION

The collected data demonstrate that there are considerable shortages in the public water supply. It is evident that the response to the public water supply shortages has lead to a search for alternative water sources. Hence, a different supply management approach was introduced. The objective was to "get more water" or at least to get a sufficient amount of water. It seems likely that the public supply shortages lead also to an adaptation to the water shortages, and therefore to a decrease in use. However, there is no independent confirming data to validate this suggestion.

Neither the quality nor the quantity of utilized groundwater is regulated. High fertilizer and pesticide use in the agricultural sector, an inefficient sewer system and limited regulation of industrial pollution, hold dangers for groundwater pollution. As groundwater in Khanka and Khiva is mainly used as drinking water, there is an urgent need to control its quality.

The collected data leave space for further analysis. Even though it is evident that water shortages occur, it is not evident what the cause of the shortages are. The survey shows that water shortages occur mainly during the irrigation season during the summer; this would suggest that the shortages are not based on problems with the infrastructure, but with problems of the amount of water available. There could be two explanations. First, the amount of water allocated to the urban sector is too small, and does not satisfy all the needs of the urban sector. Second, the urban sector utilizes a large amount of the public water supply for uses which might not have been anticipated, such as watering of gardens, and therefore itself causes supply shortages. Going back to the theory, the case study shows that there is the need for two sorts of adaptive capacity. On the one hand, end-use efficiency could be improved: change in consumer demand for water could be achieved through installation of water meters and increasing prices. This will have the consequence of more efficient use of tap-water, and therefore a consequent increase in the overall supply of tap-water. On the other hand, allocative efficiency could be imposed, as the current demand for public water outstrips its supply. It is questionable whether the focus on end-use efficiency will increase the available water to such an extent that it satisfies the needs of the urban households. Hence, there must be a readjustment of the relative amount of water allocated to the agricultural and urban sectors. Taking into the consideration the huge inefficiencies which are current in the agricultural sector, small changes towards end-use efficiency in the agricultural sector should allow sufficient supply to the urban sector.

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