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Recharge systems
for protecting and enhancing
groundwater resources

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Preface

The principle objective behind UNESCO's International Hydrological Programme (IHP) has always been to develop a solid scientific and technological base for a sound management of water resources. Both aspects of quantity and quality of water resources are being addressed in the programme, while at the same time taking into account protection of the environment.

The availability of quality freshwater resources is a decisive factor for sustained socio-economic development. Now, more than ever, it is this solid scientific knowledge base that is required to support political and economic strategies on which the future of mankind may well depend. Management Aquifer Recharge (MAR), in particular, is a flexible model that can be applied on a number of different levels: from the smallest check dams to schemes supplying some of the largest cities in the world with populations in their millions. MAR is being applied extensively through traditional techniques in many parts of the world; an accumulated set of experiences from which we can still learn. However, irrespective of the level of sophistication at which MAR is applied, scientific research is important to evaluate the performance of different MAR systems under different conditions.

We should bear in mind the importance of groundwater in sustainable integrated water resources management, especially in arid areas, where it forms the only source of fresh water. The IHP works with partners to compile inventories, develop guidelines and publish good experiences to support the sound use and careful protection of this precious resource. The use of MAR has opened up a variety of possibilities to extend inadequate groundwater resources by increasing the rate of groundwater formation or by using the capacity of soil to improve the quality of water. Particular focus should be given to the dissemination of information regarding practices, such as aquifer recharge using treated water that can be employed to increase the availability of groundwater resources. There are many uses for MAR as a means of improving the quality of our water. Increased experience in the field of water reuse will undoubtedly help us expand water resources, while avoiding any compromise to individual health.

The series of MAR symposia has established itself as an important mechanism by which to report on state-of-the-art techniques in managing aquifer recharge. The 5th International Symposium on Managing Artificial Recharge of Groundwater (ISMAR-5) in Berlin, June 2005, was a welcome and timely event that allowed us to review advancements in such techniques made in the three years following the 4th International Symposium on Artificial Recharge of Groundwater (ISAR-4), held in Adelaide in September 2002 and also endorsed by the IHP.

I wish to thank the organizers of the symposium for their tremendous effort in contributing to IHP and congratulate them on the symposium that was both geographically extensive and thematically wide-ranging.

Andras Szöllösi-Nagy
Secretary
International Hydrological Programme
United Nations Educational,
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**River/lake bank filtration
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**Alternative recharge systems:
subsurface dams, rainwater harvesting,
percolation tanks, ...**

Dynamic processes during bank filtration and their impact on raw water quality

P. Eckert, H.P Rohns and R. Irmscher

Abstract

Since 1870 Riverbank filtration (RBF) at the river Rhine, Germany has been used successfully by the Düsseldorf waterworks as the first step for treating drinking water. The production wells discharge raw water from a quaternary aquifer with a proportion of 50 to 90% of bank filtrate. In order to achieve a profound knowledge of the purification processes during bank filtration a research program was performed in the year 2003/2004. The purification processes have to be well understood to design adequate treatment steps and to define specific target values on river water quality.

Temporal changes of river water quality and hydraulics influence the natural purification processes during bank filtration. It was possible to show that the annual changes of the river water temperature trigger a string of subsequent reactions within the aquifer. Flood events are accompanied by shorter travel times and a less effective natural purification. However, even during flood events and during extreme low water, the multi protective barrier concept including both natural and technical purification has proven to be a reliable method for drinking water production.

Keywords

Riverbank filtration; hydrochemistry, modelling.

INTRODUCTION

The city of Düsseldorf is situated in the North-West of Germany, in the lower Rhine valley. The Düsseldorf waterworks supply the 600,000 inhabitants with treated bank filtrate. A multi protective barrier concept ensures the constant production of high quality drinking water. Natural attenuation processes during riverbank filtration (RBF) form the first and efficient protective barrier. The subsequent protective barrier is the raw water treatment including ozonation, biological active filtration and active carbon adsorption.

Basic studies on hydrogeochemical processes during RBF were performed in the 1980s at Swiss rivers (Schwarzenbach et al., 1983; Jacobs et al., 1988; Von Gunten et al., 1991). Mass balance of oxidised organic carbon was calculated by Denecke (1997) at the Rhine and by Grischek et al. (1998) at the River Elbe. Previous studies at the Düsseldorf site revealed the hydraulic conditions and the balancing out of fluctuating river water concentration during RBF (Schubert 2002a).

This paper presents the latest investigations with a special focus on hydrochemical reactions linked to the dynamics of RBF. The assessment of the purification processes requires the consideration of the dynamic character of riverbank filtration, which is linked to the varying chemical composition of the river water, the discharge of the river and therefore the hydraulic conditions within the aquifer. The investigation period includes the extreme low water event in the summer of 2003 and the following flood event.

METHODS

In order to assess the purification processes during riverbank filtration the monitoring of the river water together with the groundwater between the river and the production well has been successfully applied at various sites (Grischek et al. 1998, Schubert 2002b, Golwitz et al. 2003). In general the obtained groundwater consists of nearly 100 % infiltrated river water. Therefore, the observed groundwater chemistry is influenced only by the river water chemistry and the hydrogeochemical processes within the aquifer. At the test site two multi-level wells are situated between the river Rhine and the production well (Figure 1). The monitoring program includes the chemical as well as biological parameters relevant for drinking water quality. The concentration of almost all important chemical parameters in the river water show a significant temporal variation. In addition it was assumed that the hydrochemical process were not at steady state. Therefore, the monitoring expense had to consider these temporal variations. Because of the complexity of the hydrogeochemical processes during RBF the 1D-reaction transport model PHREEQC-2 (Parkhurst and Appelo, 1999) was applied to evaluate the obtained data. Hydrogeochemical transport modelling was performed successfully by van Breukelen et al. (1998), for ascertaining the validity of the proposed reaction scheme during Rhine water infiltration in the dunes of the Amsterdam water supply.
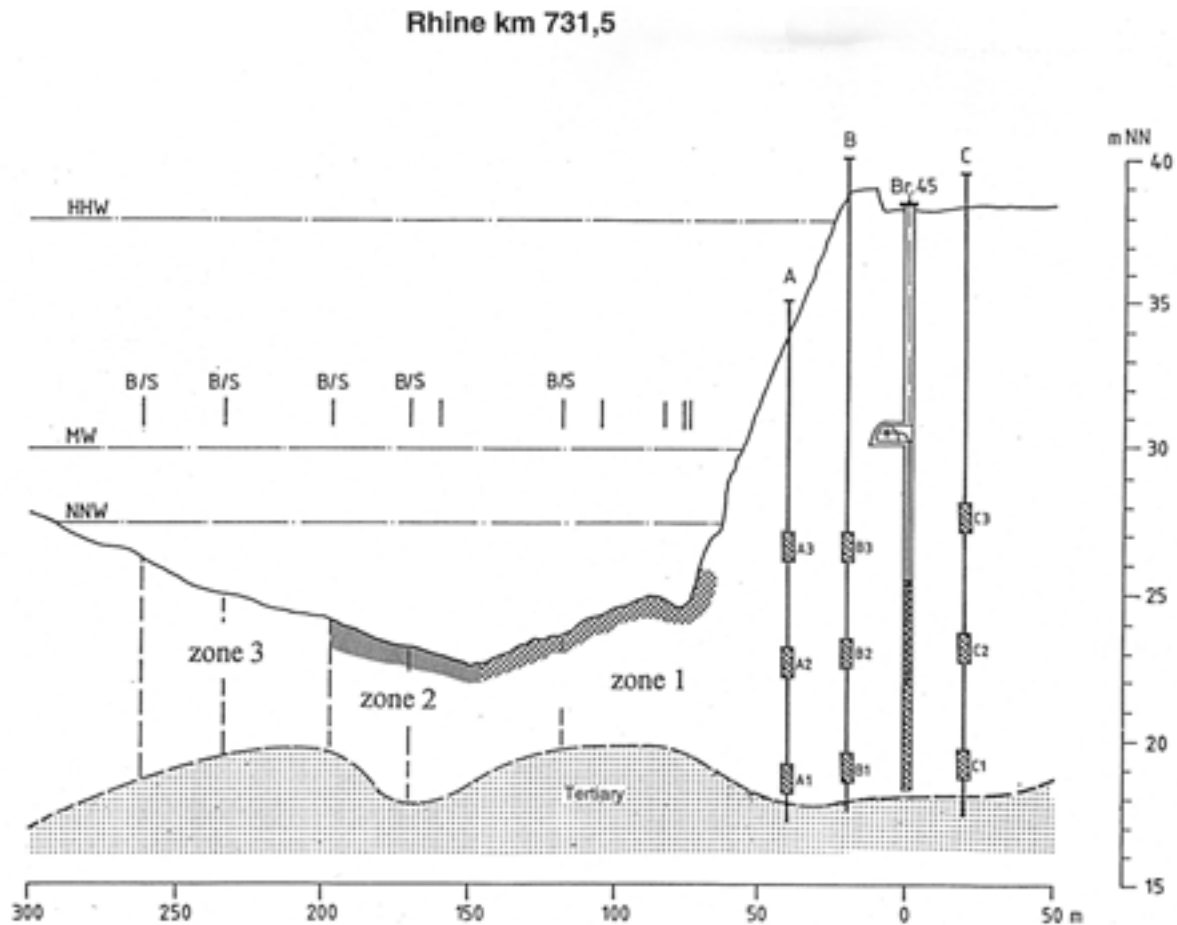
Figure 1. Cross-Section through the aquifer and the River Rhine showing the production well Br.45 and depth oriented monitoring wells A,B and C. Zones 1 to 3 indicate the different composition of the riverbed

RESULTS AND DISCUSSION

During the investigation period the temperature of the bank filtrate never exceeded 20 °C, while the river water showed several weeks during the summer, temperature values higher than 25 °C (Figure 2). The absorption capacity of the aquifer is obvious by comparing the measured temperatures of the bank filtrate with the associated conservative model results. A good agreement between the modelled and measured values was obtained by considering

diffusive heat transport within the model. The calibration of the heat transport parameters enabled the modelling of further hydrochemical processes linked to temperature.

The absorption capacity versus the temperature of the aquifer has a significant impact on the drinking water. The mixing with the landside groundwater which showed a constant temperature of 13 °C leads to a raw water temperature less than 17 °C. Despite the exceptional long period with high river water temperatures in the summer of 2003 the obtained drinking water remained cool and fresh.
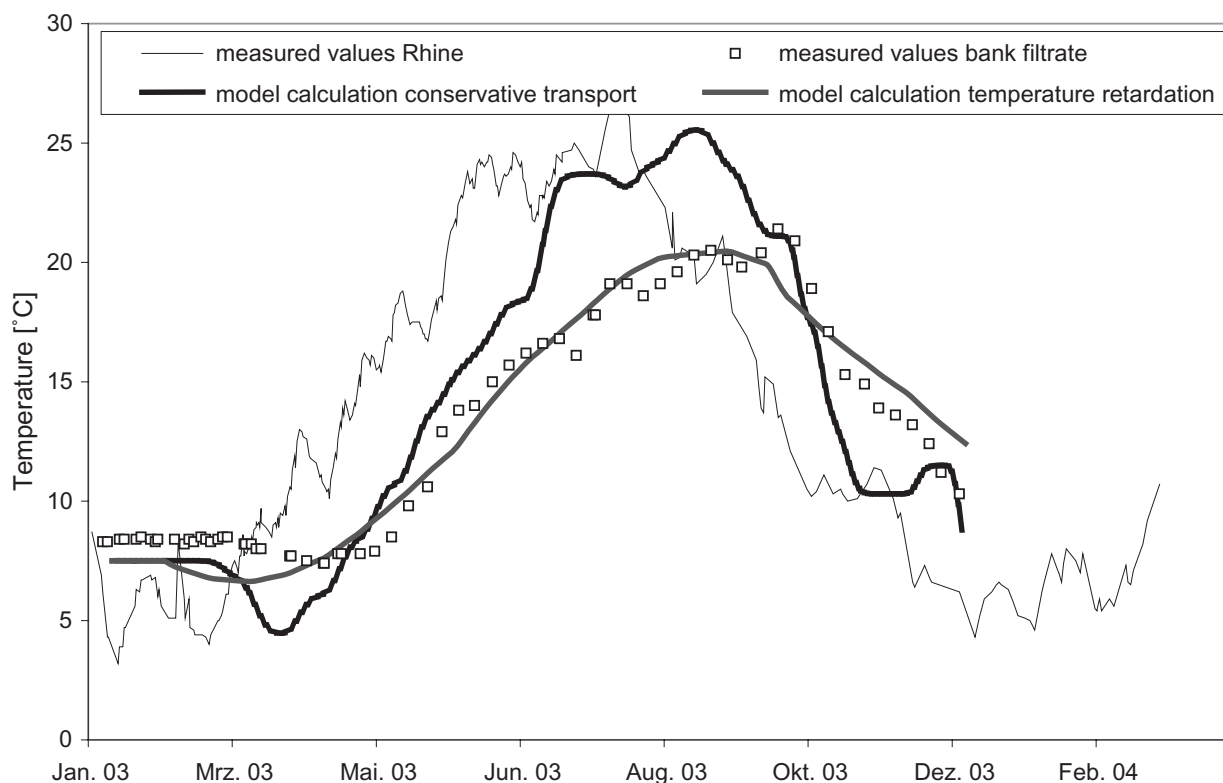
*Figure 2. Comparison of temperature in the river water, the bank filtrate,
conservative and reactive model calculation*

The yearly changing river water temperature has a direct influence on the oxygen concentration of the river water. While during the winter in the cold river water the oxygen concentration ranged between 11 and 13 mg/l, the concentration decreased to 7 mg/l in the summer due to lower solubility of oxygen in the warm river water (Figure 3). Oxygen consumption is induced by aerobic biodegradation of organic substances during bank filtration. The disagreement of the measured oxygen values of the bank filtrate with the model results while considering a fixed consumption of 5 mg/l oxygen, indicates a more complex reaction scheme between May and December. The model fitting process revealed that the microbial oxygen consumption is a function of the temperature and the oxygen concentration of the infiltrating river water. During spring time the biological activity increased with the rising temperature leading to maximum oxygen consumption of 11 mg/l. Despite of still increasing temperature during the summer the microbial activity decreased again. Obviously the lower oxygen concentration became the limiting factor. Nevertheless, the biological activity was so high that anaerobic conditions appeared within the aquifer over a period of nearly three months.

The changed redox conditions are of particular interest for the drinking water treatment. During the anaerobic period additional micropollutants were already degraded within the aquifer (Schmidt et al. 2004). Only a part of the low concentrated nitrate (7–10 mg/l) in the infiltrated river water was reduced. Because of the incomplete denitrification no dissolved iron or manganese appeared in the raw water.
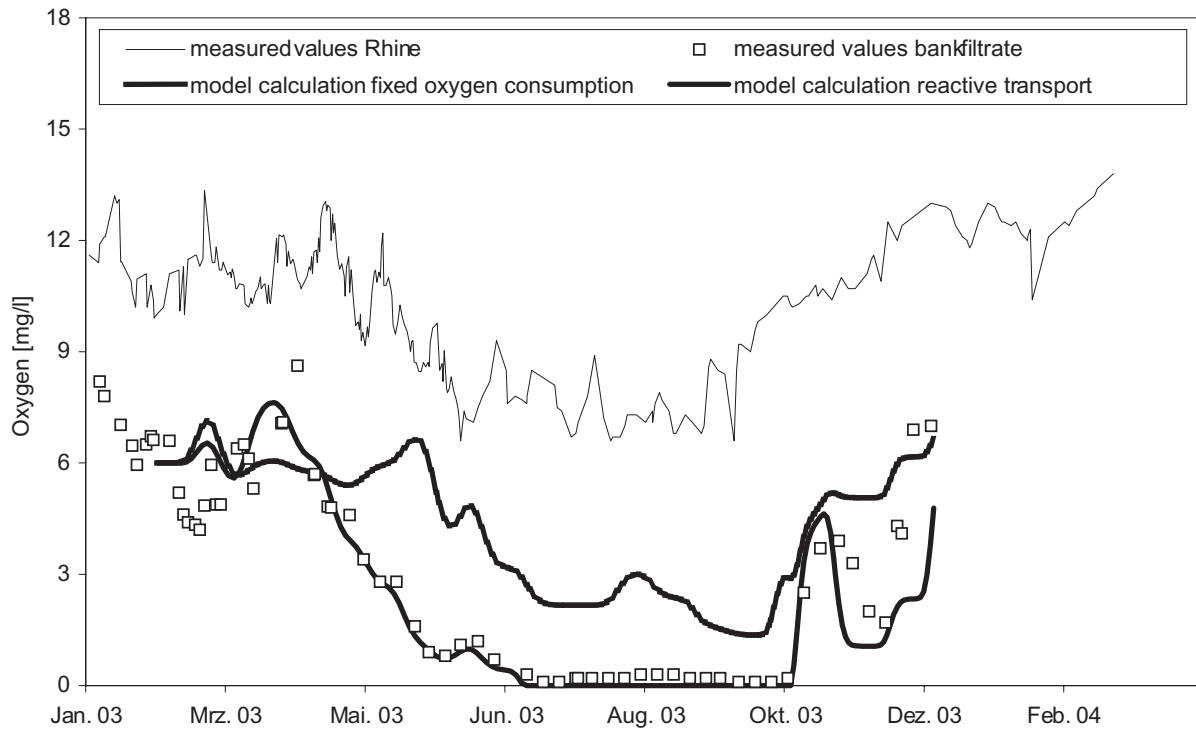
*Figure 3. Comparison of the oxygen concentration of the river water,
of the bank filtrate, the conservative and reactive model calculations*

The purification capacity of riverbank filtration is obvious in the degradation of total organic carbon (TOC). Between February and December 2003 the TOC concentration of the infiltrating river water was decreased to a level of 1 mg/l (Figure 4). Despite varying TOC concentrations in the river water between 2 and 4 mg/l the observed concentrations in the bank filtrate remained stable at a level of 1 mg/l. This suggests that the above described varia-
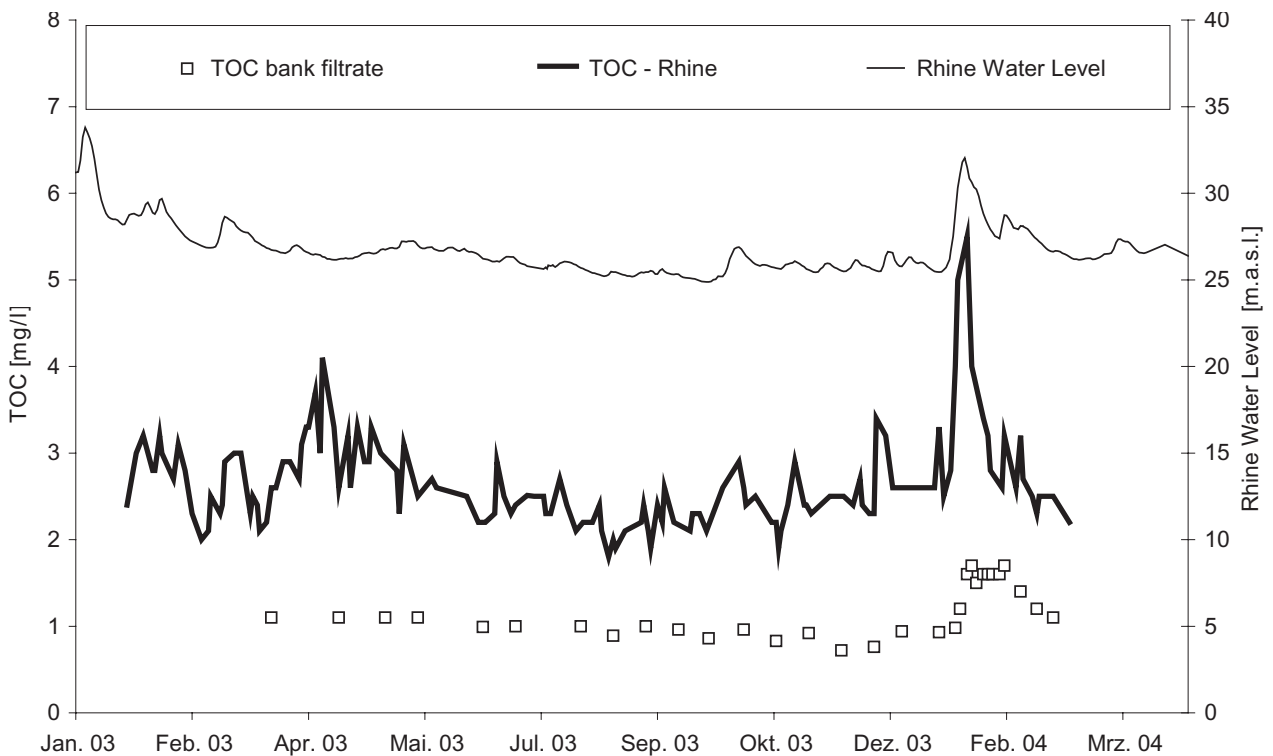
*Figure 4. Rhine water level and the total organic carbon (TOC) concentration
of the river water and the bank filtrate*

tions of oxygen consumption during RBF coincide with the varying TOC concentrations in the infiltrated river water. Obviously, the microbes are able to degrade a certain range of organic carbon flux. An increase of organic carbon in the bank filtrate appeared only following the flood event in January 2004. During the flood event the increase of the TOC concentration coincides with an increased hydraulic gradient within the aquifer. Therefore, the mass flux of TOC in the infiltrating river water exceeds for a short time the microbial degradation capacity. Based on the depth-orientated sampling the most vulnerable parts of the aquifer were detected in the high permeable gravel layer and in the upper part of the aquifer which was unsaturated prior to the flood event.

A more intense technical treatment of the raw water was induced following a flood event. Besides the increased TOC-concentration, colony counts are detectable over a period of some days (Irmscher and Teermann, 2002; Schubert, 2002b). The subsequent technical treatment including also oxidation and desinfection ensures at all times a high drinking water quality.

CONCLUSIONS

The temporal variations of the river water composition are obvious in the parameters dissolved oxygen, total organic carbon and temperature. Coupled with the extreme low water of the river Rhine was an extended period with high temperatures above 25 °C. Due to heat exchange during RBF and the mixing with groundwater the temperature of the raw water in the production well never exceeded 17 °C. The temperature increase during spring leads at first to a more efficient biological activity within the aquifer. During the summer the biological degradation of organic carbon is then limited by the decreasing oxygen concentration of the infiltrating river water. The microbiological activity has, together with the varying composition of the river water, a significant impact on the quality of the raw water. During stable hydraulic conditions the infiltrated organic carbon was decreased to a value of only 1 mg/l. Over a period of 12 weeks anaerobic conditions were observed combined with an increased degradation of micro-pollutants. While mostly the raw water already fulfills the European Drinking Water Standard, elevated colony counts were observed in the production wells following the flood event.

Temporal changes of river water quality and hydraulics influence the natural purification processes during bank filtration. They have to be well understood to design and maintain adequate treatment steps and to define specific target values on river water quality. Even during extreme low water and during flood events, the multi protective barrier concept including both natural and technical purification has proven to be a reliable method for drinking water production.

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Sustainability of riverbank filtration in Dresden, Germany

T. Fischer, K. Day and T. Grischek

Abstract

Since 1875, bank filtration along the River Elbe in the city of Dresden has been an important source for public and industrial water supply. Infiltration has been induced by pumping of wells and installation of drain pipes. Today, some of the old systems are still in operation. Periods of poor river water quality in the 70s and 80s have been overcome. Available historical data from water level measurements, clogging investigations and DOC analyses are compared with results from recent investigations. Waterworks were modernized and treatment technologies adapted.

Keywords

Bank filtration, Elbe River, Dresden, clogging, sustainability.

INTRODUCTION

Dresden is the capital of the county of Saxony in the eastern part of Germany with a population of nearly half a million people. The city is situated in a rift valley along the Elbe River, which is mainly filled with glacial deposits consisting of gravels and coarse sands. Under normal conditions, these Quaternary deposits form an unconfined aquifer as much as 15 m thick. The aquifer is partly overlain by a layer of meadow loam, 2–4 m thick. The deeper deposits are marl (Turonian formation), which have a maximum thickness of about 250 m beneath the city center. A deep aquifer beneath the marl is formed by Cretaceous sandstones (Cenomanian) and is partly artesian (Grischek et al., 1996). The sandstones crop out at the southwestern boundary of the city whilst the northern boundary is formed by the Lusatian overthrust (granitic massif).

The Quaternary aquifer is in direct hydraulic contact with the Elbe River. In Dresden, the mean flow of the Elbe River ranges from $100 - 4,500$ m^3/s with a mean of about 300 m^3/s. In general, groundwater flows from both sides of the valley towards the river with hydraulic conductivity ranging from $0.6 - 2{\times}10^{-3}$ m/s.

WATER SUPPLY OF THE CITY OF DRESDEN, GERMANY

As in many cities in Europe, groundwater resources in Dresden have been used for public water supply to a large extent with riverbank filtration (RBF) schemes along the Elbe River. At present, public water supply in Dresden is based upon 66% surface water from reservoirs and 32% bank filtration and artificial recharge.

The first waterworks in Dresden-Saloppe was built between 1871 and 1875 on the right bank of the Elbe River. Drain pipes were installed near the riverbank to abstract raw water. Due to geological boundary conditions, more than 90% of the abstracted water is bank filtrate. Today, the waterworks is still in operation and produces up to 12,000 m^3/d for industrial water supply.

Increasing water demand at the end of the 1880s exceeded the capacity of the waterworks in Dresden-Saloppe. In 1891 the city council assigned the building officer, Salbach, to write an expert's report on the future water supply of the city. Salbach proposed to build a test well at the left bank of the river, which abstracted 4,000 m³/d in 1891. Four more wells were completed in 1893 resulting in a total water abstraction at the left bank of 20,000 m³/d. Wells were connected using a siphon pipe and a collector well. Between 1896 and 1898 the second waterworks, Dresden-Tolkewitz, was constructed however further increasing water demand resulted in the construction of four more wells and a second siphon pipe in 1901 to raise the capacity to 40,000 m³/d. In the 20th century the number of wells was again increased and the water treatment facilities improved. Between 1919 and 1928 a third siphon pipe with 39 wells was built. Figure 1 shows the final system of pipes and wells at Dresden-Tolkewitz. A significant decrease in the water demand after the reunification of Germany in 1989 allowed for the closure of the water abstraction in April 1992 in order to plan a general reconstruction of the waterworks. After intensive construction works, the waterworks Dresden-Tolkewitz was put into operation again in February 2000. The water treatment technology was modernised and included a desorption chamber to remove volatile halogenated compounds (from land-side groundwater), flocculation and filtration to remove iron and manganese, active carbon filtration to remove organics and disinfection using chlor/chlordioxide. Raw water is abstracted from 72 wells using the sustainable siphon pipe system. Three pipes connect a collector well (with a pump) with vacuum well galleries. No pumps are installed in the wells. The maximum capacity is now 35,000 m³/d. Normally, only a fraction of the full potential is tapped with a certain volume of water continuously pumped to enhance stable redox conditions in the aquifer between the river and the wells. This also ensures stable mixing ratios of bank filtrate, having low nitrate and sulfate concentrations, and land-side groundwater, which has high nitrate and sulfate concentrations.
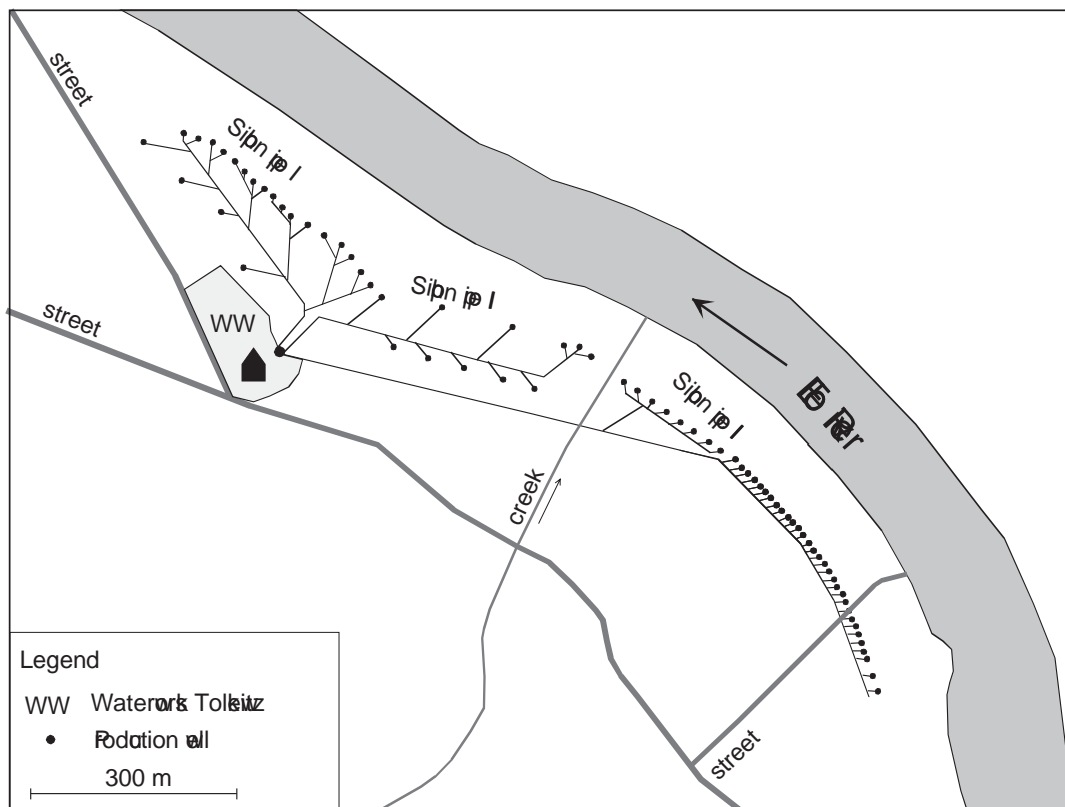
Figure 1. Location map of the bank filtration scheme of waterworks Dresden-Tolkewitz, Germany

From 1908 a third waterworks, Dresden-Hosterwitz, started operation. Whilst at the beginning bank filtration was applied, between 1928 and 1932 the main technology was shifted to artificial groundwater recharge. Pre-treated river water was infiltrated in large basins to increase the capacity to 50,000 m³/d. Increasing water demand in the

1970s induced further expansion of the infiltration basins and treatment facilities until construction works between 1983 and 1990 resulted in a final capacity of 72,000 m^3/d.

In 1946, a second raw water resource was developed when the waterworks, Dresden-Coschütz, started operation. This waterworks produces up to 120,000 m^3/d drinking water from storage reservoirs.

Now the water demand of the city of Dresden can be met either by raw water production from bank filtrate and artificially recharged groundwater or from reservoir water. These independent raw water sources are the basis for a sustainable and safe water supply. In times of low demand, the recharge basins of the waterworks Dresden-Hosterwitz are out of operation and only riverbank filtration is used.

CLOGGING ASPECTS

The proportion, and thus volume, of pumped bank filtrate strongly depends on riverbed clogging. Clogging is the formation of a layer on top of or within the riverbed which has a lower hydraulic conductivity and therefore reduces the flow rate of the filtrate through the riverbed. It is the result of the infiltration and accumulation of both organic and inorganic suspended solids, precipitation of carbonates, iron- and manganese-(hydr)oxides and biological processes. At the Elbe River in Dresden, erosive conditions in the river and floods limit the formation of a clogging layer by disturbing the riverbed via increased flow velocity and increased turbulence. At some parts in the river there is a fixed ground whilst at others gravel is dominant.

Detailed research to clogging and the stability of the clogging layer in Dresden has been undertaken by Beyer and Banscher (1976) and Heeger (1987). Due to difficulties in determining the thickness of the clogging layer, Beyer and Banscher (1976) used a term introduced as the clogging coefficient w (Eq. 1).

$$w = \frac{d_{cl}}{k_{f,cl}} = \frac{\Delta h}{v_i} = \frac{1}{L} \tag{1}$$

where w = clogging coefficient in seconds, s; d_{cl} is the thickness of the clogging layer in metres, m; $k_{f,cl}$ is the hydraulic conductivity of the clogging layer in metres per second, m/s; Δh is head drop in metres, m; and v_i is infiltration velocity in metres per second, m/s. This term is the reciprocal of the leakage coefficient L. Under specific conditions, the clogging coefficient can be calculated for RBF sites using water levels in the river and two observation wells positioned between the river and the production borehole using an analytical solution by *Girinsky* (Beims et al. 2000). Based on these levels and known pumping rates, the clogging coefficient can be determined for different river stages and measuring campaigns and be compared with former data. Furthermore, Heeger (1987) developed an empirical formula from long-term statistical analysis of water levels and related river stages and pumping rates to calculate w-values.

Between 1914 and 1930 a significant decrease in groundwater levels at Dresden-Tolkewitz was observed and discussed as a result of riverbed clogging due to the increased infiltration rates since 1901 and clogging by suspended materials. In the 1980s strong river water pollution caused by organics from pulp and paper mills in conjunction with high water abstraction caused unsaturated conditions beneath the riverbed, especially at the waterworks at Dresden-Tolkewitz. However, investigations of riverbeds using a diver chamber showed that the material responsible for the pore clogging in the gravel bed consisted of up to 90% inorganic materials (Heeger 1987).

Heeger (1987) calculated a clogging coefficient, w of 7.5×10^3 s for the riverbed without bank filtration and a mean w-value of 2×10^6 s at RBF sites in and around Dresden. The long-term process of riverbed clogging includes a series of building and destruction phases, which overlay a mean value. During floods, with sufficient hydraulic

transport energy, the riverbed is eroded and the hydraulic conductivity of the riverbed is subsequently increased. Similarly long, low-flow periods result in increasing w-values.

After improvement of river water quality in 1989–1993 the hydraulic conductivity of the riverbed increased. In 1992 similar water levels as in 1930 were observed. In 2003 groundwater flow modeling was used to analyse former assumptions on groundwater flow towards the production wells and clogging of the riverbed. From model calibration a reliable w-value of 0.1×10^6 s was determined (Table 1). Water level measurements in 2004 at low flow conditions also indicated a slight decrease in the clogging coefficient.

Table 1. Clogging coefficients of the Elbe riverbed at waterworks Dresden-Tolkewitz

Time period / Year	Clogging coefficient w in s	Remarks / Source
1960–1985	$1.5–2.2 \times 10^6$	Long term water level measurements, Heeger (1987)
1991/1992	$1.0–3.0 \times 10^6$	Water level measurements at cross section
2003	0.1×10^6	From groundwater flow model calibration
Oct/Nov 2004	0.7×10^6	Water level measurements at cross section

Looking at the long-term operation of the waterworks, it is obvious that observed clogging of the riverbed did not result in the closure of wells under the given conditions of an erosive river. After a period with additional organic pollution and observed slime at the riverbed surface (assumed to act as an organic outer clogging layer) there is a recovery of hydraulic conductivity of the riverbed and thus a slight decrease in w-values.

DOC CONCENTRATION IN ELBE RIVER WATER AND BANK FILTRATE

The industry along the Upper Elbe River valley previously discharged a wide range of organic contaminants into the river. Hence, together with urban sewage, the dissolved organic carbon (DOC) comprises a complex mixture of easily degradable and refractory substances. Among the industrial effluents paper mills, cellulose processing plants and the pharmaceutical industry played an important role in the 1980s. In 1988 to 1990 the average DOC concentration at the left bank of the Elbe River at Dresden-Tolkewitz was 24.2 mg/L and the UV-absorbance at a wavelength of 254 nm (UVA_{254}) was 55 m^{-1}. Along a flow path length of about 100 m along a cross-section at Dresden-Tolkewitz the DOC concentration was reduced to about 20% of the input concentration (Nestler et al. 1991). Problems with bank filtrate quality occurred due to the high load of organic pollutants, bad taste and odor, and the formation of disinfection byproducts. Figure 2 gives an impression of the organic load in the Elbe River in 1987–1992. It must be mentioned that the low amount of data does not allow the calculation of mean concentrations, furthermore the variation is very high and the sampling point at the left riverbank in Dresden was affected by waste water inputs some kilometers upstream.

Results from 17 measurements in 1991/92 at a cross section at Dresden-Tolkewitz showed a mean DOC concentration of 6.9 mg/L in Elbe River water and 3.4 mg/L at an observation well near a production well. From that, a reduction of DOC concentration of about 50% can be seen as effect of riverbank filtration processes. Recent investigations in 2003 at the same cross section included 7 samples. In 2003 the mean DOC concentration in Elbe River water was 5.6 mg/L and in bank filtrate at the same observation well sampled in 1991/92 was 3.2 mg/L. The mean DOC concentration in raw water from all wells was found to be 2.6 mg/L as a result of mixing with groundwater. These results prove that the period of strong pollution of Elbe river water did not limit the further use of the site.
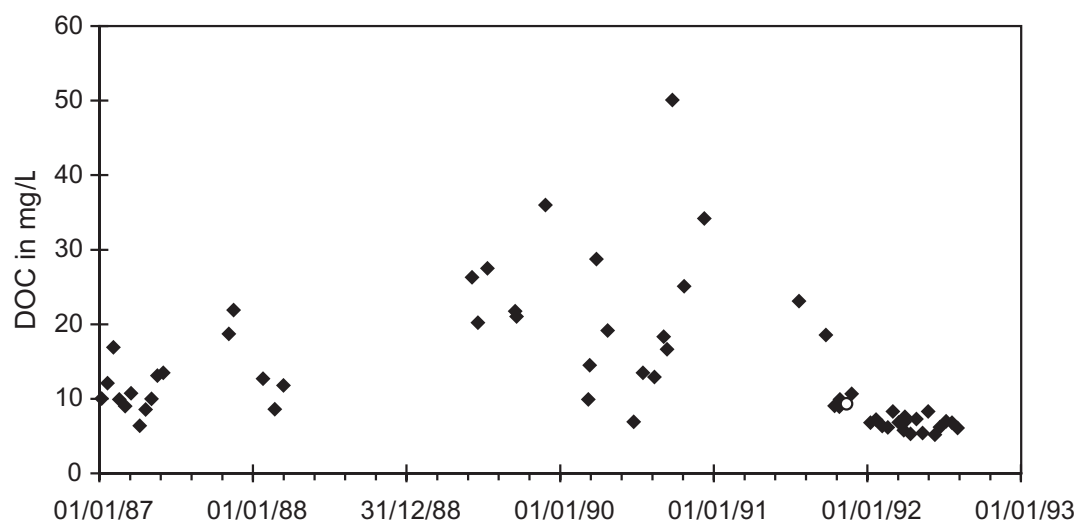
Figure 2. DOC concentration (mg/L) in Elbe River water 1987–1992 (Grischek, 2003)

SUMMARY AND CONCLUSIONS

Drain pipes at waterworks Dresden-Saloppe have been in operation for more than 130 years whilst four production wells at waterworks Dresden-Tolkewitz had to be replaced only after 60 years. Severe clogging of the riverbed occurred in the 1980s mainly due to high loads of organics from pulp and paper mills upstream. After improvement of river water quality in the 1990s, no problems with clogging of the riverbed or bad taste and odor of the drinking water have been encountered.

Raw water quality and treatment are optimized by managing specific mixing ratios of bank filtrate and land-side groundwater. Pumping rates were reduced to get longer retention times in the aquifer and higher attenuation rates of organic compounds. No indication of a decrease in attenuation capacity of the aquifer with time was observed (Grischek, 2003).

Since 1875, bank filtration along the Elbe River in the city of Dresden has been an important source for public and industrial water supply. Infiltration has been induced by pumping of wells and installation of drain pipes. Today, all of the old systems are still in operation. Periods of poor river water quality in the 70s and 80s are overcome. Waterworks have been modernized and treatment technologies adapted. Nevertheless, there is a periodical demand to prove that bank filtrate is a reliable and economic raw water resource for Dresden. Long-term experiences and results of the evaluation of historic and recent data and of investigations using modern modeling tools prove that riverbank filtration is a sustainable water resource for water supply in Dresden.

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Kaliningrad Water Services Improvement Project – Baseline investigation for potential for reduction and stabilisation of organic matter

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Abstract

As part of the international donor- and IFI-financed Kaliningrad Water and Environmental Services Rehabilitation Project, a wellfield 30 km East of the City of Kaliningrad is being upgraded from 30,000 m^3/day to 90,000 m^3/day. The wellfield relies on recharge from a system of interconnected lakes and abandoned gravel pits that are connected to the Pregol River. The planned expansion of the wellfield will result in a large scaling up of the existing abstraction, thus increasing the need to shed light gaps in the present understanding of the geochemistry and hydrology of the area. A combination of hydrological investigations and analysis of the composition, sources and variability of chemical parameters is being carried out, with particular focus on organic matter, which is critical for the design of water treatment. Organic matter as expressed by permanganate oxidisability presently varies at levels close to the drinking water standard. Preliminary data and modelling suggests that there may be potential to develop the wellfield over the long term in a manner that may stabilise and reduce organic matter.

Keywords

Infiltration, Kaliningrad, organic, oxidisability, wellfield.

INTRODUCTION

The Kaliningrad Water and Environmental Services Improvement Project (KWESIP) is a multi-component IFI-cofinanced programme to improve the provision of water services to the City of Kaliningrad with a population of approximately 500,000 and to reduce pollution of the Baltic Sea from poorly treated wastewater from the area. A pivotal component of the KWESIP is to supplant the production of drinking water from surface water with groundwater abstraction. The surface water sources are severely affected by pollution and frequent wind-induced intrusion of brackish water into the Pregol River system. The so-called Eastern Wellfield and Treatment Plant – located approximately 30 km upstream and to the East of Kaliningrad – currently provide approximately 30,000 m^3/day, relying on infiltration from a system of abandoned gravel pits and shallow lakes that are connected to the Pregol River. The wellfield was originally established during the years 1935–1943 to supply Königsberg, when the area was part of the former German province East Prussia, as a response to increasing demand and the lack of suitable groundwater in the immediate area of the City.

The wellfield project aims to increase the production from the Eastern Treatment Plant to 90,000 m^3/day and is funded by the Danish Environmental Protection Agency and the European Bank of Reconstruction and Devel-

opment. Other projects of the KWESIP are directed at reducing leakage and water demand, so that in future the City of Kaliningrad may rely mainly on this source of groundwater. The expansion of treatment capacity is being implemented in parallel with development of the wellfield. However, the large increase in abstraction amplifies some of the uncertainties regarding the future water quality, organic matter being the most critical parameter from a treatment design point of view. This research aims to provide a better general understanding of the water quality processes that may serve as a baseline for future management and monitoring of the wellfield. Specifically, the aim is to understand better the main determinants of organic matter at the wellfield with a view to establishing a complementary approach to infiltration and further treatment as wells as the potential for future reduction and stabilisation through wellfield development.
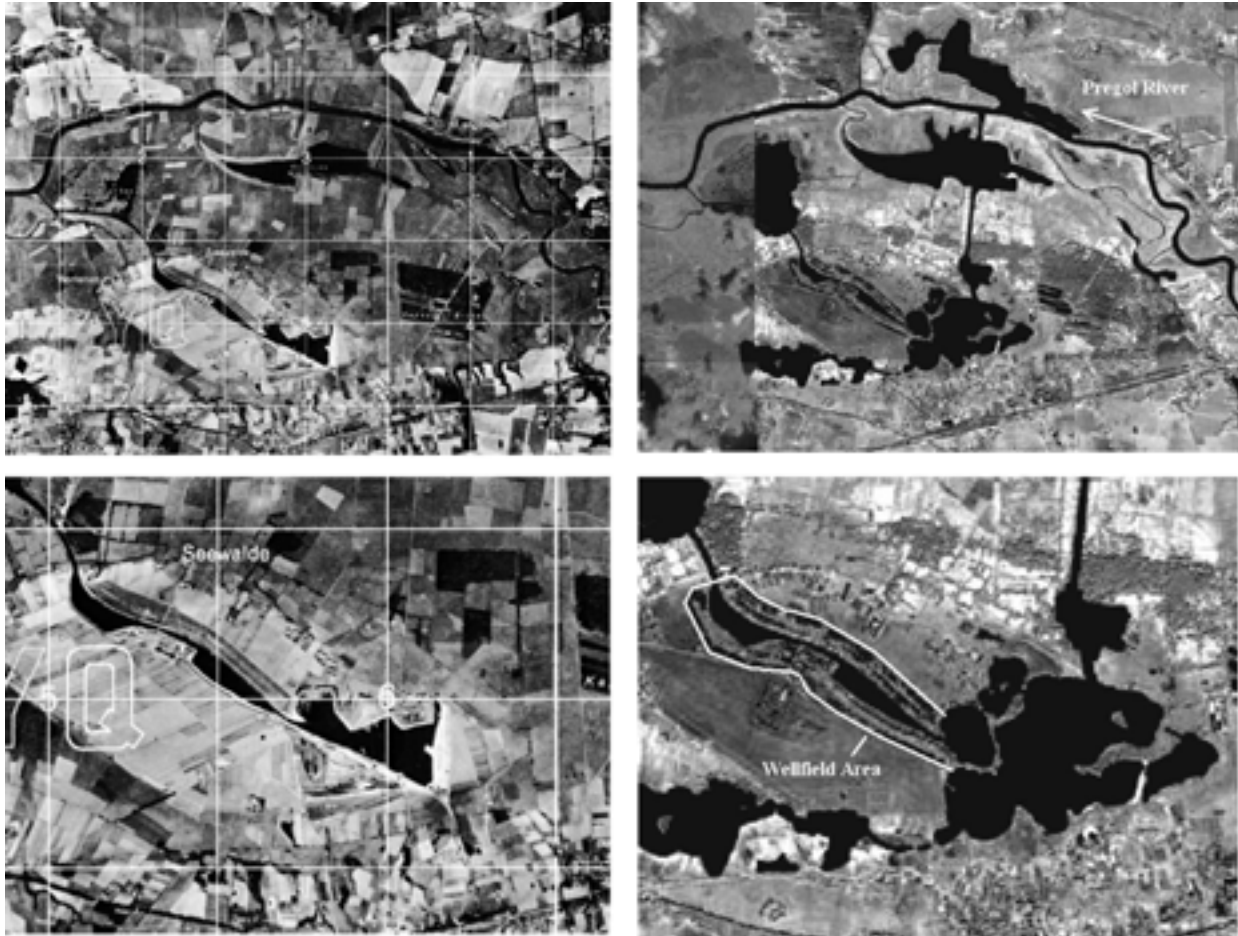
1944
Aerial photograph taken on August 23rd, 1944
(size of squares: 500 m),
provided by Luftbilddatenbank.

2001 and 2002
Composite of Ikonos Sattellite Images taken
on September 5th 2001 and May 21st 2002,
provided by Space Imaging.

Figure 1: Development of wellfield area and surface waters

METHODS

Permanganate oxidisability/COD-Mn

The Russian method for determining chemical oxygen demand in drinking water is in accordance with the method PND F 14.2:4.154-99 for the permanganate value in oxygen equivalents. The method is based on oxidation of

matter present in the water sample by a known volume of potassium permanganate in sulphuric-acid medium during boiling for 10 minutes. Correction for inorganic elements such as Nitrate and Ferrous iron is performed.

The relationship between the amount organic carbon and the permanganate oxidisability is only empirical and, ultimately, site specific since not all organic carbon is readily oxidisable using permanganate. The method is similar to other methods for determination of Permanganate Value / COD-Mn. Investigations of raw water and surface water at the wellfield of COD-Mn (Danish Standard Danish standard DS275) and Non-Volatile Organic Carbon (NVOC) showed an approximately linear relationship with a numerical ratio of approximately 1:1 between COD-Mn in mg/l oxygen equivalents and NVOC in mg/l. The Russian drinking water standard for permanganate oxidisability is 5 mg/l.

Liquid Chromatography - Organic Carbon Detection LC-OCD

The technique LC-OCD (Liquid Chromatography - Organic Carbon Detection), developed by the DOC Labor in Karlsruhe, is based on carbon mass determination following chromatographic separation in a porous gel. The latter yields a breakdown of the organic matter present according to hydrophobicity and molecular weight. Reference measurements allow identification of distinct fractions of Natural Organic Matter and the method therefore able to yield a clear picture of the presence of Extracellular Polymers, humates and various low and high-molecular intermediary 'building blocks' and degradation products. It provides a good illustration of the cycle of organics in natural waters. A more detailed description of the method is available from Huber et al. (1996).

RESULTS AND DISCUSSION

History and description of the site

In the early 1900s, the large German concrete producer Windshild und Langelott began exploiting one of Germany's best gravel deposits in the flood plain of the Pregol River in East Prussia. The gravel pit was located by the small town of Gross Lindenau, present-day Ozerki, where a concrete plant was also set up. Most of the exploitable gravel was below the groundwater table and had to be dredged at depths of up to 20 m. In order to be able to reach customers by water, in 1925 the company excavated a navigable channel through exploitable gravel deposits and a shallow natural lake at its Western end, connecting the gravel pond with the Pregol River. Spurred by the need to look further afield for suitable sources of water, the City of Königsberg was quick to accept Windshild and Langelott's proposal around 1935 to use its channel and the adjacent strips of land for the purpose of developing a wellfield, with a guarantee of ample recharge from the Pregol River. The company E. Bieske developed the wellfield from 1935 until the end of the 2nd World War. A total of 26 evenly spaced wells were drilled along the Northern bank and in 1943 an iron and manganese removal plant with a capacity of 30,000 m^3/day was commissioned.

Intensive gravel extraction continued during the Soviet period and the pits were enlarged considerably. The wellfield was not developed significantly other than through replacement of existing capacity until the Russian authorities conducted a large-scale water resource evaluation, concluding in 1982 that the greatly increased area of the gravel pits and the size of the Pregol River source would allow for a significant upscaling of abstraction. Over the entire history of the wellfield, some 130 wells have been constructed, along two lines on the northern and southern side of the channel, respectively. Nevertheless, only about 30 wells are in operation today. The distance between the most of the wells and the channel is approximately 60–70m. Attempts at increasing overall production have largely failed. This is most likely attributable to several factors, mainly poor well construction and an environment that is conducive to well clogging.

The wellfield and gravel pits (see Figure 1) are situated left of the river Pregol, 30 km East of Kaliningrad. The Pregol flows into the Kaliningrad lagoon on the Gulf of Gdansk. The wellfield is situated on a terrace at the edge of the Pregol floodplain, above which the terrace is elevated some 5–9 m. The floodplain itself has a little gradient and is rich in lakes and channels and possesses a vast system of man-made ditches and drainage channels that extends across much of the Kaliningrad region, serving to reclaim land for agricultural purposes since the 1800s. The channel itself is shallow, only 2–3 meters in depth, and there is little movement of the water. The large gravel pits beyond the Eastern end of the channel have depths of up to 20 m. The channel and the small lake at its Eastern end is separated from these pits by an embankment that is pierced by a large pipe, allowing a more or less constant flow of water from the Eastern pit into the channel. The visual appearance of the banks of the gravel pits and channel is clean and sandy. However, workers at the wellfield report that there is a layer of sludge at the bottom of the channel.

The Geological structure of importance to the water abstraction at the wellfield site is quite simple. Glacial and Fluvioglacial sediments have been deposited on the eroded surface of Middle Palaeogene strata. The middle Paleogene provides an underlying protection against upwelling of salt water from the Cretaceous deposits below. A relatively thick sequence of fluvioglacial deposits is present all over the site and provides the favourable basis of water abstraction from the area. There are large areas of peat deposits in the floodplain and around the wellfield, particularly towards the North and East, which are thought to have a heavy influence of the level of organic matter in the ground and surface waters.

Most economic activity in the immediate vicinity of the wellfield ceased in the 1990s, including agriculture and gravel abstraction. However, poorly treated wastewater from the town of Ozerki (old Gross Lindenau) with a population of approx. 2,000 discharges in to the gravel pits. Moreover, virtually untreated filter backwash is discharged from the Water Treatment Plant into the channel. Poorly treated wastewater continues to be discharged in the Pregol River from cities upstream of the wellfield. The drainage system in the Kaliningrad region is falling into disrepair and many areas are returning to their natural state as wetlands, which figures as a large unknown in the general development of water quality of the Kaliningrad region.

Modelling

In order to provide some verification of the feasibility of increasing abstraction to 90,000 m^3/day, a single layer hydrogeological model was prepared under this project. The available historical groundwater monitoring and abstraction data was not strong enough for accurate calibration. Inverse calibration and a sensitivity analysis using present groundwater levels were therefore performed in order to yield a baseline model with a lower-conservative estimate of transmissivity of 0,007 m^2/s. The estimated net precipitation of 149 mm provides only an insignificant contribution to the abstracted volume of water.

The model suggested that the contact to the channel was weaker in the Eastern half than in the Western half. This observation was consistent with the indication in existing geological data of the presence of a thin clayey layer under the channel that could act as boundary. Under the prevailing high range of feasible transmissivities, it appeared that clogging of the channel bottom or the presence of any other impermeable boundary under the channel was unlikely to have a detrimental effect on potential yield from wellfield, because water would easily flow in from other directions. Another important result was that travel times of the filtrate could be as low as days or weeks in areas with good contact to channel.

Surface water chemistry and ecology

The historical data for channel and raw water quality that could be provided by the Kaliningrad Water Company covered the period 1984–2004. Unfortunately, for the last 15 years, measurements have been intermittent. It has also been a period of large changes in the general level of pollution of surface waters and land-use around the wellfield as consequence of the economic upheavals with the break up of the Soviet Union. Moreover, the analytical methods have not been applied with equal rigour throughout the period, but it is thought that the last 10 years represent a fairly consistent set of data.

Given the inconclusive hydrogeological data, some corroboration of the model findings was sought by observing the variations of temperatures in individual production wells, as measured in April and September 2004, complemented by historical raw temperature water data. These observations yielded a picture in which wells in the Eastern half generally show an almost constant temperature of around 10 °C, whereas variation of up to a few degrees is observed in wells of the Western half of the channel. Historical chemical data confirmed that water travel times, as evidenced by the lag in certain peaks in concentrations of chlorides and other ions, may be as low as a couple of weeks.

It can be seen from Figure 2.1 that oxidisability levels have become lower and less variable since the mid-1990s. Most of the time, oxidisability in the channel varies around a level of approximately 10 mg/l. In the raw water, levels are generally between 4 and 7 mg/l. The lower limit of 3.5–4 mg/l oxidisability is rarely gone below and seems to be constant over the entire period of data. As can been seen in Figures 2.2 and 2.3, the data shows a very good match between levels of oxidisability in the North and South raw water lines, suggesting that the general level of oxidisability is determined by a common source, probably the channel. This is consistent with the hydrogeological model that demonstrated that even if clogging of the channel is quite severe, much or most of the raw water should originate from the channel. However, there is no clear correlation in the provided data between oxidisability in the channel and the raw water. One possible explanation for this apparent inconsistency may be the fact that oxidisability in the channel is measured on unfiltered samples, so that much of the measured oxidisability may be due to particulate organic matter. Another possible explanation is that the single sampling point in the channel does not reflect temporal and spatial variations inside the channel.

There is a correlation between channel oxidisability, BOD, Nitrate and Turbidity in the channel, as shown in Figure 2.4, and there is a visible algal bloom in the channel from June to September. As seen in Figures 2.4 and 2.5, nitrate is virtually depleted from the channel in summer. The data suggest that algal growth in the channel is N limited. Phosphorous seems abundant not only in the surface water but also in the groundwater. Sampling of the Pregol and gravel pits showed levels between 0,12 mg/l and 0,34 mg/l in June of 2004. The levels in wells were measured between 0,15 mg/l and 0,24 mg/l. A single measurement in the channel showed only 0,022 mg/l, however. The two main probable sources are wastewater and washing out of phosphorous from previously intensively fertilized land. There is a significant decrease in sulphate concentration in the channel during the summer months as well as sulphate-reducing conditions at the bottom of the channel, as evidenced by high measured sulphide levels in the channel and wells in summer. See Figure 2.6. This is probably induced by decaying organic matter with resulting oxygen depletion at the channel bottom during the summer months.

Figure 3 shows TOC and DOC in the surface water and two wells. A breakdown of organics in the channel and adjacent water bodies using the LC-OCD method is also shown. There is very little low-molecular weighted organic matter present – the organic matter consists of almost entirely of humates and a minor part of polysaccharides. Moreover the shape of the humates profile is very 'pure' without extended ridges. The highest level of humate is found in the Pregol, upstream of the wellfield. The lowest surface water DOC levels are found in the gravel pits south of the wellfield.
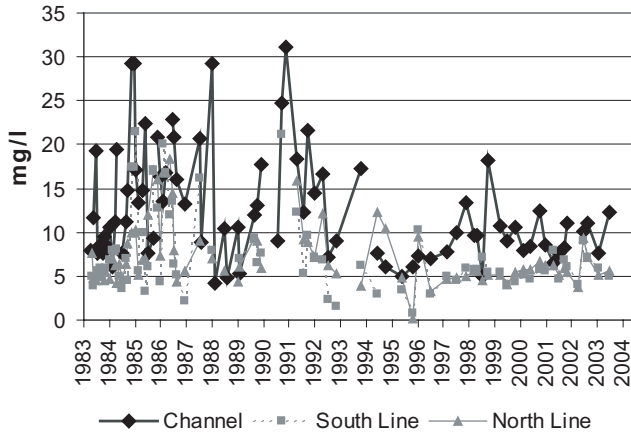
Figure 2.1.Permanganate Oxidisability in mg/l (oxygen equivalents) for the period of 1984–2004 for the wellfield channel and the North and South Raw water lines
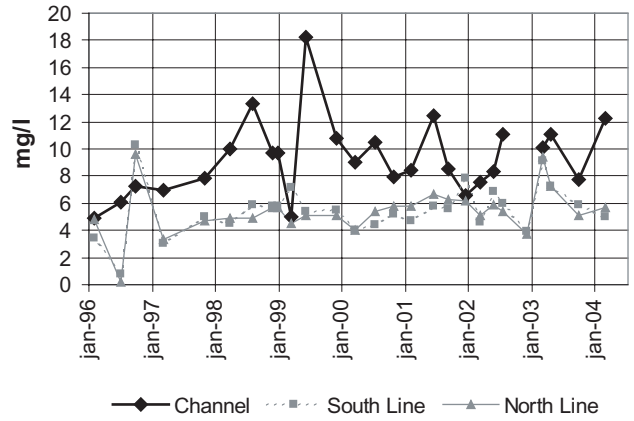
Figure 2.2. Permanganate Oxidisability in mg/l (oxygen equivalents) for the period of 1994–2004 for the wellfield channel and the North and South Raw water lines
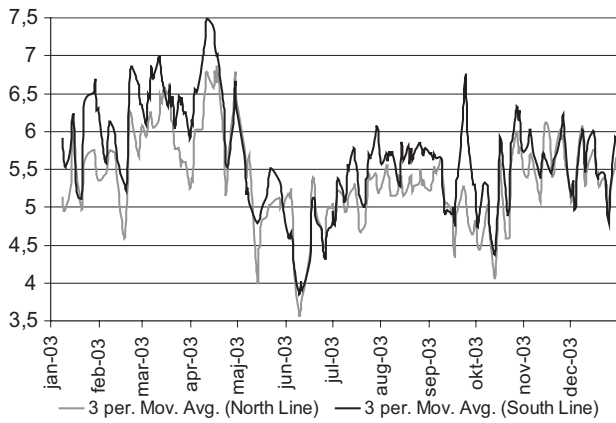
Figure 2.3.Permanganate Oxidisability in mg/l (oxygen equivalents, uncorrected for inorganics) in raw water in 2003, 3 point moving average
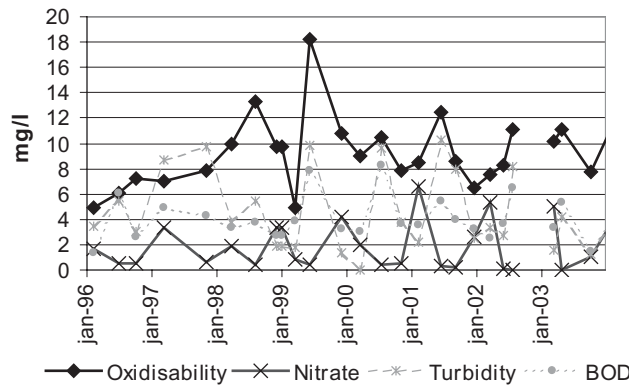
Figure 2.4. Permanganate Oxidisability in mg/l (oxygen equivalents), Turbidity, BOD and Nitrate for the period of 1994–2004 for the wellfield channel
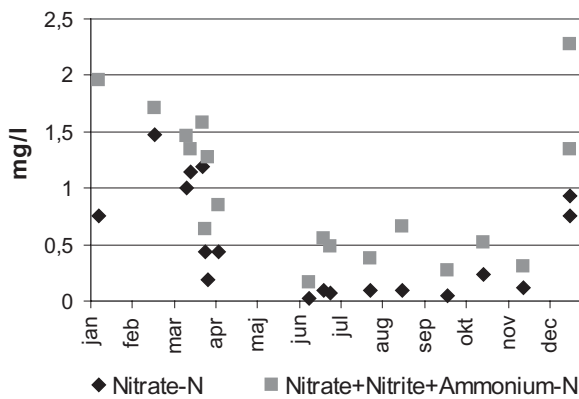
Figure 2.5. Seasonal N variation 1998–2004
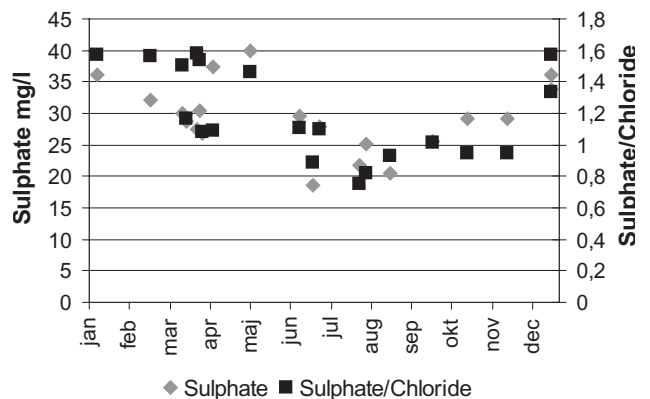
Figure 2.6. Seasonal Sulphate variation 1998–2004

	TOC (mg/l)	DOC (mg/l)
Channel	17,4	10,4
Pregolya Upstream	10,8	9,7
West Shallow Lake	10,6	9,7
East Gravel Pit	9,8	9,1
South Gravel Pit West	8	7,4
South Gravel Pit East	7,8	7,4
South Bank Well 1	7,3	6,7
South Bank Well 2	5,9	5,6
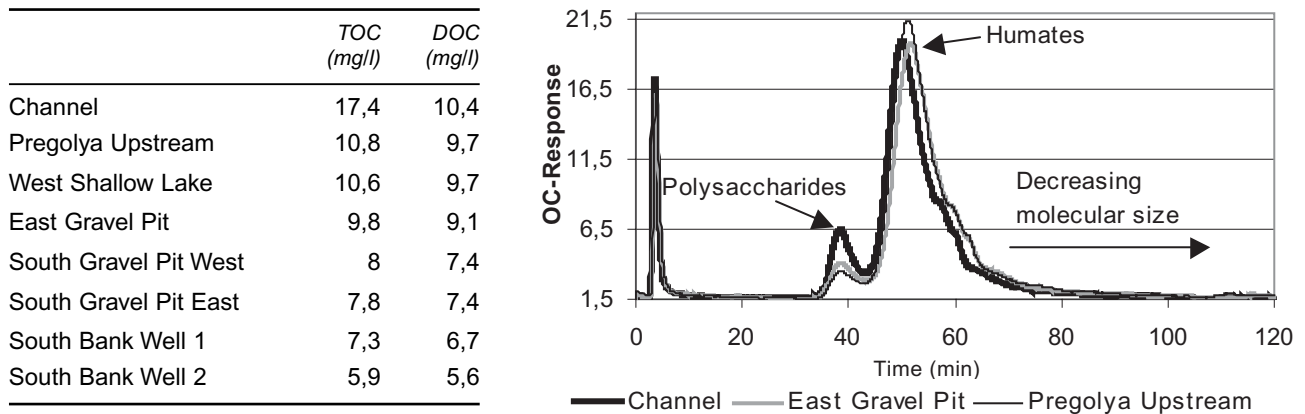
Figure 3. TOC, DOC and LC-OCD profiles of surface waters and groundwater
at the Eastern Wellfield. Samples taken on June 18th 2004

Determinants of the observed level of organic matter

The model and temperature observations suggested different degrees of contact with the channel throughout the wellfield. From the point of view of achievable quantity, this is unlikely have a detrimental effect because of the high transmissivity of the aquifer and the possibility of recharge from several other surface water bodies. However, a complex pattern with regard to water quality is likely to emerge. In the areas with good contact to the channel, average filtrate travel times may be as low as days or weeks, whereas travel times of only a couple of months may be encountered in other parts. Grünheid et al. (2004) have shown that optimal degradation of organic matter may require filtrate travel times of at least 2 weeks under aerobic conditions and over 3 months under anaerobic conditions.

The good correlation of fairly high-frequency variations (order of magnitude of weeks) in raw water from the North and South bank rows of wells suggests that the overall determinant for observed variation lies in a common source, which the model implies must be the surface water, mostly the channel. The low significance of local precipitation precludes that ground surface infiltration should be having a significant effect. The historical data indicates an apparent consistent minimum value of raw water oxidisability of approximately 3,5–4 mg/l, which is close to the Russian drinking water standard of 5 mg/l. Fairly large variations take place, with values apparently reaching up to 7–10 mg/l at certain times. Unfortunately, no measurements outside the wellfield area exist to give a conclusive picture of the background level of oxidisability/organic matter. Moreover, analytical inconsistencies between surface and raw water analyses make it difficult to determine the degree to which variation in the surface water are presently translating into variations in the raw water.

The breakdown of the dissolved organic matter in the surface water shows that it is dominated by humates, with a minor component consisting of polysaccharides. Interestingly, the humate profiles are quite smooth and narrow, without extended ridges representing intermediary states or light molecular fractions. This may be an indication of a very stable nature, which may in turn be an indication that the humate is mainly of geogenic origin. Moreover, the profiles are similar. The ubiquitous presence of peat and drained areas throughout the floodplain would be a likely source. The DOC measurements show an increasing trend in a Northerly direction towards the Pregol River, albeit there appears to be a separate source of organic matter in the channel at the time of observation in June of 2004 as seen in the difference between TOC and DOC as well as the LC-OCD profiles.

Visual inspection and historical water data indicate an annual algal bloom that lasts from June to September, depleting the channel of Nitrate. Sulphate-reducing conditions seem to exist at the bottom of the channel in summer. Moreover, staff at the wellfield report that there is a layer of suspended sludge at the bottom of the

channel, suggesting a more permanent anaerobic state at the bottom of the channel. The lack of oxygen is likely to slow down degradation of organic matter during infiltration.

The variability of oxidisability in the raw water shows that degradation inside the aquifer as well as attenuation of variations through groundwater mixing is not sufficient to stabilise levels or consistently to bring them down to below the standard of 5 mg/l. Some variation is attributable to the shifting operation of wells with different levels of oxidisability. However, the correlation between the North and South raw water demonstrates that the main cause is variation in oxidisability levels in the surface water.

CONCLUSIONS

The main factor determining raw water levels of oxidisability is the surface water. The channel makes an independent contribution to DOC, but its direct significance for raw water levels has yet to be determined. The travel times proposed by the model and evidence of anaerobic conditions in the channel and aquifer indicate that the configuration of the wellfield may not optimal from the point of view of degrading organic matter.

The potential to reduce and to stabilise oxidisability and organic matter at the Eastern Wellfield depends on the prevailing background level as well as the potential for degradation inside the aquifer. By adopting a strategy of wellfield development that pursues an increased distance of the wells to the surface water, more groundwater mixing and degradation and attenuation of variations of organic matter might be achieved. However, the ultimate potential for reduction of organic matter and oxidisability is not known with any certainty. Geological variability in the area and the limited extent of the existing geological exploration make a gradual approach to future wellfield development desirable. Given that the observed raw water levels of oxidisability are generally above, but close to the drinking water standard, margins may ultimately be important.

There is a need for determination of the background level of organic matter and oxidisability and for establishing an empirical relationship between the Russian standard for oxidisability and DOC. Further investigations should also be made into determining the underlying factors for variations in oxidisability and organic matter in the surface water as well as establishing a more accurate water balance and hydrological understanding of a wider area. In general, there is a need to establish a more consistent and systematic programme of ground- and surface water monitoring if many important questions are to be addressed. The present investigations will serve as a basis for future monitoring.

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Groundwater recharge from a lined watercourse under shallow water table condition

V. Goyal, B.S. Jhorar and R.S. Malik

Abstract

The study was conducted to quantify the field groundwater recharge dynamics. Rise and fall in water table were observed in observation wells at orthogonal distances X = 5, 25, 50 and 100 m from a 7 year old cement lined channel (Width = 0.6 m, Water depth = 0.37 m, Design discharge = 0.23 m^3/s) at different time intervals. The initial water table was at 2.54 m depths from soil surface. The height of the bottom of the watercourse (H_o) from the water table was 2.43 m.

Ground water rise R and groundwater fall F increased with time at all orthogonal distances X and followed the square root time law with r^2 values more than 0.95 (R = 0.0366 $t^{1/2}$ + 0.03514; F = −0.0474 $t^{1/2}$ + 0.0754 for 5 m). The watercourse influencing orthogonal reach (the value of X at which rise is more than 14 % of the total rise at 5m) was 100m. Groundwater rise rate RR and falling rate FR decreased with time following inverse square root law with r^2 values equals to 0.765 for X=5m. Seepage from watercourses was mainly a horizontal infiltration phenomenon under shallow water table condition. The ground water rise and fall decreased with distance exponentially with r^2 value more than 0.93 (R= 0.4092 $e^{-0.0192X}$ and F = 0.5505 $e^{-0.0114X}$). Groundwater rise R was higher than groundwater fall up to 100 hours and then the order was reversed i.e. fall was larger than rise at time more than 100 hours.

The practical applications are that lined watercourses contribute to the considerable seepage and ground water recharge up to a canal reach of 100 m even in the short time interval of 170 hours.

Keywords

Ground water recharge; lined watercourses; shallow water table.

INTRODUCTION

In order to conserve water resources, canal and watercourses have been lined at very huge costs; covering 5,000 watercourses of 20,000 km length in state of Haryana, India alone (Master Plan, 1998). The seepage could be an effective and eco-friendly measure for groundwater recharge in falling water table situations even after lining of watercourses (Anonymous, 1984).

A few studies have been conducted to quantify the seepage effect and groundwater recharge from ephemeral (Abdulrazzak and Morel-Seytoux, 1983), perennial rivers (Knappe et al., 2000) and unlined canals and watercourses (Dillon, 1968; Nagaraj and Dewan, 1972; Phogat et al., 2000 and Malik and Richter, 2000). To our knowledge little information is available (Rohwer & Stout, 1948) on quantification of groundwater recharge from lined watercourses. Therefore, the present study was conducted to quantify the groundwater recharge dynamics from the lined watercourse under shallow water table condition.

MATERIAL AND METHODS

A field experiment was conducted on a 7-year-old cement lined rectangular watercourse (width 2W = 0.6 m; constant water head h_o = 0.37 m; design discharge = 0.23 m^3/h) situated at an outlet number – RD 5500 L of farm canal at soil research farm CCS Haryana Agricultural University, Hisar. Soil samples from different layers taken during the installation of piezometers, were oven dried and ground gently with the pestle-mortar. The fraction remaining above a 2 mm sieve was identified as calcite concretions. The soil passed through the sieve was analyzed for different physico-chemical properties. Soil analysis was done with the standard methods. The relevant physico-chemical properties up to impervious layer are given in Table 1.

Table 1. Relevant soil physico-chemical properties of hisar lithology

Depth (m)	EC (dS^{m-1})	pH	Texture	k (cm/h)
0.0–9.6	1.2	8.2	Loam	0.628
9.6–12.8	1.19	8.5	Loamy sand	1.29
12.8–24.3	0.96	8.6	Clay loam	0.633

The initial water table was at 2.54 m depth from the soil surface. The height of the bottom of the watercourse H_o from the water table was 2.43. The saturated aquifer thickness e was 11.6 m and height of the bottom of the watercourse from the soil surface h_s was – 0.11 m as shown in Figure 1(a).
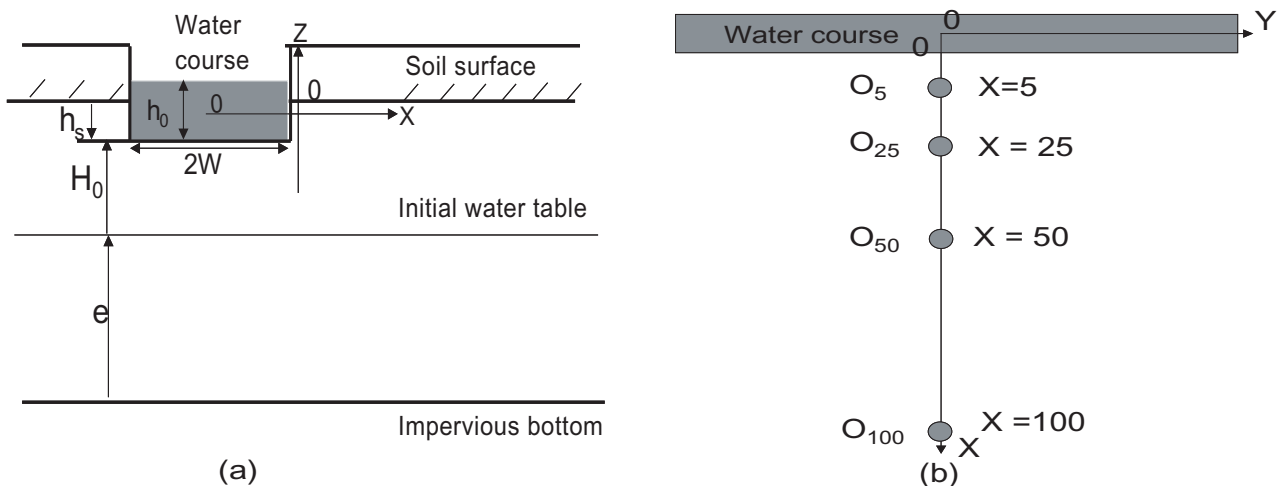
Figure 1. Schematic diagram showing (a) watercourse positional parameters in horizontal X and vertical Z plane: water depth h_o, width 2W, height of the bottom from water table H_o, height of the bottom from soil surface h_s (b) observation wells position in horizontal X and watercourse length Y planes at indicated orthogonal distances

Rise and fall in water table were observed as a function of time t in observation wells O_5, O_{25}, O_{50} and O_{100} placed at watercourse orthogonal distances X = 5, 25, 50 and 100 m as shown in Figure 1(b) at different time intervals starting from t = 2 h and 6 h for rise and fall respectively. Time t = 0 was taken just before the onset of water flow in the channel for water table rise R; and just after the closure of water flow for water table fall F respectively. Water table rise rate RR = dR/dt and falling rate FR = dF/dt at any time t were estimated by numerically differentiating R and F w.r.t. t by the central finite difference method (Forsythe and Warrant, 1960).

RESULT AND DISCUSSION

Water table rise R and fall F increased with time at all orthogonal distances X = 5, 25, 50 and 100 m (Figs. 2a and 3a). Indeed the regression line almost passed through the origin (Figs. 2b and 3b) with $r^2 > 0.95$ and little intercept (Eqns 1-8, Table 2). It meant that the groundwater flowed in horizontal direction during seepage from the watercourse under shallow water table condition. The square root time law had been reported to govern unsaturated horizontal infiltration in capillary tube (Malik et al., 1978) and in porous media (Green and Ampt, 1911; Kirkham and Feng, 1949; and Malik et al. 1978) after Darcy law. While observing groundwater recharge from seepage from an ephemeral river, Abdulrazzak and Seytoux, (1983) have reported that after establishing hydraulic connection in the soil column between saturated watercourse bed and initial water table, the infiltration rates (seepage rate) equaled recharge rate and it was a case of horizontal groundwater flow.
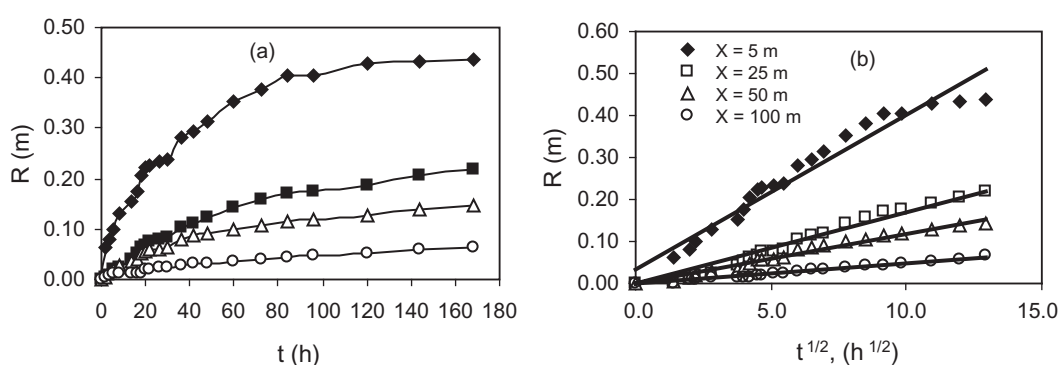
Figure 2. Water table rise R at indicated distance X
as a function of (a) time t and (b) square root of time $t^{1/2}$
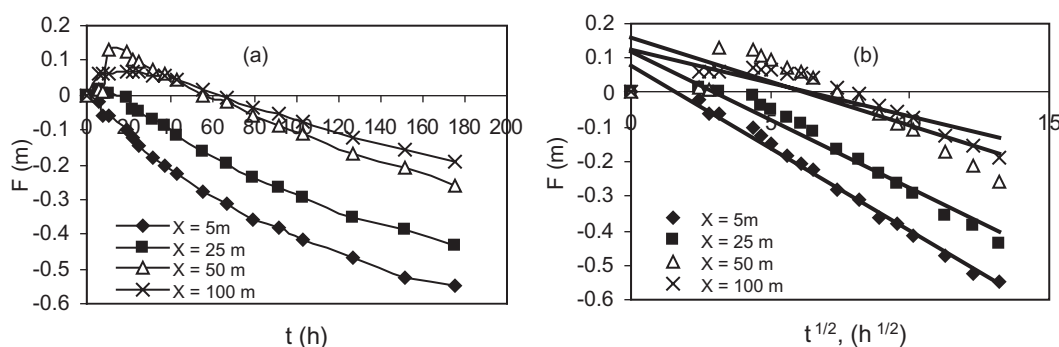
Figure 3. Water table fall F at indicated distance X
as a function of (a) time t and (b) square root of time $t^{1/2}$

The straight line relationship between water table rise rate RR and fall rate FR; and the inverse square root of time $1/t^{1/2}$ (Eqns 9-16, Table 2) are shown in Figure 4 for rise and in Figure 5 for fall at an orthogonal distance X = 5 m. The r^2 values equals to 0.765 at X = 5 m; it verified the validity of Darcy's law for recharge rates. However the correlations were not found significant at large orthogonal distances X ≥ 25 m for the rise rate RR and fall rate FR. This might be due to error introduced in determining the RR and FR by central finite difference method and also due to slightly more time lag in reaching the waterfronts to X ≥ 25 m (Table 3).

Table 2. Relationship between rise and fall in water table with time and orthogonal distance

Sr No.	Water Table Rise/Fall	Time (t)/ Distance (X)	Regression equation	n	r^2
1	Rise (R) at 5 m	$t^{1/2}$, $(h^{1/2})$	$R = 0.0366\, t^{1/2} + 0.0351$	22	0.9526*
2	Rise (R) at 25 m	$t^{1/2}$, $(h^{1/2})$	$R = 0.0169\, t^{1/2}$	22	0.961*
3	Rise (R) at 50 m	$t^{1/2}$, $(h^{1/2})$	$R = 0.0119\, t^{1/2}$	22	0.9771*
4	Rise (R) at 100 m	$t^{1/2}$, $(h^{1/2})$	$R = 0.0047\, t^{1/2}$	22	0.9728*
5	Fall (F) at 5 m	$t^{1/2}$, $(h^{1/2})$	$F = -0.0474\, t^{1/2} + 0.0754$	18	0.9825*
6	Fall (F) at 25 m	$t^{1/2}$, $(h^{1/2})$	$F = -0.0393\, t^{1/2} + 0.1167$	18	0.9388*
7	Fall (F) at 50 m	$t^{1/2}$, $(h^{1/2})$	$F = -0.0255\, t^{1/2} + 0.158$	18	0.6554*
8	Fall (F) at 100 m	$t^{1/2}$, $(h^{1/2})$	$F = -0.0193\, t^{1/2} + 0.1252$	18	0.7153*
9	Rate of rise (RR) at 5 m	$1/(t)^{1/2}$, $(1/h^{1/2})R$	$RR = 0.06\, 1/t^{1/2} - 0.0031$	21	0.765*
10	Rate of rise (RR) at 25 m	$1/(t)^{1/2}$, $(1/h^{1/2})R$	$RR = 0.0088\, 1/t^{1/2} + 0.0043$	21	0.1068
11	Rate of rise (RR) at 50 m	$1/(t)^{1/2}$, $(1/h^{1/2})R$	$RR = 0.014\, 1/t^{1/2} + 0.0023$	21	0.2643
12	Rate of rise (RR) at 100 m	$1/(t)^{1/2}$, $(1/h^{1/2})R$	$RR = 0.0136\, 1/t^{1/2} + 0.0014$	21	0.2285
13	Rate of fall (FF) at 5 m	$1/(t)^{1/2}$, $(1/h^{1/2})F$	$FF = -0.0447\, 1/t^{1/2} + 0.0027$	17	0.7993*
14	Rate of fall (FF) at 25 m	$1/(t)^{1/2}$, $(1/h^{1/2})F$	$FF = 0.0063\, 1/t^{1/2} - 0.004$	17	0.1067
15	Rate of fall (FF) at 50 m	$1/(t)^{1/2}$, $(1/h^{1/2})F$	$FF = 0.0379\, 1/t^{1/2} - 0.0062$	17	0.1051
16	Rate of fall (FF) at 100 m	$1/(t)^{1/2}$, $(1/h^{1/2})F$	$FF = 0.0192\, 1/t^{1/2} - 0.0043$	17	0.7435*
17	Rise (m)	X	$R = 0.4092 e^{-0.0192X}$	4	0.9683*
18	Fall (m)	X	$F = 0.5505 e^{-0.0114X}$	4	0.9372*
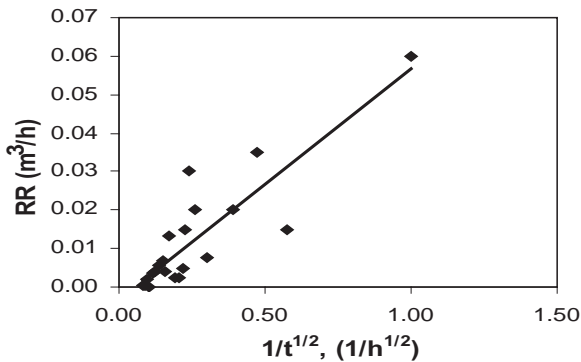
Figure 4. Rise rate RR as a function of inverse square root time $1/t^{1/2}$ in observation well O_5
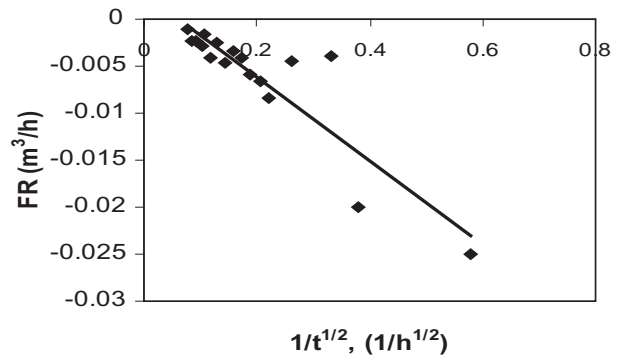
Figure 5. Fall rate FR as a function of inverse square root time $1/t^{1/2}$ in observation well O_5

Table 3. Water table rise R and fall F at indicated orthogonal distances X from watercourse at different times

Time (h)	R (m) at X= 5	25	50	100	Time (h)	F (m) at X= 5	25	50	100
2	0.064	0.004	0.0045	0.0021	6	−0.02	0.015	0.015	0.06
14	0.154	0.042	0.0347	0.0122	22	−0.125	−0.04	0.105	0.065
42	0.294	0.112	0.0861	0.0322	79	−0.36	−0.235	−0.06	−0.035
168	0.437	0.217	0.1374	0.0586	175	−0.55	−0.435	−0.26	−0.19

Water table rise R at time t = 168 h and water table fall F at time t = 175 h were found exponentially related to orthogonal distances X (Figure 6 a and b). The regression equations are given in Table 2 with r^2 values more than 0.93. Such exponential relations between mound height R and distance X at any time t has also been reported by Abdulrazzak and Morel-Seytoux, (1983) for ephemeral river seepage, Phogat and Malik, (1997) for sand box canal and Phogat et al., (2000) for farm canal.
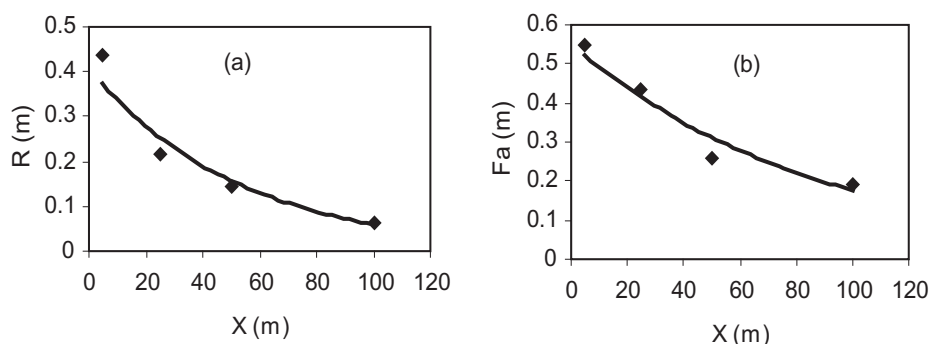
*Figure 6. (a) Water table rise R and (b) absolute fall in water table Fa
in observation wells as a function of distance X*

Water table rise R of 0.49 m at X = 5 m and 0.05 m at X = 100 m in 168 hours (Table 3) implied that there was considerable seepage even from a lined water course under field conditions because some cracks developed in cement lined surfaces in due course of time and resulted in to preferential channel flow. The watercourse under study was 7 year old and such high seepage may take place. The practical implication of having considerable seepage from a 7 year old lined watercourse is that lining should be resorted only where absolutely necessary such as sandy beds with saline ground water and there should be more frequent and effective repairs of lined water courses. The seepage could be an effective and eco-friendly measure for groundwater recharge in falling water table situations.

Comparison of water table rise R with absolute fall Fa (Figure 7) at X = 5 showed that groundwater rise R was higher than groundwater fall up to 100 hours and then the order was reversed i.e. fall was larger than rise at time more than 100 hours.
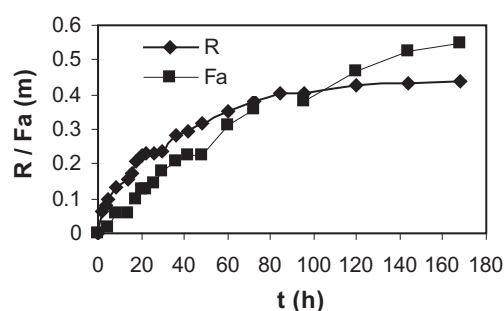
Figure 7. Water table rise R and absolute fall Fa as a function of time t at X = 5 m

It might be due to the fact that large time was taken by the advancing waterfront to reverse to receding waterfront. This time of crossing over increases as the distances increases and may be seen in Table 3.

CONCLUSION

- Water table rise R and fall F increased with time at all orthogonal distances X = 5, 25, 50 and 100 m with $r^2 >$ 0.95 and follows square root time law.
- Rise rate RR and fall rate FR of groundwater decreased with time following inverse square root law with r^2 values more than 0.76 for X = 5 m.
- The groundwater rise R and fall F decreased with distance exponentially.
- Influencing orthogonal watercourse reach was 100 m in 168 hours leading to considerable seepage from the lined watercourse.

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Artificial recharge of Baghmelak aquifer, Khouzestan province, southwest of Iran

N. Kalantari and A. Goli

Abstract

The collected hydrogeological data indicated that in order to combat progressive depletion of groundwater level in Baghmelak aquifer and meeting the growing demand on groundwater, establishment of artificial recharge can be considered as a tool, which aims groundwater resource management. The success of such adoption requires the integration of the available knowledge in many decipline. The prime importance in establishment of such engineered system where surface water is put on in the ground for infiltration requires proper selection of sites. Therefore, an integrated investigations including surface geology, well logs, infiltration test and pumping test in vicinity of three ephemeral streams draining from the conglomerate highs were carried out. The potentials of harvesting storm waters, quality of available water and suspension loads have been assessed. On this background, three sites including basin and a check dam for artificial recharge were suggested in the north and northeast of the area where coarse alluvial thickness was remarkable. On the bases of collected data, the rate of recharge that could be achieved at the three sites is approximately 2.2 million m^3 per year.

INTRODUCTION

With increasing population demand for more reliable supplies of water has increased. The situation is critical in places where groundwater is the only accessible water resource and withdrawal of water is more than the rate of recharge. Consequently groundwater deterioration and/or an imbalance in the groundwater reservoir is created and managed aquifer recharge is an attractive option (Heinzmann and Sarfert, 1995; Viswanathan and Al-Senafy, 1998; Mousavi and Rezai, 1999; Krishnamurthy et al., 2000). A limiting factor in developing artificial recharge of groundwater is site selection and lack of suitable site causing neglect of artificial recharge as a water management technique (Margaret et al., 1986).Therefore, in the first stage the efficiency of any recharge project is a function of physical characteristics of surface, sub-surface and its surrounding.

The Baghmelak agricultural plain, which is characterized as a relatively flat surface of moderately porous unconsolidated sediments, falls in temperate climatic condition and groundwater is often the only source of water supply for drinking as well as agriculture. The Baghmelak aquifer is one of the promising groundwater reservoirs in Khouzestan province in the south-west of Iran, but withdrawal of water from the aquifer is more than the rate of recharge. The growing demand for groundwater has increased contineously since at least a decade when farmers started drawing water from wells through mechanical pumping for agricultural activities and caused groundwater level depletion. The increasing demand for groundwater resources for various purposes and continuos water table drawdown (on average 7 m from 1994 to Aug 2003) in the study area necessitate integrated water resource management and artificial recharge of groundwater might help to some extent this approach. In addition to groundwater decline, artificial recharge site limitation is another challenge facing the study area. Due to lack of promising site in the course and adjacent to the main ephemeral tributaries (Galal and Paderazan) flowing in the area, investigations

were mostly focussed on three moderate ephemeral tributaries. Therefore, in order to augment groundwater supplies of the study area to some extent an integrated studies including surface and sub-surface investigations, water resource availability, water quality, infiltration capacity, hydrodynamic parameters were taken into account.

Based on the collected data three artificial recharge site options were selected in downstream of the Tapeh television (1), Kayani (2) and Sharkat (3) tributaries in the northeast and the north of the area and the general location map is depicted in Figure1. The catchment area of the above tributaries is small, but these are the only alternative, which relatively meet requirement for artificial recharge in the area. The recharge rate that could be achieved at the three sites is approximately 2.2 million m^3 per year. The estimated cost of surface water infiltration using basins and check dams for the first years of operation is relatively expensive but it is a reasonable way to augment groundwater supply in the long term.

The main purpose of this study was to determine the potential sites for artificial recharge of groundwater in the Baghmelak plain and the other objectives were:

1. To determine the suitable artificial recharge method to augment groundwater
2. To compute the annual rate of recharge from water retention schemes
3. To estimate cost of the water retention structures.
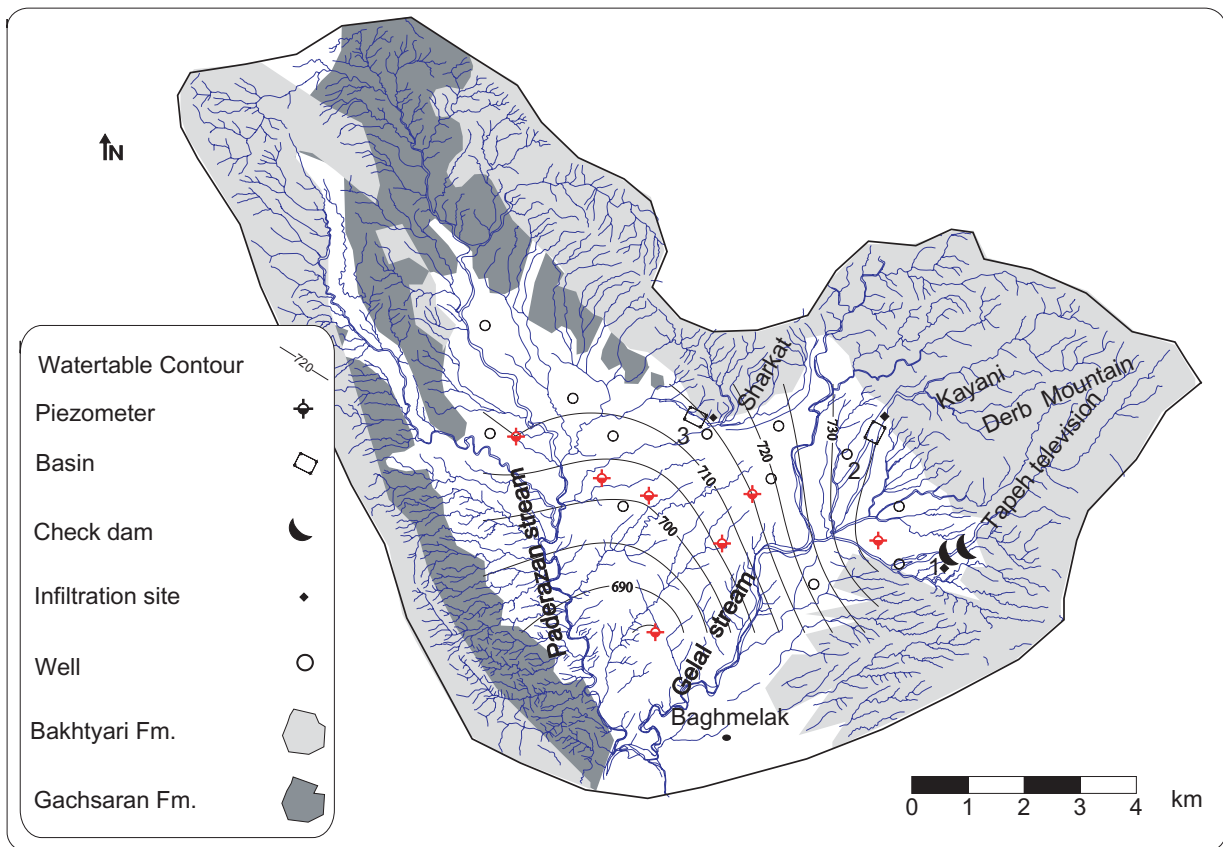
Figure 1. Location map of the study area

GENERAL DESCRIPTIONS

The Baghmelak plain is a semi- enclosed, embraced by surrounding mountain and lies in zagros structural belt. The study area occupies approximately 55 km^2 and is situated between 49° 42' to 50° 8' E longitude and 31° 26' to

$31^{°} 41^{'}$ N latitude in the northeast of Ahvaz in Khouzestan province. The average elevation of the Baghmelak plain is 725 m and receives an average annual rainfall of 640 mm. The minimum and maximum temperature is respectively 7 and 37 °C and the overall climate is temperate. The absence of high physiographic difference and presence of the fertile soil in parts of the study area makes it agriculturally rich.

The ephemeral streams known as Tapeh television (1), Kayani (2), and Sharkat (3) are rising from Derb conglomerate mountain and the former two (1 and 2) join the Gelal ephemeral stream while the later (3) is tributary of the Paderazan ephemeral stream. The drainage system is dendritic and the summarized characteristics of the Tape television, Kayani and Sharkat watersheds are given in Table 1.

Table 1. Physical characteristics of the watersheds

Watershed	A (km^2)	L (km)	S $(\%)$	BR	D (km/km^2)	H (m)
1	4.1	4.52	19.4	3.61	2.95	1325
2	5.9	5.02	15.2	2.57	3.06	1082
3	3.35	3.66	3.4	2.96	5.1	780

A = Area; L = Length: S = Net slope: BR = Bifurcation ratio: D = Drainage density: H = Average height

Geological maps and field information were used to identify the lithological units, which is depicted in Figure 1. The different rock units in the north and the northeast are Bakhtyari conglomerate formation (Pliocene), and in the remaining parts are covered by Gachsaran gypsum formation (lower Miocene). Major structural features observed in the northeast of the area are lineaments that are controlling the permeability of the conglomerate formation.

The Baghmelak plain in the north and the northeast is covered by Quaternary gravel, which is varying in size, and the remaining parts are mainly sandy loam type soil. The tube well litologs distribution indicates that the sediments are predominantly gravel but interbeded with mixed sediments at different depths. The depth to water table ranges from 12 to 37 m, thickness of the aquifer ranges from 55 to over 75 m and the bedrock is conglomerate and gypsum. The tube wells were constructed with a depth range from 70 to 110 m and the wells yield ranges from 17 to 45 L/s where the higher yields are found to be in the north and northeast. Groundwater movement is dominantly from the northeast and northwest towards the center. The annual amount of groundwater abstraction and recharge varies and depend on rainfall, but the estimated groundwater recharge from the alluvial plain, highly fractured conglomerate in the northeast and gypsum and conglomerate formations in the northwest in the year 2003 were respectively 4.2, 3.4 and 1.7 million m^3. In the same year, groundwater abstraction from 47 bore wells of the area was about 12.5 million m^3.

RESULTS AND DISCUSSION
Infiltration test and hydraulic parameters

As infiltration (I) reflects true upper soil layer drainage capacity (Abu-Taleb 1999), therefore for each location several infiltration tests were conducted with double- ring infiltrometer as well as surface ponds. Accurate determination of porosity is also crucial for recharge water but due to lack of facility in present investigation with respect to nature of geological material (sandy gravel), total porosity (n) was estimated (Table 2). In order to obtain hydraulic conductivity (K) in the selected sites inverse auger method was used. The pumping test data in alluvium were analysed by Boulton and Strelestova method (1975) for estimation of transmissivity (T) and storage coefficient (S).

Table 2. Physical characteristics of surface and sub-surface materials

Sites	I (mm /h)	n (%)	K (m/h)	T (m^2/day)	S
1	35	38	14.5	740	0.17
2	31	38	13.8	650	0.15
3	32	38	16	745	0.18

Runoff

The mean annual rainfall of the area as a whole is about 640 mm and the process of precipitation is stormy as well as continuous rain, the former has characteristics of short duration and high intensity, while the later is of long duration and low intensity. The runoff estimation is a prerequisite for design of infrastructures in artificial recharge schemes, but generally undertaken tributaries do not possess runoff data. Therefore, in such circumstances if the rainfall records of fairly long periods are available, the resulting runoff can be derived from empirical formula (Eq. 1, 2 and 3). For determination of annual run off in the study area experimental method, which is based on annual flow deficiency (D), annual rainfall of the basin in meter (P), watershed average temperature of watershed (T) in degree centigrade and the run off height in meter(R), was used .The annual discharge of the three selected ephemeral streams is depicted in Table 3.

$$D = P - \lambda P^2 \qquad (1)$$

$$\lambda = 1/0.8 + 0.14T \qquad (2)$$

$$R = P - D = \lambda P^2 \qquad (3)$$

Table 3. Annual discharge of the ephemeral streams (m^3)

Tapeh television(1)	Kayani (2)	Sharkat (3)
1.46×10^6	1.43×10^6	9.23×10^5

Water quality and suspended load

The surface and groundwater are classified as calcium bicarbonate and electrical conductivity (EC) of water resources ranged from 126 to 1,752 micro-μhos/cm in the north and northeast. The suspended loads of surface water samples were measured and analysis of mixed sediment by master sizer device indicated varying percentage of sand (39) silts (47) and clay (14). The measured average weight of runoff suspended loads in sites 1, 2 and 3 are respectively 132, 143 and 123 mg/L and the representative physico-chemical parameters are presented in Table 4.

Table 4. Physico-chemical parameters

Sample No.	T (°C)	EC (μhos /cm)	pH	$HCO3^-$ (mg/L)	$SO4^{-2}$ (mg/L)	Cl^- (mg/L)	Ca^{+2} (mg/L)	Mg^{+2} (mg/L)	Na^+ (mg/L)
1 SW	14	147	7.54	81	11	12	28	6	11.5
2 SW	13.6	150	7.52	86	13	13.7	32	7.3	14.5
3 SW	14	126	7.46	75	10.5	11	24	6.9	9.5
4 GW	23.5	481	7.6	202	72	22.7	76	21	18.5
5 GW	24	1,752	7.7	189	701	24.8	294.5	51	17.5

W = Surface water; GW = Groundwater.

Artificial recharge methods

Based on the collected data two types of planed artificial recharge including infiltration basin and check dam was suggested to conserve the water contained in flash flood of the ephemeral streams. The Tapeh television check dams with storage capacity of approximately 160,800 m^3 was suggested across the stream. The dimension of the masonry trapezoidal check dam is 60 x 3.5 x 4.5 m (Length, width and height). On the eastern side of the Kayani ephemeral stream three infiltration basins of 350 x 100 x 2.5 m suggested. On the down stream where the width of the Sharkat stream is reducing two infiltration basins of 250 x 150 x 2.5 m was also suggested. The average depth of water in the basins and backwater in the check dams are respectively 2 and 3.35 m.

Discharge time of impounded water

The required time for the total discharge of impounded water in the recharge sites was computed (Table 5) from the following relation and in this computation the vertical permeability (Vmax) was considered to be 70 % of the basic infiltration (I).

T = V/ A * Kvmax   T = Discharge time
V = Volume of water in the basins and back water in the check dam
A= Area of the reservior
Kvmax = Vertical permeability

Table 5. Parameters of discharge time computation

Sites	T (day)	V (m^3)	A (m^2)	Kvmax (m/h)
1	5.6	8 x 10^4	2.4 x 10^4	2.45 x 10^{-3}
2	3.8	7 x 10^4	3.5 x 10^4	2.17 x 10^{-3}
3	3.7	7.5 x 10^4	3.75 x 10^5	2.24 x 10^{-3}

Evaporation

An analysis of evaporation records from the evaporimeter installed at the Baghmelak meteorological station indicated that the average rate of evaporation was 1.49 mm/day from December to March. Therefore, on the bases of discharge time, evaporation rate from the recharge basins as well as backwater impoundment of the check dams during annual intakes was estimated (Table 6).

Recharge estimation

The quantitative estimation of recharge rate from surface reservoirs (recharge basins and check dams) to unconfined aquifer carried out through the following steps.
1. The actual design capacity of the sites (1,2 and 3) are respectively 1.7 x 10^5, 2.1 x 10^5 and 1.5 x 10^5 m^3.
2. In view of rainfall records, high intensity rainfall producing considerable run off occurs 5 to 6 times a year.
3. For computation of recharge water by the schemes, 5 flood events was taken into account.
4. It is assumed that in the first intake of the reservoirs 1 x 10^5, 1.3 x 10^5 and 1.1 x 10^5 m^3 are used to saturate the aeration zone below the recharge sites.
5. The percolated water volume of the recharge infrastructures was obtained from the difference of reservoir capacities and the computed evaporation rate.
6. It is estimated that the total annual recharge water flux into groundwater storage from these three sites to be about 2.2 million m^3.

Table 6. Evaporation and discharge rate from the recharge sites

Sites	Evaporation (m^3)	Recharge volume (m^3)
1	0.1×10^5	0.7×10^6
2	0.3×10^4	0.9×10^6
3	0.2×10^4	0.6×10^6

Cost

In view of tariff 2003 of planing and finance organization of Iran, the fixed costs including design, construction, hydraulic structures and management plus miscellaneous cost of the Tape television check dams, Kayani and Sharkat basin type of artificial recharge schemes are respectively 79 764, 80 588 and 52 882 American dollar.

CONCLUSIONS

Based on the surface and subsurface investigations recharge sites were selected and implementation of such infrastructures facilitates to increase groundwater storage to some extent. The total annual recharge water is about 2.2 million m^3 and mitigation of destructive floods is the side benefit. The cost of one cubic meter of infiltrated water as compare to construction cost in the short term is expensive but it is the reasonable and economic way of storing groundwater in the long term.

ACKNOWLEDGEMENTS

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Management of river bank filtration in the Elbe River Basin near Torgau, Germany

M. Krüger, K. Ende and T. Grischek

Abstract

A site-specific, scientifically based and long-term monitoring of bank filtrate and groundwater quality is the basis for an effective management of bank filtration sites and for the prediction of changes in water flow and quality. Results from a monitoring programme of more than 10 years at a bank filtration site near the city of Torgau prove a good and sustainable quality of the pumped raw water. At all sites with long flow paths, mixing ratios of bank filtrate and groundwater were found to be of main importance for the concentration of DOC, nitrate, sulfate, dissolved iron and manganese in the raw water. Thus, management of bank filtration schemes should be incorporated into wider catchment planning in order to limit potentially polluting activities in the groundwater recharge area.

Keywords

Bank filtration, Elbe River, water management, water quality monitoring.

INTRODUCTION

In Europe, over a century of experience exists in the operation and maintenance of river bank filtration (RBF) schemes. The company 'Fernwasserversorgung Elbaue-Ostharz GmbH (FWV)' was founded in 1946 as a bulk water supplier and manages four bank filtration waterworks in the Elbe River Basin near the city of Torgau, about 90 km south of Berlin, Germany. Two other waterworks produce drinking water from groundwater and reservoir water. In 2004, the drinking water production in these six waterworks was 80 Mm³, with 37.1 Mm³ of the total production taken from bank filtrate. The company supplies about 3 million people through a 700 km long distribution network.

RIVER BANK FILTRATION NEAR TORGAU, GERMANY

The abstraction of bank filtrate is of vital importance as a raw water source for the company. As an example, the waterworks Torgau-Ost, situated south-east of the city of Torgau, is presented in the following sections. The waterworks has a production capacity of 100,000 m³/day. The 42 production wells are located in an alluvial sand and gravel aquifer with a thickness of 40 to 60 m, covered by a 2 to 5 m thick layer of meadow loam. The meadow loam provides an important protection against pollution and infiltration e.g. during flood events. Due to erosive conditions in the river, there is only low clogging of the riverbed. The good hydraulic connection between the River Elbe and the adjacent aquifer ensures stable water abstraction even during low flow conditions.

The proportions of bank filtrate and groundwater in the pumped raw water vary between 45% and 65%. At most wells, the mean proportion of bank filtrate is higher than 50%. Retention times of bank filtrate in the aquifer range from 80 days to more than 300 days.

Raw water is purified by aeration and deacidification, pre-purification by sedimentation basins, and fine purification by open sand filters (Fig. 1). Iron, manganese and carbon dioxide are the main constituents of raw water that require treatment. These constituents are eliminated by adding hydrated lime. Prior to pumping into the public water supply network, the water is further treated with small amounts of chlorine and chlorine dioxide to prevent bacteriological deterioration during the transportation process to the consumer.
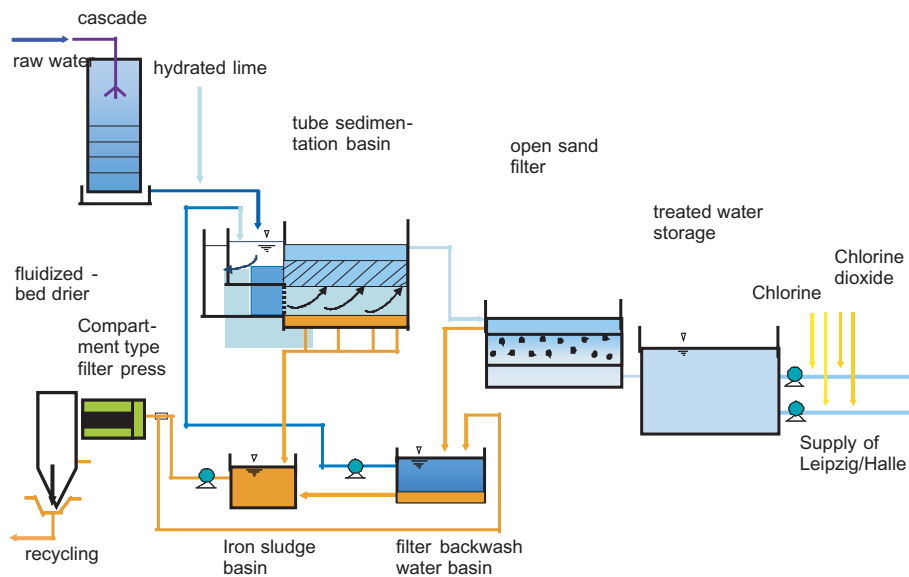
Figure 1. Technological scheme of the waterworks Torgau-Ost

MONITORING OF WATER QUALITY

Two monitoring profiles have been installed between the river and the extraction well at 300 m from the shore (location see Fig. 5), to measure the changes of the quality of the river water along its flow paths (Nestler et al., 1996). Next to this, sampling points are present at the landside of the wells to measure the quality of the approaching groundwater inside the wider capture zone for each well group separately. Up to five sampling depths were available at various spots (Fig. 2).

Between 1992 and 1998 intensive studies were carried out to investigate groundwater flow, retention times, the behaviour of dissolved organic carbon, major cations and anions and relevant organic trace compounds. To distinguish between bank filtrate and groundwater, investigations included measurements of chloride, ethylenediaminetetraacetate, persistent aromatic sulfonate and pharmaceutical and ^{18}O concentrations (Nestler et al., 1998). Since 1998 a routine monitoring programme has been established to monitor water flow and quality. The frequency of water sampling and the extent of the hydrochemical analyses and parameter spectrum were defined depending on retention times of bank filtrate in the aquifer. Hence, monthly, quarterly and biannual sampling is organised for selected observation wells along the flow path.

The results of the monitoring programme allow for short-, middle- and long-term control of bank filtrate quality as well as a quick response to extreme events. Data gained from more than 10 years of water sampling is used to show trends in water quality of the bank filtrate and changes in water flow.
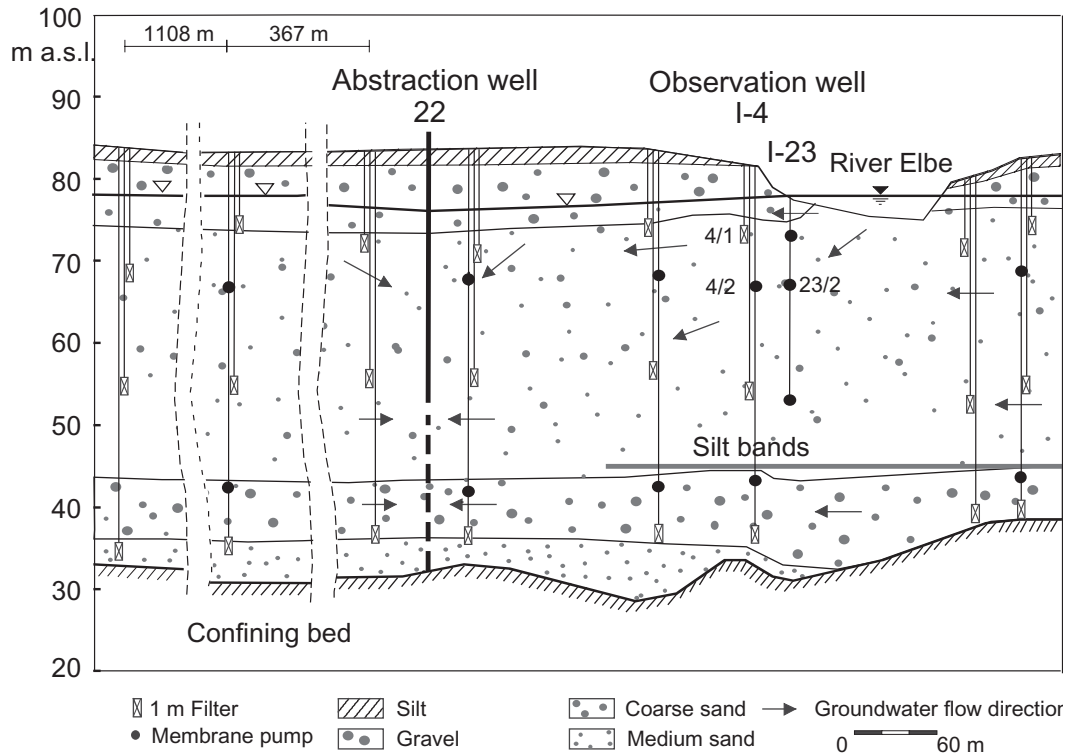
Figure 2. Cross-section Torgau-Ost A

The dissolved organic carbon (DOC) figure is of primary interest at the waterworks. Up to 50% of the decrease in DOC concentration occurs in the first meters of the flow path (Fig. 3). A significant attenuation of DOC along the flow path was observed for the time period 1994 to 2004. The DOC concentration in the pumped raw water was 2.2 to 2.4 mg/L (n=140) whereas the DOC concentration in River Elbe water was about 5.5 mg/L (n=157). Near the river bank some changes in the attenuation of DOC can be seen in Fig. 3.
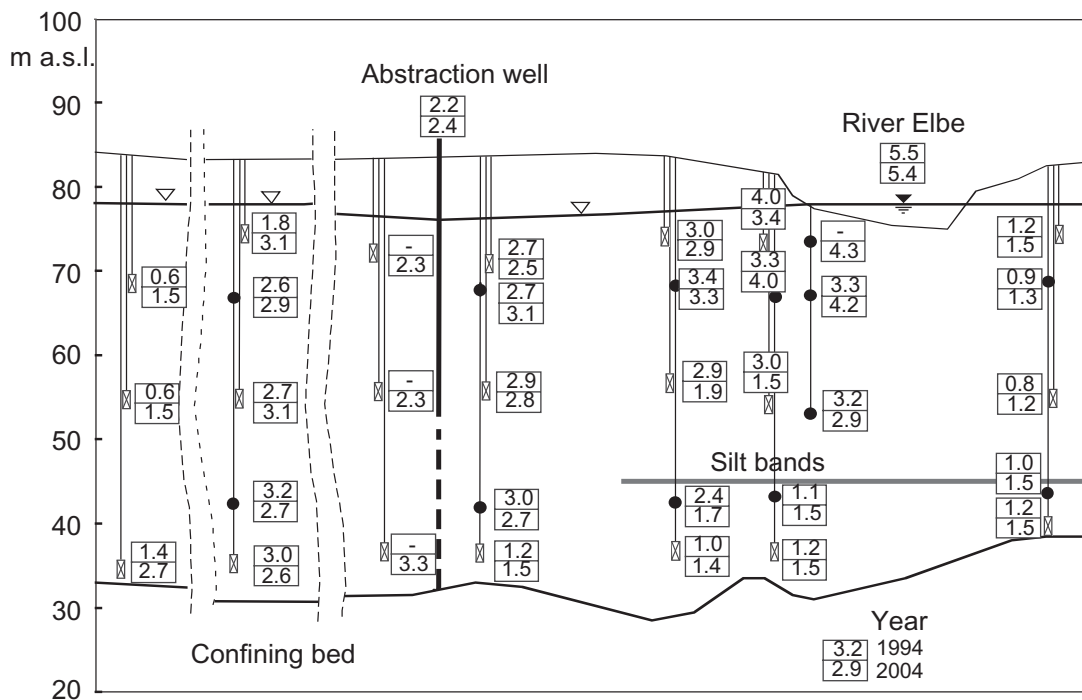
Figure 3. Mean DOC concentrations (mg/L), cross-section A, 1994 and 2004

Whereas the DOC concentration at observation well OW 4/1 is slightly decreasing from 4 mg/L to 3.4 mg/L (see also Fig. 4), there is an increase at well 4/2 and 23/2. This indicates the possible changes of DOC concentrations near the river bank. A conclusion about the relation between the hydraulic conductivity (which is higher in the uppermost layer) and observed changes in DOC concentration can not be drawn yet. A general increase in DOC concentration is observed for the landside groundwater but not for bank filtrate. Some changes can be explained by the different number of exploited samples.

Results from investigations during floods are of special interest to the company to clarify the risk of floods affecting raw water and drinking water quality. The extreme flood event in August 2002 caused a strong increase in turbidity, organic carbon concentration, and number of microorganisms in the River Elbe water. An increase in DOC from 5 mg/l to more than 8 mg/l in river water did not affect the DOC concentration in the abstracted water (Fig. 4). Biodegradation and mixing in the aquifer prevented an increase in DOC concentration in the raw water and an increase in the concentration of trihalomethanes during disinfection. The quality of abstracted bank filtrate was never at risk during the flood in August 2002 due to the design of bank filtration sites and long flow paths and retention times (Krueger and Nitzsche, 2003).
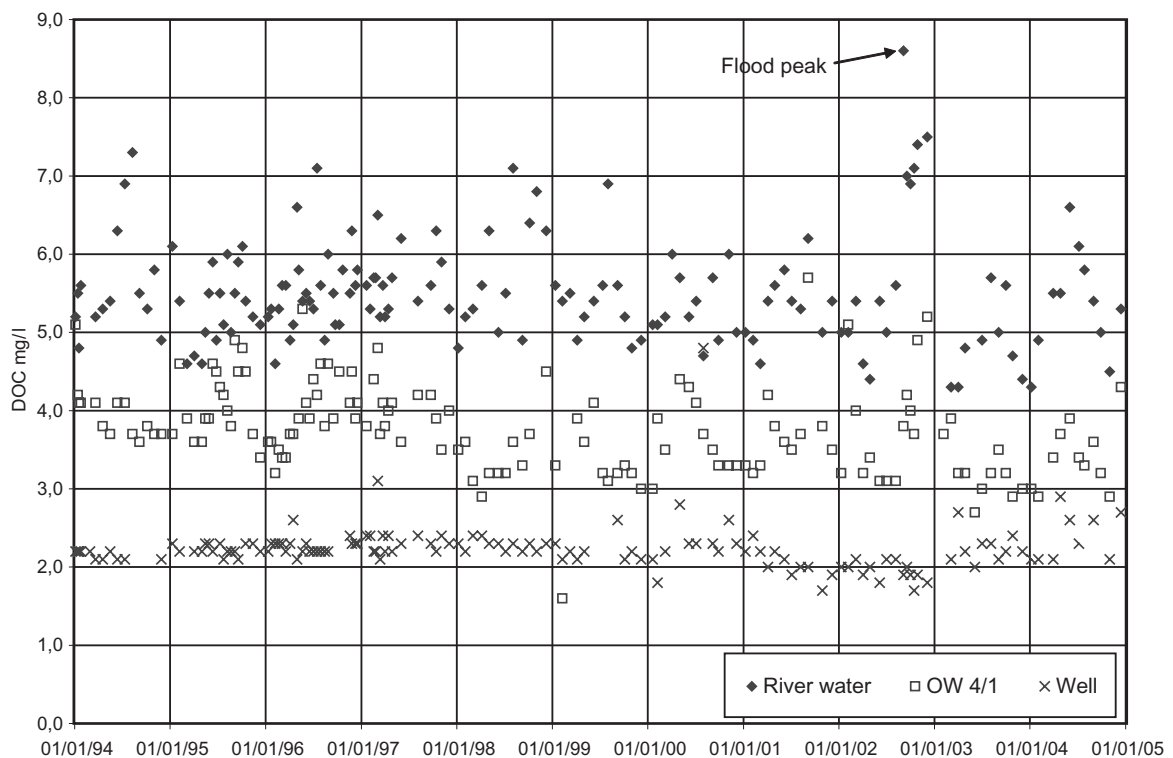
Figure 4. DOC concentration (mg/L) in River Elbe water, bank filtrate, and raw water, 1994–2004. For location of observation well 4/1 and abstraction well see Fig. 2.

MANAGEMENT

The management of the waterworks depends on various limiting factors. Satisfying the fluctuating water demand is of prime importance which, however, is often opposed to the preferred continuous operation of wells. Another important factor is the mixing of river bank filtrate and groundwater to obtain an optimum quality with regard to raw water treatment. The main aims of water quality management include the maximum attenuation of organic compounds during aquifer passage and low concentrations of DOC, dissolved iron and nitrate in raw water.

The resulting well management is therefore a compromise and an ongoing optimization procedure regarding abstraction regime, quantity of abstraction, number of wells operated and the water quality of operated wells. To obtain comparable data, the production wells situated within the sampling profiles at Torgau-Ost are operated almost continuously. This is of advantage for lower concentrations of dissolved iron in raw water. Furthermore, periodic well operation stimulates well clogging. The operation of the number of wells at Torgau-Ost depends on the drinking water demand at a given time.

Due to the long distance between the wells and river bank (> 250 m), a change in the well filter depth has only negligible effects on groundwater flow and quality (Grischek, 2003a).

The most important factor in water quality management is the selection of abstraction wells according to their land-side groundwater catchment zone. Based on a dense net of observation wells, it was possible to identify zones with different concentrations of DOC and dissolved iron in landside groundwater. Figure 5 shows the range of DOC concentrations in groundwater in the catchment zones behind the wells.
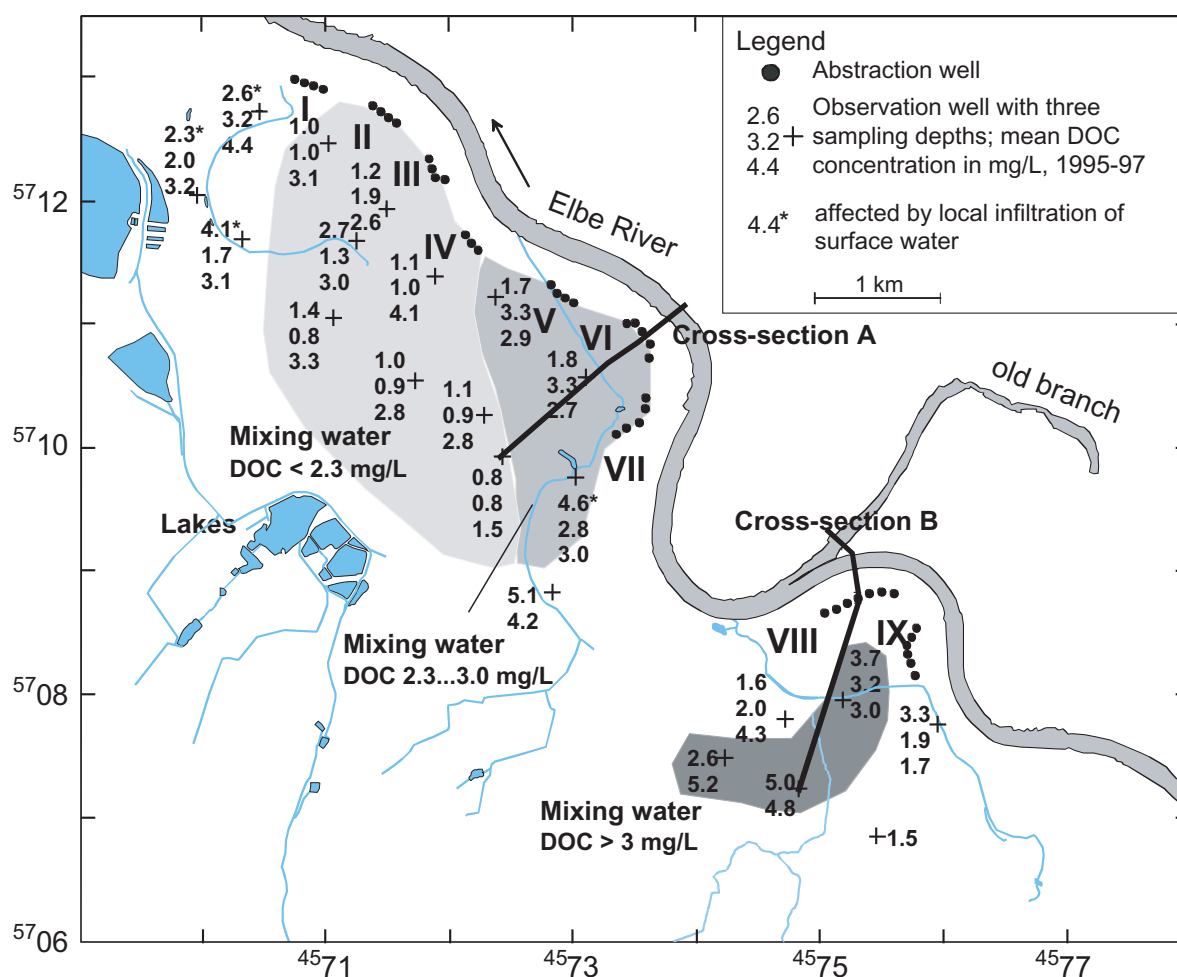
Figure 5. Mean DOC concentration in mg/L in groundwater
in the catchment zone of the waterworks Torgau, 1995–1997 (n > 6).

The mean concentration in the mixed water (see Fig. 5) was calculated from the concentrations determined at different depths in the aquifer. Depth-dependent groundwater sampling was a key factor in understanding groundwater flow and quality changes in the catchment zone. The operation of Well Groups II to IV results in lower DOC concentration in the raw water compared to the operation of Well Groups VIII and IX. Fortunately, low DOC

concentrations are associated with low dissolved iron concentrations. Thus, the advantage of appropriately selecting abstraction wells covers both parameters. Since the water quality change for bank filtrate was very similar in all wells, an improvement of raw water quality can be achieved mainly by the selection of wells abstracting the proportion of landside groundwater with the best quality. At the Torgau waterworks, the preferential operation of these wells has already resulted in cost savings, especially for the removal of dissolved iron during the water treatment process that requires iron sludge disposal (Grischek, 2003b).

Another method was chosen at the Mockritz waterworks, located north of the city of Torgau. To adapt the operation of wells to the decreasing water demand, to allow almost continuous well operation and to use the whole aquifer as a reactor for pre-treatment, the installed pump capacity of the wells was reduced.

CONCLUSIONS

A site-specific, scientifically based and long-term monitoring of bank filtrate and groundwater quality is the basis for an effective management of bank filtration sites and the prediction of changes in water flow and quality. The understanding of the pattern of contaminant migration and location of biogeochemical reaction zones is a prerequisite to an effective determination of pumping strategies to optimise the system for best quality at lowest cost. The main aim of using river bank filtration as a pre-treatment of River Elbe water is the attenuation of organic compounds and low concentrations of DOC. Long flow paths and retention times promote the attenuation of organics, but were found to have only a relatively small effect on iron and manganese concentrations. From a water quality point of view, continuous pumping of selected wells is preferred over periodic operation.

At all sites with long flow paths, mixing ratios of bank filtrate and groundwater were found to be of main importance for the concentration of DOC, nitrate, sulfate, dissolved iron and manganese in abstracted water. Due to spatially varying groundwater quality within the whole catchment zone, a selection of wells having a catchment zone with good groundwater quality at the land side offered a significant improvement in raw water quality. A detailed investigation of groundwater flow conditions and proportions of bank filtrate in the raw water is very important for deciding effective water quality management measures. Thus, management of bank filtration schemes should be incorporated into wider catchment planning in order to limit potentially polluting activities in the groundwater recharge area.

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Statistical description and analysis of a bank filtration system

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Abstract

The transect of the bank filtration site at Lake Tegel is characterized with regards to their redox conditions using a Cluster analysis. Four different groups of observation wells could be found, enabling the derivation of a redox zoning with horizontal boundaries, which are moving downward during winter time. At the same site, Regression analysis served to examine influencing variables on the reduction of the pharmaceutical Carbamazepin during bank filtration. Two different regression models for summer and winter time were found, with each of them including the standardized temperature and the travel time as influencing variables. Whereas during winter time the redox conditions seem to have a significant influence on the reduction of Carbamazepin, the same influence could not be found for the reduction of Carbamazepin during summer time.

Keywords

Redox zones, pharmaceutical Carbamazepin, Berlin/Germany, Cluster analysis, Regression analysis.

INTRODUCTION

The statistical analysis introduced here is part of the NASRI project of the KompetenzZentrum Wasser Berlin (2001–2005). In the framework of this project, the transect of the bank filtration site at Lake Tegel was sampled and studied intensively by different research groups. The large amount of data received was summarized in one database, enabling a statistical analysis from an overall point of view.

The aim of this analysis was to complete or verify the interpretations given by the different research groups and to uncover connections not yet examined between different variables. The variables chosen to be included in the statistical analysis are the redox indicator variables O_2, NO_3, Mn, Fe, SO_4, NH_4 and Eh, the temperature, the estimated travel time and the pharmaceutical Carbamazepin.

Taking seasonal differences of bank filtration processes into consideration, the complete data set was split into two data sets, one containing bank filtrate from summer and one containing bank filtrate from winter. Summer and winter are defined according to hydrological seasons. The statistical analysis was then carried out separately for each of these two data sets.

METHODS

Two widely used multivariate statistical methods were applied to the two data sets: a Cluster analysis and a Regression analysis.

The aim of a Cluster analysis is to arrange multivariate data objects in groups with homogenous properties. Differences within the groups should be as small as possible and differences between the groups should be as large

as possible. The two fundamental steps of each Cluster analysis are the choice of a proximity measure and the choice of a group building algorithm (see Härdle 2003, Dillon 1984 for details). In this analysis the observation wells of the transect at the bank filtration site at Lake Tegel should be grouped with regards to their redox conditions. Thus, the redox indicator variables O_2, NO_3, Mn, Fe, SO_4, NH_4 and Eh served as variables for grouping the observation wells. For measuring the proximity between objects the squared Euclidian distance and the City Block distance were used. The groups were built by use of the Single Linkage, Complete Linkage, Average Linkage and Ward algorithms. For a comparison of the resulting groups, Parallel Coordinate Plots were used. Instead of plotting observations in an orthogonal coordinate system, the Parallel Coordinate Plot uses a system of parallel axes, where the variables are mapped onto the horizontal axis and their corresponding values on the vertical axis. The observations for one row of the data matrix are then connected over all variables, which means that one line represents one measurement.

A Regression analysis serves to examine connections between an endogenous variable Y and one or more exogenous variables X_j. Assuming a linear relationship between endogenous and exogenous variables, the regression model can be written as: $Y = \beta_0 + \beta_1 X_1 + ... + \beta_p X_p + u$, where u is a random error term. By means of this model and with the concentrations of Carbamazepin as endogenous variable, the reduction of Carbamazepin during bank filtration at Lake Tegel was analyzed. The estimated travel time, standardized temperature, Eh and Dummy variables for the redox groups S2, S3, W2 and W3 from Cluster analysis (the first group is the reference group) and for the infiltration year (the year 2002 is the reference group) as well as interactions between these variables were used as exogenous variables. The influence of groundwater mixing on the reduction of Carbamazepin was tried to be eliminated by the exclusion of observations from observation wells with a T/He-age above 6 month (see KompetenzZentrum Wasser Berlin 2003). The unknown parameters β_j were then estimated by ordinary least squares (OLS). Variables are selected using backward elimination and heteroscedasticity robust t-ratios. The robust estimation methods least trimmed squares (LTS) and least median squares (LMS) were applied to the resulting models to examine the impact of outlying observations on the resulting regression equations.

RESULTS AND DISCUSSION

The solution of the Cluster analysis for the summer time data is given in Figure 1. It shows a horizontal rather than a vertical redox zoning of the transect. Figure 2 contains the Parallel Coordinate Plot for this solution. In the first two zones O_2 and NO_3 are reduced wheras in the last two zones Mn and Fe are appearing in rising concentrations.

Figure 3 shows the solution of the Cluster analysis for the winter time data. Like for the summer time data, a horizontal rather than a vertical redox zoning is visible. The boundaries of the redox zones derived from the solution of the Cluster analysis for the winter time data lie below those derived from the solution of the Cluster analysis for the summer time data. Nevertheless, the four groups built by Cluster analysis for the winter time data show similar patterns like the groups for the summer time data, which can be seen by means of the Parallel Coordinate Plot in Figure 4.

Regression analysis led to two different models for the summer and the winter time data. The variables included in these models as well as their coefficients with corresponding t-ratios contain Tables 1 and 2. The coefficients of the Regression model for the summer time data have the same sign after each estimation method, which means they characterize quite well the influence of the exogenous variables on the reduction of Carbamazepin. By way of comparison, the coefficients of the Regression model for the winter time data for the variables D_2003, D_W3, D_2003*travel time and D_2003*temperature are changing their signs, estimating them by different estimation methods. Apparently, outlying observations mainly from the infiltration year 2003 lead to the presented regression model.
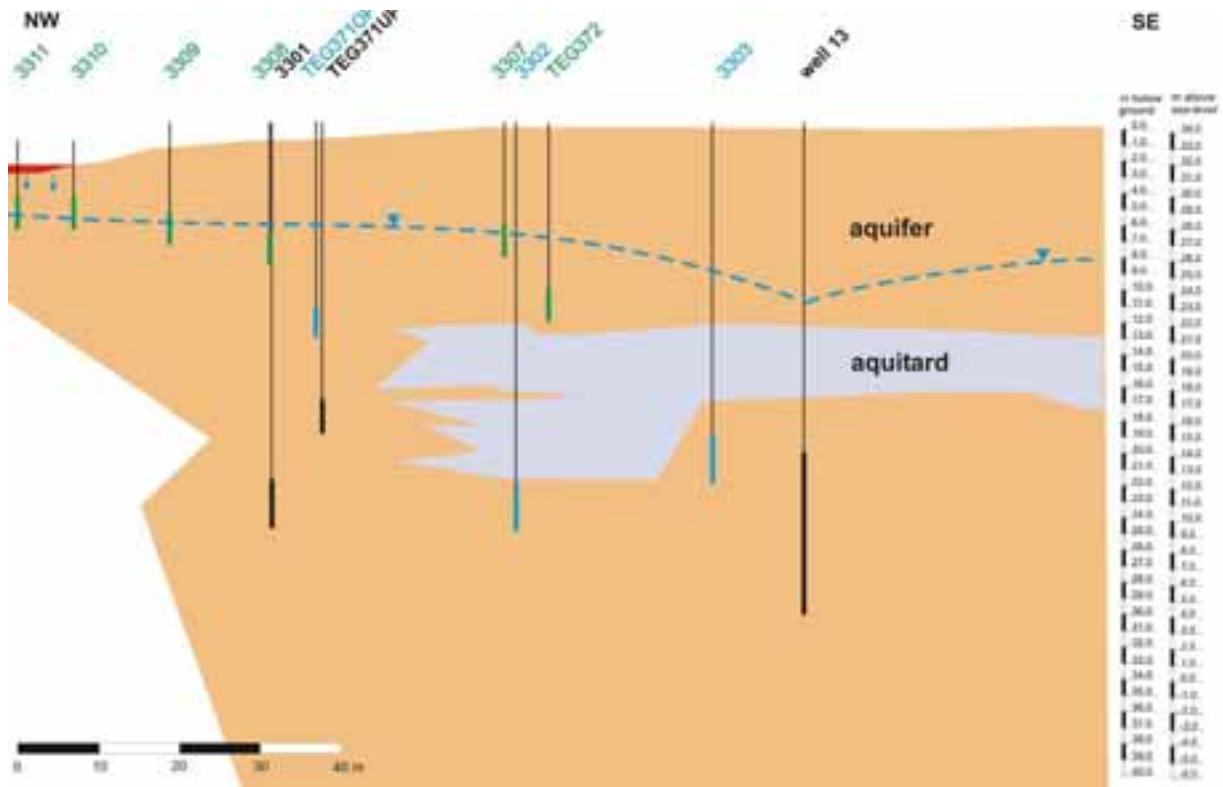
Figure 1. Solution of the Cluster analysis for the summer time data
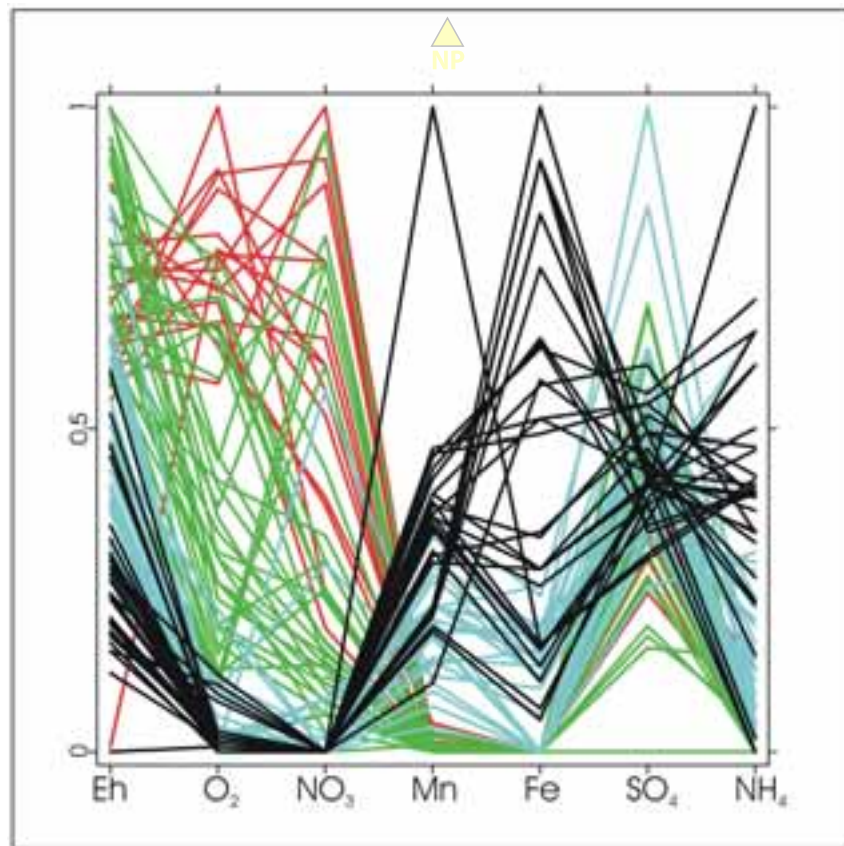
Figure 2. Parallel Coordinate Plot for the solution of the Cluster analysis for the summer time data
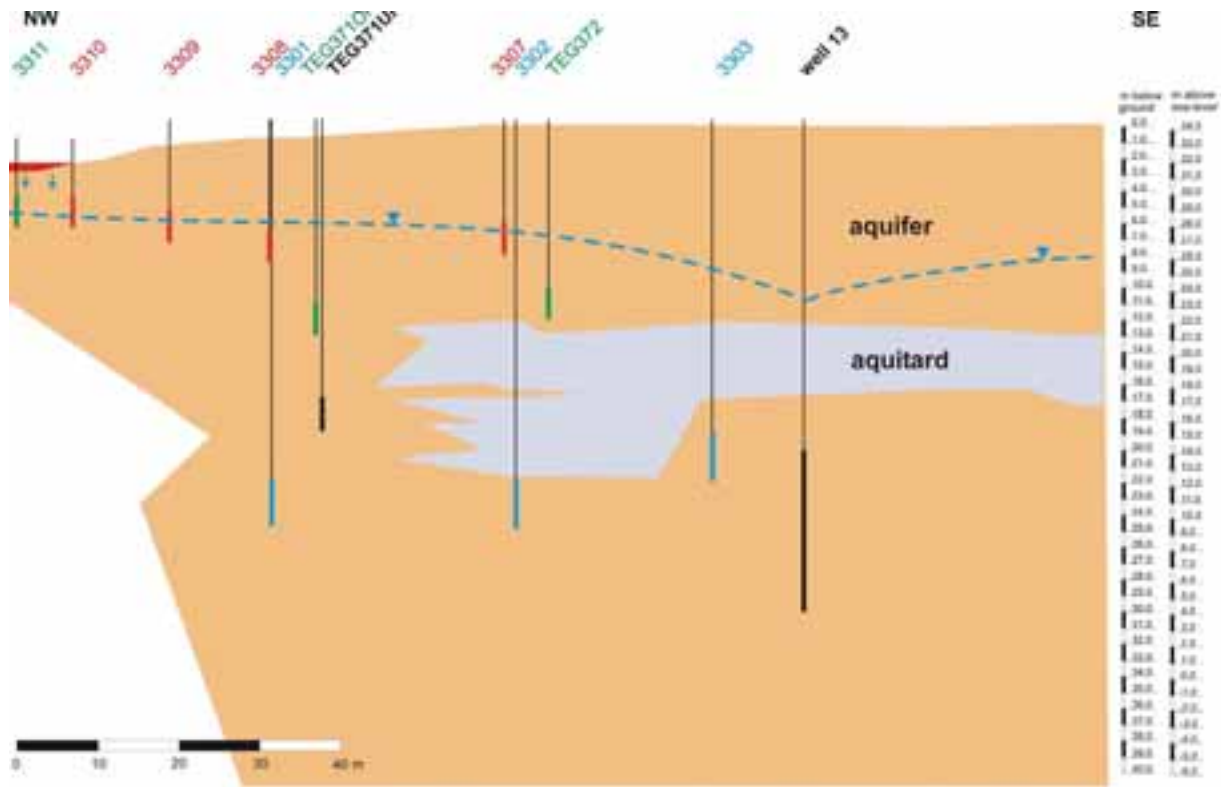
Figure 3. Solution of the Cluster analysis for the winter time data
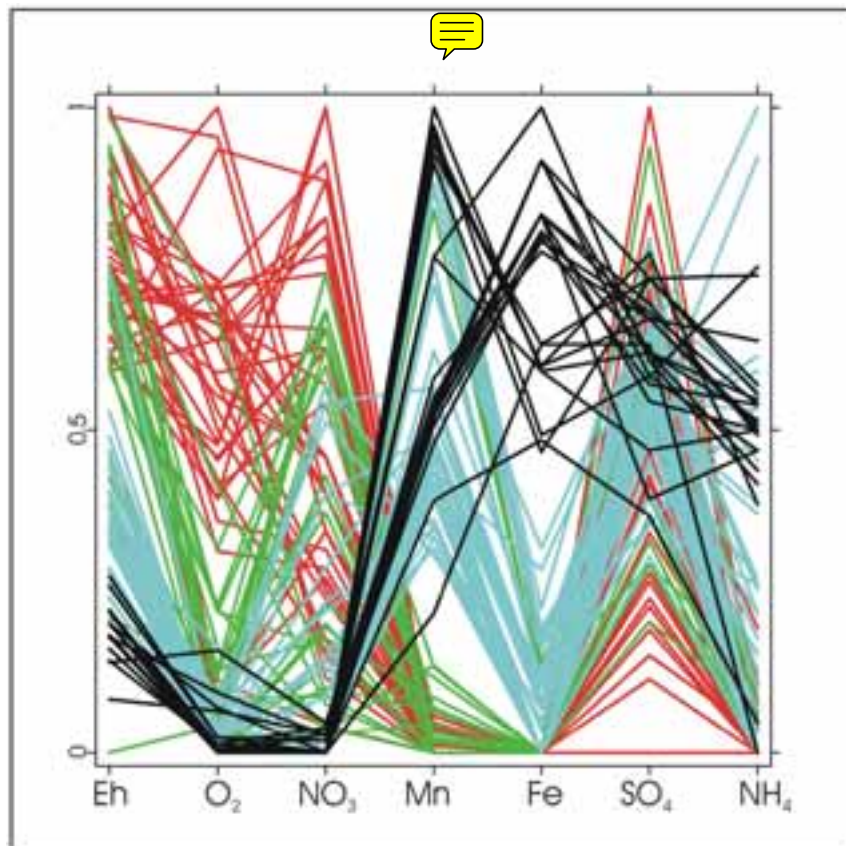
Figure 4. Parallel Coordinate Plot for the solution of the Cluster analysis for the winter time data

The coefficients of the Regression model for the summer time data can be interpreted as follows: negative deviations from the mean temperature go ahead with lower concentrations of Carbamazepin, higher concentrations of Carbamazepin were measured in the years 2003 and 2004 compared to the year 2002 and longer travel times in the year 2003 lead to a reduction of Carbamazepin. Notice that the coefficient for the travel time in year 2003 (D_2003*traveltime) is not significantly different from zero after the LTS- and LMS-estimation methods.

For the Regression model for the winter time data interpretations will only be given for the OLS-coefficients. As in the summer data Regression model, negative deviations from the mean temperature go ahead with lower concentrations of Carbamazepin and higher concentrations of Carbamazepin were measured in the years 2003 and 2004 compared to 2002. Longer travel times lead to a reduction of Carbamazepin, with positive deviations from the mean temperature additionally intensifying this effect. Eh as well as the Dummy variables for the redox groups W2 and W3 from Cluster analysis have a negative effect on the concentrations of Carbamazepin. Notice that after LTS only the coefficient for D_2004 and after LMS only the constant and the coefficients for temperature and D_2004 are significantly different from zero.

Table 1. OLS-, LTS- and LMS-coefficients and their corresponding t-ratios of the Regression model for the summer data set

Variable	OLS-coefficients	t-ratio	LTS-coefficients	t-ratio	LMS-coefficients	t-ratio
constant	340,935	16,445	272,720	9,660	258,710	10,021
temperature	57,347	3,921	64,758	3,580	56,704	3,114
D_2003	195,205	5,834	129,850	3,138	115,990	2,784
D_2004	152,293	4,344	231,870	5,347	214,000	4,902
D_2003*traveltime	−3,641	−7,164	−1,326	−2,110	−1,057	−1,670
D_2004*temperature	82,688	2,369	98,697	2,287	115,240	2,652

Table 2. OLS-, LTS- and LMS-coefficients and their corresponding t-ratios of the Regression model for the winter data set

Variable	OLS-coefficients	t-ratio	LTS-coefficients	t-ratio	LMS-coefficients	t-ratio
constant	498,611	6,177	212,900	1,701	362,050	3,084
temperature	181,031	4,375	87,789	1,370	177,610	2,954
Eh	−0,316	−1,878	0,153	0,587	−0,125	0,510
D_2003	392,268	6,367	−48,778	−0,510	−17,073	−0,190
D_2004	331,538	6,330	258,820	3,188	395,430	5,192
D_W2	−116,993	−3,518	20,272	0,393	−87,419	1,808
D_W3	−151,088	−2,899	−52,208	−0,646	−103,940	1,371
D_2003*traveltime	−6,244	−6,076	1,048	0,657	0,686	0,458
D_2004*traveltime	−4,560	− 4,353	−2,159	−1,329	−2,775	1,821
D_2003*temperature	199,360	3,980	−8,831	−0,114	−60,180	0,826
temperature*traveltime	−2,182	−3,742	−0,354	−0,392	−1,055	1,243

CONCLUSION

Cluster analysis showed the horizontal proceeding of redox reactions during bank filtration with downward moving boundaries of the corresponding redox zones during winter time. An influence of these redox conditions on the reduction of Carbamazepin could only be found for winter time data, which indicates a microbial reduction of Carbamazepin for low temperatures. Robust estimation methods do not support this result, so that further investigation is needed. Nevertheless, it could be proved that lower temperatures result in lower concentrations of Carbamazepin for the summer as well as for the winter time data.

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Evaluation of the hydrochemical conditions during bank filtration and artificial recharge in Berlin

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Abstract

Hydrochemical conditions were evaluated at both bank filtration and artificial recharge sites in Berlin. All bank filtration sites show a strong vertical age stratification. Rather than showing a typical redox zoning with more reducing conditions in greater distance from the surface water, the redox zones are horizontally layered, with more reducing conditions in greater depth. This is believed to be an effect of the strongly alternating groundwater-levels and by the age stratification. The redox conditions are generally more reducing at the bank filtration sites, mainly as a result of the longer travel times and operational differences. Redox conditions at all sites vary seasonally in particular at the artificial recharge site, which is mainly caused by temperature changes.

Keywords

Bank filtration, artificial recharge, redox conditions, biodegradation, age dating.

INTRODUCTION

The drinking water for the 3.4 million inhabitants of metropolitan Berlin originates from the local groundwater reservoirs. Production well galleries are located adjacent to the surface water system and artificial infiltration ponds, and 70% of the abstracted groundwater is bank filtrate or artificially recharged groundwater (Pekdeger and Sommer-von Jarmerstedt 1998). Hence, besides other factors, the quality of the abstracted water is influenced by hydrochemical changes occurring during the process of infiltration. Monitoring and understanding of these water quality changes is therefore of great importance in order to ensure the sustainability of the drinking water production system.

The natural attenuation of contaminants during bank filtration includes the elimination of suspended solids, particles, biodegradable compounds, bacteria, viruses, and parasites as well as the partial elimination of adsorbable compounds caused by biotic and abiotic processes such as physical filtration, biodegradation, adsorption, chemical precipitation, and redox reactions (e.g. Hiscock and Grischek, 2002). Redox changes during infiltration are of particular importance, since they cause the appearance of the undesired metals Fe^{2+} and Mn^{2+} (Bourg and Bertin, 1993), influence the behavior of a number of organic pollutants such as pharmaceutically active substances (Holm et al., 1995), halogenated organic compounds (Bouwer and McCarthy, 1993; Grünheid et al., 2004) and effect the pH and calcite solution capacity (Richters et al., 2004). Several authors have found that the most significant chemical changes related to organic matter degradation take place during the first few meters of flow (e.g. Jacobs et al., 1988; Bourg and Bertin, 1993; Doussan et al., 1997)

METHODS

In the past years, several transects were installed at representative sites in Berlin, following the flow direction from the lake or artificial recharge pond to production wells. The transects contain a number of observation wells of

various depths, usually one below the lake, some between lake and production well, and some beyond the well (example in Figure 1). Between May 2002 and August 2004, surface water, observation wells, and abstraction wells have been sampled monthly and analysed for a large set of parameters within the NASRI project (KWB, 2004). Analysis for anions and cations was done in the laboratories of the Berlin Water Company (BWB). Anions in water were measured with ion chromatography, DX 500 (Dionex Coop.). Cations in the water were determined with an ICP (OES) IRIS (Nicolet). The DOC measurements were carried out with a 'high TOC' TOC/DOC-analyzer (Elementar). The tritium-helium age dating method uses the ratio of the concentration of radioactive tritium (^{3}H, half-life 12.32y) and its decay product helium (^{3}He) in the groundwater to determine the groundwater age (Schlosser 1988). The ^{3}H concentration in surface waters is governed by current concentration in precipitation. As soon as the surface water infiltrates, the ^{3}He from ^{3}H-decay is trapped and accumulates in the aquifer. The analysis and data evaluation of the groundwater samples was carried out at the noble gas laboratory at the University of Bremen. Stute et al. (1997) and Beyerle et al. (1999) used this method to date bank-filtrates.

RESULTS AND DISCUSSION

Age stratification of the bank filtrate

A surprising result of the travel time evaluation was the fact that the age stratification within the uppermost unconfined, glacio-fluvial, sandy aquifer (Figure 1) is much larger than previously assumed. This is reflected in a number of observations. While the shallower observation wells at all sites reflect the seasonal changes of the lake-type rivers, the deeper observation wells show no seasonality at all. A determination of travel times with the peak shifts of tracer breakthrough curves was therefore not possible for these wells. In addition, the deeper wells differ strongly from the shallow ones with respect to some minor organic compounds, thereby reflecting the historical changes of the surface water composition (data not shown).
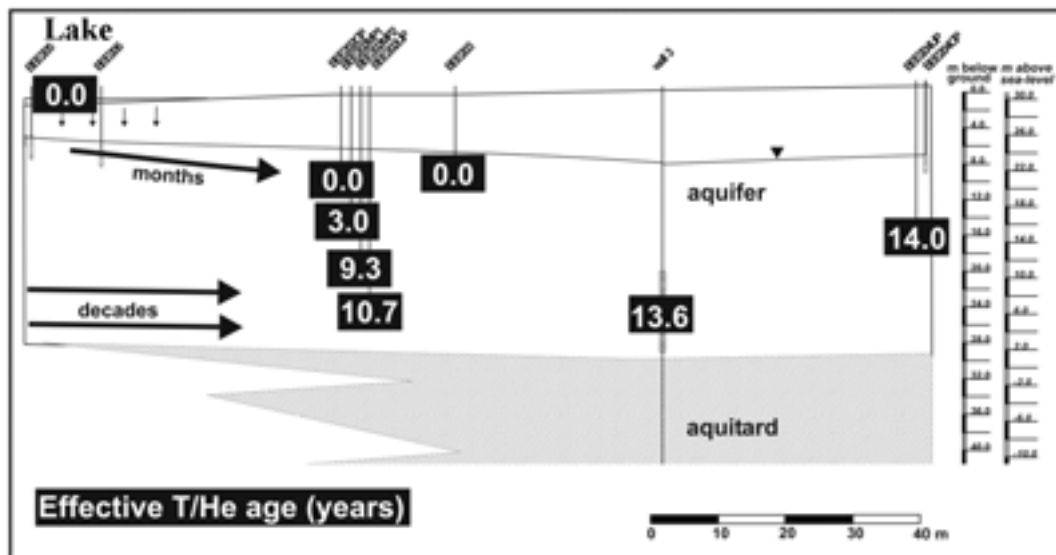
Figure 1. Mean T/He ages at the transect 2 at the field site Wannsee (uncertainty 0.5y or 10%)

The ^{3}H/^{3}He-dating finally confirmed that water abstracted from the deeper observation wells is decades rather than months old. In contrast, the ages determined for the shallower wells were modern, which, at these sites, is equivalent to a travel time of less than 6 months (resolution of the method). An example is given in Figure 1 for the transect 2 at Lake Wannsee. The effective ^{3}H/^{3}He-ages are plotted at the position of the filter screens. Production well 3 has a mean ^{3}H/^{3}He age of 13.6 years, which is the result of mixing of water from 3 different aquifers (the two deeper aquifers are not shown in Figure 1) from 2 sides. The groundwater from the inland side displays a similar age. The bank filtrate between the lake and well 3 in the upper aquifer parts has an age of less than

6 months. In the deeper parts, water ages increase up to 10.7 years. This indicates that bank filtrate must be flowing underneath the lakes, which are therefore not water divides. Intermediate ages are thought to be the result of mixing between these two sources.

Redox processes during bank filtration

Redox zones characterised by the disappearance of reactants (oxygen, nitrate, sulphate) or the appearance of reactants (iron, manganese) as suggested by Champ et al. (1979) were drawn into Figure 2. While the shallower (younger) wells contain oxygen and nitrate throughout most of the year, the deeper observation wells are free of these but contain manganese (Mn^{2+}) and iron (Fe ($^{2+}$).
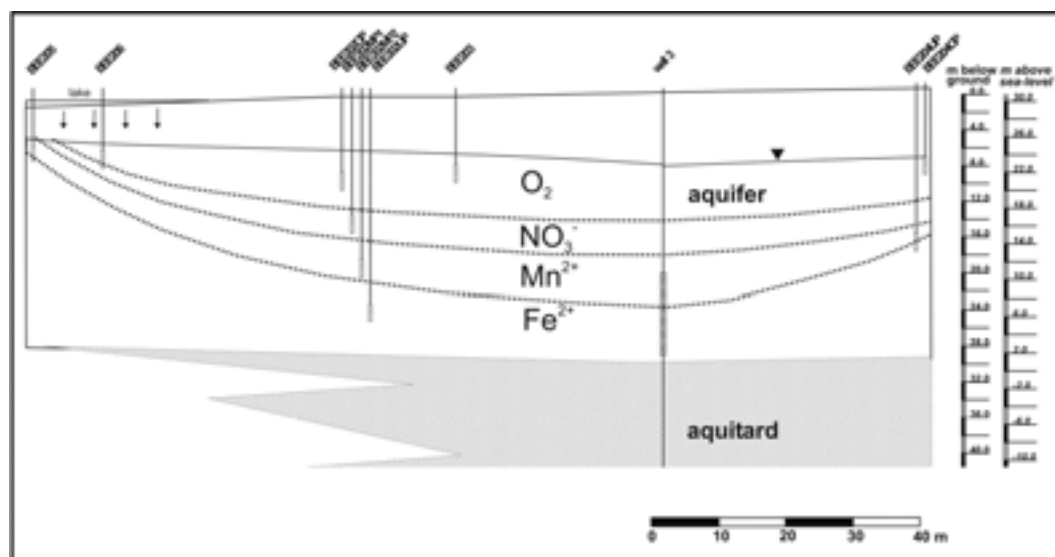
Figure 2. Approximate redox zoning at the bank filtration site Wannsee

The two observation wells in the shallow water are quite different from each other with respect to their hydrochemistry. While BEE206 closer to the shore is still aerobic, BEE205 is mostly oxygen and nitrate-free. The bottoms of Lake Tegel and Lake Wannsee are covered with very thick layers of mud with a very low hydraulic conductivity (Pachur and Röper, 1987). At the lake borders, the lake bottom is free of mud but the sands are strongly clogged. Hence, with increasing distance from the shore the hydraulic conductivities decrease and the organic carbon content increases, which results in longer travel times and more reducing conditions at BEE205 in comparison to BEE206. The vertical redox zoning may be a result of the different groundwater ages. However, it may also be caused by re-oxidation of more reducing bank filtrate by oxygen penetrating though the permeable unsaturated zone, possibly enhanced by the water-level fluctuations caused by the irregular pumping regime. Bourg and Bertin (1993) investigated biogeochemical changes during bank filtration at the Lot River. They observed a similar reduced anaerobic zone close to the river followed by aerobic conditions further along flow direction. They concluded that the observed redox processes were reversible and oxidation caused by aeration through the permeable unsaturated zone close to the river. Similarl to our findings, Richters et al (2004) describe a vertical redox zonation at a bank filtration site near the Rhein river.

However, these zones are not immobile and redox boundaries move seasonally. The younger bank filtrate (age < 3 months) undergoes strong seasonal temperature changes of up to 25 °C, depending on the distance from the lake. Because redox processes are microbially catalysed, these changes lead to differences in microbial activity (e.g. David et al., 1997; Prommer and Stuyfzand, in press). As a result, oxygen and nitrate disappear at times when temperatures are highest which is in summer or autumn, depending on the respective time lag to the well. It is interesting to note that nitrate concentrations in the lake itself decrease to zero in summer, probably due to consumption

by algae. The disappearance of nitrate in the observation wells therefore appears to reflect the input signal rather than a reduction in the aquifer itself (Figure 3).
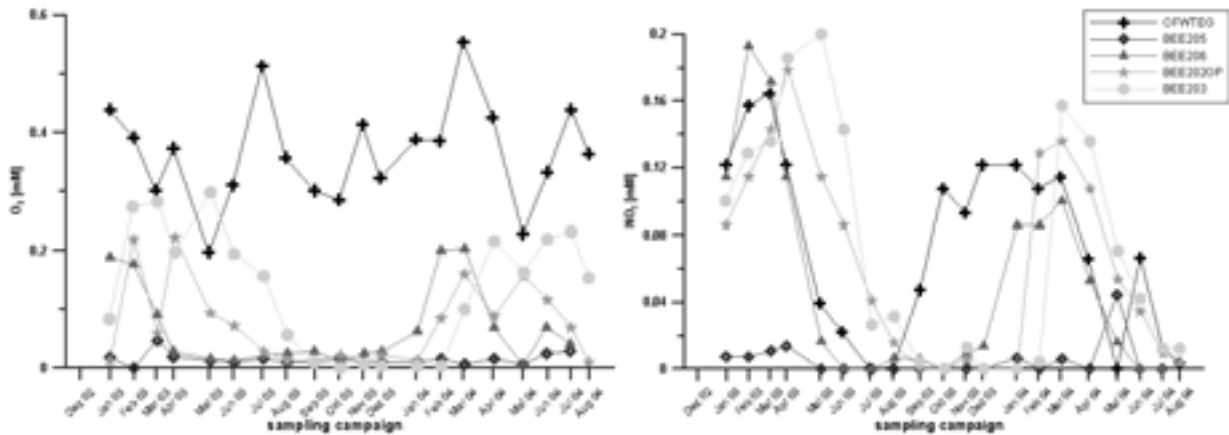
Figure 3. Time series for oxygen and nitrate in shallow observation wells

Hydrochemical changes during artificial recharge

At the artificial recharge (AR) site, redox zones show a similar seasonal variation to the bank filtration site (Massmann et al., this volume). The main difference between the 'natural' lake sites and the AR site is the way they are operated. At the AR site, the infiltration pond is cleared of the developing clogging layer regularly at intervals from weeks to 3 months, while the bank filtration sites are never redeveloped. The result is a cyclic regime at the artificial recharge site with variable saturated and unsaturated conditions directly below the pond (Greskowiak, in press). In Figure 4, pond water-level (a) and groundwater level (b), infiltration rate (c) and calcium concentrations in the pond and the two shallowest observations wells close to the pond (d) are given. The changes from unsaturat-
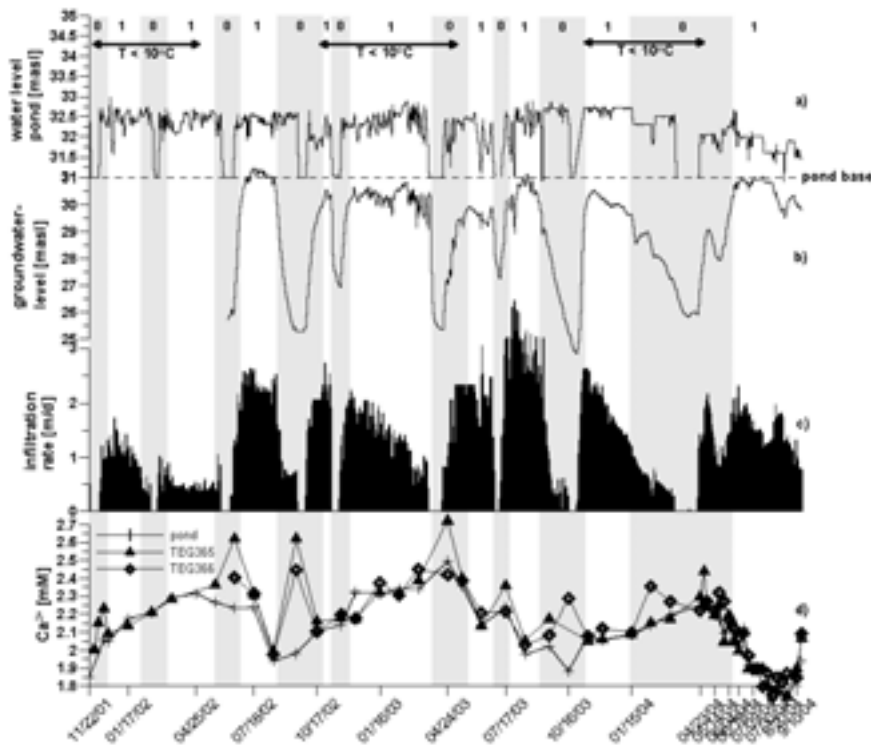
Figure 4: pond water-level (a) and groundwater level (b), infiltration rate (c) and calcium concentrations in the pond and the two shallowest observations wells close to the pond (d)

ed (grey areas, annotation 0) to fully saturated conditions (reached when the groundwater-table is ~1 m or less below the pond; white areas, annotation) are indicated.

Detailed investigations directly below the pond showed that penetration of atmospheric oxygen from the pond fringes during unsaturated conditions leads to rapid oxidation of filtrated labile organic matter and thus to enhanced calcite dissolution causing bicarbonate and calcium to peak in the groundwater (Figure 4d, Massmann et al., submitted; Greskowiak et al., in press).

Massmann et al. (in press) illustrated that the order of magnitude of the reaction velocity is similar at the bank filtration and artificial recharge sites for O_2 reduction but the conditions are generally more reducing at the BF site due to longer travel times.

CONCLUSIONS

The infiltrating surface water undergoes hydrochemical changes during bank filtration and artificial recharge. The redox conditions are influenced by (i) operational factors, (ii) seasonal temperature changes affecting the microbial activity and (iii) travel times of the infiltrate. In addition, the bank filtrate is becoming increasingly older with depth, thereby reflecting the historical changes of the surface water composition with regard to some minor water constituents.

ACKNOWLEDGEMENTS

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Removal capacity of riverbank filtration and conclusions for the operation of water abstraction plants

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Abstract

Riverbank filtration (RBF) can be used as a step in water purification during drinking water abstraction and is a sustainable, low-cost and energy efficient procedure. As a part of a joint research project of the German Ministry of Education and Research (BMBF) a literature study and an inquiry among water supply companies was made in which 33 case studies of RBF sites in middle-Europe, comprising 19 different surface water sources were collected. By cluster and discriminant analysis four different characteristic site groups, regarding hydrochemical and hydrogeological parameters of RBF, could be distinguished. Further consideration of the dissolved organic carbon (DOC) degradation capacity of subsurface passage by multi-variate correlation and regression analysis revealed as determining parameters the initial DOC concentration in the surface water, the residence time of the riverbank filtrate in subsurface passage and the transmissivity of the adjacent aquifer. A multiple, non-linear regression equation can explain almost 75 % of the variance in the whole sample and can be used for the estimation of the DOC-elimination at new planned sites, as well as for optimizing operation processes of existing abstraction plants using RBF.

Keywords

Cluster analysis, DOC, removal capacity, riverbank filtration, water treatment.

INTRODUCTION

The riverbank filtration (RBF) with a subsequent underground passage has been used in Europe as a close to nature, low cost and reliable purification method for drinking water supply for more than 100 years. The RBF induces surface water, like river or lake water, to flow downward through sediment and into a pumping well. During this process potential contaminants are reduced from the water by various purification processes, significantly improving the water quality. The riverbank filtration will be only an effective step for cleaning up surface water by adaptation to the local boundary conditions of a specific site.

Investigations in the past showed for various RBF systems, that not only the behaviour of substance transport und elimination but also the immobilization of substances in the underground depend on many hydrogeological, hydraulic and physical-chemical factors. The specific local distinction of oxidizing and deoxidizing conditions has a central importance.

Up to now there is a lack for purification orientated guidelines for the planning, dimensioning and operating of RBF sites for drinking water production. Particularly the reaction of RBF to surface water with rather different concentrations of chemical substances than in central Europe are not well understood as well as the impact of differing extreme climate conditions. In the project 'Technical concepts and adjusted operational modes for an optimal adaptation of riverbank filtration to local boundary conditions', as part of a joint research project of the German Ministry of Education and Research (BMBF), fundamental variables for planning, dimensioning and running water production plants were compiled. For this reason it was referred to the long term practical experience in operating riverbank filtrate abstraction plants, which especially prevails in Germany.

MATERIALS AND METHODS

Knowledge from literature was supplemented with a representative data inquiry among water supply companies in Germany, France, Austria and Switzerland. Thus 33 case studies of RBF sites in middle-Europe, comprising 19 different surface water sources were collected.
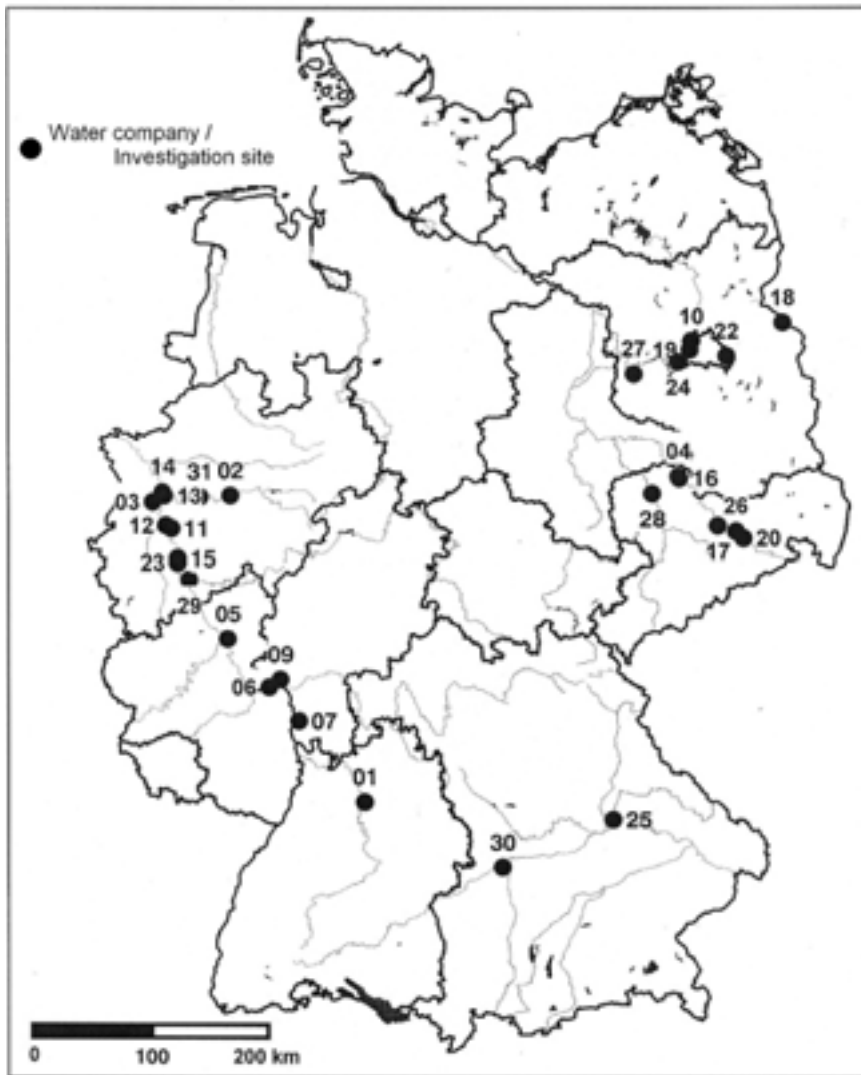
Figure 1. Water companies and investigation sites with at least 50% riverbank filtrate in the water of the well and monitoring wells recorded in the data study

Only RBF sites with significantly more than 50% bank filtrate in the water of the abstraction wells or monitoring wells were taken into consideration. Many different types of surface waters and geological areas were registered like sites at the middle and lower Rhine or various surface waters in the region of Berlin which have a higher sewage load in comparison to other sites. Figure 1 shows the recorded water companies and investigation sites with RBF in Germany.

The questionnaire contained among others questions about (i) the local boundary conditions of the sites like the surface water quality and the local hydrogeological situation, (ii) the monitored purification performance of the riverbank filtration and underground passage (DOC, NO_3, O_2 a.o.) and (iii) data of the travel time and length of the underground passage as well as the redox situation.

The aim of the study was to classify central European type situations of RBF sites on the basis of selected local boundary conditions. On the background of that classification correlations should be calculated between local boundary parameters and the regarded systems purification capacity to relevant water solutes (e.g. DOC, NO_3, and O_2). All the data from the literature and the survey were interpreted using various multi-variate statistical methods.

RESULTS AND DISCUSSION

The classification to RBF type situations was done on the base of the local boundary conditions of the sites recorded in the survey and literature study. The following local hydrogeological site parameters were considered:

hydraulic conductivity, thickness and transmissivity of the aquifer. Local surface water was characterized by following parameters: temperature, pH-value and concentrations of O_2, NO_3, NH_4, SO_4, and DOC. These parameters determine the redox induced transformation process of substances during the infiltration and the subsequent underground passage.

The classification of the middle-European RBF type situations was computed by algorithms of a hierarchic cluster analysis. The euklidic distance was used as a proximity measure and the WARD method as a merging algorithm. Before calculation the raw data were standardized to avoid implicit significances and to guarantee a dimensional comparison between the single variables. The single-linkage-algorithm was used to identify data outlier, which possibly could falsify the merging process. These data were excluded from the data matrix. The quality of the calculated cluster classification was rated by T- and F-values.

By the hierarchic cluster analysis four typical middle-European different characteristic site groups, regarding hydrochemical and hydrogeological parameters of RBF, could be classified (Fig. 2).

Each cluster group has characteristic combinations of the considered local boundary conditions. There are significant differences for each group between the chemical situation of the surface water and the characteristics of the adjacent aquifer. Table 1 contains the variable values of the cluster centres and the numbers of bank filtration sites for every hydrochemical-hydrogeological cluster. So, for example, the chemical situations in the sites of cluster 4 are distinguished especially by high DOC- and sulphate concentrations and moderate nitrate concentrations of the surface waters compared to the total sample. Referring to the local hydrogeological conditions site group 4 shows aquifers with a high thickness and low hydraulic conductivity

Figure 2. Cluster tree (WARD; n=27) based on the local boundary conditions (FS = case study)

compared to the values of the total sample. This cluster contains mainly riverbank filtration sites at the river Elbe and the Berlin region. Riverbank filtration sites from the river Rhine are found in the second cluster group. Compared to cluster 4 they rather have more average values for the parameters mentioned above.

Table 1. Variable values of the cluster centres and numbers of bank filtration sites for every hydrochemical-hydrogeological cluster

	Cluster 1	Cluster 2	Cluster 3	Cluster 4
k_f- value [m/s]	0,0083	0,0046	0,0044	0,00074
aquifer thickness [m]	11,2	17,1	8,7	33,9
T(max.) [°C]	20,6	23,9	20,6	22,6
pH-value [–]	7,9	7,7	7,7	8,1
DOC [mg/L]	3,2	4,1	4,6	6,7
O_2 [mg/L]	10,1	8,9	9,7	10,5
NO_3^- [mg/L]	11,0	14,2	26,4	15,1
NH_4^+ [mg/L]	0,2	0,4	1,0	0,3
SO_4^{2-} [mg/L]	46,2	63,5	85,5	104,7

Generally the significant differences between the cluster groups justify the classification of the whole sample into the 4 riverbank filtration site groups on the basis of the hydrochemical and hydrogeological criteria. The quality of this classification was corroborated by the result of a discriminant analysis. In excess this analysis yielded a coefficient set for the linear combination of the considered variables in order to prognosticate the classification of new planned sites.

The reduction of organic compounds is one of the central goals of the water purification for drinking water purposes. The RBF as a pre-treatment step in the purification process for drinking water and as a part of a multi-barrier concept has essential advantages compared to the direct treatment of surface waters. Organic substances will be reduced and degraded in the biological high active infiltration zone and in the subsequent underground passage. By this it is possible to reduce the amounts of needed energy and chemical substances at subsequent treatment steps.

From the 33 recorded RBF case studies 67 hydrochemical sample results of suitable abstraction and monitoring wells were selected to quantify the spatial and temporal development of soluble substances of the surface waters during the underground passage (Figure 3). The degradation capacities of the subsurface passages were then characterized using bi- and multivariate correlation and regression analysis. As the essential determining variables for the measured dissolved organic carbon (DOC)-elimination at the examined locations, (i) the initial DOC concentration in surface waters, (ii) the residence time of the riverbank filtrate in subsurface passage and (iii) the transmissivity of the adjacent aquifer could be found.
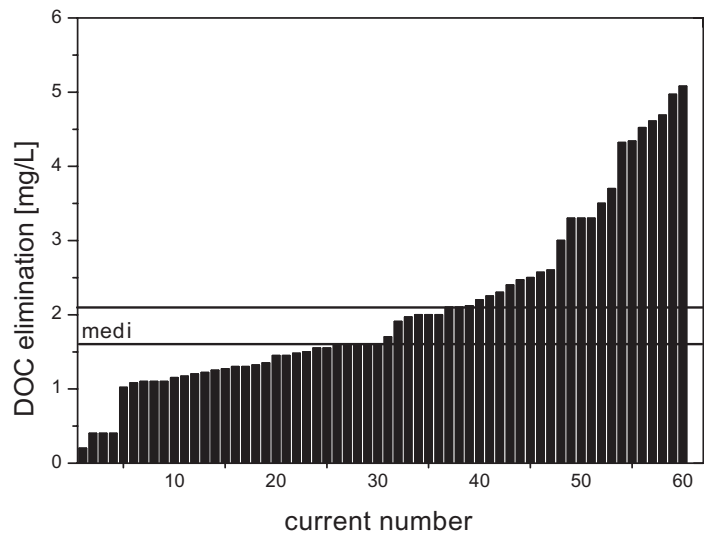
Figure 3. DOC elimination [mg/L] during the underground passage at the recorded abstraction and monitoring wells

The best fitting bivariate models were used in the search for a combined multiple, non-linear regression function explaining the capacity of the DOC-elimination. The following function can explain 74% (n = 43) of the monitored variance in the sample regarding the DOC-elimination and referring to the whole population:

$$Y = -0{,}503 + 0{,}811 * \ln(X_1) + 7{,}428 * (X_2) + 0{,}236 * (X_3)^{0{,}437} \qquad (1)$$

with: Y = DOC-elimination [mg/L]; X_1 = DOC-concentration of the surface water [mg/L]
        X_2 = transmissivity of the aquifer; X_3 = residence time in the underground [d]

A restriction to RBF sites with a subsequent aerobic subsurface passage results in a slightly different regression function for the objective DOC-elimination which explains 87% (n = 23) of the monitored group variance for the sub-sample:

$$Y = -0{,}614 + 1{,}370 * \ln(X_1) + 4{,}856 * (X_2) + 0{,}026 * (X_3)^{0{,}937} \qquad (2)$$

These regression equations serve also as a prognosis model which can estimate the degree of DOC-elimination at unknown sites.

The structure of the non linear functions with its logarithmic and potential terms results from the different degree

of the DOC-elimination during the infiltration and the underground passage. At the beginning of the underground passage the more degradable DOC-fractions as a major part of the total DOC amount can be oxidized in the biological high active infiltration respectively clogging zone. Then, on the flow path during the subsequent underground passage the DOC-elimination decreases compared to the first centimetres and meters beyond the infiltration point. The elimination capacity reaches a plateau level and the more poorly degradable DOC-fraction consisting of high molecular carbon acids or anthropogenic organic trace substances begins to degrade. In that manner the most significant reduction of the DOC concentration occurs at the beginning of the underground passage. So, only suitable dimensioned underground passages have a sufficient elimination capacity for the refractory DOC fractions.

In a similar way the decrease of oxygen and nitrate removal in the passage between surface water and abstraction well as well as the development of the redox environment were investigated. In partial correlation analysis was found that the oxygen depletion during the underground passage is mainly determined by the DOC concentration in the surface water. The higher the starting DOC concentration is the higher will be the depletion of oxygen. This may result in anoxic conditions during the subsurface passage with oxygen concentrations below 0,5 mg/L in the abstraction wells. Due to the highest DOC concentration in surface water the RBF sites of cluster group 4 show the greatest proportion of anoxic subsurface environments of all regarded sites. Nitrate concentration is influenced by the oxidation of ammonium during the underground passage. The reverse process, the ammonification of nitrate, will only occur under anoxic conditions. A substantial nitrate reduction may take place when initial oxygen concentration have already decreased. In the average, under anoxic conditions 80 % of the initial nitrate concentration is degraded after a residence time of 50 days or a travel distance of 45 m.

The resulting set of empirical regression models can be used as a practical guideline for the estimation of the development of hydrochemical parameters at new planned sites, as well as for optimizing operation processes of existing abstraction plants using RBF. As an example Figure 4 shows the results of the computed prognosis functions (equation (1)) for the DOC-elimination depending on the local boundary conditions summarized in nomograms. The range of the variables used for these calculations has to be regarded: (i) initial DOC concentration in surface waters 1,4 – 9,0 mg/L, (ii) residence time 0,01 – 210 d, (iii) length of the underground passage 0,15 – 310 m and (iv) transmissivity 0,003 – 0,230 m^2/s. Outside of these ranges the used functions are not valid.

Figure 4. Nomogram for a prognosis of the DOC elimination at the riverbank filtration and underground passage on the base of equation (1) (SW = surface water, T = transmissivity)

CONCLUSIONS

By cluster and discriminant analysis the considered case studies can be classified to 4 typical site groups which show a significant distinction of its hydrogeological site properties and the hydrochemical characteristics of the infiltrated surface water. The discriminant analysis provides also a set of coefficients by which unknown sites can be classified.

Essential purification processes like DOC-elimination are determined to a high extent by three parameters: the initial DOC concentration in the surface water, the residence time of the riverbank filtrate in the subsurface passage and the transmissivity of the aquifer. In a non-linear model DOC-elimination can be described by a multivariate regression equation. The empirical regression function can be used in a prognostic way for the estimation of the DOC-elimination at new planned sites. However, the range of variables used for the calculation has to be respected in any transfer case. A sufficient residence time respectively a long travel distance presumed also a part of the refractory DOC-fraction can be degraded.

The summary of these results can be used by engineers as a practical guideline for planning purposes, and dimensioning drinking water production plants with RBF as well as to adapt them to the local situation.

ACKNOWLEDGEMENTS

A model study of the proposed Everglades Restoration Hillsboro ASR Pilot Project

Christopher J. Brown and Richard Nevulis

Abstract

In support of Everglades Restoration in South Florida, USA, the U.S. Army Corps of Engineers and the South Florida Water Management District are currently engaged in the execution of four ASR pilot projects located throughout the Everglades region. It is envisioned that thorough testing of the four projects will enable the project team to better grasp the technical uncertainties associated with implementing ASR on a grand scale. The Hillsboro ASR Pilot Project is one of the four pilot projects planned for construction and testing starting in 2005. A numerical model was developed to aid the project team in estimating potential performance along with eval uating potential impacts from the Hillsboro project. The model provided reasonable estimates of ASR well drawdowns and mounding effects, as well as a range of predicted ASR recovery efficiencies. Impacts to existing FAS well users were deemed insignificant with only one ASR well pumping, however, simulations using the proposed full-scale 30 well system revealed some long term design challenges. The simulations also indicated that future refinements to the model should probably include finer resolution at the ASR well location.

Keywords

ASR; aquifer; model; recharge.

INTRODUCTION

The Everglades Ecosystem, located in Southern Florida, is a myriad of wetlands, marshes, cypress domes, estuaries and coral reefs. The Everglades is a broad, flat expanse of wetlands inhabited by innumerable plants and animals. Dubbed the 'River of Grass' by Marjorie Stoneman Douglas (Douglas, 1947), the Everglades is in trouble. The distribution of water, its timing, its quality and its quantity, have all been radically changed over the last 100 years (Davis, 1994).

The *Central and Southern Florida Project Comprehensive Review Study* (USACE and SFWMD, 1999) – developed jointly by the South Florida Water Management District (SFWMD) and the U.S. Army Corps of Engineers (USACE) – presents a framework for Everglades restoration. Now known as the Comprehensive Everglades Restoration Plan (CERP), this plan contains 68 components, including structural and operational changes to the Central and Southern Florida (C&SF) Project. The overarching purpose of CERP is to restore the Everglades by improving the quantity, quality, timing and delivery of water for the natural ecosystems of south Florida. A key component in the overall restoration strategy is the provision of more dynamic storage of freshwater.

The CERP water storage strategy proposes the use of both above-ground storage reservoirs and underground storage via deep wells. The deep wells proposed are Aquifer, Storage and Recovery (ASR) wells. The use of ASR is increasing in the United States and abroad and has been documented by Pyne (1995). The CERP relies heavily upon ASR technology to provide additional system storage. The proposed use of ASR for this project is unprecedented with 333 large-capacity wells proposed to store over 3.8 million cubic meters of water per day.

The performance of an ASR system is a complex process that may be affected by operational design, subsurface heterogeneity, density-dependent flow processes, and biogeochemical processes. Numerical models can be helpful in

simulating these complex processes. Specifically, the models are helpful in evaluating system performance, evaluating extent of drawdown or mounding effects, and identifying key uncertainties that should be explored further through cycle-testing or other data collection efforts. Due to the technical uncertainties involved, the USACE and the SFWMD proposed a series of ASR pilot projects to be constructed and tested prior to large-scale implementation (USACE and SFWMD, 2004). One of the pilot projects is located in Palm Beach County, Florida USA along the Hillsboro Canal. The Hillsboro ASR Pilot Project has a planned capacity of 19,000 cubic meters per day and will include an associated water treatment plant designed to remove suspended solids and pathogens from the source water prior to ASR recharge. The storage zone is located in a carbonate section of the Floridan Aquifer System (FAS) overlain by a massive confining unit consisting of clay, silt, and mudstones, collectively referred to as the Hawthorn Group (Miller, 1997). Figure 1 depicts the project location along with general hydrogeologic information for the project area (SFWMD, 2001).
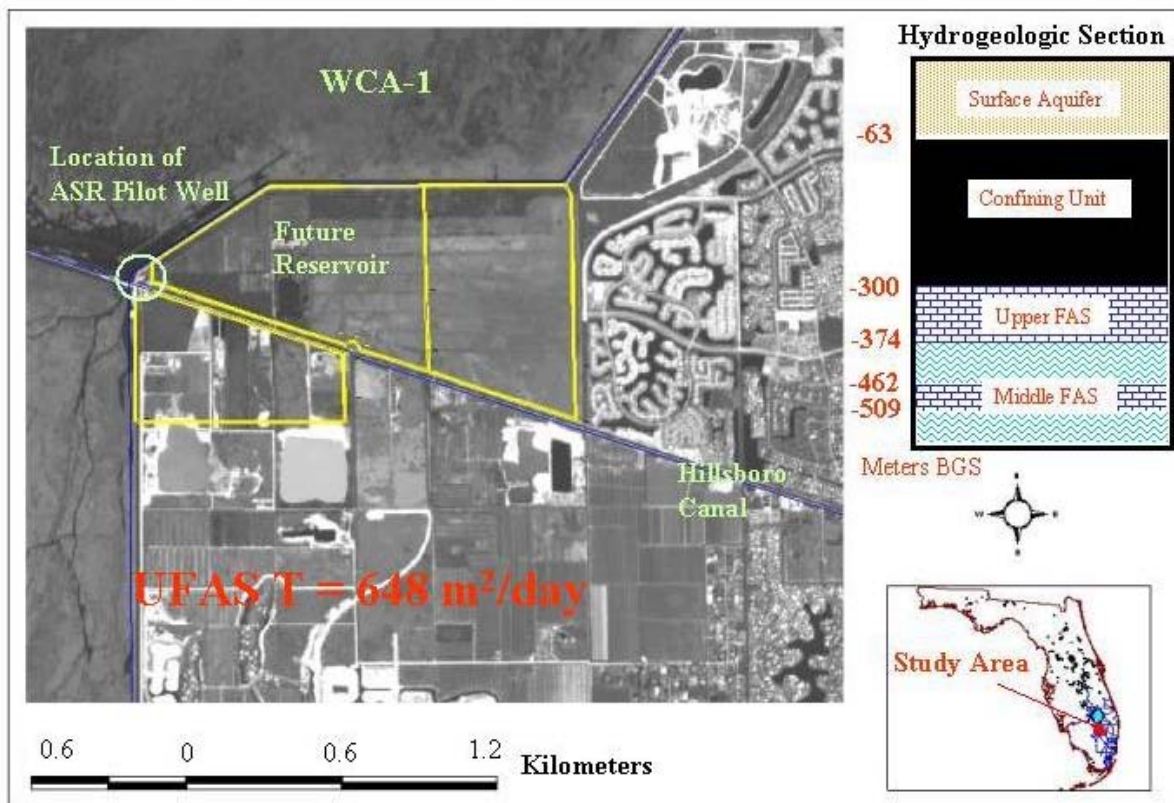
Figure 1. Hillsboro Pilot well study area and hydrology

As part of studies supporting the ASR Pilot Projects, five separate numerical models were developed to provide additional information regarding extent of drawdown or mounding effects at each proposed project site. System performance could not be estimated accurately at all sites due to lack of site-specific hydrogeologic and groundwater quality data; however, these data were available at the Hillsboro site. At the Hillsboro site, the modeling effort included both evaluation of drawdown or mounding effects as well as estimates of system performance. The model was intended to provide preliminary estimates of project performance.

METHODS

Various investigators have undertaken modeling of ASR systems in order to explore assorted controls on system performance. Merritt (1986), Missimer et al (2002), and Pavelic et al (2002), have all developed models of brackish-water ASR projects. Khanal and Pavelic utilized conventional two-dimensional models to explore the expected

performance. Pavelic elected not to utilize a fully-density dependent model to develop performance estimates. Merritt and Missimer utilized fully-density dependent three-dimensional models to evaluate a choice of realistic operational scenarios and the effects of various independent variables (porosity, vertical hydraulic conductivity, density of ambient groundwater, etc.) upon system performance. Streetly (1998) and Anderson and Lowry (2004) have investigated ASR performance in near-potable aquifers using numerical models. The Hillsboro ASR Pilot Project model was designed for the purpose of making preliminary evaluations of system performance; however, it was designed with the intent of converting it to a fully density-dependent model later. Ongoing efforts are in the process of refining the Hillsboro model to incorporate newly developed aquifer data as well as providing additional model functionality.

The Hillsboro ASR Pilot Project model consists of several flow models developed from MODFLOW (McDonald and Harbaugh, 1988) linked to contaminant transport code MT3DMS (Zheng and Wang, 1999). Due to the large recharge/recovery rate anticipated for the Hillsboro pilot test, the model boundary included a large part of Palm Beach County, Florida USA. The initial model grid is 64 kilometers x 64 kilometers and consists of an irregularly spaced grid with 30 meter grid resolution at the ASR well location and 1,250 meter resolution at the model boundary. The initial model (version 2.0) included 7 layers and included the Surficial Aquifer System, the Hawthorn Group, the upper FAS (2), the middle confining unit, the Middle FAS, and the Lower Floridan. Initially, smaller domains were tested to determine the artificial influences induced by the boundary conditions; however, all smaller domains tested introduced significant induced error.

Calibration of the model included both steady-state and transient calibration efforts. The steady-state calibration matched regional groundwater stage information available in SFWMD (1999). The transient calibration matched observed drawdowns recorded during an aquifer performance test at three onsite monitoring wells. Figure 2 depicts transient calibration results from two versions of the model during the development process. The figure compares simulated drawdown versus field data collected at one observation well located 100 meters from the ASR well.
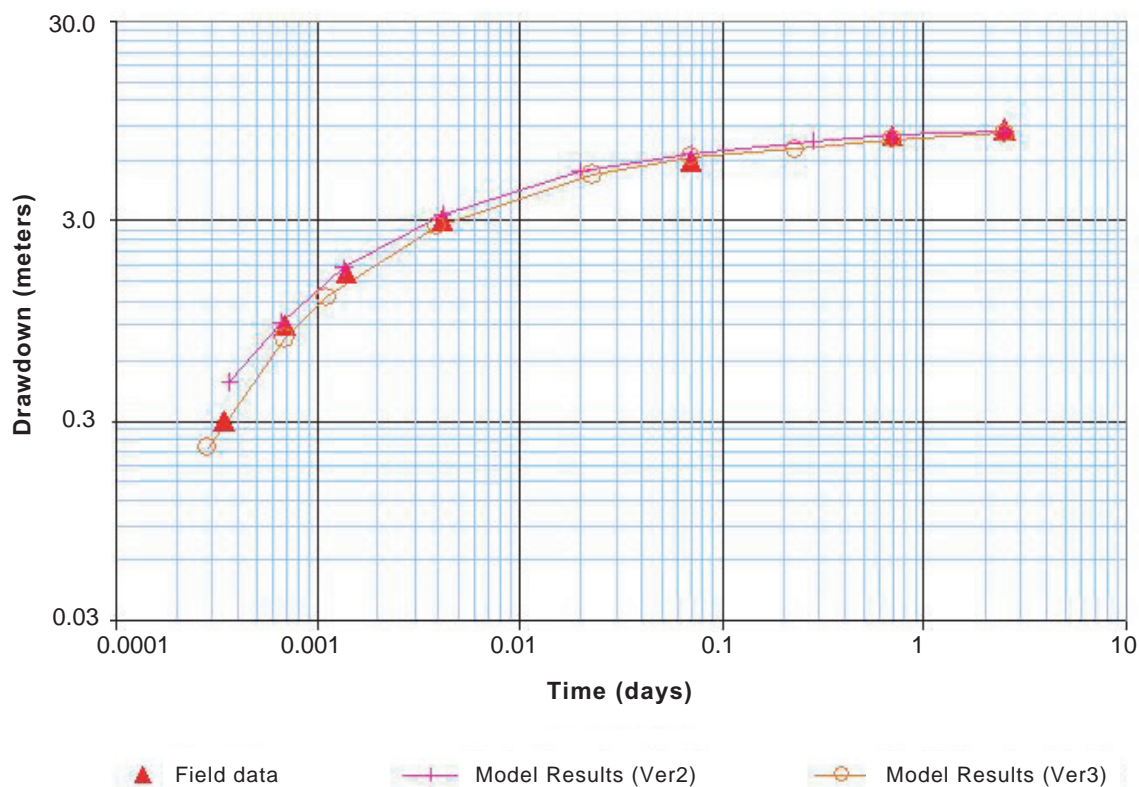
Figure 2. Hillsboro ASR Transient Model calibration

Ultimately, both model versions were utilized for different predictive simulations. Model version 3.0 was more refined at the ASR well head (3 meters) and did provide slightly improved calibration results as depicted on Figure 2, however, run times were very long. Version 2.0 was used for evaluation of regional drawdown impacts. Version 3.0 was used for providing estimates of ASR recovery efficiency. Version 3.0 was also more stable when investigating advection dominated transport simulations where the grid Peclet number was small and the flow gradient steep. Typically, model version 2.0 experienced convergence or instability problems with similar conditions due to 30 meter grid resolution specified at the ASR well.

The model was later verified against observed transient drawdown from another nearby ASR well located approximately 8 kilometers east of the Hillsboro ASR Pilot Project. The 'Eastern Hillsboro' ASR project is being jointly developed by Palm Beach County Water Utilities and the SFWMD, and completed extensive aquifer testing in 2003. These data provided an excellent data set to verify the model against. A full description of the calibration results is presented in the *Pilot Project Design Report* (USACE and SFWMD, 2004).

RESULTS AND DISCUSSION

The model provided reasonable estimates of ASR well drawdowns and mounding effects, as well as a range of predicted ASR recovery efficiencies. The base case simulation consisted of a steady-state simulation with a recovery rate of 19,000 cubic meters per day at the site. The maximum drawdown at the ASR well was –33 meters, and the 1.25-meter drawdown contour was within an approximate 4,000 meter radius from the recharge well. The closest facility to the site is the Eastern Hillsboro ASR site, approximately 8,000 meters downstream along the Hillsboro Canal. Approximately 0.5 meters of drawdown was predicted at the location of the Eastern Hillsboro site, however, it is not envisioned that the anticipated drawdown will impede pumping capacity at the Eastern Hillsboro site unless both sites are pumping simultaneously. In addition, it may be useful to continuously monitor the Eastern Hillsboro FAS wells during future cycle testing at the Hillsboro Pilot location. Data collected at both the pilot location and Eastern Hillsboro will help to validate the numerical model and permit additional future improvements to be made. Finally, it is noted that the drawdown contours were asymmetric, with drawdown extending further to the west than east, which is attributed to the shallower target-zone thickness in the western direction. Following completion of hydraulic evaluations for one ASR well, contaminant transport and mixing evaluations were completed with MT3DMS. Model simulations included advection, mixing, and hydrodynamic dispersion. As noted by Merritt (1986), hydrodynamic dispersion was determined to be one of the most important performance variables at the Hillsboro site.

Allowing dispersivity to range from 1 to 10 meters varied the simulated hydrodynamic dispersion. The 'standard' recovery efficiency of the initial test cycles was predicted to range from 3% to 15%. The recovery efficiency as defined by Reese (2002) is expressed as the percentage of the volume of water recovered that meets regulatory standards versus the volume of water injected over a cycle of recharge and recovery. The recovery efficiency is typically presented as a percentage from 0% (no recovery volume) to 100% (full recovery volume) as compared to a 250 mg/l chloride standard. Since recovered water from the Hillsboro site is to be utilized for Everglades Restoration, the recovered water will be discharged into the Hillsboro Canal. Unfortunately, the regulatory standards (Florida Class III Surface Water Standards) for the Hillsboro Canal may be more restrictive than potable standards since the poor condition of the Hillsboro Canal source water has already been documented (SFWMD, 2003). Whereas, the Class III standard for specific conductivity is 1,275 umhos/cm, the source water already contains up to 750 umhos/cm. In contrast, a typical south Florida potable ASR system starts with source water with 50 mg/l chloride and can recover water until 250 mg/l chloride. It is evident that the Hillsboro ASR site is already much closer to exceeding the pertinent regulatory standard than similar potable sites. Given the higher source-water specific conductance concentration at Hillsboro and the higher ambient groundwater specific conductance values, it was expected that recovery percentages at the site would be relatively low for a number of cycles. The modeling

completed confirmed this suspicion. The model simulations revealed that the ASR recovery efficiency at the site would improve over subsequent recharge and recovery cycles but at a slow rate due to the brackish nature of the storage zone. Based upon the programmed two-year cycle testing plan, recovery efficiencies are expected to range from 25 to 40%. Cycle testing beyond the two-year duration proposed would be expected to improve the recovery efficiency of the Hillsboro ASR site.

Following the pilot project model simulations, several simulations were conducted to explore long-term CERP plans for the site. The CERP plan proposes 30 ASR wells to be located at the Hillsboro site in conjunction with a 6.7 km^2 surface water reservoir. Model simulations using the Hillsboro Pilot model indicate that well spacing of these 30 wells will be an important design consideration to ensure aquifer pressures are not enormous. Model runs using a well spacing of approximately 300 meters revealed potential drawdown of 130 meters. More widely spaced wells resulted in much lower drawdown but at reduced system recovery performance due to increased mixing. Future efforts will need to evaluate these design features in more detail in order to develop a feasible system.

Model hydraulics for the Hillsboro Pilot project are quite challenging due to the large recharge and recovery rates planned for operations. Model resolution is a key consideration in constructing any ASR simulator. Simulated drawdown at the Hillsboro ASR well head is very steep using model version 2.0 with 30 meter grid resolution; however, checks against existing analytical solutions revealed that the model could be improved through incorporation of a finer grid resolution. Ongoing model tests of model version 3.0 confirm that 3-meter grid resolution at the ASR well location provides both an accurate estimate of aquifer drawdown and a numerically stable model, although run times are considerably longer. The finer grid resolution also provides more accurate estimates of recovery efficiency of the Hillsboro ASR pilot site.

CONCLUSIONS

Modeling of the Hillsboro ASR Pilot project was very useful in understanding system constraints, potential impacts to nearby users, and expected performance. The model provided realistic estimates of actual operating conditions and was an excellent planning tool. Model predictions could be improved if additional grid resolution is included in the model or if additional model functionality such as density-dependent capabilities, is included. These improvements would likely lead to superior estimates of site performance.

A small number of model simulations of the proposed full ASR 30-well system were completed. These model simulations point to future design challenges that need resolution including final well spacing and locations. For ASR sites involving recharge of natural systems, the recovered water quality is a key-determining factor in project performance. In fact, due to water quality limitations in the receiving surface water body, the actual ASR recovery efficiency may be less than if the site was utilized for potable supply. The ASR pilot project at the Hillsboro site should be completed in order for the site to provide additional site-specific field data for future design efforts. This data could also be utilized to improve the existing numerical model of the site.

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Development of fresh water sources in some parts of the arsenic contaminated areas of West Bengal (eastern India) through artificial recharge: a remote sensing and geographical information system based approach

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Abstract

Chronic arsenic toxicity (arsenic concentration > 0.05 mg/l) is reported from the state of West Bengal in eastern India. In this context this present investigation in a part of Nadia district of West Bengal state, involves remote sensing application aided by geographical information system (GIS) techniques to generate a composite map (scale, 1:25,000) to reveal the sites for (i) artificial recharge (ii) water harvesting structure and for (iii) fresh groundwater exploration. Older meander belts, geomorphic lows and palaeochannels /meanders (excluding that of older meander belts) are suitable sites for artificial recharge, water harvesting structures and for fresh water exploration respectively. Deeper aquifers present in the relatively older flood plains consisting of clay beds are also suitable for fresh water exploration. Here the shallow aquifers are unconfined and deep aquifers are confined. In fact, the aquifers change gradually from open to semi-confined character towards the south. The meanders, levees and flood plains with their sediment characteristics vary from sandy, silty to clay size with varying landcover/landuse practice are clearly discernible in the satellite imageries (IRS IC/ID LISS III and PAN merged data) and are verified by field checks.

Keywords

Arsenic toxicity, artificial recharge, composite map, GIS.

INTRODUCTION

India is highly endowed in respect of water resources. The country is criss-crossed with rivers and receives high precipitation mainly due to the south-west monsoon, which accounts for 75 per cent of the annual rainfall. The major river basins along with the medium and minor river basins account for almost 91 per cent of the country's entire drainage area (Biswas, 2003). Yet, in spite of nature's bounty, paucity of potable water is an issue of national concern. In recent times, the practice of irrigated agriculture supported by groundwater development, which has been proved to be the most dependable source for irrigation even during periods of water stress. Groundwater supports more than 50 per cent of the total irrigated area in India. Groundwater contributes 85 per cent of India's rural water supply and is significant in the drinking water sector (Kittu, 2003).

Arsenic in groundwater

Unfortunately about a million people are drinking arsenic contaminated water and over 200,000 people are suffering from arsenic related diseases in the eight districts of West Bengal State in eastern India. Apart from India, the presence of arsenic (>0.05 mg/l) in ground water has been reported from several countries of the world like the U.S.A, China, Chile, Nepal, Thailand and Bangladesh.

The first instance of arsenic toxicity in ground water of West Bengal, eastern India was reported in the year of 1978. Arsenic contamination in groundwater beyond the permissible limit of 0.05 mg/l has been found to be present in the shallow aquifer (20–150 m bgl) located in the part of Bengal delta, covering the southern portion of West Bengal state of India and Bangladesh. The higher concentration of arsenic is restricted mainly the upper delta plain within older (abandoned) exposed meander belt. Composition of the sediment changes laterally across the delta plain. This may explain the discontinuity of the aquifer. Sediment from shallow aquifers in the Bengal delta of eastern India has been found to contain arsenic. Geomorphological domains with different depositional pattern regulated the style of distribution of area with varying content of arsenic in ground water .The high arsenic zones occur as narrow sinuous strips confined to channel deposits (Sengupta et al. 2004).

Ground water recharge movement and accumulation are governed by geomorphology, geology, soil/sediment, land cover and land use of a region. So, the information of these aspects provides valuable clues for locating the sites for ground water exploration, artificial recharge and water harvesting structures. Palaeochannels (early Holocene age) situated near to the present course of the Hugli-Bhagirathi river are generally arsenic free. The convex part of the cut off meanders is arsenic affected due to its reducing environment.

In fact, the geometry of the abandoned drainage course closely matches with the geometry of the high arsenic zones suggesting a link between creation of such zones and the activity of the fluvial system.

Solution to the problem of excess arsenic in groundwater through artificial recharge

Artificial recharge is considered as a type of water harvesting in which surface water is stored and transfered into the subsurface during the rainy season and can be utilized in the other seasons. Artificial recharge can be used for augmenting groundwater reserves and improving water quality. It can be considered as one of the most widely practiced methods of water harvesting. In India, systematic application of artificial recharge is still at an initial stage though India has an age-old tradition of water harvesting. Surface water generally contains very low concentration of fluoride and arsenic. The total dissolved solids (TDS) level is also invariably less than that of ground water. Therefore, artificial recharge can be used specifically for reducing the brackishness or the high concentration of harmful substances such as fluoride and arsenic in the groundwater. A mixture of the surface water with the aquifer water reduces the concentration of various chemical radicals.

Mandal et al. (1997) suggested the use of surface water stored in many ponds in the arsenic affected areas of West Bengal state of eastern India. The average annual rainfall (a.a.r.) in the area under study is about 2,000 mm. A pilot experiment was carried out by a group of workers in the Habra block of N-Parganas district, which is adjacent to the present study area. The experiment lasting over a period of 90 days brought the local groundwater arsenic level from 1.28 mg/l to 0.001 mg/l (Athavale, 2003). Different methods of artificial recharge have been practiced in many parts of the world, but the right choice of site for artificial recharge and water harvesting structure is important. In recent years, a number of studies have concentrated on the site selection process especially with the help of remote sensing techniques (Sharma, 1992, Ramaswamy and Anbazhagan, 1996).

The complexity and diversity of water resources related problems demand the recording of an ever-increasing range of different types of data with greater precision on spatial and temporal scales. In some cases, conventional methods may take months and years for recording one parameter, whereas remote sensing makes it possible to record not only one time data but also repetitive data. Remote sensing data have been effectively used in geological, geomorphological, groundwater and land use/land cover studies for more than two decades. Information on geology, geomorphology and groundwater conditions are essential for water shed planning, management, and development (Rao, 2003). Use of remote sensing with geographical information system (G.I.S.) methods is helpful in decision making for better water management. Integrated information on geology, terrain conditions, land use/land cover,

extracted from satellite imagery is useful in selection of sites for construction of reservoirs and artificial recharge structures for improving surface and ground water resource. Multi-layer remote sensing data involving information on morphological and morphometrical details of the terrain, rock types, soil/surface materials, drainage, land use/ land cover help in catchment characterization. This leads to better modeling of surface water resources by determining rainfall run off ratio with reasonable accuracy (Reddy, 2003). In GIS both graphic (spatial) and textual (non-spatial) data are used.

OBJECTIVES

In the above context the present study has been carried out in the area bounded by longitude $88°27'30''$ E and $88°37'30''$ E and latitude $23°$ N and $23°7'30''$ N. The objective of the present study has three main components. The first is the identification of the site suitable for construction of artificial recharge structures using IRS data. The second aspect is to suggest suitable locations for surface water reservoir/basins aiming towards conjunctive use of surface and groundwater. The third objective is to identify the suitable areas for arsenic contamination free fresh water exploration.

DATA USED

Three types of data sets have been used in the present study:

a. Remotely sensed data, viz. false colour composite (FCC) of Indian Remote Sensing Satellite (IRS) IC/ID LISS (Linear imaging self scanning sensor)-III and PAN merged geocoded data, scale 1:25,000 of 19 February, 2001. Date of acquisition of remotely sensed data has been so chosen as this is the peak time of growth of winter crops (Rabi) and dry season vegetation is an indicator of groundwater.

b. Existing map, viz. Survey of India (SOI) topographical sheets and published geological map have also been consulted.

c. Published bore hole data and arsenic concentration in the different wells of the study area (Sengupta et al., 2004) have been consulted. Arsenic concentration values have been divided into two categories for subsequent GIS analysis purpose. One is groundwater zones with greater than 0.05 mg/l arsenic (unsafe zone), another is groundwater zone with less than 0.05 mg/l arsenic (safe zone).

METHODOLOGY

It involves the following steps:

i. Reconoitery visit to the field;

ii. Remote sensing approach: It involves preparation of thematic maps on fluvial morphology, landform, land use and land cover through visual interpretation of satellite imageries aided by field verifications. The image interpretation keys like tone, texture, colour, association, shapes, size and pattern have been used to decipher the different units/features;

iii. GIS approach: It involves overlay analysis of different pertinent thematic layers in the GIS environment. In the present study digitization, layer creation and analysis have been performed using the Map Info Professional version 7.5 GIS software package. The produced composite map (Figure 1) which reveals the desired areas and sites.

Legend

- Settlement in older flood plain
- Natural Levee, As>0.05 mg/lt, Artificial recharge site
- Meanders in clay covered zone

- Settlement
- Natural levee , As>0.05mg/lt. Artificial recharge site
- Meanders in older flood plain

- Sites for water harvesting in agricultural filed
- Sites for water exploration in clay covered older flood plain area
- Meanders in clay coverd area
- Meanders, As>0.05 mg/lt.
- Arable land
- Ponds
- Sites for Water Exploration
- Sites for artificial recharge
- Natural Levee, As<0.05 mg/lt.
- Natural Levee, As>0.05 mg/lt.
- Agricultural field

- Water harvesting sites in agricultural fields
- Cut off meanders in agricultural fields
- Oxbow lake
- Natural levee, As<0.05 mg/lt.
- Meanders
- Meanders in younger flood plain
- Vegetal cover
- Traces of palaeo channels
- Ponds
- Hugli River
- Natural Levee with huge settlem
- Natural levee, As>0.05 mg/lt.
- Water in huge oxbow lake
- Agricultural & Arable land
- Water in smaller oxbow lake

0   2.5   5
kilometres

Figure -1

The central to north-western part of the map shows the younger flood plain of Hugli river, the rest is older flood plain.

OBSERVATIONS

Field study during the present investigation reveals that the arsenious zones are confined along the levees of these drainage channels. Palaeochannels (early Holocene age) situated near the present course of Hugli-Bhagirathi river (Figure 1) are generally arsenic free. The convex part of the cut off meanders (Figure 1) is arsenic affected due to its reducing environment. In fact, the geometry of the abandoned drainage courses closely match with the geometry of the high arsenic zones suggesting a link between creation of such zones and the activity of the fluvial system. Field verification confirms that the arsenious zones are confined along the levees of these drainage channel. The major drainage in the area under study, Hugli (a stretch of Bhagirathi) river has a low sinuosity value of 1.14 which is significantly less than a meandering river with the sinuosity value of 1.50. This particular aspect has direct correlation with the location and distribution of high arsenic concentration in the groundwater (Sengupta, 2002). The arsenic concentration is mostly high (>0.05 mg/l) in the levees of abandoned meander belt (Figure 1). The relatively younger flood plains (Figure 1) do not show high concentration of arsenic in the groundwater. Again, within the older flood plains, the formation, which is consisting of almost continuous clay bed/layer, is free from the high arsenic concentration.

DISCUSSION

Visual interpretation of IRS data aided by field checks has been proved very useful in the mapping of the present fluviogeomorphlogical pattern. The aquifer sediments show an upward fining cycle with variable thickness. The sandy lithofacies are dominantly channel fill and the finer muddy lithofacies is over-bank levee deposits formed during floods.

Arsenic laden FeOH finegrained sediments are deposited in the organic rich estuaries and wet lands of the Bengal basin. The top part of the sequence encountered in the bore holes from the two zones, shows certain differences. In the safe water zone, the sand and silt layers covered by a 25–30 m thick capping of clay with occasional peat bands. In contrast above the arsenious water zone, the clay capping is either partially preserved or totally absent. Clay along with the peat in the upper part possibly represents a reducing environment. McArthur (2001) suggested that the buried deposits supply arsenic to groundwater through microbial degradation. Since the microbes are more active at the air-water interface, the sediments having higher porosity in the top of the sedimentary column will be more suitable sites for proliferation. Now, the sandy lithofacies and muddy lithofacies or capping present here show a marked difference in vegetal cover and crop pattern and which is clearly discernible in the standard FCC of IRS IC/ID LISSIII and PAN merged data.

Fluviogeomorphological patterns, which are extracted through visual interpretation of the satellite data, offer clues through sinuosity. Sinuosity (ratio of arc length and axial length) of the river channels show a direct correlation with the location and distribution of high arsenic in groundwater. Since agricultural practice demands high volumes of water, agricultural land use patterns guide the location of water harvesting structures. The fresh groundwater, which can be exploited from below the thick clay capping, is to be used in conjunction with the fresh surface water stored in the demarcated areas (Figure 1) adjoining to the agricultural field.

CONCLUSION

The surface water stored in many ponds of the West Bengal countryside becomes almost dry in summer months. A proper watershed management approach is to be taken to make them perennial and the water can be used after proper treatment. Information layers derived through visual interpretation of IRS IC/ID LISS III and PAN merged data aided by field verifications have been proved useful for analysis in GIS environment aiming towards

conjunctive use of surface and ground water. Artificial recharge of arsenic contaminated aquifers by fresh water will reduce the arsenic level up to the safe limit through dilution. The dugwell (situated in phreatic aquifer) water is more hygienic than surface water and, therefore, it is desirable that at least domestic water needs are to be met from the dugwell as a source. In cases where the phreatic aquifer has arsenic above the permissible limit, the excess arsenic will get co-precipitated with the iron on being exposed to aeration (oxidation) in the cistern of the well.

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Water quality improvements during aquifer storage and recovery at ten sites

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Abstract

Literature reviews and field evaluations at up to ten aquifer storage and recovery sites in USA, Australia and The Netherlands were undertaken to determine the fate of natural organics, disinfection by-products, selected endocrine disruptors and pathogens during subsurface storage. This paper summarises American Water Works Association Research Foundation (AWWARF) project 2618 which evaluated attenuation of these species and incorporated this knowledge in several types of models.

The paper briefly characterises the field sites and explains the formation and role of geochemical conditions in the aquifer that influence attenuation rates of these species. Organic matter degradation occurred most rapidly for larger molecular weight materials, including polysaccharides, and organics containing functional groups susceptible to microbial degradation. Low molecular weight acidic material was found to persist and had a signature indistinguishable from native groundwater. Trihalomethanes were degraded under anaerobic conditions with degradation of chloroform requiring reducing conditions and occurring fastest under sulphate reducing and methanogenic conditions. Haloacetic acids were degraded under aerobic and anaerobic conditions. Of five endocrine disrupting chemicals tested, two were degraded under aerobic conditions and none under anaerobic conditions in the absence of a co-metabolite. A selection of pathogenic viruses, bacteria and protozoa were found to inactivate during residence within the aquifer with fastest inactivation under aerobic conditions. It is evident that passage of water through aerobic and anaerobic conditions maximises the opportunity for contaminant attenuation.

At two sites three models of varying complexity were used to describe changes in water quality on a range of time scales. While two of the models are not validated they do reflect the observations for a large number of water quality parameters, suggesting that the process descriptions incorporated may be reasonable, and that an understanding of the biogeochemical interactions during ASR is emerging.

INTRODUCTION

In many countries high quality drinking water supplies are drawn from aquifers that are recharged naturally by continuously polluted surface waters. With increasing knowledge of groundwater contamination from point sources, there is also an increasing reliance on passive remediation for some classes of contaminants. From these two observations the hypothesis has emerged that while pollution should be avoided, there are sustainable treatment processes at work in aquifers, and so long as assimilative capacity of an aquifer ecosystem is not overloaded, an aquifer potentially can provide sustainable water quality improvement during Management of Aquifer Recharge (MAR). An international project supported by the American Water Works Association Research Foundation was conducted (Dillon et al., 2005a) to establish rates of attenuation of a limited suite of some of the most ubiquitous pathogens and organic substances. These rates were measured at ten field sites and, for a few emerging contaminants of concern, in laboratory studies.

The range of aquifer and injectant types covered is broad so it is likely that proponents of new ASR projects will find at least one case study, documented in Dillon et al. (2005b) relevant to their situation. Monitoring is important, not just for compliance but to build an understanding of subsurface processes that improve or impair the quality of recovered water.

The project involved a consortium of ten organizations, six with Aquifer Storage and Recovery (ASR) (recovery from injection well) operations or trials and four with Aquifer Storage Transfer and Recovery (ASTR) (injection and recovery via separate wells). The sites were located in USA, Australia and The Netherlands and in each case the quality of injected and recovered water had been monitored to determine changes. The first goal was to determine the fate of disinfection byproducts and selected pathogens at sites where these were monitored. The second objective was, with the assistance of literature and the field observations, to categorize and determine factors affecting attenuation rates and to perform laboratory tests to fill in some gaps. Laboratory tests also included determining attenuation rates of selected endocrine disruptors for one aquifer. This led to the third goal, to fit models to describe the data with a view to building confidence in our understanding of processes and as a first step towards prediction of sustainable contaminant attenuation.

METHODS

The characteristics of the ten field sites are given in Table 1. Source waters included reclaimed water, (treated sewage effluent given advanced additional treatment), stormwater, surface waters with various levels of treatment and groundwater. Seven of the sites, including all USA sites, used disinfected injectant. Aquifers in all but one case were confined, but three of the confined sites contained aerobic groundwater. Four were limestone aquifers, and the other six were sand and gravel with different degrees of consolidation. There were various water quality issues at all sites but the dominant issue from USA sites was the fate of disinfection byproducts and in particular trihalomethanes which at some sites had increased in concentration during storage. At two Australian sites and the Netherlands site injectant was not disinfected and pathogen monitoring was undertaken. Several sites had unique water quality issues but these were not pursued either because of lack of appropriate monitoring data, or because the issues (e.g. mobilisation of metals, hydrogen sulfide production, reaction of chlorine with natural organic matter, and mixing with mineral or radon-rich groundwater), fell outside the scope of the project which was focussed on water quality improvements, and are subsequently being addressed in a sequel project.

Recognising that there are many other ASR and ASTR sites where relevant data may have been collected, an effort was made to contact 30 operators of other ASR sites, to categorise water quality issues. A water quality questionnaire was appended to that a questionnaire developed by Dr Ivan Johnson for ASCE, converted to electronic form, web mounted, circulated by email and by mail, but aside from the ten existing partners, this met with a poor response. One of the key criteria sought was for monitoring of at least one conservative water quality parameter which could be used to differentiate the proportions of injectant and native groundwater in each recovered water sample. Those parameters are shown in Table 1. Ultimately the only additional data used was that on DBPs from an ASR project at Oak Creek Wisconsin (Miller, 2001). Adsorption and biodegradation of five endocrine disrupting species were evaluated in laboratory batch studies using Bolivar aquifer material inoculated with fresh groundwater and in sterile controls, with methods described by Ying et al. (2003).

Of the ten sites for which information was available it became clear that an input-output analysis was only possible for selected parameters at each site. For example DBP data was available for 7 of these 10 sites, and pathogen attenuation data were available at 4 of the sites (all of which were outside the USA). The simplest form of modelling, a risk index approach was applied at all sites. It became clear that the data available at Bolivar and Someren provide the strongest basis for evaluation of processes, hence these are the two sites at which the more advanced process models were applied.

Study Site	Source water	Aquifer *	Water Quality Considerations #	Mode and Scale of Operation	Conservative Tracers
Bolivar, South Aust.	Effluent, 2ndary treated + lagoon + DAF/F + chlorination	Confined, Anaerobic Limestone +sand (Core)	Pathogens, OC N, Fe, Mn	Single ASR well research site with 14 obs. wells < 300 m. 200 ML/ year started 1999.	Cl, Br, F, deuterium, oxygen-18
Talbert Gap, Orange County, CA, USA	Effluent, 2nd ary treated + lime clarification + recarbonation + filtration + RO (1/3) or GAC (2/3)	Confined, Aerobic Gravel + sand with clay layers (Cuttings)	OC – DBP's, PhAC's	Separate injection and recovery wells (ASTR), many obs wells, 20 GL/yr since 1975.	Cl
East Bay, Oakland, CA, USA	Sierra Nevada aquiduct, plus local runoff, filtration and chloramination	Confined, Aerobic Gravel and sand, some clay (Cuttings)	N + nitrifiers, Radon, OC-THM's, Mn	Single ASR well pilot started 1998, obs wells, 3@ r=15 m, 1 @ 60m	Cl
Las Vegas, NV, USA	Lake Mead, Colorado R. (incl. some treated effluent discharge-ed to river) with coagulation, filt-ration & chlorination	Confined, Aerobic Cemented gravels and sands (no samples)	OC – TOC, perchlorate Viruses and crypto Hardness	Separate injection and recovery wells (ASTR) and some ASR wells, up to 33 GL/yr for 10 years	Cl
Charleston, SC, USA	Edisto River and Black River, treated	Confined, Anaerobic Limestone & fractured sandstone (no samples)	THM, HCO_3, H_2S, Cl	Single well ASR started 1994, 10 ML , aiming for 440 ML/yr emergency supply	Cl
Someren (DIZON), Netherlands	Canal water, treated by flocculation, flotation, rapid sand filtration, AC, slow sand filtration	Confined, Anaerobic Sand(Core)	Fe, Mn, N, SO_4, temp, As, Ba, Co, Ni, Zn, pathogens	Separate injection and recovery wells (ASTR) 98m apart, 200 days residence, started 1996 4 obs wells	Cl, HCO_3, Temp
Jandakot, W.Aust.	Surface water or shallow groundwater, chlorination	Confined, Anaerobic Sand and sandstone (no samples)	DOC, DBP's , patho-gens, Fe, Mn, H_2S, N, turbidity, temperature	Trials commenced 1999/2000, aiming for 15-30 GL/yr in 10-20 years	Cl
Clayton, South Aust.	Lake water, filtration	Unconfined, Aerobic Karstic limestone (no samples)	Pathogens, Hardness Salinity	Separate injection and recovery wells (ASTR) 20 ML/yr recovered since 1997	Cl (EC)
Andrews Farm, South Aust.	Urban runoff, detention pond, screening	Confined, Anaerobic Limestone (Core)	Pathogens, Pesticides, Salinity	ASR well 1993-98, 3 obs wells at 25,65 & 325m. Mean 50 ML/yr	Cl
Memphis, TN USA	Ground-water untreated	Confined, Anaerobic Sand alluvium (no samples)	THM's	Single ASR well, commenced 1998	Fl

*   only sites where cores were available had mineralogical analyses
\#   many water quality parameters were monitored besides those listed and frequency and locations of measurements varied between and within sites.

WATER QUALITY IMPROVEMENTS

When water is injected into an aquifer biogeochemical processes change the quality of water that is subsequently recovered. In anaerobic aquifers, ASR with aerobic water provides a redox gradient that is the reverse of that encountered at sites where organic contaminants pollute naturally aerobic unconfined aquifers. When injected water progresses through a redox gradient this allows exposure of the water's constituents to a succession of microbial communities each of which may be competent in reducing inorganic species, such as nitrate or sulphate, and of degrading different organic contaminants. Where injectant contains particulate degradable organic carbon, highly reducing conditions can occur adjacent the injection well during storage periods, further enhancing the range of species degraded (e.g. including chloroform). In cases where redox gradients are not observed, the attenuation processes are not as comprehensive for organics but may be at least as effective for pathogens. Related changes such as changes in pH, dissolution of minerals and cation exchange also contribute to water quality changes observed at ASR sites.

Table 2 summarises the findings of the study. Readers are referred to (Dillon et al., 2005b) for a comprehensive summary of water quality improvements at each site, and to (Dillon et al., 2005a) for a detailed account of the attenuation of each species, including process descriptions and references to a large body of literature. Due to space constraints only a very cursory summary of results is given in Table 2 for the major species and processes. There are many subtleties so readers are referred to the original report for more detail.

Table 2. Time[a] (days) to attenuate 90% of viable micro-organism numbers or concentrations of species injected into aquifers under aerobic and anaerobic conditions (after Dillon et al., 2005a)

Species	Aerobic aquifer (days)	Anaerobic aquifer[b] (days)
Pathogens[c]		
Viruses	1–10 (2)	8–143 (4)
Protozoa	ND[d]	21 (1)
Bacteria	<1–7 (2)	2–8 (4)
Disinfection byproducts		
Chloroform (CHCl3)	190 – UD[^] (3)	60–1130 (3)
Total Trihalomethanes	260 – 1000[e] (3)	33–530 (3)
Endocrine disruptors		
Estradiol (E2)	6 (1)	400 (est at 70d) (1)
Ethynlestradiol (EE2)	560 (est. at 70d) (1)	UD (1)
Bisphenol A (BPA)	UD (1)	UD (1)
4-t-Octylphenol (OP)	UD (1)	UD (1)
4-n-Nonylphenol (NP)	25 (1)	UD (1)

a. To convert t_{90} (one $\log_{10}$ reduction) to half life, divide by 3.32.

b. Anaerobic conditions here may be taken as supporting denitrification but not sulphate reduction. A wider range of conditions for degradation of THMs is reported in Dillon et al. (2005a).

c. Composites of several enteroviruses, protozoa and bacteria in diffusion chamber and lab studies using groundwater from a wider range of aquifers, including the Halls Head MAR site reported in Toze et al. (2004).

(n) Indicates number of sites from which these data were derived, where n is between 1 and 4.

d. ND = Not Done.

UD^ = Undegraded in 130 days.

e. Upper value estimated at 130 days.

UD = Undegraded in 70 days.

Temperature range for pathogen studies was 21 to 28 °C and for DBP studies; Aerobic: 18–21 °C; Anaerobic: 13–26 °C.

Disinfection By Products (DBPs)

The fate of DBPs, principally trihalomethanes (THMs), was determined at eight ASR sites, including Oak Creek, Wisconsin. Mass balance calculations were performed to assess the proportion of injected DBPs that were recovered. DBPs were reduced within the aquifer at five of the sites. Microbial attenuation of DBPs such as THMs and haloacetic acids (HAAs) has been demonstrated both in the laboratory and at ASR sites. HAA removal can occur relatively quickly under both aerobic and anaerobic conditions. Removal of chlorinated THMs requires anaerobic conditions. The literature based laboratory studies suggest that removal of chloroform occurs only under sulfate – reducing or methanogenic conditions. The more highly brominated THMs are more readily removed under all conditions, with bromoform removed most rapidly. The most important factor for attenuation of THMs is the redox state of the aquifer. Adsorption and hydrolysis are not major removal pathways for THMs and HAAs, unlike haloacetonitriles (HANs) which are removed by hydrolysis. At two sites (Memphis and Las Vegas) there was significant *in situ* formation of chloroform within the aquifer, and existing DBP formation models from pipe networks did not fit the data so further work is underway.

THM attenuation rates were found to vary more than two orders of magnitude (t_{90} of <3 to >400 days) across the sites. Chloroform was found to be considerably more persistent than other THM compounds as was expected. Evidence for reduction in the DBP formation potentials (FP) occurred at the two sites where data were available. At the Bolivar site reduction in FP was the result of the removal of precursor material, but an increase in the FP per unit weight of TOC occurred as predicted from the evaluation of the fate of aromatic organic carbon (Pavelic et al., 2005).

Natural Organic Matter (NOM)

A conceptual model was developed to describe the fate of fractions of organic carbon (OC) and appears to match the observed processes at the Bolivar site. That is, filtration of particulate organic carbon, sorption of higher molecular weight dissolved organic matter (DOM) and bioaccumulation of 0-alkyl-C occur near the injection well. These contribute to active biological growth and the rapid demand for electron acceptors clearly observed during storage periods.

During injection at Bolivar, hydrophilic materials, retained on XAD-4 resin, are preferentially lost, probably through microbial decomposition. The higher molecular weight, more acidic materials are preferentially sorbed on the aquifer matrix. The more hydrophobic materials retained on XAD-8 resin are less strongly sorbed to the mineral matrix and are transported more readily through it. After an initial period of sorption, breakthrough of DOM occurred at observation wells at 4, 50 and 75 m with partial residual loss of DOM. Aromatic OC decreased at a slower rate than most DOM during residence and DBP formation potential declined at a slower rate than would be predicted based on attenuation of TOC alone. Breakthrough of reclaimed water at a well 50 m from the injection well at this site demonstrated that the organic carbon profile reverts very quickly to having nuclear magnetic resonance (NMR) spectra similar to those of natural waters (Skjemstad et al., 2002).

Endocrine Disrupting Chemicals (EDCs)

Sorption and degradation of the five selected EDCs; bisphenol A (BPA), 17β-estradiol (E2), 17α-ethynylestradiol (EE2), *4-tert*-octyl phenol (4-*t*-OP) and 4-*n*-nonyl phenol (4-*n*-NP), were investigated in the laboratory using core material and ambient groundwater from Bolivar (Ying et al. , 2003). The sorption coefficients measured on the porous media were in the following order: 4-*n*-NP > 4-*t*-OP > EE2 > E2 > BPA, with K_{oc} ranging from 39,000 to 780 L/kg, and approximating as linear isotherms for the limestone aquifer at Bolivar translated to retardation factors from 330 to 8 respectively. Degradation experiments of the five EDCs showed that E2 and 4-*n*-NP degraded quickly in the aquifer material under aerobic conditions while the other chemicals remained almost unchanged.

Little or no degradation of the five EDCs was observed in ambient groundwater within 70 days under anaerobic conditions. Hence, as with DBPs, redox status is an important factor in EDC degradation, but in this instance aerobic conditions favour degradation of E2 and 4-*n*-NP. Further biodegradation experiments using reclaimed water are planned to evaluate the potential for removal by microbial communities capable of co-metabolising EDCs. This pathway for degradation was not evaluated in this study.

Pathogens

A review of the literature indicated that there is a wide variation in the survival of any given microbial pathogen among different studies at a range of sites, as well as differences between unlike organisms at the same site (Toze, 2004). Thus, the degree of movement and survival of any microbial pathogen is expected to be site-specific. Much of the information used in the review details microbial pathogen survival in relatively quiescent environments. However, at ASR sites, injected water is subject to a range of environmental conditions, such as redox state, nutrient supply and temperature change. Literature and laboratory studies indicate that enteric pathogens compete poorly with indigenous microbiota in aquifers and that inactivation consistently occurs, although the rates differ. Very frequently pathogen inactivation could be described by an exponential function requiring only one parameter, the time for one-$\log_{10}$ removal. No attempts were made to delve into other mathematical descriptions of removal rates, but the processes by which pathogen inactivation occurs were explored in some depth. For the purpose of this research the subsurface biosphere was divided into two zones – the microbial biofilm colonising the injection well and nearby environment, and the aquifer further from the well where advection rates, mass fluxes of nutrients and microbial biomass are lower.

Varying temperature, oxygen and nutrient levels were tested in the laboratory in the presence and absence of indigenous groundwater microorganisms to determine the predominant factor influencing inactivation of the viruses. Inactivation of *E. coli* and the bacteriophage MS2 were detected by loss of culturable cells/particles whilst poliovirus and coxsackievirus loss was determined by lack of detection by RT-PCR. The results indicated that the most influential factor affecting the inactivation of the introduced viruses and indicator microorganisms was the presence of indigenous microorganisms in the groundwater. The results also imply that temperature, the presence of oxygen and nutrient levels indirectly affect virus and indictor microorganism survival, by influencing the activity of the indigenous microorganisms. All viruses and indicator microorganisms tested were inactivated most rapidly under aerobic conditions at elevated temperatures and without the addition of nutrients in the presence of indigenous groundwater microorganisms (Gordon and Toze 2003). It is interesting to note that pathogen inactivation is most rapid under conditions where disinfection byproduct (THM) formation and persistence is most problematic.

The harbouring effect of the well biofilm on pathogen survival was successfully explored using a green Fluorescent Protein gene (GFP) as a bacterial cell marker with propidium iodide (PI) staining to identify *E. coli* in a mixed population and distinguishing between viable and dead or dying *E. coli* cells. Using these techniques it was shown that *E. coli* was able to integrate into mixed-species aquatic biofilms and persisted longer than in the absence of biofilm, but for a shorter duration in the presence of reclaimed water than in groundwater (Banning et al, 2002). Further work is warranted on the effects of biofilm development on the fate of other microbial pathogens.

The role of indigenous groundwater microorganisms in the inactivation of the enteroviruses poliovirus type 1 and coxsackievirus B1 in groundwater was also investigated. A membrane with an approximate 250,000 molecular weight cut-off was used to separate viruses from the indigenous groundwater microorganisms to determine if physical contact was required for viral inactivation to occur. Poliovirus inactivation occurred with or without direct contact with groundwater microorganisms. This suggests that virucidal compounds less than 250,000 molecular weight, could have been produced by the indigenous groundwater microorganisms. Coxsackievirus required closer contact, possibly direct contact, with groundwater microorganisms for an increased inactivation rate to be observed,

indicating that coxsackievirus decay was caused by compounds that were either larger than 250,000 molecular weight or that were closely associated with the groundwater microbes. The differences in observed inactivation rates between poliovirus and coxsackievirus suggest that the indigenous groundwater microorganisms may be producing more than one virucidal compound and that these compounds may also be virus type- or species- specific (Wall 2002). While the activity of the groundwater microorganisms and the presence of extracellular enzymes are important, there are other as yet unidentified factors that influence pathogen inactivation.

MODELS OF WATER QUALITY CHANGES

Having explored to the extent possible within the project the attenuation of selected synthetic organics and pathogens, the research then focussed on incorporating this knowledge into models that could be used to assess the risks associated with reliance on subsurface treatment processes in ASR and ASTR projects. Three models were used. The simplest, ASRRI, was developed within this project and is intended for use as a screening tool. The others were existing models EASY-LEACHER (Stuyfzand, 1998) and PHT3D (Prommer et al., 2003), and were used to describe, integrate and analyse the processes inferred from observations at the two most intensively monitored sites: Bolivar and Someren. EASY-LEACHER, a spreadsheet model, was particularly useful for providing longer-term prognoses of mineral dissolution and breakthrough of constituents (macro-ions, trace elements, organic pollutants and radionuclides), especially for dual-well, basin recharge and river bank filtration systems. PHT3D is a general purpose three-dimensional MODFLOW/MT3DMS-based reactive multicomponent transport model that incorporates the USGS model PHREEQC-2 (Parkhurst and Appelo, 1999) to compute a wide range of biogeochemical reactions. It is therefore suitable to be adapted to complex geohydrological and geochemical problems. Obviously, depending on the complexity of the conceptual model and its numerical implementation the application of PHT3D requires a correspondingly greater effort.

ASRRI

A simple 'risk index' model, ASRRI (Version 1), was constructed to allow data on degradation rates to be assembled in a convenient form to predict, on a conservative basis, the length of storage time in a single-well system for organic and microbial contaminants to reach acceptable values in the aquifer. Similarly the minimum travel length and the maximum effective porosity for which recovered water quality would meet required standards, can also be estimated for a dual-well system (Miller et al. , 2002). Default values used for contaminant attenuation were derived from Table 2. When ASRRI was applied to the case study sites it correctly predicted complete attenuation of pathogens at the 4 sites that had data. However it failed to account for DBP formation within aquifers so it under-predicted DBP concentrations of recovered water in 3 of 8 sites. A model for DBP formation in aquifers is currently under development to address this. At the other 5 of 8 sites, ASRRI over-estimated DBP concentrations in recovered waters, enabling it to be considered as a potential screening model for regulatory purposes in these circumstances. Users can apply ASRRI for any species for which there is information on adsorption coefficients and attenuation rates.

EASY-LEACHER

The simple analytical 2D-spreadsheet model EASY-LEACHER 4.6 (Stuyfzand, 1998) was demonstrated to be capable of simulating the dynamic quality changes of an infiltrating, oxidizing solute, which leaches an anoxic aquifer. The main processes simulated at Someren were: displacement of the native groundwater and the leaching of exchangeable cations, organic matter and pyrite. The main reactions included were: pyrite oxidation, oxidation of organic matter and cation exchange. The general leaching sequence was, starting with the first to be leached in the most permeable zones: exchangeable cations (10-30 pore volumes), labile organic matter (LOM) (20-550), pyrite (400-2,700) and 'tough' (more recalcitrant) organic matter (TOM) (1,000-20,000 pore volumes).

When EASY-LEACHER 4.6 was applied to the Bolivar ASR trial some modifications were required to suit both the different quality of injectant and the geochemical setting. Observation wells 4 m and 50 m from the ASR well allowed the effect of multiple pore flushes (>200) at the 4 m radius to be compared with fewer pore flushes (<5) at the 50 m radius. The dominant processes were the oxidation of injected dissolved organic matter, by injected oxygen and nitrate, which occurred predominantly within 4 m, and the dissolution of calcite. Cation exchange is observed to be an important contributor to calcium concentrations during breakthrough of injectant to the 50 m radius. Leaching of calcite is predicted in 1,000 to 10,000 years at the 4 m radius, so well stability is unlikely to be an issue in the economic life of the ASR well. EASY-LEACHER was useful in explaining the sensitivity to cation exchange capacity of the mobility of ammonium and potassium in the aquifer. It was also valuable in differentiating adsorption from biodegradation in explaining organic carbon depletion in the aquifer.

PHT3D

PHT3D was successfully applied to simulate the major physical and chemical processes of the Someren field experiment, whereby the temperature-dependent pyrite oxidation by dissolved oxygen and nitrate was identified to be the dominant reactive process that was responsible for most of the observed water quality changes. The temporal and spatial evolution of the redox state and pH are very well represented by the calibrated model, both being key parameters with respect to determining the potential mobility and fate of micropollutants such as herbicides, pharmaceuticals, trace metals as well as pathogenic viruses. Ion exchange was shown to having no significant impact on water quality changes, whereas organic matter mineralisation was estimated to contribute 15% of the electron acceptor consumption. A detailed description of the modelling study is given in Prommer and Stuyfzand (2005a) and a summary is given in Prommer and Stuyfzand (2005b).

PHT3D was also used as a quantitative framework for analysing the observed data at the Bolivar site. The simulations provide a detailed quantitative description of the hypothesised physical and biogeochemical processes at the Bolivar site and reproduce very well the observed major ion chemistry including redox state and pH. The modelling study demonstrated that the locally observed significant hydrochemical changes during the storage period (i.e., at the injection/extraction well) can be well explained by microbial activity, in particular by the effect of biomass decay. In contrast, on a larger scale the observed hydrochemical data could also be successfully modelled without the detailed simulation of microbial dynamics. Ion exchange and calcite dissolution were shown to be the dominant reactive processes for the observed water quality changes at the 50 m well. The flexible nature of the modelling tool allowed rapid adaption of the numerical model to test different conceptual models and their suitability to explain the observed data. More details on the modelling study are given in Greskowiak et al.(2005).

CONCLUSIONS

The goals of the project were met to a high degree. Fate of pathogens and DBP's were determined at all relevant sites. DBP and EDC attenuation was found to depend on the redox status of the aquifer, and for pathogens biotic processes dominated attenuation rates and these processes are being further explored. The third goal, to develop models of attenuation to fit field data and increase confidence in capability of predicting attenuation rates was met for inorganic species. This also resulted in the development of a screening model that could be used by regulators to determine the likelihood of contaminants exceeding given guideline values and to predict the number of log-reductions of pathogens between injection and recovery from ASR or ASTR operations.

Quantitative information and a screening model derived from this project are expected to facilitate design and regulation of ASR and ASTR projects to achieve the required water quality targets for parameters currently considered. Observations and process-based models revealed that near the injection well resilient ecosystems establish that adapt to a range of redox conditions depending on the injection/ storage/ recovery phase of the ASR cycle.

Degradation and inactivation rates (Table 2) depend on these local conditions as well as more stable conditions further out in the aquifer. ASTR projects offer greater residence time of injectant in the aquifer than ASR projects, and hence give greater opportunity for degradation/inactivation prior to recovery. ASR has other advantages such as cost and easier maintenance to prevent clogging. The study shows that ASR and ASTR can be considered as a reliable part of the water treatment train for all pathogens and a number of the organic species studied.

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Feasibility of ASR for surface water storage in Haarlemmermeer (Netherlands)

B.C. Drijver and A. Willemsen

Abstract

The surface water system in the western part of The Netherlands is not able to deal with the current demands on the system: the risk of flooding due to storm water peak flows is too high, and the water quality in summer is below the Dutch standards due to salinization and low supply of good quality surface water. Enlarging the surface water system to transport extra water to and from the problematic areas is costly. To create extra storage facilities for water the water board of Rijnland has planned large bodies of extra surface water. However, the surface area required for extra surface water is very hard to find. ASR has been studied as an alternative to storage in surface water. The main conclusion is that ASR is technically feasible for storage of storm water peak flows (peak ASR) and for seasonal storage of good quality surface water from winter to summer (seasonal ASR). Seasonal ASR is economically very promising compared to surface water storage, also when the price of the required land is not included in the cost comparison. Peak ASR is economically attractive in urban areas, where the price of land is relatively high.

Keywords

Water management, water storage, ASR, feasibility, seepage.

INTRODUCTION

The surface water system in the western part of The Netherlands is not able to deal with the current demands on the system: the risk of flooding due to storm water peak flows is too high, and the water quality in summer is below the Dutch standards due to salinization and low supply of good quality surface water. Enlarging the surface water system to transport extra water to and from the problematic areas is costly, and the best option is to create extra storage facilities for water: both for storm water peak flows and for summer droughts. Therefore the water board of Rijnland has planned large bodies of extra surface water. However, the western part of The Netherlands is densely populated, and the surface area required for extra surface water is very hard to find. The Water board of Rijnland has calculated that around the Haarlemmermeer polder (the polder that also holds Amsterdam Airport Schiphol) a storage reservoir for 1,000,000 m^3 is needed to be able to handle storm water peak flows (peak storage). Seasonal storage of 2,000,000 m^3 is also needed in that area to meet the need for water in the nature areas surrounding the location of the peak storage facility. A plan was made to solve the need for both peak and seasonal storage by creating 2.5 km^2 of extra surface water in the form of a lake with a varying water level with dikes around the lake. However, the surface area available in this region is limited, and the demand for surface area for urban development is increasing steadily. Also, the growth of Amsterdam Airport Schiphol increases the demand for land in The Haarlemmermeer. This puts a large political pressure on the water board to minimize the demand for surface area for water storage. A possible solution for this problem is ASR. Research was done to find out if ASR is a technically en economically feasible alternative for storage in surface water.

GEOHYDROLOGY

The area of study is located in the polder Haarlemmermeer. The land surface lies below sea level, which implies that the land has to be kept dry by pumping out all net precipitation and water originating from upward seepage. Only

the shallow groundwater is fresh and the rest is brackish or salt. The groundwater table is approx. 1 to 2 m below surface level. A system of ditches every few 100 m is in direct hydraulic contact with the ground water. During wet periods fresh water is discharged and during dry periods fresh water must be supplied from elsewhere, which illustrates the benefits of storage. The geohydrology at the site of study is schematically given in Table 1.

Table 1. Geohydrology at the site

Depth (m)	Type of sediment	Layer name	Transmissivity (m²/d) or hydraulic resistance (d)	Storage coefficent
0 – 10	clay and peat	covering layer	1,000 d	0.15
10 – 83	medium fine to coarse sand	first and second aquifer	1,500 m²/d	0.0016
83 – 90	clay	second aquitard	1,500 d	
90 – 125	medium - coarse sand	third aquifer	1,000 m²/d	0,0004
125 – 133	clay and peat	third aquitard	1,000 d	
133 – 170	fine - medium sand	fourth aquifer	600 m²/d	0,0004

General guidelines for ASR (EPA, 2004) mention that ASR should be located in areas with groundwater tables at depths larger then 5 m. Such low groundwater tables are however not found in and around The Haarlemmermeer. ASR in areas with high groundwater tables require extra attention to the impact on seepage/infiltration and changes in groundwater levels.

SEASONAL ASR

Efficiency of seasonal ASR

The efficiency of seasonal ASR depends mainly on the water quality injected, the natural water quality in the aquifer, the geochemical reactions in the aquifer and the demands on the water quality produced.

Density driven flow will occur because the infiltrated fresh water (100 mg/l Cl) has a lower density than the native salt groundwater (approximately 10,000 mg/l Cl). The maximum acceptable chloride content in the extracted groundwater is 200 mg/l. Calculations indicate an efficiency of 50% after a few years (or earlier when a large amount of fresh water is injected in the first year).

The water produced from an ASR system for surface water storage should be suitable for discharge into surface water. To prevent the necessity of water treatment after production, the produced water should be oxic, and only contain negligible amounts of iron and manganese. Depending on the presence and reactivity of components as organic carbon and pyrite, the water injected into an aquifer may become anoxic. Especially during the first years these redox processes could influence the efficiency of the seasonal ASR. In cases of reactive aquifers the impact of water quality changes due to redox processes can influence the feasibility of seasonal ASR as demonstrated in the case of the DIZON pilot described by Stuyfzand et al. (2002). What the efficiency-limiting process will be: redox reactions or buoyancy flow, is unknown. For now the assumption is made that an efficiency of 50% is feasible.

System setup

The source of water for infiltration is surface water. For the design of the ASR system it was assumed that 4,000,000 m^3 will be infiltrated during 4,000 hours (5.5 month with an infiltration rate of 1,000 m^3/h). The amount of wells needed is dependent on the properties of the aquifer, the infiltration rate, the amount of water infiltrated per year and the clogging potential of the infiltrated water.

A parameter that is often used to estimate the clogging potential of water is the Membrane Filtration Index or MFI. Using the MFI, the clogging rate can be estimated using the theory described by Buik and Willemsen (2002). The surface water is estimated to have an MFI of at least 1,000 s/l^2. Water with such a high MFI will rapidly clog infiltration wells. Therefore the infiltration water has to be pretreated before infiltration. In this case the use of a natural channel bed drainage system is assumed for filtering the water. A comparable type of filtering is also satisfactory used by the Amsterdam Water Supply as described by van Duijvenbode and Olsthoorn (1998, 2002). The MFI of the filtered water is assumed to be 20 s/l^2 at most. To minimize the impact on the phreatic groundwater and to maximize the infiltration capacity per well, the wells are situated in the third aquifer. Calculations show that 14 wells are required.

Calculations

Model calculations were performed to quantify the impact of the seasonal ASR on upward seepage and on phreatic groundwater levels. At the top of the model a drainage resistance of 300 d was used. First 4,000,000 m^3 is infiltrated through 14 wells during 5.5 month. After two months without pumping, 2,000,000 m^3 is extracted during 3 months.

Upward seepage

Figure 1 shows the calculated change in upward seepage and the total flow rate of the wells. From the start of infiltration the upward seepage starts to increase. At the end of the infiltration period the flow rate of extra upward
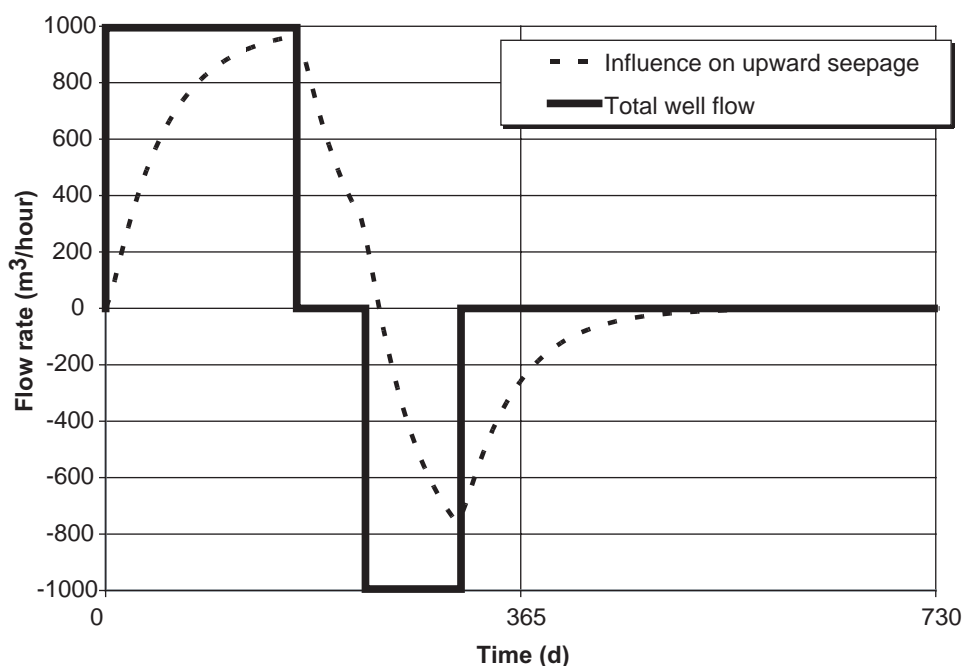
Figure 1. Impact of seasonal ASR on change in flow rate of upward seepage compared to well flow rate

seepage is nearly equal to the flow rate of infiltration. Figure 2 shows the change in cumulative upward seepage as a consequence of the seasonal ASR.

Figure 2 shows that the change in total upward seepage is equal to the net amount of water infiltrated. This means that the amount of water present in the subsoil is in the end equal to the amount that was present before infiltration. The ASR system does not store water quantity. It stores water quality. The ASR system will cause extra seepage of (brackish) groundwater in winter, and a reduced seepage in summer (or even infiltration).
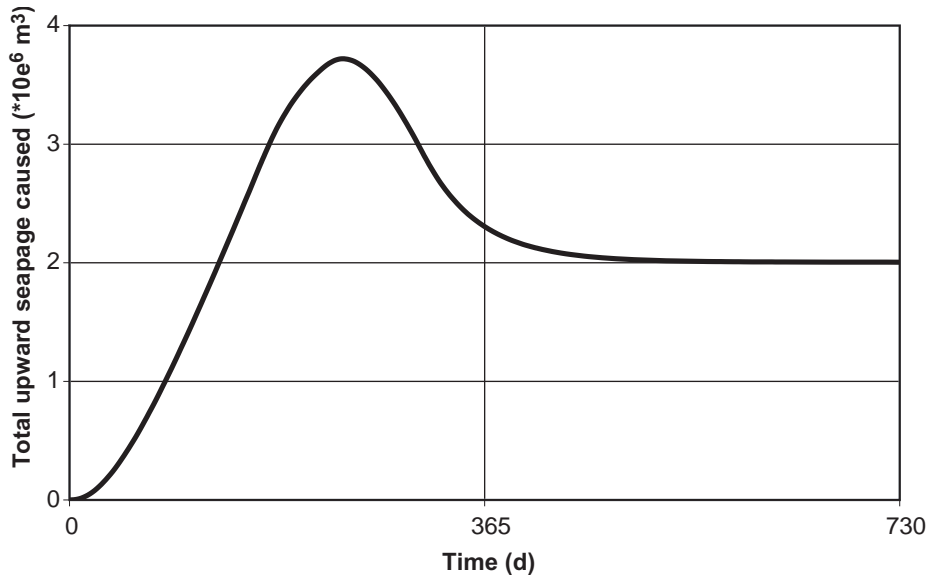
Figure 2. Cumulative change in upward seepage to surface water system caused by seasonal ASR

Groundwater table

The groundwater table in the area is typically situated 1 or 2 m below surface level. A small impact on the ground-water table can have significant consequences. In Figure 3 the calculated influence on the groundwater table is
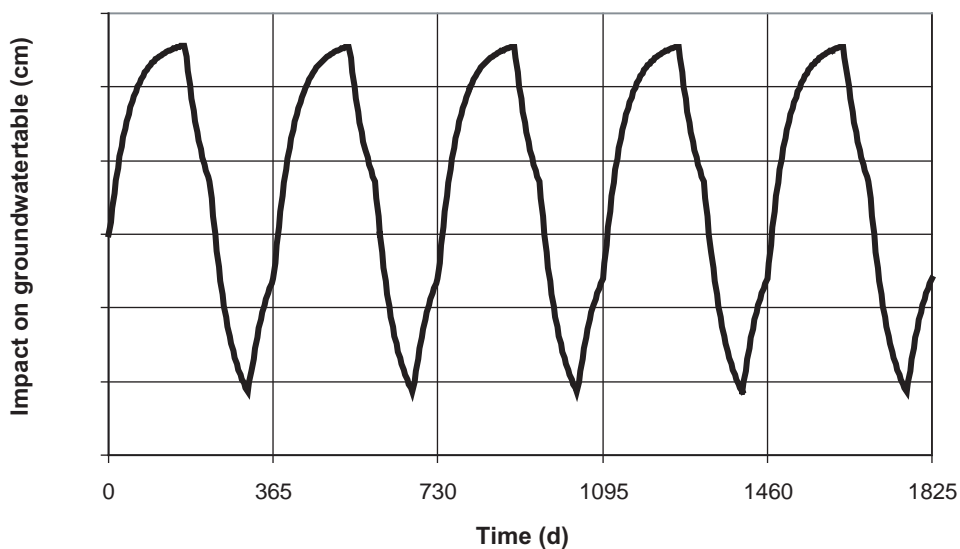
Figure 3. Calculated impact on groundwater level during five years of seasonal ASR

presented. The impact of the seasonal ASR on the groundwater table shows the same pattern every year. Calculations indicate that the maximum influence on the groundwater table is reached at the end of the infiltration period and amounts 0.13 m. This change is not expected to have significant impacts.

PEAK ASR

In case of peak ASR the goal is to temporarily store 1,000,000 m^3 of surface water in a time frame of 18 hours, which corresponds to an average flow rate of 15 m^3/s (56,000 m^3/d). On average, the peak ASR will be necessary once every 10 years.

System setup

Because of the large flow rate, water treatment before infiltration is not realistic. Also, treatment is not needed, because the clogging of the infiltration wells is not the determining factor for the design of the peak ASR: the wells clog only to a limited extent during the very short period of time that they are taking up water. The most critical parameter in the design is the maximum increase in hydraulic head without breaching the confining layer. According to the theory of Olsthoorn (1982) that was extended by Oostveen (2004) the increase in hydraulic head caused by an infiltration well should not exceed the horizontal grain pressure. Using this rule 140 infiltration wells will be needed for the peak ASR in the third aquifer.

Calculations

Upward seepage

The calculations performed for the seasonal ASR indicate that the infiltration of water is eventually fully compensated by upward seepage. Peak ASR is only useful if the amount of water that is infiltrated is larger than the amount of water that seeps upward during the peak flow event. The calculations for the seasonal ASR indicate that the upward seepage needs time to reach the flow rate of the infiltration. This time lapse between infiltrated volume and seepage volume is caused by the fact that for extra flow from the groundwater to the surface water, a rise in ground water table is required. The rise in ground water table takes place with a phreatic storage coefficient of 15%. The drainage resistance determines the head difference for a certain flow between ground water and surface water. A time lapse between infiltration and seepage requires storage, and the only significant storage is found in the phreatic groundwater level, therefore the magnitude of the drainage resistance will determine the success or failure of peak ASR.

The magnitude of the drainage resistance in The Haarlemmermeer is however not very well known. When only the surface water is taken into account, a drainage resistance of 300 d seems realistic. However, when the groundwater level is high, a series of shallow drains might become active, which will lower the effective drainage resistance significantly. In that case a drainage resistance of 10 d might be more realistic than the 300 d used for the seasonal ASR calculations, which means that the seepage of groundwater to the surface water system will respond a lot quicker.

Figure 4 shows the results of calculations performed for the peak ASR with respect to the change in total upward seepage caused by infiltration of 1,000,000 m^3 in 18 hours. The figure clearly shows the significance of the magnitude of the drainage resistance. Other parameters are less sensitive. But even for a low value of the drainage resistance of 10 days, the maximum seepage flow rate is only 11% of the infiltration rate. This illustrates that peak ASR can be a solution for temporarily storing large volumes of water within a short amount of time.
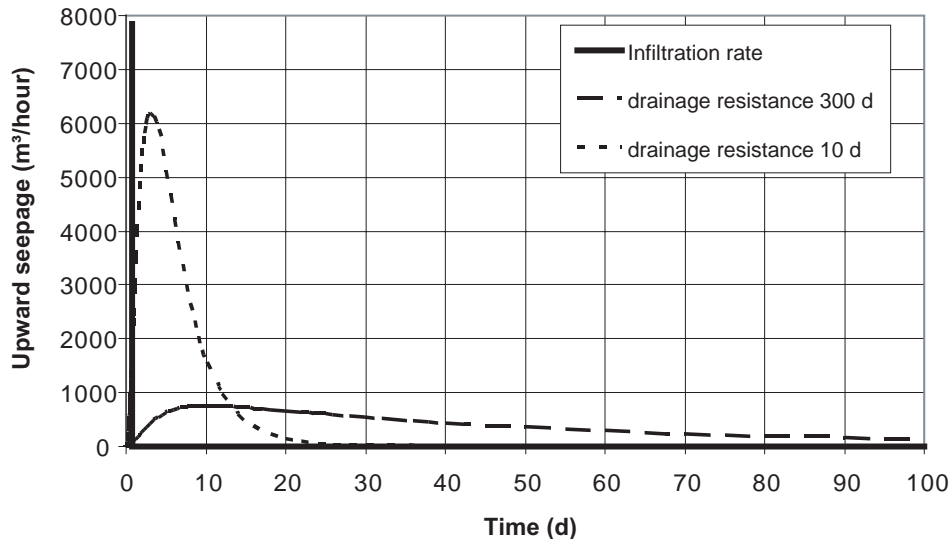
Figure 4. Impact of peak ASR on change in total upward seepage at different drainage resistances

Groundwater table

The calculated impact on the groundwater table in case of a drainage resistance of 300 d is given in Figure 5. The calculated influence is very small. The calculated maximum rise of the hydraulic head in the first aquifer amounts to 1.44 m. This rise of hydraulic head may cause the upward pressure to exceed the vertical grain pressure below ditches. This aspect requires further research. In this feasibility study it was assumed that a short time with high pressures would not harm the integrity of the covering layer.
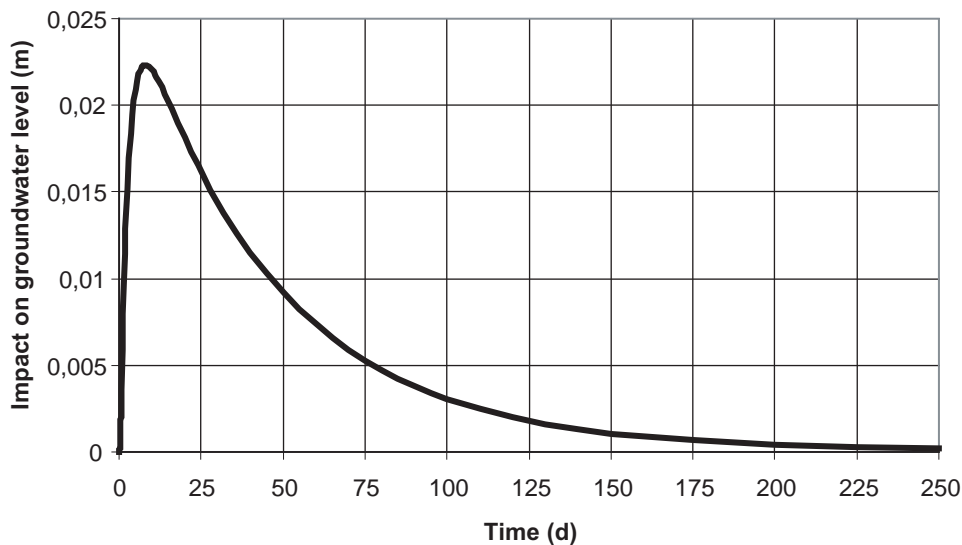
Figure 5. Calculated impact of peak ASR on groundwater level

FEASIBILITY

The major advantage of ASR as compared to storage in surface water is the minimum amount of space needed. Other advantages of ASR are the freedom to choose the locations of the ASR and the fact that an ASR is never full. Disadvantages might be the increase of upward seepage in certain periods and the potential impact of redox processes in the aquifer on the quality of the stored water. An item not discussed yet is the potential presence of

pollutants in the surface water. The cost for peak and seasonal storage (combined) in 2.5 km^2 of surface water is estimated to be 35,000,000 (9 million for peak and 26 million for seasonal storage when split up). This price is based on the assumption that the land is obtained free of charge, because the land for the storage reservoir would be bought by the state to increase the area of nature in The Netherlands. The cost of seasonal ASR amounts to 6 million and to 29 million for peak ASR, so also 35 million for a combination. It appears that the cost of seasonal ASR is much lower than the cost for seasonal storage in surface water. Peak ASR is more expensive compared to the peak storage in surface water, as long as the price of the land is low. Peak ASR is becoming economically attractive when the price of land is larger than approx. 30/m². Normal prices in urban areas are higher than 100/m^2. Agricultural land in The Haarlemmermeer has a price of approx. 10 /m^2.

EVALUATION

Seasonal ASR is economically very attractive compared to seasonal storage in surface water. Peak ASR is expected to be an attractive option in urban areas. Because of the advantages of ASR compared to storage in surface water it was recommended to proceed with a pilot. The most important aspects that should be subject of study during this pilot are:

- can natural channel bed filtration be an effective method for filtering the water for seasonal ASR?; what will the MFI of the water be after filtering?;
- what MFI values can be expected in surface water?; what is the relation between MFI and the rate of clogging at the high MFI values found in surface water?;
- what is the efficiency of seasonal ASR: what is the influence of buoyancy flow and redox processes?;
- is the impact of ASR acceptable (influence of pollutants in surface water; breaching of the covering layer, settlements, upward seepage, groundwater level, etc.);
- social aspects (acceptation by the population in The Haarlemmermeer);
- legislation and permitting;

ACKNOWLEDGEMENTS

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NLARS: Evolution
of an Artificial Recharge Scheme

Sally Harris, Marcus Adams and Michael Jones

Abstract

The North London Artificial Recharge Scheme (NLARS) was developed as a drought management tool, however emphasis on the Scheme's use is now changing. NLARS takes advantage of the large dewatered volume in the confined Chalk and Basal Sands aquifers that resulted from historical pumping. Additionally, the Lea Valley reservoirs and the New River aqueduct provide inexpensive methods of transferring the raw water to the treatment works. Recharge trials in the 1950s and 1960s led to an operational artificial recharge scheme in the late 1970s. During the early 1990s further developments resulted in what is known now as NLARS. It consisted of 35 boreholes, and was capable of 150 Ml/d output with a recharge capability of 45 Ml/d. The Scheme was further extended between 2001 and 2003 to 41 boreholes and an output of 200 Ml/d, with recharge capability increased to 78 Ml/d. In the next 3 years NLARS drought abstraction capability will be further increased by 30 Ml/d. NLARS is now evolving to allow the management of shorter-term operational requirements. Some boreholes have recently been equipped to provide direct supply to the network during annual peak demands. Should this mode of operation prove successful then it may be adopted elsewhere in the Scheme.

Keywords

Artificial Recharge, Chalk, Groundwater Resource Management, London, NLARS.

INTRODUCTION

NLARS, North London Artificial Recharge Scheme, is currently the only large-scale operational artificial recharge scheme in the UK. Located along the Lea Valley of northeast London (Fig. 1), boreholes and wells provide a supply of raw water during periods of drought. Aquifer storage is then recovered at times of low demand by artificial recharge using potable water through the same sources (Fig. 1). The Scheme in its current form has been licensed and operational for 10 years, but its history and evolution can be traced back over 100 years. NLARS comprises 41 boreholes and wells (Fig. 2) abstracting groundwater from the confined Chalk and Basal Sands aquifer. The sources either abstract directly into the Lea Valley Reservoirs system (Lea Valley Sources) or to a 400 year old aqduct, the New River, further west (New
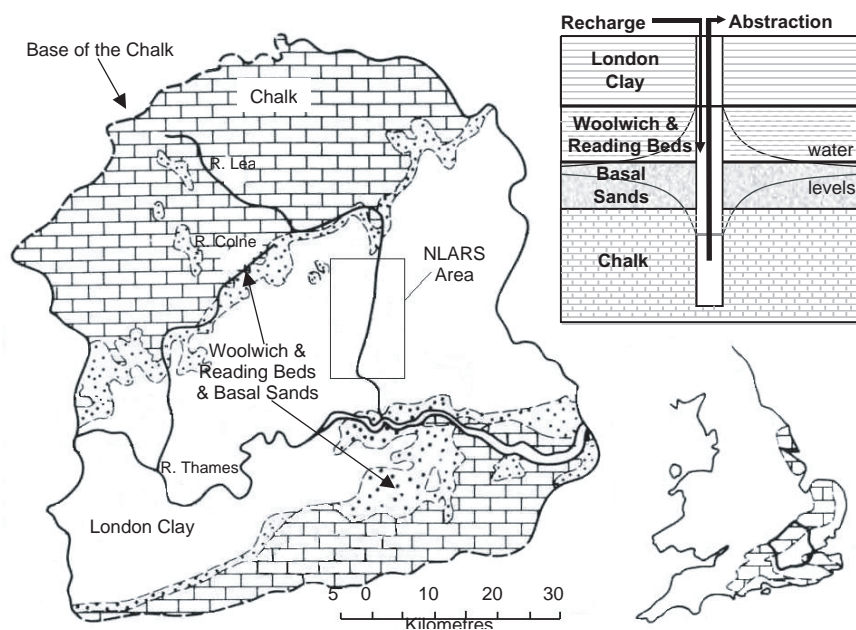
Figure 1. Simplified geological map showing location of NLARS

River sources). Water then gravitates to the major Coppermills Water Treatment Works at Walthamstow or the smaller Water Treatment Works at Hornsey, negating the need for a raw water network. A number of the sources are equipped for artificial recharge with connections to the water mains network. During periods of low demand, excess water can be recharged to the aquifer at a rate appropriate to maintain network pressures.

GEOLOGY AND HYDROGEOLOGY

NLARS is located within the London Basin (Fig. 1). The geological succession comprises alluvium resting upon a Tertiary sequence of London Clay, Lambeth Group and Thanet Sands, in turn resting on Cretaceous Chalk. The strata have an overall dip to the south. The Chalk and sandy sequence (Basal Sands) above comprise a major aquifer system, with a general hydraulic gradient to the south. Natural recharge in to the area is very limited as a result of the distribution of low flow boundaries produced by low permeability Chalk and faulting. This limits flow into the confined aquifer. It is also reflected in source transmissivity values ranging from 80 m^2/day to >2,000 m^2/day. Historical over-abstraction has de-

Figure 2. Borehole locations within the NLARS Scheme

watered the Basal Sands producing a potential zone for storage by artificial recharge. The abstraction and recharge points comprise an amalgamation of old hand-dug wells with adits and drilled boreholes. These vary in diameter between three metres and 760 mm, extending to a depth of up to 130 m, and produce yields of between 20 Ml/d and 1.3 Ml/d.

LICENSING

The Scheme is currently regulated by an abstraction licence and an Operating Agreement. The licence restricts daily output to 275 Ml/d, but this is not a constraint as the total peak output is just over 200 Ml/d. There is also an annual abstraction limit of 55,000 Ml, equivalent to 150 Ml/d. The Operating Agreement describes the trigger for Scheme use in a drought, as defined by flows in the River Thames and levels of reservoir storage. The Agreement also specifies a water level monitoring network to assess the degree of aquifer storage. It also stipulates a

programme of water quality monitoring put in place to identify at an early stage any adverse impacts on ground-water, potentially associated with the recharge process.

HISTORICAL DEVELOPMENT

The history of artificial recharge in the Lea Valley can be traced back to the end of the 19th century following decades of over-abstraction from the confined Chalk aquifer. The associated decline in abstraction capability of groundwater sources and the formation of a de-watered zone led to pioneering work at a series of well stations to investigate the viability of artificial recharge to recover storage. Since then, the gradual emergence and evolution of an artificial recharge scheme into its current form can be sub-divided into a series of phases:

- Phase I   Detailed investigations carried out in the 1950s
- Phase II   Implementation of the Lea Valley Scheme in the 1970s
- Phase III Development of North London Artificial Recharge Scheme 1990s
- Phase IV Augmentation of NLARS 2001–2003
- Phase V   Further expansion 2005–2008

Phase 1

Although some artificial recharge trials were carried out before World War 1 in the 1890s, it was not until the 1950s that controlled experiments were carried out using a group of adited wells at the southern end of the Lea Valley (Boniface, 1959). In 1953, Lea Bridge Well investigations using filtered and chlorinated river water at rates of up to 18 Ml/d were followed by trials on the Coppermills Well system at 9 Ml/d over the winter of 1954/55. High demands over the rest of 1955 and into 1956 required an extended period of abstraction from which it was seen that some 30% of recharge water was recovered and that a previously declining trend in water level had been reversed in some areas. No long-term adverse impacts on water quality were observed. A further recharge trial ensued with the inclusion of two additional well systems and the overall percentage recovery in recharge water rose to 37%. The whole area experienced either a cessation in decline or slight increase in groundwater levels. The experiments were therefore considered a success.

Phase II

Detailed experimental work was carried out between 1972 and 1974; treated water was injected through an existing Lea Valley well, and through specially drilled boreholes into the Chalk and Basal Sands at a New River site. This led to a pilot scheme consisting of six existing wells and seven new boreholes (Hawnt et al., 1981). This gave the scheme an abstraction capacity of 80 Ml/d. The scheme (then known as the Lea Valley Scheme) was formally licensed in 1977 and recharged over a five-month period between December 1977 and April 1978. 5,800 Ml of treated water was recharged at a peak rate of 53 Ml/d and an average of 41 Ml/d with recharge rates at individual sites of 1.3 Ml/d to 11.5 Ml/d. Abstraction capability was tested for 10 weeks starting in September 1978, abstracting a total volume of 2,900 Ml at an average rate of 37 Ml/d, although not all of the sites were available for abstraction. Further recharge was carried out during the winters of 1978 and 1979, resulting in a total recharge volume of 21,000 Ml by the end of 1980 (Flavin and Joseph, 1983).

Phase III

In the early 1990s a major phase of borehole drilling was invoked to greatly increase the abstraction and recharge capacity of the Scheme (O'Shea et al., 1995). A total of 14 new boreholes were drilled and tested along the New

River. These boreholes increased the maximum yield to 150 Ml/d. Five of the new sites were equipped for artificial recharge and connected to the mains network whilst the remainder were partially equipped for possible connection to the mains system at a later date. At each site, a pair of observation boreholes were drilled, one deep hole into the Chalk, the other shallower into the Basal Sands only. In this way the hydraulic connectivity between the two parts of the aquifer system could be fully investigated and the local response of the aquifer to abstraction and recharge observed. The Scheme was formally licensed as the North London Artificial Recharge Scheme in 1995.

Phase IV

Between 2001 and 2003, further Scheme augmentation was implemented in two parts. Part one involved the re-furbishing of seven existing well and borehole sites to maximise their abstraction capability, and in part two, five new boreholes were drilled, tested and commissioned. Four of the latter were sited along embankments of the Lea Valley reservoirs whilst the fifth was located along the New River. Only one of the new sites was equipped and connected for recharge, but additional recharge capacity was achieved by connecting four older sites, which were previously abstraction only. As in Phase III, several observation boreholes were drilled to help provide a better understanding of the hydrogeology and to allow changes in aquifer storage to be tracked. At the end of Phase IV, the abstraction potential had been increased to in excess of 200 Ml/d, whilst the theoretical recharge capacity had been raised to 78 Ml/d. However, network pressure restrictions and general resource availability meant a more realistic maximum of 60 Ml/d of artificial recharge.

Phase V

Phase V is currently being planned. So far, 11 new borehole sites have been identified, which it is hoped will add a further 30 Ml/d abstraction capability. This includes a disused well source on the New River, which has recently been tested and its viability demonstrated. Up to six of the new sites will be recharge-capable whilst one existing operational site in a key area of the Scheme will be connected to the network. This Phase of work is scheduled to start in summer 2005 for completion in 2008.

NLARS OPERATION SINCE 1995

Since the licensing of NLARS in its current form in 1995, the Scheme has seen two major abstraction events, in 1997 (O'Shea and Sage, 1999) and 2003 (Fig. 3). These events yielded 10,700 Ml and 9,700 Ml respectively. On
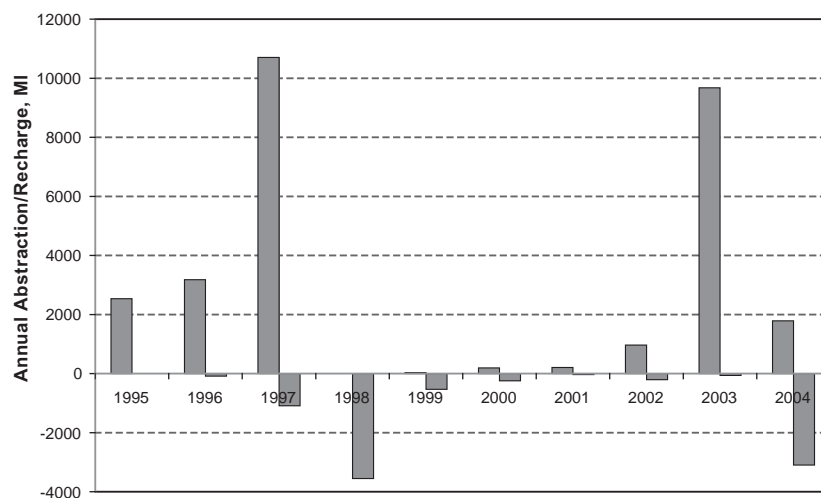
Figure 3. Operation of NLARS since 1995

both occasions a significant recharge period followed to recover storage as quickly as possible and return the Scheme to its pre-abstraction capability.

POTENTIAL FOR GREATER FLEXIBILITY IN USE

The original concept for NLARS was as a drought management tool to be kept in reserve for major, albeit infrequent, water resource shortages in the Lea Valley. This remains valid, however, over the last two to three years it has become clear that NLARS has the potential to play an important role in supporting more immediate, shorter-term water supply operations. The Scheme, or sub-sections of the Scheme, has shown that there is potential for NLARS to be operated with more flexibility to offset local raw water supply problems, for example, recent supply reductions associated with algal blooms, groundwater contamination and Thames Water's own engineering activities.

Annual algal blooms in reservoirs

The surface water stored in the raw water reservoir system is prone to a sharp deterioration in water quality towards the end of winter and into spring as a consequence of the growth of algae. The algae clog up sand filters rapidly such that the need for cleaning increases dramatically and output from the water treatment works has to be reduced. Critically, this can occur in winter at times of high demand when water mains burst rates are at their peak. One mitigation measure used has been the use of selective NLARS sources, particularly from the Lea Valley sub-scheme. Groundwater from these sources is used to blend out the reservoir water close to the main intakes for Coppermills and at Hornsey thereby reducing the impact of the algae.

CHARS: Chingford Artificial Recharge Scheme. The annual impact of algal blooms under peak demand conditions has highlighted that demands can stretch the capability of the Coppermills and Hornsey Works, due to significant filter bed outages. One solution being put in place is the construction of a new treatment works at Chingford. Using reservoir water, this works will boost the treated water capability of the area in times of high demand. However, the reservoir water quality will remain a potential issue as with the other surface water plants. As a mitigation measure, a group of NLARS boreholes along the Lea Valley has been adapted to directly supply 20 Ml/d to the new works at critical periods of the year. These boreholes will operate at times of poor reservoir water quality for one or two months per year. The new treatment works has the added advantage of providing an additional source of recharge water. The combined recharge capacity of the boreholes is around 8.5 Ml/d, thus to recover storage, post- abstraction, the recharge phase will probably need to last for 3 to 5 months. With adequate recharge the boreholes will retain their full abstraction potential in a drought.

Lea Valley Groundwater: bromate contamination

Thames Water also supplies an area to the north of NLARS along the Lea Valley. Although all the treated water is supplied from the Coppermills Works to the south, a major raw water supply of approximately 100 Ml/d is obtained from this area from the unconfined Chalk. These sources lie along the northern stretch of the New River (the Northern New River Wells) and pump directly into the aqueduct. Since 2002, these sources have experienced significant bromate contamination, which has infiltrated into the groundwater system. The suspected source of pollution is a former factory, some 20 km to the west, which manufactured bromine-based chemicals. The impact of the contamination varies at each source both in terms of bromate concentration and timing of its seasonal peak, but the outage of the wells due to bromate has at times been significant. A major outage in these wells during a dry weather event in late 2003 was largely offset by the strategic use of NLARS to support flows to the New River.

HARS: Hornsey Artificial Recharge Scheme. Bromate removal at Hornsey Treatment Works is currently being considered, but it is unlikely to be commissioned before 2008. As a mitigation measure, three NLARS sources, all

located at Hornsey, are being connected to the treatment works intake by a raw water pipeline to provide a bromate free water source to blend with the contaminated New River supply. Blending is limited to particular times of the year when bromate concentrations peak. At other times, artificial recharge will recover storage and ensure the boreholes can still provide their full drought yield.

Engineering works on reservoirs and the New River

Since 2000, the Lea Valley Water Supply Zone has been subject to multiple engineering works, mostly pre-planned but also as a result of unforeseen problems. A particular example of the former has been the drain-down of the King George V reservoir for inspection. Drain-down commenced in the summer of 2004, but the poor winter rainfall 2004/2005 has produced relatively low flows in the River Lea reducing the possible refill rate. To support the re-filling by river water, specific NLARS sources have been operated to supplement New River flows and thereby allow additional water resources to be diverted to the King George V reservoir. Similarly, in early 2004, an unexpected use of specific NLARS boreholes was required when a section of the New River embankment failed. Although New River flow was pumped around the sections being repaired, maintaining adequate supply to Hornsey works was problematic. Consequently several NLARS sources along the New River north of Hornsey were used in a controlled, but flexible manner to provide sufficient flow to meet demand at Hornsey Works.

London Resilience. As part of a London-wide initiative to improve security of supply, particularly in times of major, unexpected outages of treated water capacity, all of the NLARS sources are being considered for their potential to provide an emergency supply using temporary treatment. It is hoped that a substantial number of NLARS sites will be able to fill the criteria of water quality, land access, mains access and yield such that temporary treatment plants could be made available to them at short notice and additional treated water made available to the network.

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ASR well field optimization in unconfined aquifers in the Middle East

Rolf Herrmann

Abstract

Storage of large quantities of potable water has become more important over the last years in the Middle East. Unconfined aquifers allow the storage of potable water for strategic purposes. The operational requirements for the recovery of water in emergency situations involve high abstraction rates over long durations. The approach utilized involves numerical simulation, groundwater information analysis, GIS capabilities and extended aquifer characterization tools that are linked to a single shared database. A 3-dimensional hydrogeological model was developed to characterize the aquifer conditions. Subsequently a 3-dimensional numerical model was generated and calibrated with the newly derived field data.

A typical surficial aquifer in the region may display a relatively thin saturated thickness compared to the overall thickness of the aquifer including the unsaturated zone. This fact can limit recovery with high abstraction rates for long durations. A thin saturated thickness raises concerns over wells running dry on recovery.

In order to maximize recovery rates, an ASR well field optimization must be carried out. A 3-dimensional numerical model was generated including the thick unsaturated zone and populated with aquifer properties developed from advanced geophysical logging. The number of wells needed, well spacing, pumping and injection rates were determined for large well fields by numerical modeling. The key factors that need to be accounted for and their effects on the operational scheme of the aquifer storage and its economical feasibility will be described. This work allows insight into the requirements necessary for the design of large ASR well fields in unconfined aquifer conditions.

Keywords

ASR, aquifer recharge, well field optimization, unconfined.

INTRODUCTION

ASR systems storing freshwater in unconfined aquifers rarely exist world-wide (Pyne, 1995). The extensive surficial aquifer systems that exist in the Middle East have been identified as potential aquifer storage and recovery sites. The unconfined aquifer conditions and the complex alluvial geologic setting made it necessary to develop and apply a new hydrogeologic technical approach to evaluate the ASR feasibility and to design an operational scheme.

A new integrated aquifer characterization workflow was developed and linked with a numerical simulator that is capable of simulating variably saturated and density-dependent groundwater conditions (Herrmann et al., 2004).

Unconfined aquifer conditions will add more complexity to an ASR system. Particularly the dynamic aquifer conditions during the injection period are of special interest, because of a mound that will extent into the unsaturated zone. ASR well field optimization is governed by several factors that will be explained in this paper.

INTEGRATED AQUIFER CHARACTERIZATION

The newly developed workflow offers a set of aquifer characterization, modeling, and simulation programs to help develop the most efficient and cost-effective solutions to water management problems. The workflow allows the

building and evaluation of geologic and hydrogeological models representing saturated and unsaturated flow conditions. Hydrogeologists, hydrochemists, geologists, and water managers can share data and results within the same environment, which encourages collaboration between disciplines. The workflow can be used to develop and test solutions to comprehensive water management issues, with unique capabilities to address all aspects of a problem, from initial characterization and model building to simulation, monitoring, and model calibration and updating.

The workflow tools provided are scalable and therefore can efficiently incorporate as many or as few of the advanced tools available as required. As a result, the workflow is ideal for both small- and large-scale water management projects, addressing the most basic to the most complex problems. A wide variety of field data can be combined with models to verify correlations and validate interpretations.

The workflow allows the integration of the following disciplines:
- geophysical interpretation,
- surface imaging and mapping,
- log interpretation and well correlation,
- complex fault and fracture modeling,
- facies and geophysical modeling,
- hydrodynamic test analysis,
- uncertainty analysis,
- surface and subsurface interaction,
- upscaling processes and property population,
- flow and mass transport simulation.

Advanced tools for aquifer characterization (Schlumberger-Technoguide, 2003) allow the development and exploration of realistic solution scenarios, reducing uncertainty and risk. Inconsistencies that may be difficult to identify in two dimensions (2D) are immediately apparent in 3D. By discounting conceptual models that do not fit the available data, uncertainty in the interpretation is reduced, resulting in a more robust model. The process is then concluded by the development of a 3D hydrogeological model that represents the conceptual model used to generate a 3D hydrogeological grid (gridding process).

The numerical aquifer simulator ECLIPSE (Schlumberger-Geoquest, 2003) is an integrated part of the workflow tools, providing a fully implicit, density-dependent, multiphase 3D flow and mass transport solution. The simulator is based on the proven technology of twenty years of experience as the reservoir simulation software for the oil and gas industry (Ellis et al., 1996). Aquifer systems can be more fully understood with the simulation of variably saturated conditions, flow and mass transport modeling, and density-dependent modeling for brine or coastal aquifers. Processes that may influence the well performance such as clogging and flow dependent skin can be addressed. The workflow platform allows building and updating of large aquifer models in near real time.

STORING WATER IN UNCONFINED CONDITIONS

A large number of surficial aquifers in the Middle East consist of Alluvial and Eolian sediments. The total thickness of the sediments varies between 5 and 100 meters. The saturated aquifer is generally less than half of the total thickness of the sediments. The aquifers are relatively heterogeneous and heavily dominated by paleo-channel systems generated by surface run-off from large rainfall events through wadi systems during deposition of the alluvial sediments. Native groundwater quality varies from freshwater to saline.

Storing water in unconfined conditions tends to be more complex compared to storage in confined conditions due to a number of additional considerations that have to be evaluated when investigating the aquifer. Unconfined

storage systems are more vulnerable to surface contamination because of their hydrostatic setting. Generally abstraction of groundwater is dominated by drilling hundreds of shallow wells for agricultural purposes, which obviously influence the hydrodynamic behavior of the aquifer. Sediments are unconsolidated, challenging well construction techniques to meet high well efficiencies and to control sanding issues. Generally the groundwater velocity is larger in unconfined aquifers and must be regarded as one of the critical suitability constraints. Specific yield must be taken into account to evaluate the transient behavior of the cone of depression of the wells.

Injecting high volumes of water into an aquifer will result in a mound of water (Figure 1) at the injection well. An undesired effect that develops during injection is the mound that creates a radial temporary hydraulic gradient, which allows injected water to move away from the ASR well. The fact that water raises above the static water table results in re-wetting of the unsaturated zone. However, very limited hydraulic information can be gathered from standard investigative methods to obtain unsaturated zone properties (e.g. moisture content, relative permeabilities, capillary pressure and water saturation) to predict the movement of water through this zone.

Because of well interference the maximum drawdown must be evaluated with care in large well fields. Especially in a hydraulic setting with a relatively thin saturated aquifer thickness, it may not be feasible to sustain the high recovery rates desired for emergency cases for long durations.
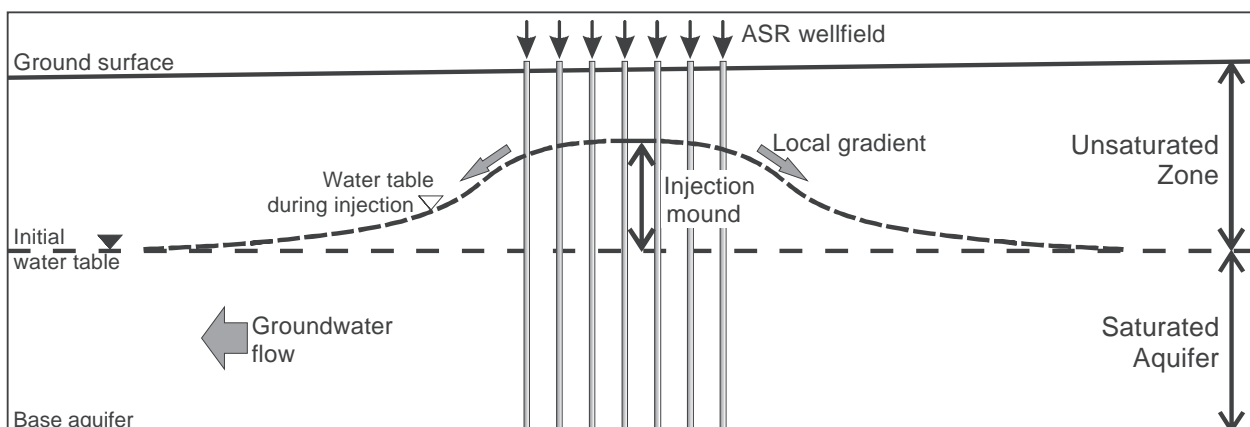
Figure 1. Cross-section showing a typical water table profile
of a surficial alluvial aquifer ASR wellfield during injection

ASR WELL FIELD OPTIMIZATION

The operational performance of the ASR well field will subsequently reveal the actual effective recovery efficiency of the ASR system, which is used to measure the success of the ASR system (Pearce, 2001). Careful design of the well field is needed to optimize the recovery efficiency, the total recovery rate, the area of influence and the area covered by the well field. Generally geologic and hydrogeologic heterogeneities are not known at the design stage of an ASR project, which will have a large impact on the optimization.

The following factors have a key influence on the success of the well field:
• Heterogeneities of the aquifer (hydrodynamic dispersion),
• Movement of the groundwater,
• Well performance,
• Well field layout.

Practical experience reveals that unexpectedly low ASR efficiency is mainly due to the lack of full understanding of

the aquifer characteristics. Macro-heterogeneities may be identified from further site investigations by means of drilling wells and/or resistivity surveys. Hydrodynamic dispersion is a particularly major unknown.

In unconfined aquifers the groundwater velocity is a major concern because the injected freshwater bubble may move away from the well field. In addition to the movement of the aquifer water, a groundwater velocity will be temporarily superimposed due to a freshwater mound during the injection period.

The well performance (efficiency of the well) is generally over-estimated during the optimization of an ASR well field using numerical simulations. In unconsolidated sand aquifers in the Middle East, clogging effects that build up over time during injection and recovery must be considered.

It has been demonstrated in the literature that the well field layout has a considerable effect on the recovery efficiency of an ASR system. Optimizing the ASR well field efficiency aims mainly to reduce the mixing that occurs between the injected water and the native groundwater in the aquifer. Mixing occurs generally at the edge of the interface between the freshwater bubble and the native water. Mixing can be reduced if the area of the interface is reduced. It is therefore desirable that the freshwater bubbles of each well in an ASR well field connect to create a single freshwater bubble at depth. If a reasonable connection of the freshwater bubbles of each ASR well is not guaranteed, saline native water may be trapped between the freshwater bubbles, having a negative impact on the recovery efficiency.

In order to design a large ASR well field, the arrangement of the wells must be optimized. The distance between ASR wells must be evaluated with great care for the following reasons:
• A large distance will reduce interference effects.
• A small distance will ensure efficient connection of the freshwater bubbles.

The objective of the optimization is to find a maximum connection of the bubbles with a minimum well interference.

RESULTS

Injection and recovery cycles are simulated for three ASR wells that are operating with a constant rate and equal duration simultaneously. The aquifer is unconfined, homogeneous and a hydraulic gradient exists to the northwest.

The salinity and pressure of the system was monitored at the center-point of the wells for a distance of 300 m and 150 m between the wells. The simulation results are displayed for the final cycle period.

At a distance of 300 m between the three wells the freshwater bubbles will not connect (Figure 2 left) at the end of the simulated duration. At the center-point between the three wells saline native water is trapped which will impact the recovery efficiency of the ASR system.

In order to optimize the recovery efficiency the wells should be moved closer to each other. Reducing the distance between the wells to 150 m will result in a combined single freshwater bubble (Figure 2 right).

The breakthrough curve of the aquifer salinity for a single injection period at the center-point of the wells clearly displays the improved mixing that has occurred for the 150 m case (Figure 3).

Varying the distance of wells in a well field will effect the head distribution due to well interference. The com-
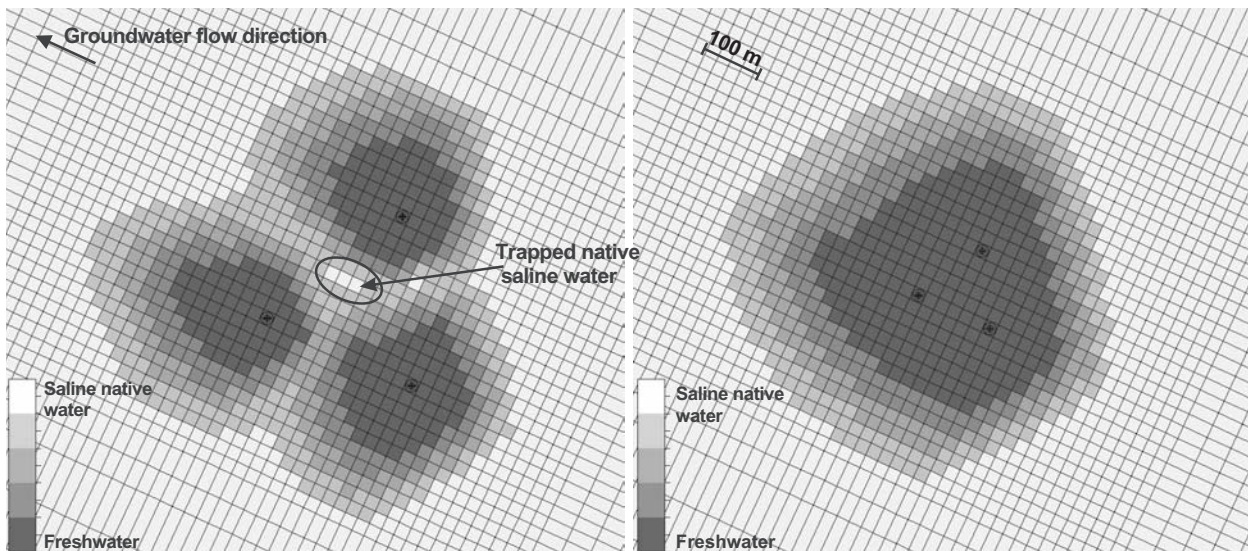
Figure 2. Development of freshwater bubble after injection into three ASR wells
spaced 300 m (left) and 150 m (right) apart
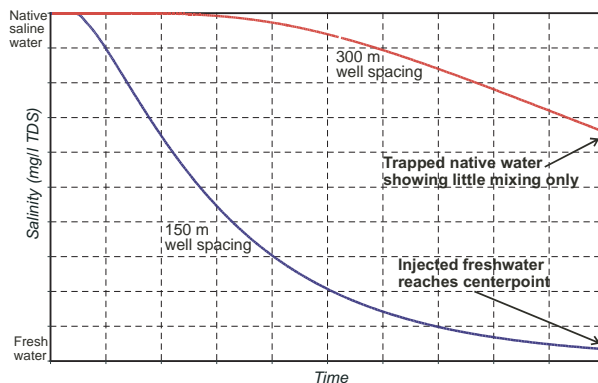
Figure 3. Comparison of breakthrough curves of the aquifer salinity at the center-point between the three ASR wells

parison of the total head at the center-point of the wells for the two simulated cases shows a 2 m increase for the 150 m case. During recovery similar results have been achieved. The total drawdown at the center-point will be larger for the 150 m case. Figure 4 shows the dynamic behavior of the water table for an injection and recovery period crossing the well field.

Generally the maximum drawdown is of special interest to prevent any pumps to run dry in the wells.
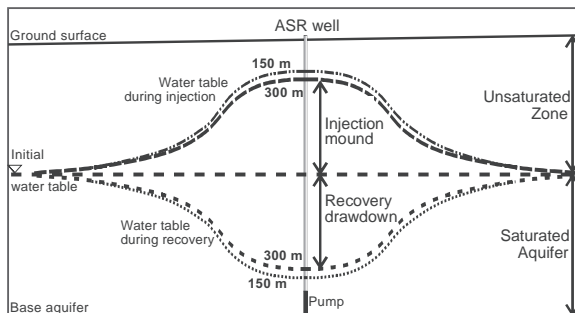
Figure 4. Schematic cross-section displaying dynamic water table for injection and recovery period
for 300 m and 150 m case

CONCLUSIONS

The recovery efficiency of an ASR well field can be optimized if mixing between freshwater and native water is minimized. This can be achieved by allowing the freshwater bubbles of each ASR well to connect to a single freshwater bubble at depth.

This can be achieved by finding the optimum spacing for each ASR well resulting in the maximum connection of the bubbles with a minimum well interference.

By optimizing the spacing of the ASR wells to create a single water bubble, the recovery efficiency can be substantially improved.

Special care must be taken to evaluate the interference between the wells that will effect the depth of the water table during injection and recovery. Well field optimization allows preventing undesirable storage effects like trapped native water and increasing drawdown causing pumps to run dry.

A new integrated aquifer characterization workflow was developed and linked with a numerical simulator that is capable of simulating complex hydrogeological conditions including variably saturated and density-dependent groundwater systems.

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Geochemical and microbial processes in the unsaturated zone at the Arrenæs artificial recharge trial plant

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Abstract

At the Arrenæs AR trial plant, Zealand, Denmark, investigations of the geochemical and microbial processes occurring in the unsaturated zone when recharging water from Lake Arresø have been performed. Focus was on ensuring a continuously stable quality of the abstracted AR water. The investigations revealed that the primary changes in the geochemical composition and the removal of bacteria below both an irrigated area and an infiltration basin occur in the uppermost part of the soil. The difference in the decalcification front between the irrigated and a non-irrigated area is app. 3 m. Simulations show that the calcite buffer capacity in the unsaturated zone is not depleted even in a long-term perspective, indicating that problems with aggressive CO_2 should not be expected. A removal capacity at the AR plant of up to 97-99% of the bacteria was observed; however, break through of coliform bacteria was seen twice in the abstracted AR water. It is estimated that only 17% of all break through with coliform bacteria in the abstraction wells is found.

Keywords

Artificial recharge, unsaturated zone, geochemical composition, bacterial removal, filter capacity, modelling.

INTRODUCTION

Since 1995, Copenhagen Energy has run a trial plant based on artificial recharge (AR) to determine the potential for use of this technology in Denmark to ensure the supply of drinking water to Copenhagen. (Copenhagen Water Supply and County of Frederiksborg, 1995; Copenhagen Energy, 2000). The results of running the trial plant have been positive and a large-scale production plant is therefore proposed at the same location (Brandt, 1998; Hartelius et al., 2001). The Arrenæs AR trial plant consists of four infiltration basins and two trough systems (1,000 m^2 each – in operation all year), and a grass-covered irrigation area (20,000 m^2 – in operation from April to November), see Figure 1, (Passow, 1996). The amount of runoff water from Lake Arresø used as recharge water is app. 100,000 m^3/year to the irrigation area and 300,000 m^3/year to the infiltration basins and trough systems, resulting in a production of app. 270,000 m^3/year, which is currently not used for drinking-water purposes. The unsaturated zone consists of glaciofluvial sand and is app. 25–26 m thick. Figure 1 shows a map of the area and also the water table. The hydraulic heads are affected by abstraction well I1 and an abstraction well located just north of trough system West.

With focus on ensuring a continuously stable quality of the abstracted AR water, the geochemical and microbial processes occurring in the unsaturated zone when recharging lake water were
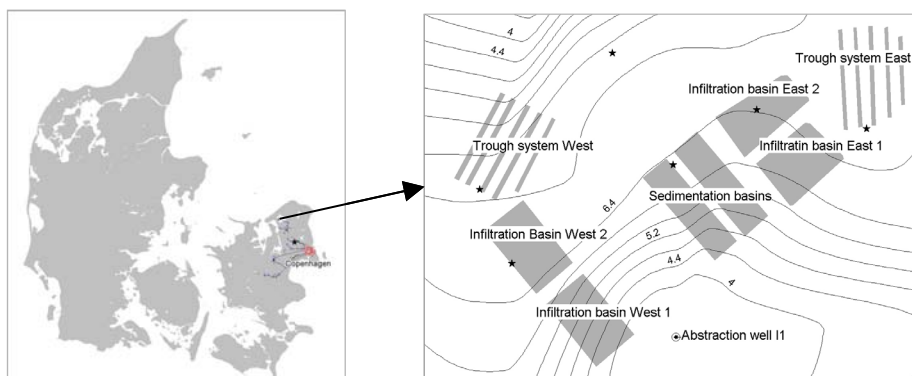
Figure 1. The Arrenæs AR trial plant, Zealand, Denmark

investigated. This includes investigation of the difference in geochemical composition and bacteria content in sediment samples from an irrigated area, a non-irrigated area (reference area), and an infiltration basin, and the risk of break through of pathogenic indicator organisms in an abstraction well. These investigations performed within the EU-project ARTDEMO constitute the background for establishing a management strategy including an early-warning system for an AR plant.

METHODS

The investigations include continuous monitoring (1994–2005) of the quality of the input water, soil water and abstracted AR water. Additionally, sediment samples from 0–30 m below surface (m.b.s.) were collected in October 2004 from an irrigated and a non-irrigated area using the solid flight auger drilling method. In March 2004, sediment samples were collected from an infiltration basin at depths of 0.01 (filterskin), 0.07 and 0.35 m.b.s. Geochemical analyses performed on the soil material included pH (NEN 6411), EC (NEN 6412), non-silicate bound calcium (ICP-MS analysis on HNO_3 extract), calcite (Ca-extraction using HCl), organic carbon and total N (pyrolysis of decalcified sample and subsequent gas detection by LEKO analyser) and grain size distribution (<2mm, by a FRITSCH Laser Particle Sizer A22). Chemical modelling (PHREEQC) was done using chemical parameters analysed in water samples according to the Danish Standards. Microbiological analyses of both water and sediment samples included colony-forming units (CFU) at 22°C and 37°C (DS/EN ISO 6222), coliform bacteria and thermotolerant coliform bacteria (DS 2255), sulphite-reducing clostridia (NMKL 56) and *Clostridium perfringens* (DS 2256). Enterococci were analysed in both water (ISO 7899/2 mod.MST98) and sediment samples (DS 2401). The total amount of bacteria in sediment samples from the irrigated and non-irrigated area was determined using the Acridine Orange Direct Count (AODC).

RESULTS

The geochemical results from sediment samples (0–30 m.b.s.) collected at the irrigated and non-irrigated area at the Arrenæs AR trial plant show that the difference in geochemical properties between the two areas is significant in only the upper 2 m of the unsaturated zone, see Figure 2. In the deeper parts from 2–26 m.b.s., the geochemical properties are similar for the two areas. The water table is located in app. 26 m.b.s., which is reflected in the geochemical properties.

Accumulation of organic carbon is occurring in the topsoil, see Figure 2. Due to bioturbation and degradation, the content of organic carbon is decreasing in the upper 2 m.b.s. The degradation of organic carbon is reflected in a decreased pH compared to the deeper layers, where pH is controlled by calcite equilibrium.
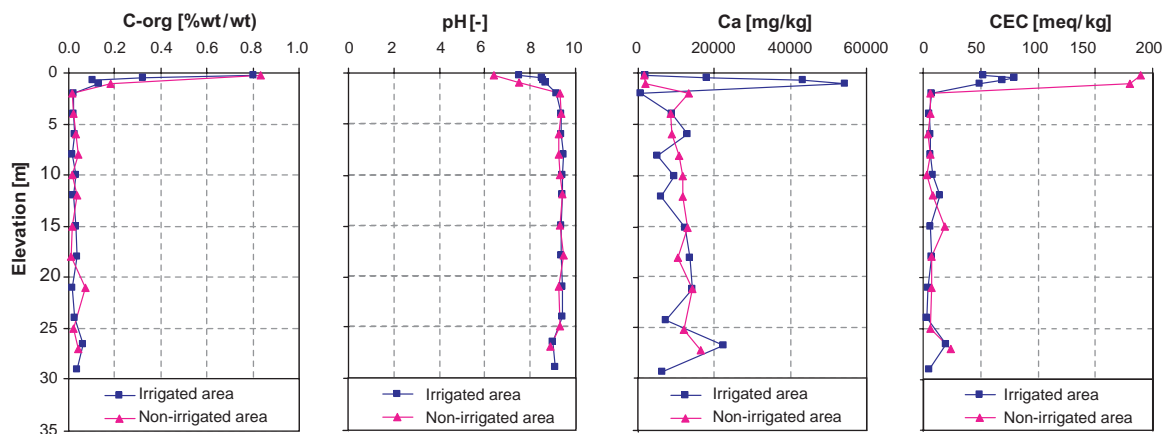
Figure 2. Organic carbon, pH, calcium, and cation exchange capacity (CEC)
at the irrigated and the non-irrigated area

Data show a significant increase in the content of calcium from 0–2 m.b.s. at the irrigated area. This calcium is primarily present as calcite. From the cation exchange capacity, Figure 3, it is estimated that ion exchange does not have a significant influence on the calcium content in the sediment. The accumulation of calcite in the upper 2 m is primarily a result of irrigation with lake water supersaturated with calcite, and evaporation. In addition, algae growth in the topsoil and loss of CO_2 gas may cause precipitation of calcite. Figure 2 indicates a decalcification front located app. 1–2 m.b.s. at the non-irrigated area and below the calcite rich soil horizon in app. 3–4 m.b.s. at the irrigated area. Whether this difference can be explained by artificial recharge needs further investigations of the location of the decalcification front, as natural geological heterogeneities also have an impact on the difference shown in Figure 2. Below the leaching front, geological variations may influence the carbonate equilibrium.

Similar investigations were performed in the upper 0.35 m of the soil below an infiltration basin. Figure 3 shows the content of calcite and organic carbon together with electrical conductivity. The high calcite content in the filter-skin (0.01 m.b.s.) may be explained by algae growth in the infiltration basin causing a decrease in CO_2 and thus increasing the calcite saturation index (Schuh, 1990). The organic carbon content in the sediment below the infiltration basin decreases from 0.095 to 0.031% in the upper 0.07 m of the soil, see Figure 3. The degradation of organic carbon is reflected in a concurrent decrease in calcite content and electrical conductivity.
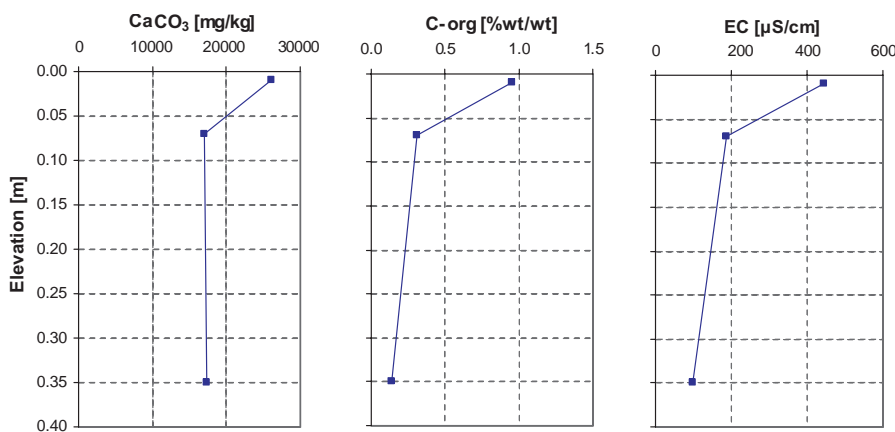
Figure 3. Calcite, organic carbon and electrical conductivity (EC) in the soil below an infiltration basin (n=2)

The geochemical processes in the unsaturated zone may also be affected by complicated flow patterns such as preferential flow, as revealed in studies at the Arrenæs AR trial plant by Brun and Broholm (2001).

The microbiological results from the sediment samples collected at the irrigated and non-irrigated area (0–30 m.b.s.) indicate that bacteria are primarily present in the upper 2 m, see Figure 4.
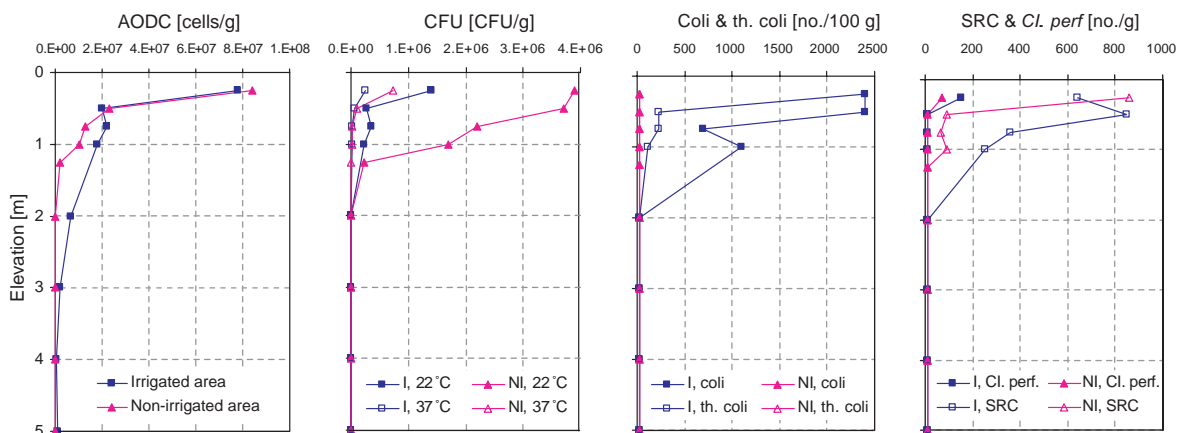
Figure 4. AODC, CFU, coliform bacteria, thermotolerant coliform bacteria, sulphite-reducing clostridia (SRC) and Clostridium perfringens *from 0–5 m.b.s. at the irrigated and non-irrigated area*

CFU at 22°C and 37°C detected using the yeast extract agar are higher in the upper 2 m at the non-irrigated than at the irrigated area. The total number of bacteria (AODC) is similar at the two locations, indicating the presence of bacteria cultures not accounted for using the yeast extract agar. As expected, the content of pathogenic indicator organisms is generally higher at the irrigated than at the non-irrigated area. Sulphite-reducing clostridia and *Clostridium perfringens* were present at 0–2 m.b.s. at both the irrigated and non-irrigated area, whereas coliform bacteria, thermotolerant coliform bacteria, and enterococci only were detected at the irrigated area from 0–2 m.b.s., Figure 4. As a total, the results show a reduction of up to 99% of the pathogenic indicator organisms in the upper 2 m.
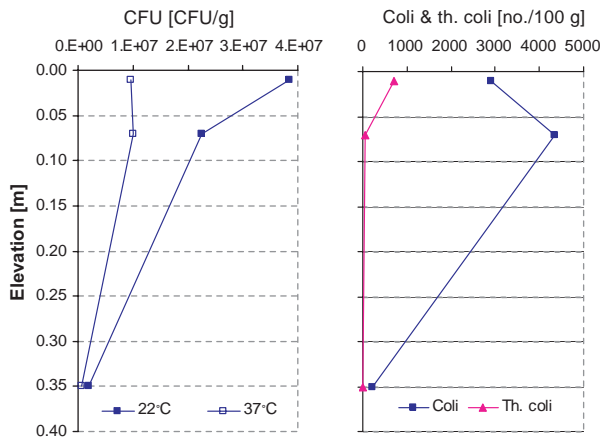
Figure 5. CFU, coliform bacteria, and thermotolerant coliform bacteria in the soil below an infiltration basin (n = 2)

The studies also included analysis of sediment samples from in an infiltration basin. The results show a large reduction of bacteria in the upper 0.35 m (see Figure 5), where the content of coliform bacteria and thermotolerant coliform bacteria is reduced by up to 97%. This is similar to findings at the Arrenæs AR trial plant by Jørgensen (2001).

The pathogenic removal capacity is similar for both the irrigated area and the infiltration basin. However, the filter efficiency is higher in the infiltration basin, due to the formation of a filterskin.

DISCUSSION

Depletion of the calcite buffer capacity in the unsaturated zone

In a long-term perspective, recharging water from Lake Arresø may cause depletion of the calcite buffer capacity in the unsaturated zone and thereby problems with the formation of aggressive CO_2. The analyses of sediment samples from the Arrenæs AR plant have shown that the soil has a large calcite buffer capacity. In addition, the input water is supersaturated with calcite. However, Table 1 shows that there is a significant difference in the organic carbon content between the input water and the abstracted AR water, indicating degradation in the unsaturated zone, which induces calcite dissolution. This was also evident from the sediment samples as discussed previously. The abstracted AR water has fairly constant pH, calcium and bicarbonate levels, indicating that buffering processes occur in the unsaturated zone.

The risk of depletion of the calcite buffer capacity in the unsaturated zone at the irrigated area was investigated using the programme PHREEQC and data are given in Table 1. In the calculations, equilibrium with calcite, and oxidation of organic matter by oxygen and nitrate using first order reaction kinetics is assumed. Ion exchange is not included in the calculations.

The composition of the soil water in the unsaturated zone at the irrigated area after 10 years of operation was simulated in PHREEQC and shows results similar to the field data from 2–30 m.b.s. Simulation of the processes occurring in the upper 2 m has, however, not been possible using the assumptions given above. Additionally, alternating calcite content over depth due to geological variations cannot be reflected in the simulation, as the unsaturated zone is assumed uniform. Simulations show that the decalcification front has moved app. 2 m downwards after 60 years of operation compared to the present location of the front in app. 3–4 m. Problems connected with production of aggressive CO_2 due to a depletion of the calcite buffer capacity are therefore not expected in a time perspective of 60 years.

Investigation of the potential risk for pathogenic break through

Break through of pathogenic indicator organisms in an abstraction well have been registered twice – 35 coliform bacteria/100mL in August 2002 and 17 coliform bacteria/100mL in August 2004. This break through of pathogenic indicator organisms might have been caused by changed operational procedures in 2001, where the recharge in the beginning of the operation periods of the infiltration basins was increased. This causes a lower removal of pathogens, as the clogging layer is not fully developed.

Table 1. Selected water quality parameters in the input water, soil water and abstracted AR water

Parameter [mg/L]	Arresoe – input water Average 1995–2005	Tensiometer well T9 – 1m (soil water) 19/04/1995	Tensiometer well T9 – 5m (soil water) 19/04/1995	Tensiometer well T9 – 15m (soil water) 19/04/1995	Abstracted AR water Average 2001–2005
pH at 12°C	8.9	9.1	7.6	7.5	7.5
O_2	9.3	–	–	–	4.7
Ca	46	26	38	55	87
Cl	65	29	6	35	59
HCO_3	132	140	184	132	236
NO_3	0.269	3.7	10	80	4.7
NO_2	0.03	0.17	0.42	0.19	<0.01
NH_4	0.1	0.48	0.25	0.1	0.03
NVOC	15.2	–	–	–	3.9

Unregistered contamination of the abstraction wells may, however, have occurred. The probability of a contamination of an abstraction well being found is estimated as discussed by Boe-Hansen et al. (2003). It is assumed that a contamination has a duration of N days with probabilities $P(N)$, $N = 1,2,...$ and as an example, it is assumed, that N varies between 1 and 13 days with equal probabilities $P(N) = 1/13$ (mean(N) = 7). The annual rate of contaminations is called λ_y, and the sampling frequency is S_y (currently 12 samples/year). To repeat a positive test d days (in this study $d = 2$) are needed, and the detection limit is 1 bacterium in a 100 mL sample. The mean number of bacteria per 100 mL in a contamination is called μ_c. Under these assumptions the proportion of contaminations detected and confirmed can be estimated:

$$Q\left(S_y, \mu_c\right) \cong \left(1 - \exp\left(-\mu_c\right)\right)^2 \cdot S_y \sum_{N>d} P(N) \cdot (N - d)/365$$

where $\sum_{N>2} P(N) \cdot (N - 2)/365 = 5.076/365 = 0.0139$ in the given example. If further $\mu_c > 3$, $\left(1 - \exp\left(-\mu_c\right)\right)^2 > 0.90 \cong 1$, the variation of μ_c between contaminations is thus of little consequence, except for very low-level contaminations. The contaminations experienced to date have shown $\mu_c \gg 3$ in both cases. The annual rate of confirmed findings is $\lambda_y \times Q(S_y, \mu_c)$. The formulas apply for moderate S_y and moderate length of contamination, that is when the time between sampling is long compared to the mean duration of contamination ($365/S_y \gg$ mean(N)) and the occurrence rate is low. Using the above $P(N)$ and d the detection proportion, $Q(S_y, \mu_c)$ is computed, see Table 2.

Table 2. The proportion of detected and confirmed contaminations using P(N) = 1/13 and d = 2

Contamination	Samples per year = S_y		
	12	18	24
$\mu_c = 1.0$	0.07	0.10	0.13
$\mu_c = 2.0$	0.12	0.19	0.25
$\mu_c = 3.0$	0.15	0.23	0.30
$\mu_c \gg 3.0$	0.17	0.25	0.33

Since the detection rate is proportional to the occurrence rate λ_y, the table can be applied to estimate λ_y. If, for example, 2 contaminations are found in a year using the present $S_y = 12$ and $\mu_c \gg 3$, the contamination rate may be as high as $\lambda_y = 2/0.17 \cong 12$ per year or more. It is therefore important to increase the sample collection frequency to detect a possible break through. In this context, online monitoring equipment for detection of pathogenic indicator organisms could be used as a

management tool in the operation of the AR plant, and contamination of the abstraction wells could thereby be limited. Furthermore, operational changes must be made to avoid pathogens in the abstracted AR water.

CONCLUSIONS

The investigations in this study show that the primary changes in geochemical properties and removal of bacteria at the Arrenæs AR trial plant occur in the uppermost part of the soil. Interpretation of the geochemical data shows that the difference in the decalcification front between the irrigated and non-irrigated area is app. 3 m. Simulations show that the calcite buffer capacity in the unsaturated zone is not depleted even in a long-term perspective, indicating that problems with aggressive CO_2 should not be expected. A removal capacity of up to 97–99% of the bacteria was observed at both the irrigated area and in the infiltration basin. The filter efficiency is the highest in the infiltration basin, due to the formation of a filterskin. However, break through of coliform bacteria in an abstraction well was detected twice. Estimations show that only 17% of all break through with coliform bacteria in the abstraction wells is found. This shows that online monitoring equipment for detection of pathogenic indicator organisms is an essential tool to ensure a sustainable operation of an AR plant. To further examine the geochemical and microbiological processes occurring in the uppermost part of the soil at the recharge areas, detailed investigations need to be performed from 0–1.5 m.b.s.

ACKNOWLEDGEMENT

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Physical and chemical interactions in an ASR cavity well in a brackish aquifer

R.S. Malik, D.K. Yadav, B.S. Jhorar, R.K. Jhorar and T. Streck

Abstract

Rain water (22 m^3) and bleaching powder treated canal water (173 m^3) were gravity injected sequentially in a cavity type aquifer storage recovery ASR well having highly brackish native water. Injection rate did not decrease with time indicating no serious clogging during ASR operation. Residence time of 9 days was given in the aquifer. The recovered water was analyzed for various physical and chemical properties as a function of recovered volume. The recovery efficiency at a targeted quality in terms of electrical conductivity EC = 200 mS/m was 24% for instantaneous recovery and 43% for integrated recovery. Potassium was released from aquifer minerals; bicarbonates were produced from native water, and borates were consumed from injected water; and heat was consumed from that of injected water. Up to target water quality of 200 mS/m of recovery water, 7 kg of potassium and 1.2 kg of nitrate were mined out from one-hectare irrigation of 0.6 m depths. The ASR technique may safely be applied to improve the quality of ground water for irrigation and to meet the partial crop requirement for potassium and nitrate.

Keywords

Aquifer Storage Recovery, ASR, cavity well, brackish aquifers, aquifer storage volume.

INTRODUCTION

The improvement in quality during Aquifer Storage and Recovery (ASR) process depends upon the nature and intensity of site-specific physical and chemical interactions between the injected and native ground water. The nature of these interactions has been identified (Faust and Vecchioli, 1974; Berner, 1978; Pyne, 1995; Martin and Dillon 2002; Vanderzalm et al., 2002) at some ASR sites. Most ASR wells are cavity type in north India and were not found to clog when they were used to inject fresh water even with high, 900 mg/l, sediment load (Anonymous, 1993 and Malik et al., 2002). Cavity wells are shallow wells installed in aquifers (15 to 100 m deep) where an empty space is formed below the impermeable layer called a cavity; thus, the well is named a cavity well (Malik et al., 2000).

The paper quantifies recovery efficiency and physical and chemical interactions in a cavity type ASR well in brackish aquifer.

MATERIALS AND METHODS

ASR facilities

Rain water (2.223 m^3) and bleaching powder chlorinated canal water (17.3 m^3) were sequentially gravity injected employing siphon system in to an ASR well installed at Regional Research Station, Balsamand (Fig. 1).

The ASR water (21.425 m^3) was recovered after allowing residence time of 9 days. Water samples as a function of recovery time and that of injected and native water were analyzed for electrical conductivity, pH, temperature, organic carbon (O.C), cations (Na$^+$, K$^+$, Ca^{2+}, Mg^{2+}, NH$_4^+$, Zn^{2+}), and anions (CO$_3^{2-}$, HCO$_3^-$, Cl$^-$, SO$_4^{2-}$, BO$_3^{3-}$) by standard methods.
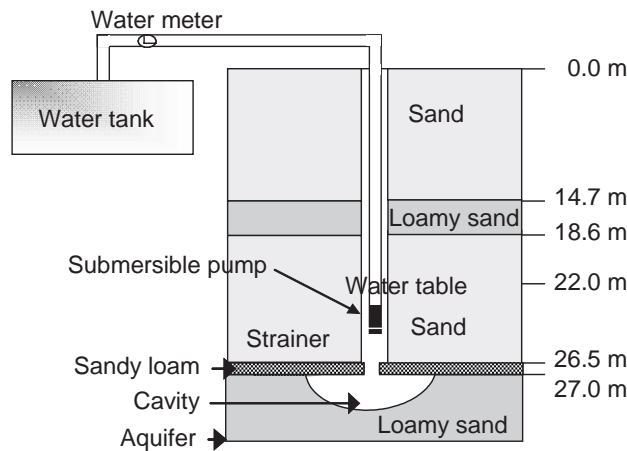
Figure 1. Schematic diagram of a cavity well at Balsamand ASR site (not on scale)

INTERPRETATION OF RECOVERY DATA

Recovery efficiency RE

Recovery percentage I may be defined as the percentage recovered water volume V$_r$ at any recovery time tr$_2$ to the total volume injected V$_i$ as:

$$I = 100 \left[\frac{\int_{tr1}^{tr2} qr(t)dt}{\int_{ti1}^{ti2} qi(t)dt} \right] = 100 \left[\frac{V_r}{V_i} \right]$$

Where t_{i1} = time that injection starts, t_{i2} = time that injection ends, t_{r1} = time that recovery starts, t_{r2} = time that recovery ends, q$_r$ (t) = recovery rate as a function of time, q_i (t) = injection rate as a function of time, V_r = volume recovered between recovery time t_{r1} to t_{r2} and V$_i$ = volume injected between injection time t_{i1} to t_{i2}.

The instantaneous recovery efficiency *IRE* represents the recovery percentage *I* at target time t_r* to meet the target instantaneous EC_r (t) criteria for the recovered water. The integrated recovery efficiency *CRE* would therefore, represents the recovery percentage (*I*) at target time t_r** to meet the target integrated EC_{rw} (t) for the recovered water. It may be expressed mathematically as:

$$RE = 100 \left[\frac{\int_{tr1}^{trt} qr(t)dt}{\int_{ti1}^{ti2} qi(t)dt} \right] = 100 \left[\frac{V_r^{**}}{V_i} \right]$$

where V_r** = total recovered volume at target time *trt*.

In this paper desired water quality (electrical conductivity, EC) of the recovered water for irrigation purpose was taken as 200 mS m^{-1}.

Extent of mixing and physical and chemical reactions

Mixing percentage M (t) of native water with injected water, as a function of injected water recovery percentage (I) defined in line with [Ragone and Vecchioli, 1975; Pavelic et al., 2002] was utilized to quantify the RE and the extent of geo- physical and chemical reactions. The instantaneous mixing percentage M_1 and the integrated mixing percentage M_2 for any of the quality parameter was estimated as:

$$M_1(t) = \frac{C_r(t) - C_i}{C_n - C_i} \times 100 \quad \text{and} \quad M_2(t) = \frac{C_{rw}(t) - C_i}{C_n - C_i} \times 100$$

Where M_1 (t) and M_2 (t) are the percentage native water in the instantaneous and cumulative recovered water volume in time t; C_r (t) and C_{rw} (t) are instantaneous and integrated (weighted average) concentrations of given parameters as function of time t in the instantaneous recovered water sample ΔV_r and in the cumulative recovered volume of recovered water V_r; and C_i, and C_n are concentrations of given parameters in injected and native water. The C_{rw} (t) can be estimated as:

$$C_{rw}(t) = \left[\frac{\int_{tr1}^{tr2} C_r(t)qr(t)dt}{\int_{ti1}^{ti2} qr(t)dt} \right] = \frac{\sum C_r(t)\Delta V_r}{\sum \Delta V_r}$$

Where ΔV_r is the instantaneous recovered water volume in any given time interval. The M_1, M_2, C_{rw} and related parameters were also calculated for EC and temperature as the former represents the salt concentration and the latter measures the heat concentration.

Instantaneous mixing (M vs. I) curve is useful to estimate recovery efficiency when the recovered water is put to direct use such as for drinking or irrigation. The integrated mixing curve is useful to estimate recovery efficiency when the recovered water may be stored in the storage tanks just before use.

Chloride was taken as an indicator for quantifying the mixing process between native and injected water because chloride is supposed to behave conservatively in the aquifer. The integrated mixing percentage (M_2) explained in previous section can also be used to quantify the physical and chemical processes. Let M_2 for chloride at 100% recovery is $M_2*(Cl^-)$ and let the M_2 value for any other quality parameter X at 100% recovery is $M_2*(X)$. Any water quality parameter that show the $M_2*(X)$ value close to $M_2*(Cl^-)$ value (in this study a critical limit is within 10% of $M_2*(Cl^-)$ value) is considered to have gone through the process of mixing only (no physical nor chemical reaction). However, $M_2*(X)$ values beyond the range $M_2*(Cl^-) \pm 0.1* M_2*(Cl^-)$ means that some other interactions have taken place in addition to simple mixing as explained in Table 1 .

Table 1. Possibility of different physicochemical processes between native groundwater and injected water (see text for the definition of different symbols)

Mixing percentage	Parameter concentration	Physicochemical process
$0.9[M_2*(\,Cl^-)] \leq M_2*(X) \leq 1.1[M_2*(Cl^-)]$	–	Simple mixing, no physicochemical reaction
M2*(X) > 1.10 [M2*(Cl-)]	Cn(X) > Ci(X)	Production/release/dissolution
M2*(X) > 1.10 [M2*(Cl-)]	Cn(X) < Ci(X)	Consumption/precipitation/dissipation
M2*(X) < 0.9 [M2*(Cl-)]	Cn(X) > Ci(X)	Consumption/precipitation/settling
M2*(X) < 0.9 [M2*(Cl-)]	Cn(X) < Ci(X)	Production/release/dissipation

RESULTS AND DISCUSSION

Injection and recovery rates were equal and remained fairly constant with time at an average value of 0.76 L/sec indicating no clogging. Mixing M increased with recovery I for all parameters and as an example it is given for EC in Table 2. The mixing percentage values of 7.1 means that 7.1 per cent of the native water is mixed with 92.9 per cent of injected water to reach the target water quality of 200 mS m^{-1}. The observed recovery efficiency (Fig. 2) at 7.1 per cent mixing (target water quality 200 mS m^{-1}) was 24 per cent for instantaneous recovery and 43 per cent for integrated recovery.

Table 2. Concentration of groundwater (C_n), injected water (C_i); integrated at 100% recovery and (C_{rw}) integrated at the target recovery (C_{rw}**) and mixing % at 100% recovery M_2*.*

Parameter	C_n	C_i	C_{rw}* (M_2*)	C_{rw}**
EC (mSm^{-1})	2470	25.45	677.8 (26.7)	220
pH	7.4	7.9	–	–
Cl (mmol L^{-1})	277.0	1.5	64.6 (22.9)	15.5
SO$_4$$^{2-}$ (mmol L^{-11})	0.63	0.008	0.16 (24.0)	0.04
HCO$_3$$^-$ (mmol L^{-1})	5.00	1.75	3.70 (60.0)	3.2
BO$_3$$^-$ (mmol L^{-1})	0.015	0.045	0.027 (60.0)	0.033
NO$_3$$^-$(mmol L^{-1})	2.00	0.04	0.43 (19.9)	0.14
Na$^+$ (mmol L^{-1})	150.00	0.35	34.7 (22.9)	7.7
K$^+$(mmol L^{-1})	1.30	0.10	0.61 (42.5)	0.30
Ca^{+2} (mmol L^{-1})	17.4	0.63	4.65 (24.0)	2.0
Mg^{+2} (mmol L^{-1})	62.9	0.98	15.4 (23.3)	4.4
Organic carbon (mg/L)	200.0	Nil	51.9 (25.9)	1.0
Dissolved Oxygen (mg/L)	0.200	0.350	0.31 (26.7)	–
Temperature (°C)	31.0	33.0	31.6 (70.0)	32.3
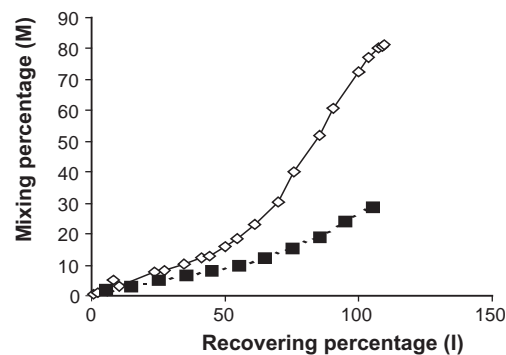
Figure 2. Recovery percentage I as a function of instantaneous mixing percentage M_1 (solid line) and integrated mixing percentage M_2 (dotted line) for Electrical Conductivity (EC)

Physical and chemical interactions

Bicarbonate formation

Bicarbonate concentration in the native groundwater exceeded that in the injection water. The (M_2*) for HCO_3^- was 60%, this is more than 22.9 ± 2.3 implying that additional bicarbonate was produced during the mixing of injected water with the native groundwater. The bicarbonate formation might be due to dissociation of carbonic acid (H_2CO_3), when injected water of higher pH of 7.9 reacted with native groundwater of lower pH of 7.4, into bicarbonate (HCO_3^-) and a proton (H^+) as per Equation 7.

Borate consumption

Injected water contained more borate than the native groundwater and its (M_2*) was higher than that of chloride implying dissipation/consumption of borate during the mixing of injected water with the native groundwater. The reduction of borate concentration in aquifer can be due to adsorption.

Potassium release

The (M_2*) value was also much higher than that for chloride and the concentration of potassium in the native water was much higher than the injected water. It implies that additional potassium was produced during the mixing of injected water with the native groundwater. It is likely that potassium was released from the potassium bearing clay minerals from its adsorbed/non-exchangeable state to the solution form due to freshening of the brackish ground water system.

Heat transport

The heat (M_2*) value was higher than that for chloride and the temperature of the native water (31ºC) was lower than that of injected water (33ºC). It implies that heat was transported radially away from ASR well in addition to simple mixing of injected and native water. This could be due to 1) heat conduction to the layers below and above the target aquifer, 2) heat exchange between water and grains in the aquifer during injection and recovery.

The M_2* values for sodium, calcium and magnesium, sulphate, nitrate, organic carbon and dissolved oxygen ions were with in M_2*(Cl^-) ± 0.1* M_2*(Cl^-) range (Table 2). It implies that these cations have undergone the process of mixing only and there were no additional sink and source processes occurring in the aquifer. The origin of native water appears to be marine, rich in organic compounds and salinity. High organic content of native water (200 mgL^{-1}, Table 2) supports the above observation.

Mining of fertilizer elements nitrogen N and potassium K

The native groundwater had very high concentrations (2 and 1.3 mole L^{-1}, Table 2) of two essential plant nutrients i.e. N and K. It means a mining of 1.2 kg of N and 7.0 kg of K in one irrigation of 0.06 m depth of such recovered water to one hectare.

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Windhoek, Namibia: Designing a large-scale borehole injection scheme in a fractured aquifer

E.C. Murray

Abstract

The city of Windhoek is currently upgrading their borehole injection scheme in order to accommodate a planned maximum injection capacity of ~8 Mm^3/a. The quartzitic aquifer is a highly complex fractured system consisting of preferential flow paths and poorly linked 'compartments'. This heterogeneous system is the result of impermeable amphibolite and schist layers and various episodes of faulting and thrusting. The design of the injection scheme required a sound conceptual flow model and a thorough geological analysis. The geological study took the tectonic history of the area into account as well as the local deformation history. Several areas were identified where the artificial recharge scheme could be expanded, and where borehole siting and drilling should take place.

Keywords

Borehole injection, fractured aquifer, domestic water use, scheme design.

INTRODUCTION

Windhoek, the capital city of Namibia is located in the centre of the country with the Namib Desert to the west and Kalahari Desert to the east. The only perennial rivers are about 700 km to the north and south, and are shared with Angola and South Africa. More that 75% of the city's water requirements, which is about 20 Mm^3/a, comes from surface water; the remainder comes from reclaimed water and groundwater from the Windhoek aquifer. The reliability of the surface water/dam sources are vulnerable due to erratic runoff, and this situation has created chronic water shortages over the past decade. Artificial recharge by means of borehole injection is considered the most favourable (and cost effective) option for providing water security to the city.

Windhoek is currently in the second phase of implementing the artificial recharge scheme. The first phase saw the equipping of five injection boreholes with a maximum capacity of 2.8 Mm^3/a. The second phase aims to add another four injection (or ASR) boreholes to the scheme, and the third phase another six boreholes. The target injection capacity of each borehole is 0.4 – 0.8 Mm^3/a. In order to identify optimum locations for the new injection boreholes, a detailed geological and hydrogeological analysis was undertaken. This involved developing a thorough conceptual flow model and detailed fracture trace mapping. This paper presents the results of the fracture trace mapping studies and the issues that were considered in expanding the scheme.

THE HYDROGEOLOGY OF THE WINDHOEK AQUIFER

This introduction to the Windhoek aquifer has been adapted from Murray (2002).

Geology of the Windhoek aquifer

The geology is extremely complex as a result of the geological processes associated with intracontinental rifting and continental convergence. Formations within the study area have been overturned in the process of orogenesis, and they have been subjected to a number of episodes of faulting including thrusting and rifting. The Windhoek well-field is located in interbedded quartzites and schists of about 500 Ma which lie north of the Auas Mountain Range. These east-west striking metasediments dip ~25° to the north and are overlain by sandy colluvium in the Windhoek Basin. The Windhoek aquifer consists of relatively pure quartzites from the Auas Formation, and impure, micaceous quartzites from the Kuiseb Formation. A simplified geological map is provided in Figure 1.
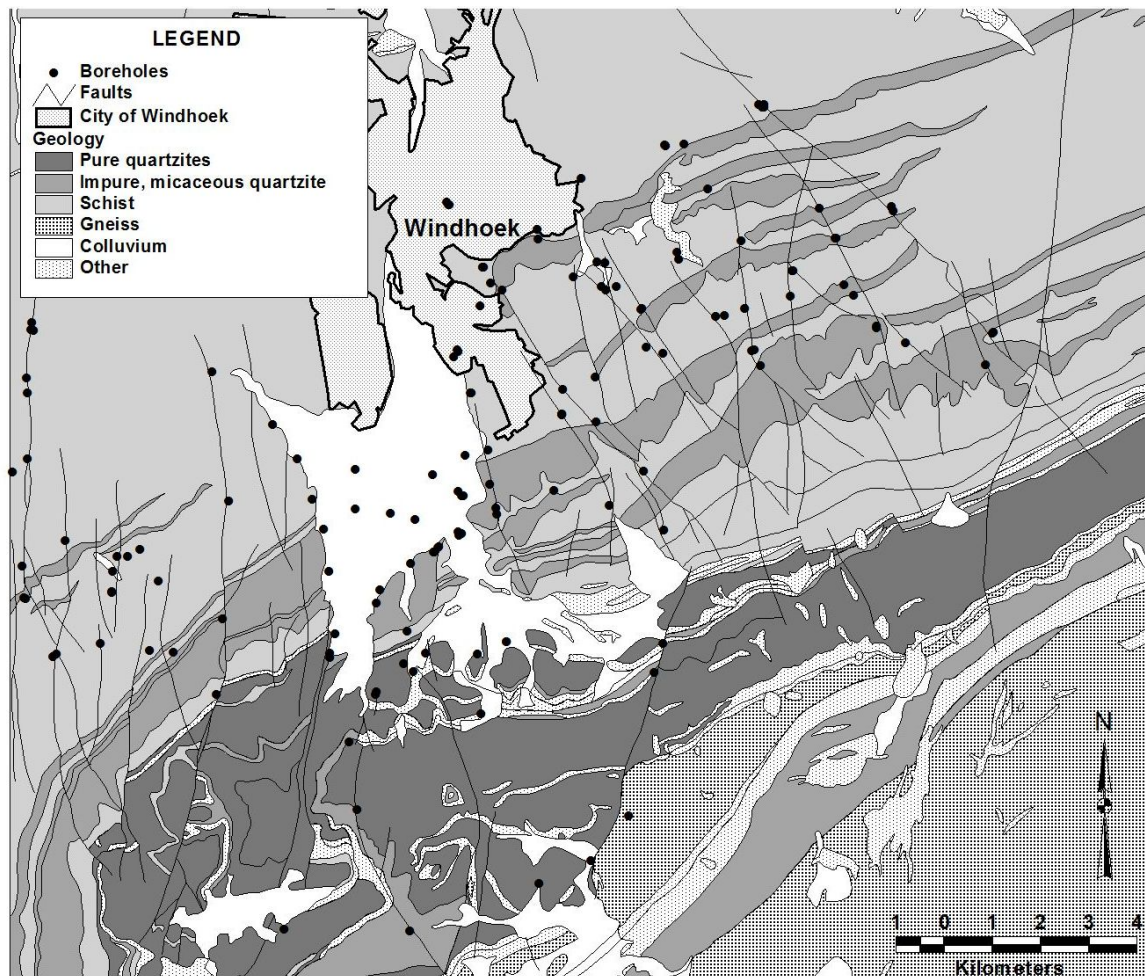
Figure 1. Simplified geological map of the Windhoek area

Groundwater flow

The flow dynamics are controlled by both lithology and structure. The quartzites, being brittle, are highly fractured as a result of folding and faulting and have developed secondary porosity and permeability. The schists on the other hand are ductile and do not have well developed secondary permeability. Both the schists and the quartzites appear to have no primary porosity.

The dominant groundwater flow direction in the Windhoek Basin is from the Auas Mountains in the south (where natural recharge is highest) to the city in the north, and then towards the Aretaragas River in the north-west. This

flow is dominated by the numerous faults and fracture zones that transect the area. Whilst they act as conduits for flow, they have also caused (in places) the low permeability schists to lie adjacent to the more permeable quartzites, thereby compartmentalising the aquifer. Thus the flow regime is complicated both by faulting and the interbeded nature of high- and low-permeability formations.

Aquifer characteristics and parameters

In the pure quartzites, groundwater flow is along highly permeable fractures associated with brittle quartzites and a single porosity system. The fractures seem to be linked to a sizeable groundwater reservoir which itself is likely to consist of a vast network of fractures. The transmissivity of the fracture zones range up to about 10,000 m^2/day and the transmissivity of the formation as a whole is in the order of 70 m^2/day. The storativity ranges between about 0.006 – 0.01. In the micaceous quartzites, the fracture transmissivity is on average about 400 m^2/day, and the formation transmissivity varies significantly depending on how impure (micaceous) the quarzites are. The range is from about 15 – 70 m^2/day. The storativity ranges between about 0.004 – 0.008.

Geochemistry

The groundwater from both the pure and impure quartzites are reasonably well clustered around the calcium-carbonate part of the Piper's rhombus. The electrical conductivity (EC) of the pure quartzites ranges from about 20 – 60 mS/m, and 50 – 80 mS/m for the impure quartzites. The ^{14}C values for the pure quartzites are generally between 40 – 70 pmc, and 20 – 40 pmc for the impure quartzites (which are located down-hydraulic-gradient of the pure quartzites).

THE IMPLEMENTATION PHASES

Phase 1 of the artificial recharge project focussed primarily on recharging the impure quarzites. Most of the well-fields are located in these micaceous quartzites, and the aim of recharging these areas is to rapidly replenish the dewatered areas created by large-scale abstraction. Four injection boreholes are located in this area. The areas associated with the different Phases are shown in Figure 2.
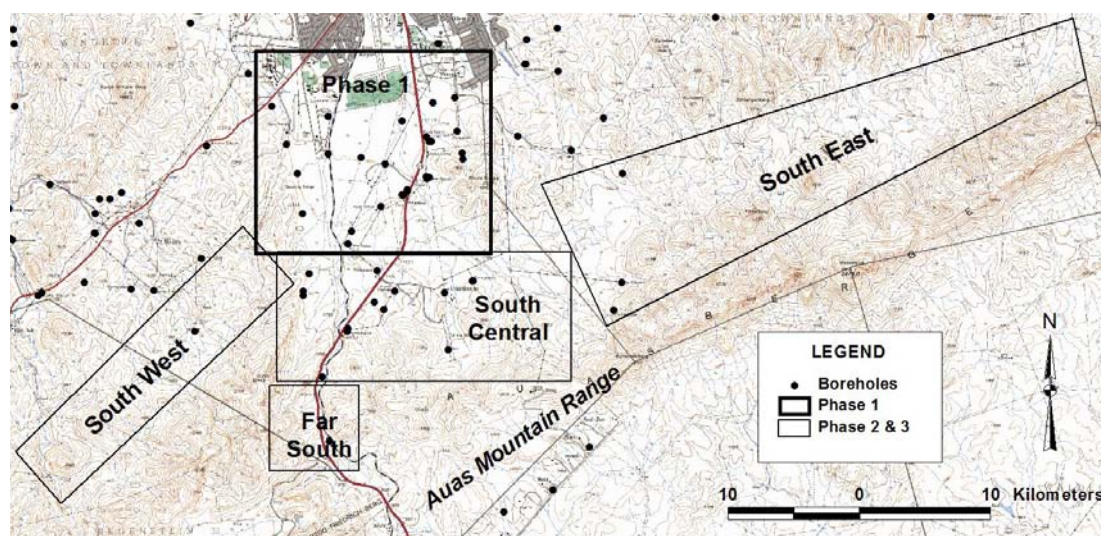
Figure 2. Location of the Phase areas

The intention of Phases 2 and 3 are to replenish the more permeable pure quartzites. The challenge lay in identifying prime areas for injection and recovery. In order to do this an extensive remote sensing study was undertaken

which included satellite and air photo interpretation, and airborne (helicopter) geophysics that was carried out by the Bundesanstalt für Geowissenschaften und Rohstoffe (BGR). This information was then added to and compared with previous fault and fracture-zone mapping exercises undertaken by Gui (1967), Halbich (1977) and the Geological Survey of Namibia (1988), and the prime target areas were identified for field mapping and field geophysics.

THE CHALLENGE IN DESIGNING THE BOREHOLE INJECTION SCHEME

The challenges in designing the expansion of the scheme in this Formation can be grouped into:

A. Hydrogeological issues:
   • Assessing the potential for large-scale abstraction, and
   • Identifying prime target areas for both injection and abstraction.
B. Technological issues:
   • Establishing the optimum drilling depth and borehole design, and
   • Establishing a drilling approach that can successfully complete large diameter boreholes to the desired depth.

The remainder of this paper focuses on describing how the prime target areas for injection and abstraction were identified.

IDENTIFYING PRIME TARGET AREAS FOR BOTH INJECTION AND ABSTRACTION

Criteria for target identification

The broad criteria that governed the identification of potential artificial recharge and deep groundwater abstraction sites were:

• New sites must be close to the Auas Formation (the 'pure' quartzites). Water injected or abstracted from new boreholes should access or replenish the higher permeability and storage of the Auas Formation.
• The boreholes need not necessarily be started in the Auas Formation, but they should preferably intersect the Auas Formation at depth.
• The new boreholes should be located on major faults that transect the Auas Formation.
• The orientation of the faults should be roughly between NE and NW, as these reflect the tensional orientations during the time of deformation.
• The fault pattern in the area should preferably show a high density of interconnected faults.
• The sites should be within a reasonable distance of existing infrastructure and be reasonably accessible.
• The sites should preferably lie on land owned by the City of Windhoek.

In relation to the last two points, the southern and eastern areas are considered to be preferable areas for wellfield expansion, so long as the hydrogeological conditions prove favourable (these are the South Central and South East Areas in Figure 2).

Remote sensing study

In 2002, Murray identified sites for large scale injection. These were located mostly in the eastern parts of the Auas Formation, and were based on the location of mapped faults (Geological Survey, 1988), on his interpretation of the

Landsat 1998 satellite image and on the position of existing infrastructure. In 2000, Murray requested (though the Department of Water Affairs, Namibia) that the BGR, who were engaging in remote sensing studies in northern Namibia, undertake an airborne geophysical survey of the Windhoek aquifer. This was kindly agreed to and the survey included the southern outcropping areas of the Auas Formation in the eastern and central parts of aquifer. The BGR data confirmed the areas previously identified, and also indicated that new or expanded areas need to be considered for further exploration.

As part of developing the artificial recharge and deep drilling implementation strategy (Windhoek Aquifer JV Consultants, 2005), the BGR data was interpreted by a Namibian geophysicist, Dr B Corner. Three pertinent observations were made:

- There are numerous faults in the eastern area with lengths of several kilometres;
- The fault interconnectivity in the eastern areas is high;
- The fault density in the eastern areas is high (although this does not confirm storage potential, it nevertheless suggests that fracture storage could be high in the eastern areas).

The interpretation of the BGR data by Dr Corner was compared with previously mapped faults and lineaments. Four published maps of the geology (including faults) of the Windhoek area are of interest. They are: Gevers (1934), Gui (1967), Hälbich (1977), Geological Survey (1998). These maps are considered to be the most accurate reflection of ground conditions as they were not only based on remote sensing techniques, but also field mapping. After overlaying the interpreted BGR data with the published geological maps, it was evident that there is a reasonable correlation between the two types of data sets. The value of the airborne geophysics was that additional faults could be delineated from this data – including those covered by colluvium in the Windhoek Basin.

Delineating prime areas for large-scale injection and abstraction

In order to derive a preliminary location of the drilling target areas, all remote sensing data was overlaid and briefly assessed. This resulted in delineating Broad Target Areas (Phase 2 and 3 areas in Figure 2) and Exploration Target Areas for detailed exploration (Figure 3).
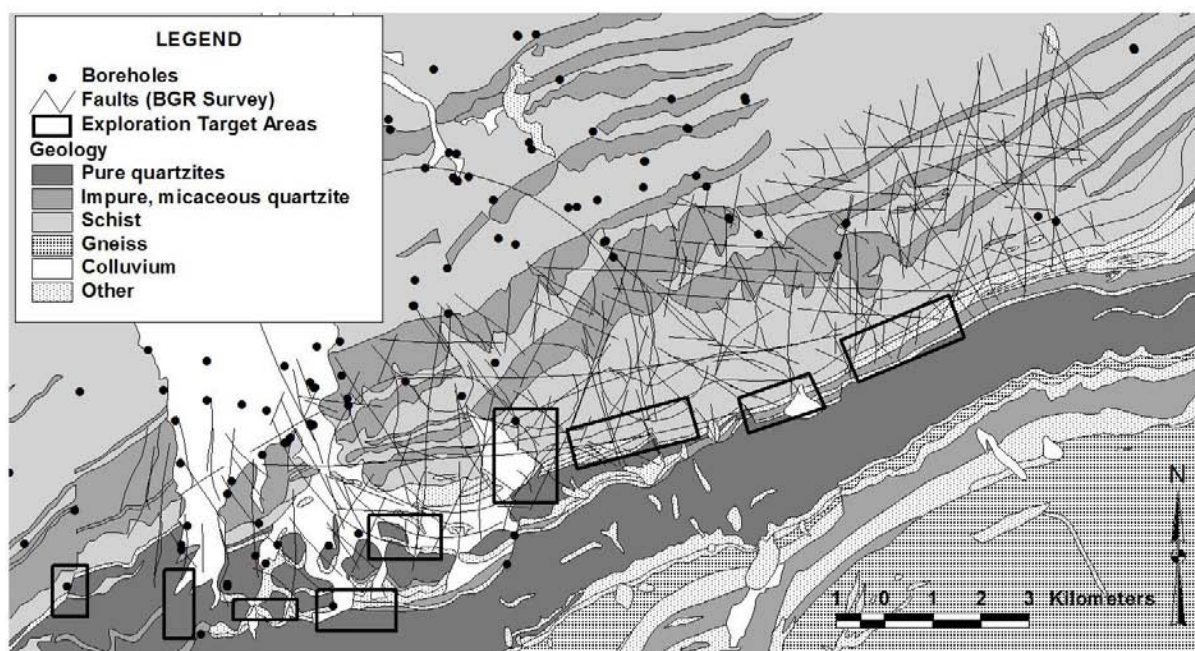
Figure 3. Exploration Target Areas

The prime target areas will first be 'tested' with exploration boreholes. Should they prove successful, they will be drilled for production purposes.

CONCLUSIONS

The City of Windhoek has chosen to manage aquifer recharge and storage in order to provide the water security they need. This is their preferred choice after detailed studies on various water supply options were considered. It is also the cheapest option. A phased approach to implementing the artificial recharge scheme was adopted, and the first phase has been completed with a maximum injection capacity of 2.8 Mm^3/a. Prior to implementing the second and third phases, a detailed assessment of the prime target areas was undertaken. This involved reviewing the tectonic history of the area (assessing tensional geologic environments) and doing an extensive remote sensing study that included satellite and air photo interpretation, and airborne geophysics. The results of this study confirmed previous recommendations about extending the artificial recharge scheme to the southern foothills of the Auas Mountains, east of the existing wellfields, and it resulted in the identification of potentially favourable areas immediately south and west of the existing wellfields. Prime 'Exploration Target Areas'. The target injection capacity of each new borehole is 0.4 – 0.8 Mm^3/a, and the expected full-scale injection capacity, after completing the second and third phases is ~8 Mm^3/a. This is about 40% of the city's current use.

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Modelling of well-field design and operation for an Aquifer Storage Transfer and Recovery (ASTR) trial

Paul Pavelic, Peter Dillon and Neville Robinson

Abstract

Defining the optimal number and arrangement of injection and recovery wells for a full-scale ASTR trial with wetland-treated urban stormwater must take into account mixing with the brackish ambient groundwater and residence time to allow for biodegradation to occur. Numerical simulations were performed with FEFLOW to explore the dynamics of the injected water plume for a range of possible ASTR scenarios. The analysis also took account of the quantity and sequence of flushing, orientation and magnitude of hydraulic gradients, uncertainties in aquifer properties, the proportion of water available for recovery, time-lag between injection and recovery, and the relative efficiency of two-, four- and six- well systems. Our results reveal that both the salinity and travel time constraints could be met with a 6-well system arranged within a quadrilateral domain with a uniform inter-well separation distance of 75 m. However this requires verification once local aquifer parameters and the risks associated with preferential flow have been better defined. A semi-analytical method for predicting the distribution of injected water 'fronts' in confined aquifers was found to compare favourably with the numerical approach suggesting this offers a potentially useful and robust design tool for testing different ASTR scenarios.

Keywords

ASTR; biodegradation; mixing; modelling.

INTRODUCTION

'Aquifer Storage Transfer and Recovery' (ASTR) seeks to demonstrate that wetland-treated urban stormwater, when injected into an aquifer and recovered from dedicated recovery wells, can create a safe and reliable drinking water supply (Rinck-Pfeiffer et al. 2005). Unlike ASR, which utilizes the same well for injection and recovery, the groundwater flow system established through ASTR by having separate injection and recovery wells offers more uniform residence time and travel distance in the aquifer, which is likely to lead to more predictable levels of chemical and microbial attenuation of contaminants necessary for the provision of water of potable quality. Further information on ASTR at the site selected for investigation on the Northern Adelaide Plains is presented by Rinck-Pfeiffer et al. (2005).

Establishing the ASTR well-field relies on identifying the optimal number and arrangement of injection and recovery wells that meet the various hydrogeological, operational and regulatory constraints. In this respect, a number of previous field and modelling studies have informed and influenced this study (eg. Bear and Jacobs, 1965; Trefry and Johnston, 1996; Dillon et al. 2002). For this study, groundwater flow and solute transport modelling was undertaken to test a range of possible ASTR well-field configurations and operational strategies. Two models were used, FEFLOW (Diersch, 2004) and a new semi-analytical model that tracks the movement and shape of the injected water fronts was developed to validate the numerical results. This paper summarizes a report by Pavelic et al. (2004).

WELL–FIELD DESIGN CRITERIA

In this system where the ambient groundwater is brackish and contaminants may be injected into the aquifer, there are two primary constraints to consider: (1) the proportion of ambient groundwater recovered is sufficiently small

that the salinity of recovered water is acceptable; (2) there is sufficient residence time in the aquifer to allow any contaminants to degrade to acceptable levels. Figure 1 demonstrates the competing needs to keep the separation distance of injection and recovery wells small enough to flush the aquifer with fresh water in the so-called 'transfer zone' around the well-field, and large enough to extend travel time to allow adequate time for contaminant attenuation.
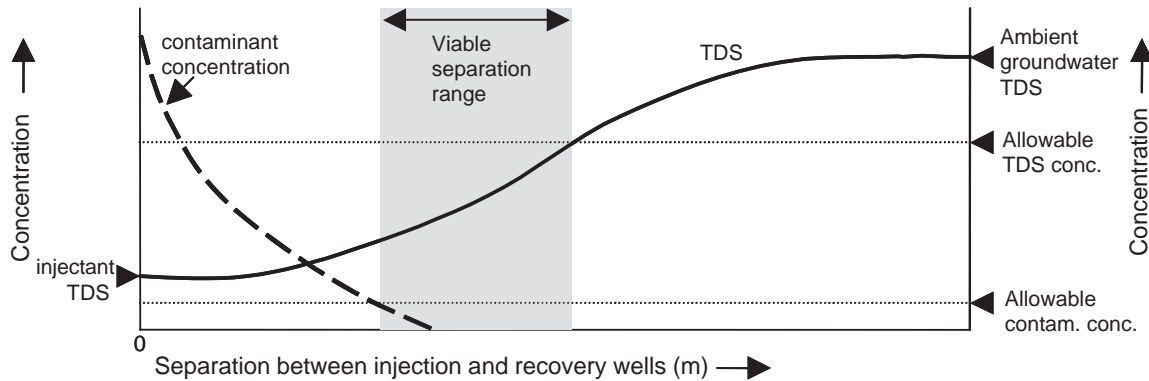
Figure 1. Schematic illustration of the effect of salinity and travel time constraints on the viable range of separation distances between injection and recovery wells

The injectant has an average TDS of 150 mg/L; the local ambient groundwater is 1,900 mg/L; and the maximum permissible concentration for the recovered water has been established at 300 mg/L, and therefore the minimum permitted mixing fraction (f) is ~0.9 (TDS is reasonably conservative in this system). This indicates that recovery is tightly controlled by mixing as the pumped water may contain no more than 10% ambient groundwater.

Wetland-treated urban stormwater runoff may contain a variety of constituents that, from a reuse perspective, can be of concern to human health or the environment. For the purposes of this study, microbial pathogens represent the single greatest risk with respect to the protection of human health (Toze, 2004). A minimum effective residence time of the injectant in the groundwater system of 300 days is proposed to ensure at least several log-removals of the most persistent microorganisms. Whilst this criterion is perhaps overly-conservative, given typical inactivation rates of pathogens and criteria in use elsewhere, such a barrier is justified from a risk management perspective to account for the slower rates of attenuation of other potential contaminants. This also recognises the uncertainty in knowledge of aquifer parameters and the potential for preferential flow. It also provides a realistic time-frame for the sampling and analysis of groundwater from intermediately positioned observation wells to give early warning in the event of unforseen water quality problems.

NUMERICAL MODELLING

The FEFLOW simulation package was used to predict the movement and mixing of injected waters in the aquifer. Briefly, the model was 2D in plan-view, with the ASTR operation situated at the centre of the 10,000 m x 10,000 m domain. The 52 m thick T2 sandy limestone aquifer targeted for storage was assumed to be homogeneous and isotropic with a dispersivity value consistent with that for the Bolivar ASR site, also in the T2 aquifer on the Northern Adelaide Plains (Pavelic et al. 2002). A regional gradient of 0.0015 was included. Injection and recovery rates were set at 25 L/s. The simulated time-scale was 10 years (beyond 3–5 years salinity in recovered water reached a steady state). An operating schedule was devised that takes into account two modes of operation, whereby, in the first year a greater than average volume of water is injected in order to flush the aquifer of ambient groundwater and no recovery occurs, then from the second year onwards, injection and recovery is more typical. Two plausible scales applying from years two to ten were tested that took into account the limited volumes of available recycled stormwater: one with 400 ML/yr injection and another with 600 ML/yr. In both cases only 80% of the

injected volume was withdrawn and the remaining 20% was intended to counteract losses due to regional drift. Further details on the conceptual model and numerical framework are given in Pavelic et al. (2004).
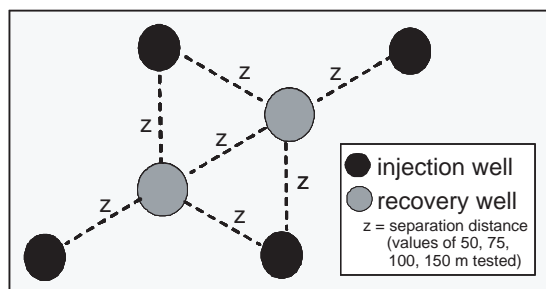
Figure 2. Schematic depiction of the six-well arrangement (the spacing is indicated in the key; regional groundwater flow is from right to left)

A six-well system was chosen after first considering and then eliminating two- and four- well systems since they failed to meet the two key constraints, including the total required recovery rate of 50 L/s. As seen in Figure 2, the six-well configuration is contained within a rhombic (i.e. diamond shaped) domain that maintains uniform separation distances between the injection and recovery wells, and its modular structure conceptually accommodates further expansion, as necessary.

Separation distances of 50, 75, 100 and 150 m were examined to determine the smallest distance at which the two constraints could be met for both scales of operation. Salinities were averaged from year 3 onwards as the year after flushing often had a small increase in salinity before stabilising. Figure 3 shows that the 75 and 100 m separations are the only spacings to consistently meet both constraints ($f > 0.9$, t > 300 days). At the 100 m scale increasing the volume injected from 400 to 600 ML/yr reduces the mixing fraction and the effective travel time, although both still exceed their target values. Reducing the separation improved the salinity of the recovered water (as suggested in Figure 1), however at the 50 m spacing the travel time constraint was not met. The largest separation distance failed due to high salinity. Of the two constraints, salinity is by far the more important (due to the high cost of desalination), whilst travel time can, to some degree, be manipulated by operational management, and the recovered water disinfected at relatively low cost if necessary.
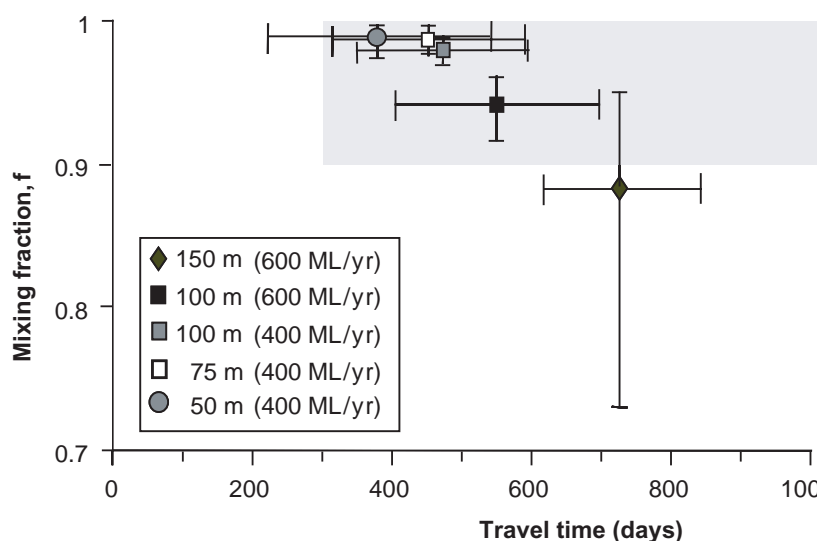
Figure 3. Mixing fraction (from year 3 onwards) versus travel time for 50, 75, 100 and 150 m separations (the bars identify the maximum and minimum times and concentrations; grey shading indicates the zone where both constraints are met)

SENSITIVITY ANALYSIS

The modelling also took into account other factors that are summarised as follows:

Quantity and sequence of flushing: A volume of no less than 1,000 ML was required to initially flush the transfer zone, and this required injections to be sequenced, including use of recovery wells to minimise the volume of ambient groundwater entrained.

Regional hydraulic gradient: Orienting the well-field such that the main transect of wells was approximately aligned with the background hydraulic gradient produced the lowest salinity in recovered water. The loss of injectant due

to drift was estimated to be 2 to 4%, depending on operational scale. The most pronounced impact on salinity was from transient local gradients that can exceed the regional gradient by a factor of 5 due to an ASR scheme situated one kilometre away.

Uncertainties in aquifer properties: Heterogeneity of the T2 aquifer was handled by adjusting effective porosity and aquifer dispersivity; the two parameters that strongly influence ASTR viability and are not sufficiently well defined. For all porosities tested, ranging from 0.1 to 0.4, recovered water quality met the mixing fraction target for the 75 m separation, 600 ML/yr scenario. Increasing porosity caused salinity to deteriorate slightly as plume size contracted and increased the residence time by approximately 100 days. Simulations of solute breakthrough at the nearby ASR site show that the effective porosity may be marginally lower than the base-case value used of 0.25. If so, this would imply the reduction in travel time to recovery wells and higher than anticipated volumes of water to flush ambient groundwater from the transfer zone. The base-case dispersivity value of 5 m was compared to 0.5 m and 50 m. Only the highest dispersivity value tested failed to meet the salinity target, although a value of this magnitude is considered unlikely for the T2 aquifer (Pavelic et al. 2002). The potential for significant preferential flow in the aquifer represents the single biggest risk to ASTR viability since it challenges both constraints by reducing travel times and mixing fractions. Tighter definition of the local aquifer hydraulic properties is required before further modelling is undertaken to verify the well-field design and operational strategies.

Proportion of water recovered: Withdrawing 80% of the injected volume, as modelled, was presumed to be an appropriate balance between maximising recovery efficiency and maintaining a buffer against the brackish groundwater. Values that exceed 80% could be tolerated in the short-term, provided the long-term average was maintained. When 100% was recovered, for instance, deteriorations occur in the longer term as some of the injectant was irrecoverable due to down-gradient migration.

Time-lag between injection and recovery: The residence time of recovered water is, in part, a function of the duration of the rest periods when neither injection nor recovery occurs. This in turn, is dependent on patterns of rainfall and demand for the water. As a worst-case scenario, removing the time-lag entirely reduces the maximum and minimum residence times. For the 75 m spacing this would reduce residence time to less than 300 days, however the four month time lag used as the base-case is considered a more realistic scenario.

Relative efficiency of an alternative six-well arrangement: A rectangular arrangement of wells was tested but resulted in higher mixing fractions than the rhombic pattern due to the greater volume in the transfer zone caused by the larger areal coverage of the well-field. The rhombic pattern consisting of wells at the apexes of two adjoining equilateral triangles with centroidal recovery wells is the more efficient of the two configurations.

COMPARISONS BETWEEN NUMERICAL AND SEMI-ANALYTICAL MODELS

The basis for the semi-analytical modelling is largely drawn from the conceptual and theoretical work by Bear and co-workers, which has been reported in Bear (1979). The movement of injected water is determined by tracking the position of the injected water particles along streamlines projected in up to 360 radial increments of 1 degree around each of the wells. The velocity distribution in the aquifer is determined by the Theis solution for well drawdown and is the net effect of the individual wells that are active at any given time. Because of the large exponential damping with time within annual cycles for flow fields determined by Theis solution, the steady state solution is also a good approximation. Front movement is determined by the displacement that occurs over successive timesteps and takes into account the drift due to regional groundwater flow. The semi-analytical method deals only with advective flow, unlike the numerical model where dispersive transport is included.

Figure 4 shows the injected water plume for a 100 m spacing at an operational scale of 400 ML/yr at two different stages: one at 125 days where injection into the four outer wells has just been initiated after completion of injection into two central (recovery) wells, and the other at 1,095 days after the recovery in the third year has concluded. Semi-analytical fronts were calculated and overlain on the FEFLOW-generated isofringe contours of solute distribution. The results show excellent agreement between the isofringes and fronts at these and all other stages of the simulation. The semi-analytical method demonstrates it can capture all the distortions to the fronts that arise due to interactions between the six wells. The method also clearly delineates between the major front arising from the current year of injection from the residual front that was not previously recovered.
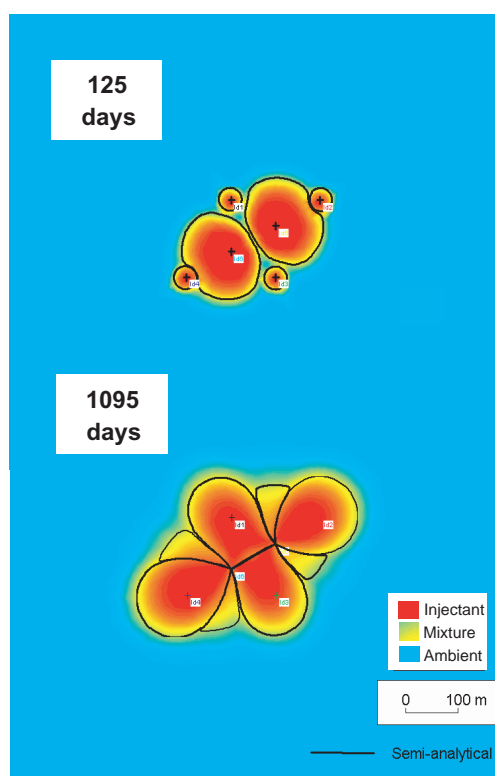
Figure 4. Comparison between FEFLOW solute isofringes and semi-analytical fronts during initial flushing at 125 days and after third-year recovery at 1,095 days

CONCLUSIONS AND RECOMMENDATIONS

This study has used two different modelling techniques to identifying the optimal number and arrangement of injection and recovery wells for a proposed ASTR trial at an operational scale of 400–600 ML/yr. Numerical simulations for a six-well system were found to meet the salinity and travel time constraints for the assumed conditions over the long-term with inter-well separation distances of 75 and 100 m (values of 50 and 150 m were also tested). The 75 m separation met the criteria above with the lowest annual volume of injectant for the assumed aquifer parameters at the site. The analysis also took account of the quantity and sequence of flushing, orientation and magnitude of hydraulic gradients, uncertainties in aquifer properties, the proportion of water available for recovery, time-lag between injection and recovery, and the relative efficiency of two-, four- and alternative six- well arrangements.

Aquifer heterogeneity is the highest risk to ASTR viability, and simulations of solute breakthrough at a nearby ASR site suggest that the effective porosity may be lower than the base-case used to model the ASTR well-field due to suspected preferential flow in the aquifer. If so, this would imply that there is a reduction in travel time to recovery wells and higher than anticipated volumes of water to flush ambient groundwater from the transfer zone. Further

modelling work is recommended to verify the 6-well arrangement chosen once local aquifer parameters have been collected.

A semi-analytical model was developed that uses particle tracking methods to determine the position of the moving fronts of injected and recovered water. Comparisons between the numerical and semi-analytical methods demonstrate that the FEFLOW results are accurate in defining fronts at all stages of injection and recovery. Therefore, the semi-analytical method offers a useful and robust tool for assessing the distribution of injected water bodies in confined aquifers for different ASTR scenarios.

ACKNOWLEDGEMENTS

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Recent advances in ASR Technology in the United States

R. David G. Pyne

Abstract

Aquifer storage recovery (ASR) is implemented at more than 70 sites in the United States. The WateReuse Foundation is conducting research and field sampling at four sites to investigate the fate of microcontaminants of wastewater origin during storage in ASR wells. The AWWA Research Foundation is conducting a research project entitled 'Design, Operation and Maintenance Considerations for Sustainable Underground Storage Facilities.' Work involves development of an updated site inventory of surface recharge, ASR and other well recharge projects; selection of case studies for analysis of design and operation experiences; and development of design criteria, long term operation procedures and maintenance requirements. Design is underway on what will probably be the world's first directionally-drilled ASR well, storing drinking water in a thin, brackish, confined sand aquifer to meet peak and emergency water demands. Research has recently been completed regarding an extensive laboratory and field investigation regarding the fate of bacteria, viruses and protozoa during ASR storage in aquifers with different temperatures and salinities. An analysis has recently been completed regarding the occurrence and attenuation of arsenic during ASR storage at 13 operating ASR wellfields in the southeastern United States.

Keywords

Aquifer storage recovery, artificial recharge, ASR, arsenic, groundwater recharge, reuse, wells.

INTRODUCTION

Aquifer storage recovery (ASR) technology is applied widely in the United States, with over 300 wells in operation at about 70 wellfields in 18 states, and many more in development. Most of these wells are storing seasonally available treated drinking water during months when demand is below peak levels or water quality is relatively good. Water is typically stored in deep, confined aquifers with a broad range of geologic settings: sand, sandstone, clayey sand, limestone, dolomite, basalt, glacial till, alluvium and conglomerates. A few ASR wellfields store water in unconfined aquifers. The stored water is recovered from the same wells to meet peak and emergency demands, or to meet a growing variety of other water supply needs. The largest ASR wellfield, at Las Vegas, Nevada, has 42 ASR wells with 157 MGD (557 Ml/D) recovery capacity. The deepest ASR well is at Des Moines, Iowa, at 2,700 ft (823 m). Although not now in operation, ASR success was demonstrated in a sea water aquifer at Marathon, Florida. Storage volumes range from about 30 to 3000 million gallons (MG) (0.1 to 11.4 Mm^3). Individual ASR well production capacities range from 0.5 to 8 MGD (2 to 31 Ml/d). Planned regional ASR programs include New York City (225 MGD, 852 Ml/D) and the Everglades Restoration Program in Florida (1700 MGD, 6,434 Ml/D).

About one third of these wells store water in brackish aquifers with total dissolved solids concentrations up to 18,000 mg/l, while many of the remainder store water in aquifers with poor ambient water quality that would require treatment for one or more constituents in order to achieve water quality standards. Increasingly ASR technology is being utilized to store water from other sources, including high quality reclaimed wastewater, treated surface water, and groundwater from overlying, underlying or nearby aquifers. Rapid implementation of ASR has occurred during the past 20 years, stimulated by its cost-effectiveness relative to other water supply and treatment alternatives, its demonstrated success, and its environmental benefits. A map and list of the operating ASR wellfields with contact information is available at http: // www.asrforum.com.

With the growing reliance upon ASR technology for cost-effective water storage, several new issues have arisen requiring scientific research, product development and advances in engineering design in order to meet evolving needs and regulatory concerns.

TARGET STORAGE VOLUME (TSV)

After initial development of a buffer zone around an ASR well, it is normal to achieve close to 100% recovery efficiency, as determined by dividing the recovery volume that meets target water quality standards by the volume recharged on the same ASR cycle, and excluding the initial buffer zone volume. The buffer zone has historically been formed over several operating cycles by leaving a portion of the stored water underground in each cycle, achieving steadily increasing recovery efficiency. In recent years experience has indicated that a simpler approach achieves the same goal. The buffer zone is placed into the well immediately following completion of well construction, following which the volume intended for recovery is stored. The well is operated in such a way as to avoid recovering the buffer zone.

The 'target storage volume' (TSV) surrounding an ASR well comprises the buffer zone volume and the stored water volume that will be recovered. Determination of the TSV depends upon several criteria, however it is typically presented in terms of 'days of recovery at the design production capacity of the well.' Typical TSV values from experience to date range from 50 to 350 days. The low end of the range might apply to a thin, confined, sand aquifer containing slightly brackish water, storing drinking water or reclaimed water intended to provide up to 30 days supplemental peaking supply to a community. The high end of the range might apply to a thick, semi-confined, karst limestone aquifer containing more brackish water, storing drinking water for a community with an unreliable surface water supply that requires sufficient water storage to sustain its needs for periods up to seven months.

A 'treatment zone' close to the ASR well essentially functions as a biological-geochemical reactor. Where sufficient carbon and nutrients are present, microbial and geochemical activity in this zone may be substantial, causing the redox level to fall dramatically, such as from +400 mv to −400 mv. The pH may also be reduced in this treatment zone. Where recharge carbon levels are low, such as through high level pretreatment of the recharge water, the treatment zone may extend further from the well. The treatment zone radius is not well known but is probably limited to a few meters.

Data has been generated regarding usually beneficial water quality changes during ASR storage (1, 2, 3). Attention is now starting to focus in the United States upon the treatment processes occurring during ASR storage, particularly in deep, anoxic aquifers. Typically the recharge water is from surface water or reclaimed water sources and contains organic carbon. It also often contains phosphorus, which is added to drinking water as a corrosion inhibitor for distribution system piping; and ammonia, which is typically added along with chlorine to form chloramines and thereby provide a disinfectant residual. Water quality changes occurring in the treatment zone include dissolution of metals such as iron, manganese and arsenic, and also reprecipitation, adsorption and biotransformation of these metals. Over a period of several ASR operating cycles at the same volume of recharge and recovery, the metals are leached from the aquifer matrix near the well and redeposited further away from the well, within the radius defined by the storage bubble.

ARSENIC ATTENUATION DURING ASR STORAGE

The limestone, artesian Floridan aquifer occurs in the southeastern United States. A recent study (4) of arsenic occurrence and attenuation at 13 operational ASR wellfields in this aquifer with a total of 65 ASR wells showed elevated initial arsenic concentrations occurring in the recovered water for 7 of these wellfields. Recovered water

concentrations from individual samples have in some cases exceeded 50 µg/l, compared to ambient groundwater concentrations of about 3 µg/l and drinking water standards of 10 µg/l. However after 3 to 6 operating cycles the arsenic concentrations in the recovered water had attenuated to below 10 µg/l at 3 of these 7 wellfields. Significantly, 16 of the 17 storage zone monitor wells located 150 to 450 ft (46 to 137 m) from the ASR wells showed no elevated arsenic concentrations. Further research is underway to better define the mechanisms for arsenic attenuation during ASR storage, however preliminary findings suggest the following:

- Arsenic is dissolved from arsenopyrite minerals in the limestone, most likely present in the flow pathways of the aquifer. Dissolution results from oxidation of the minerals.

- Carbon and nutrients in the recharge water stimulate subsurface microbial activity close to the ASR well. Within a few days, microbial activity and geochemical reactions reduce the pH and eliminate the dissolved oxygen content of the stored water around the well, driving the redox potential from $+300$ to $+400$ mv in the recharge water, down to -200 to -400 mv in the aquifer. The radial extent of this 'treatment zone' is not well understood but may be less than 10 m.

- Where the total organic carbon (TOC) content of the recharge water is low, such as below 2 mg/l, microbial activity may be reduced and subsurface reactions may require a longer time period to reach completion. Where the TOC content is high (15 to 20 mg/l), such as might occur with reclaimed wastewater, microbial activity around the ASR well will be greater, accelerating the treatment process.

Through dissolution, biotransformation, adsorption, precipitation and probably other mechanisms, the arsenic that is not produced from the ASR well during recovery is moved laterally away from the ASR well during successive cycles at approximately the same storage and recovery volume. Ultimate location of the arsenic is not well understood, however a reasonable hypothesis, based on field data obtained to date, is that it will accumulate within the buffer zone surrounding the ASR well. If ASR operations are conducted to avoid recovery of the buffer zone, such as during extended droughts, arsenic concentrations in the recovered water should be acceptable after a few operating cycles. Initial formation of an adequate TSV is probably the key to achieving this goal.

ATTENUATION OF MICROCONTAMINANTS OF POTENTIAL WASTEWATER ORIGIN DURING ASR STORAGE

ASR wells may potentially serve not only as a storage option for reclaimed water but also as an additional barrier to protect public health in those parts of the United States where reuse of highly treated wastewater is practiced for non-potable purposes. The WateReuse Foundation (WRF) is conducting an investigation of the fate of microcontaminants of wastewater origin during ASR storage. Carollo Engineers is the prime contractor for this work, which should be completed during 2006. A sampling list comprising more than 74 analytes has been developed, including endocrine disruptors, pharmaceuticals, personal care products, metals, disinfection byproducts, pesticides, pathogens, radioactivity, nutrients and general minerals. Sampling is underway at four operational ASR sites storing reclaimed water: Chandler, Arizona; Manatee County, Florida; Englewood, Florida, and Adelaide, Australia. Previous research at the Adelaide site (2) has shown that higher molecular weight constituents tend to be removed closer to the ASR well.

DESIGN, OPERATION AND MAINTENANCE OF SUSTAINABLE UNDERGROUND STORAGE FACILITIES

The American Water Works Association Research Foundation (AWWARF) has initiated this project, the primary

objective of which is to create an easy-to-use, practical document that will aid in the efficient design and operation of sustainable underground storage facilities, including both surface recharge and well recharge. ASR Systems LLC is the prime contractor for this project. A variety of geographic locations, capacities, geological settings and operational methods will be evaluated at operational ASR sites, providing a basis for updating and expanding existing 'managed aquifer recharge (MAR)' guidelines (5, 6). This project is scheduled for completion during 2007.

DIRECTIONAL DRILLING FOR ASR WELLS

Finding new well locations is becoming increasingly difficult, reflecting competing demands for different land uses, environmental and institutional constraints, aquifer contamination, setback requirements, legal restrictions and aesthetic considerations. Where suitable sites are available it is increasingly important to develop their maximum yield. Building upon prior directional drilling experience in the petroleum industry since the 1940s, the pipeline industry since the 1980s and the environmental remediation industry since the 1990s, directional drilling for the water industry is in an early stage of development in the United States. Fewer than 10 sites are known, some of which have been successful and some have not.

What may be the first directionally drilled ASR well is being designed by ASR Systems LLC for the City of Corpus Christi, Texas. Drinking water will be stored during winter months when system demand is low and supplies are plentiful, and will be recovered during summer months when peak demands occur or during emergencies. The storage zone is a confined sand aquifer at depths of 500 to 750 ft (152 to 228 m) with an ambient TDS concentration of 15,000 mg/l. Two wells are planned, extending in opposing directions from a common wellhead. These will be 'blind' wells, terminating underground. The well screens will be about 1,000 ft (300 m) long, inclined from the top to the base of the storage aquifer. Screens will be naturally developed, with an expected slot size of 0.008 in (0.2 mm). Target yield from the two wells is about 2.5 MGD (9.5 Ml/D). Screen and casing diameter, and materials of construction, are being selected carefully to withstand collapse and tensile pressures while also providing a suitable radius of curvature to accommodate a submersible pump. Allowable ground subsidence in this coastal area due to wellfield operations is a significant constraint. A vertical well would be limited to approximately 200 gpm (1 Ml/D) production rate to comply with the 0.5 ft (150 mm) subsidence limit. The horizontal well approach disperses the drawdown over a much wider area, achieving significantly higher production rates.

FATE OF MICROBIOTA DURING ASR STORAGE

Extensive investigations of the fate of pathogenic microbiota during ASR storage (bacteria, viruses and protozoa) have been conducted (7, 8). Results are also available for downloading from the www.asrforum.com website. These were primarily laboratory investigations under controlled conditions, supplemented by field samples of representative groundwater and surface water from selected ASR wellfield sites. A comprehensive scientific literature search was included as part of the laboratory investigation. Attenuation rates were evaluated within temperature ranges extending generally higher than most previous investigations so that results would be useful for warmer Florida ground and surface water temperatures. Different ambient groundwater salinities were also considered, ranging from 500 to 3,000 mg/l TDS. Attenuation rates, expressed in terms of days/log cycle, are presented for several microbiota, selected to represent conservative indicators of pathogenic contamination that might be expected in the aquatic environment. PRD-1 was included as a conservative reference point for viral attenuation since this constituent has no public health significance and is commonly used as a conservative tracer in lab experiments.

This work was supplemented by a summary of field investigations prepared by ASR Systems LLC, also on the same website. The summary report was based upon well-documented bank filtration sites, deep injection monitor wells,

ASR wells, sinkholes and drainage well investigations from which data has been obtained regarding attenuation of microbiota.

Of considerable significance is that the storage times associated with ASR wellfields, which are typically seasonal and range from weeks to years, is usually sufficient to achieve several log cycles of pathogen attenuation. Storage of surface water from a reasonably high quality source, in conjunction with filtration to remove particulates and source water monitoring to avoid recharge of highly contaminated water, may be sufficient in many cases to achieve microbial attenuation prior to ASR recovery, and also to protect groundwater quality for adjacent wells. The combination of bank filtration of surface water to improve source water quality, plus ASR seasonal storage deep underground, is a potentially powerful combination of proven, viable and cost-effective technologies that has yet to be applied for water management purposes. For many communities, particularly in developing countries, this will probably emerge as a useful water management strategy.

REGULATORY POINT OF COMPLIANCE WITH WATER QUALITY STANDARDS

The regulatory framework for ASR wellfields in the United States is evolving slowly, and generally moving in a direction that facilitates more efficient application of this technology while protecting public health, groundwater quality and the rights of adjacent groundwater users. State laws, combined with different water management needs, constraints and opportunities tend to drive the development of the ASR regulatory framework differently in each state.

All ASR sites measure compliance during *recovery* at either the individual ASR wellheads or at a point in the well-field collection system piping where the blended water from several ASR wells is sampled prior to distribution. A key issue has been the determination of the point of compliance measurement with applicable groundwater quality standards during ASR *recharge*. Where the compliance point is established at the ASR wellhead prior to recharge, as in Florida, experience is showing that the cost of pretreating the water to meet these standards is substantial. For recharge with filtered surface water, or groundwater pumped from an overlying surficial aquifer, UV disinfection pretreatment is required in order to reduce low levels of coliform bacteria to levels complying with drinking water standards, even though the storage zone is a brackish (TDS = 6,000 mg/l), artesian aquifer in which natural subsurface microbial processes would achieve the same treatment objectives at far less cost. Ammonia is added to high quality, chlorinated reclaimed wastewater prior to ASR storage in a brackish, confined aquifer so that THM concentrations of the recharge water will not exceed drinking water standards, even though THM attenuation would occur naturally in the storage zone. Pretreatment of drinking water prior to ASR storage to remove oxygen is under serious consideration at some ASR wellfield sites so that arsenic is not mobilized during ASR storage, even though available data shows that arsenic attenuation occurs naturally in the aquifer. Such pretreatment substantially increases the cost of ASR storage with little real benefit other than compliance with regulatory standards. An assessment of relative risk, benefit and cost is needed to achieve a balance that is deemed acceptable.

Other states have taken the position that compliance with groundwater quality standards will be evaluated at a monitor well, not at the wellhead prior to recharge. This is the situation in Arizona, Wisconsin and North Carolina, where ASR storage zones are fresh, not brackish. Allowable monitor well distances in Arizona and Wisconsin are up to 700 ft and 1,200 ft, respectively, (213 and 366 m) while in North Carolina they are determined on a site-specific basis. In these states the regulatory framework for ASR provides the opportunity for natural subsurface treatment processes to occur.

Most of the focus to date in these three states has been on the fate of disinfection byproducts (DBPs) during ASR storage. Previous researchers (1, 2, 3) have shown consistently that DBP's attenuate during ASR storage primarily

due to microbial reactions. Haloacetic acids (HAAs) and their formation potential are eliminated in a few days due to aerobic microbial reactions. Trihalomethanes are typically eliminated in a few weeks of storage due to anaerobic microbial reactions, and their formation potential also tends to be reduced. THM reduction may be small or negligible where the ASR storage zone is aerobic, such as in unconfined aquifers. However, most ASR storage zones are in confined, anaerobic aquifers.

The rate of DBP attenuation during ASR storage is probably dependent upon the TOC of the recharge water and the chlorine disinfection residual concentration. Where the TOC is reduced to very low levels by high levels of pre-treatment and then the water is disinfected prior to recharge, bacterial activity in the storage zone may be inhibited around the well. As a result more time would be required for bacterial activity to reduce THM concentrations. A key issue in the United States is that allowable THM concentrations for drinking water are 80 µg/l whereas for groundwater recharge through wells some states measure compliance with much lower standards for individual THM species. For example, THM concentrations in the recharge water at Oak Creek, Wisconsin, ranged from 11.9 to 18.3 ug/l during cycle testing, well within the 80 µg/l regulatory standard (9). However bromodichloromethane (a THM constituent) concentrations in the recharge water ranged from 4.3 to 6.5 ug/l, exceeding the state standard of 0.6 µg/l for ASR recharge. Compliance is measured at a monitor well up to 1,200 ft (366 m) away, not at the ASR well, providing the opportunity for natural subsurface treatment processes to occur, in addition to dilution. Cycle testing demonstrated such attenuation.

CONCLUSIONS

ASR has become an important water management tool in the United States, and globally, providing enhanced water supply security and sustainability at relatively low cost. Improved understanding of the biological and geochemical processes governing beneficial water quality changes during ASR storage is a suggested path forward for future research and development .

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Aquifer Storage Recovery (ASR): An economic analysis to support use as a strategic managerial tool to balance a city's desalinated water production and demand

Nauman Rashid and Asam Almulla

Abstract

Most cities around the globe are faced with the same challenge – the year round availability of adequate freshwater. The quality, quantity, reliability and economics of the supply become measuring sticks for success. In the city of Sharjah in the UAE, as in other regional countries, the creation of large buffer storages for its desalinated water production is a well-recognized need for optimizing its water resource management. In the face of expensive surface tanks, the use of Aquifer Storage Recovery (ASR) has been evaluated to support economic decisions towards its use in balancing the city's desalinated water production and demand.

To evaluate the use of ASR in this manner, the production versus demand charts for the city were analyzed, daily and seasonal variations plotted and a number of scenarios created for strategic and seasonal objectives. Detailed economic evaluations carried out included the storage size, facility costs, the recharge water source and related plant and operating costs. To meet the challenges, options evaluated using decision tree software included utilizing surplus production, dedicated desalination units or purchasing from a national grid.

The conclusions drawn clearly demonstrate the viability of Aquifer Storage Recovery as a reliable and economic building block of an overall water resource management strategy.

Keywords

Aquifer Storage Recovery; desalination technologies; techno-economic evaluation; water demand and production.

INTRODUCTION

In the United Arab Emirates (UAE), desalination is the main source of potable water. The use of this source is steadily increasing. This paper deals specifically with water storage as a water management and cost reduction tool. Due to unexpected circumstances, interruption in water production may occur. A storage facility for water can play a major role in providing uninterrupted water supply to consumers and overcome any crisis or emergency situations. The objective of this work is to investigate, cost and present the operation of a strategic storage facility in the city of Sharjah to handle crisis or emergency scenarios and to reduce the overall cost of water production.

WATER PRODUCTION AND STORAGE

Having a water storage facility has the following advantages:

- Handles operational disturbances, providing a smooth, continuous water supply;
- Offset shortages due to a sudden unplanned plant or equipment shut down;
- Strategic planning to secure the water supply following major crisis;
- Reduce over design capacity and thus capital cost of desalination plants, which are usually designed to meet peak demand, a condition that is reached for only part of the time. At off-peak hours when the water demand is less than the nominal (design) demand, water can be produced and stored. This water can be used later to meet peak demand.

Types of storage

The main types of storage that can be utilized in Sharjah include surface tanks, lined ponds and Aquifer Storage Recovery (ASR). The first type is the most expensive; the second requires a huge area of land. The third type, ASR, will be discussed in this work as a strategic water storage facility for the city of Sharjah. ASR is a proven technology for storing large volumes of water (Maliva, Missimer, 2003).

Sharjah water production

Sharjah supplies water to its customer from two main sources: desalinated water (about 60%) and ground water (about 40%). The current storage capacity of potable water for Sharjah is only 17.5 MIG (Million Imperial Gallons). Water is mainly produced in Sharjah in conjunction with power cogeneration plants. The electricity demand varies by about 70% from winter to summer. The water demand has a uniform trend throughout the year.

Sources and scenarios for water supply/recharge

Sharjah has a desalinated water production of about 42 MIGD (Million Imperial Gallons per Day). The possible methods to source water for a storage facility are:

- Exploit the existing desalination units by running them at higher or full capacity.
- Build a new seawater reverse osmosis (SWRO) desalination plant that could utilize the idle power in the winter or spinning reserve to produce water at a very low cost.
- Buy water from large water producing companies or from a national water grid.

Decision making techniques will be utilized in a later section to determine the type of scenario that would be optimal for storage. The size of the storage facility will be determined by the situation it will be required to address. Two situations have been identified: major crisis including main plant outage and improving overall cost.

Strategic planning for a major crisis

Major crisis is assumed to interrupt water production for a full week. Based on a flowrate of 55 MIGD, the needed storage size is 385 MIG at a theoretical 100% recovery and assuming adequate deliverability. It is assumed that a crisis can occur at least once every five years but the recharge completed within one year.

Cost improvement planning

This scenario assumes that all desalination plants are designed at nominal capacity. The storage will have enough water at all time to cover any load above the nominal capacity. Table 1 represents the monthly water production for Sharjah. It is clear that the plant is running six months of the year above the average production rate. The maximum accumulated capacity is around 300 MIG.

Two scenarios for sources of recharge are considered: using existing desalination plants and using a new reverse osmosis desalination plant.

Using existing desalination plants

Since the average demand is used to calculate the overload, the plants should have the capability to provide enough loads at off-peak times that match the year round overload. Table 1 also shows when water can be stored and when water can be recovered. From this Table, the size of the storage facility is determined to be 300 MIG.

Table 1. Monthly water production calculations*

Month	Daily demand	Daily surplus/ shortage	Monthly surplus	Recharge	Recover	Accumulated capacity
January	48.11	−2.4	−73.3	−73.3		−162.0
February	48.73	−1.9	−57.5	−57.5		−219.6
March	49.45	−1.2	−35.9	−35.9		−255.5
April	49.20	−1.4	−43.4	−43.4		−298.9
May	52.45	1.8	54.0		54.0	−244.9
June	55.05	4.4	132.1		132.1	−112.8
July	53.32	2.7	80.1		80.1	−32.7
August	51.43	0.8	23.5		23.5	−9.2
September	50.96	0.3	9.2		9.2	0.0
October	49.56	−1.1	−32.6	−32.6		−32.6
November	51.68	1.0	30.9		30.9	−1.7
December	47.75	−2.9	−87.1	−87.1		−88.8
Average	50.64		Total	331	−331	

* All numbers are in million imperial gallons (MIGs).

Using new reverse osmosis desalination plant

It is assumed that the desalination plant generates 1 MIGD and runs for 330 days per year with an average production rate of 27.7 MIG per month. Table 2 shows how much water needs to be in the storage facility for each month to meet the overload of water demand. The capacity of the storage facility is around 184 MIG.

Table 2. Monthly Water Production Calculations Using SWRO*

Month	Daily demand	Daily surplus/ shortage	Monthly surplus	Recovered amount	Capacity SWRO	Accumulated capacity
Jan	48.11	−2.4	−73.3		27.6	−81.4
Feb	48.73	−1.9	−57.5		27.6	−53.8
Mar	49.45	−1.2	−35.9		27.6	−26.2
Apr	49.20	−1.4	−43.4		27.6	0.0
May	52.45	1.8	54.0	54.0	27.6	−26.4
Jun	55.05	4.4	132.1	132.1	27.6	−130.8
Jul	53.32	2.7	80.1	80.1	27.6	−183.3
Aug	51.43	0.8	23.5	23.5	27.6	−179.2
Sep	50.96	0.3	9.2	9.2	27.6	−160.9
Oct	49.56	−1.1	−32.6		27.6	−133.3
Nov	51.68	1.0	30.9	30.9	27.6	−136.6
Dec	47.75	−2.9	−87.1		27.6	−109.0
Average	50.64		Total	331	−331	

* All numbers are in million imperial gallons (MIG).

COST ANALYSIS

Equivalent Annual Cost (EAC) is used to compare the various options.

$$EAC = \text{Total Annual Costs} + \text{Total Capital Cost} \frac{i(1+i)^n}{(1+i)^n - 1}$$ with i = interest rate, n = time

The cost of a 5 MIGD RO plant was taken as US$ 20.0M. Water in-take cost was added to capital cost and a 40% recovery assumed. Table 4 (Awerbuch, 2004) gives the cost breakdowns.

Table 4. Cost details for RO technology

Cost	RO	Unit
Capital Cost	4.0	US$ M/MIGD
Capital Cost - Intake/Outfall	0.1	US$ M/MIGD of Feed water
Power Consumption	5	kW.hr/m^3 permeate
Chemical Cost	50,000	US$/yr per MIGD
Labor	50,000	US$/yr per MIGD
Membrane replacement cost	10%	Of original capital cost repeated every 5 years

Storage alternatives

Considering a major crisis only. For a 385 MIG ASR (US$ 10.3M), the recharge options can be:
• New 1.2 MIGD SWRO (US$ 6.8M) with EAC of US$ 0.97M (using an interest rate of 6% and a life time of 20 years as quoted by SEWA). The total EAC of this is US$ 1.867M
• National water grid with no further major capital investment and water cost US$ 8,150/MIG. The total EAC is US$ 3.952M.

Considering overload only. There are three recharge alternatives:
• Use existing desalination plants. For a 300 MIG ASR (US$ 8M) with desalination plant annual operating costs of US$ 0.815M, the EAC is US$ 1.515M.
• Build a new 1 MIGD SWRO plant with EAC of US$ 0.83M.
• National water grid. The EAC of this is US$ 3.0M.

Combination of annual overload and crisis. There are three different alternatives:
• Annual overload and 100% crisis (of the maximum capacity of 385 MIG). For this alternative, the size depends on the recharge option.
  - Existing desal plants plus new 1.2 MIGD SWRO with EAC of US$ 0.97M. This means a 685 MIG ASR (US$ 18M) with EAC of US$ 1.57M. With desalination plant annual costs of US$ 0.815M, the total EAC is US$ 3.355M.
  - Existing desal plants plus national grid. For 685MIG ASR, the EAC is US$ 5.522M.
  - 2.2 MIGD SWRO. For a 596 MIG ASR (U$ 16M), the total EAC is US$ 3.023M.
  - Use only national grid. The EAC is US$ 5.543M.
• Annual overload and 50% crisis.
  - Existing desal plants plus new 0.6 MIGD SWRO with EAC of US$ 0.537M. This means a 493 MIG ASR (US$ 13M) with EAC of US$ 1.134M. With desalination plant annual costs of US$ 0.815M, the total EAC is US$ 2.486M.
  - Existing desal plants plus national grid. For 493MIG ASR, the EAC is US$ 3.518M.
  - 1.6 MIGD SWRO. For a 376 MIG ASR (U$ 10M), the total EAC is US$ 2.111M.
  - Use only national grid. The EAC is US$ 4.020M.
• Annual overload and 25% crisis
  - Existing desal plants plus new 0.3 MIGD SWRO with EAC of US$ 0.3M. This means a 396MIG ASR (US$ 10.55M) with EAC of US$ 0.92M. With desalination plant annual costs of US$ 0.815M, the total EAC is US$ 2.035M.
  - Existing desal plants plus national grid. For 396MIG ASR, the EAC is US$ 2.52M.
  - 1.3 MIGD SWRO. For a 280 MIG ASR (U$ 7.5M), the total EAC is US$ 1.7M.
  - Use only national grid. The EAC is US$ 3.228M.

DECISION MAKING

A decision tree analysis process (Clemen, Reilly, Reilly, 1999) was used to determine the size of the ASR and recommend the best water source option. A decision tree was constructed based on decision and chance nodes. In a decision node, the software chose the best economical alternative. The chance node involved probabilities and the software calculated the expected monetary value (EMV) of each, choosing the best option. The probabilities were assumed as:

• Crisis only, probability: 5%;
• Annual overload and full crisis, probability: 5%;
• Annual overload and half crisis, probability: 10%;
• Annual overload and quarter crisis, probability: 50%;
• Annual overload only, probability: 30%.

Decision tree outcome

Based on the above assigned probabilities, the best alternative was found to be annual overload plus 25% crisis, using only SWRO. The decision tree is presented in Figure 1.

Sensitivity analysis

A sensitivity analysis was carried out to investigate how changing the probabilities affected the final decision. For a probability of 40.5% or higher, the annual overload only option was selected.

Operational scenario

Based on the alternative selected by the decision making process, an operational scenario for storing water in the off-peak demand times and recovering during peak times can be developed. At the time of a crisis, the water recovery rate will increase by the average value and no recharge would occur.

CONCLUSION

ASR is considered in this work as a cost-effective, water managerial tool towards building a storage facility to support strategic planning and in its production cost improvement. Several scenarios for storage capacity and water source options were considered and evaluated. Decision tree analysis and risk assessment were used to determine the capacity of ASR and to determine the water source option. The optimum ASR capacity for the case of the city of Sharjah was determined to be 284 MIG. This capacity accounts for overloads throughout the year and additionally caters to a minimum of 25% of volumes required in the case of a major crisis. A new RO plant with a capacity of 1.3 MIGD is recommended, partly for providing the feedwater for building the ASR capacity.

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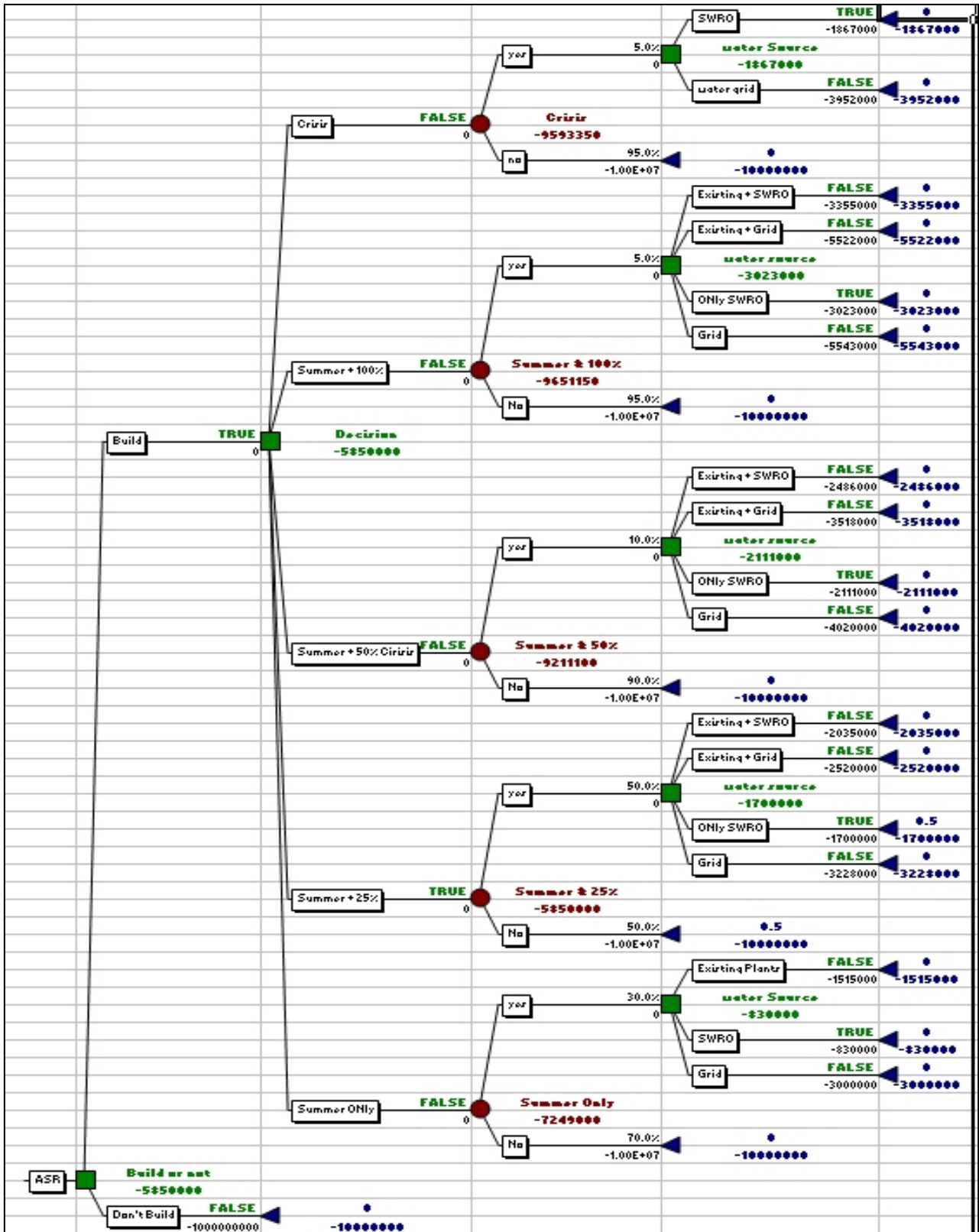
Figure 1. Decision Tree Analysis

Stormwater ASR in practice and ASTR (Aquifer Storage Transfer and Recovery) under investigation in Salisbury, South Australia

Stephanie Rinck-Pfeiffer, Colin Pitman and Peter Dillon

Abstract

The existing Parafield stormwater ASR scheme on the Northern Adelaide Plains, South Australia, has been successful in harvesting 1,100 m^3 yr^{-1} urban stormwater from a 1,600 Ha catchment by diverting it via a weir in a trunk drain to a 50,000 m^3 capacity capture basin. From there, it is pumped to a similar capacity holding basin, from which it gravitates to a 2 Ha reed bed to improve its quality for use directly, or via two ASR wells. These store and recover the treated stormwater in a confined limestone aquifer 160–200m below ground surface to balance demand and supply. The water is used to supply a wool scouring plant with water of lower salinity than the mains water supply. Success with this and other projects using stormwater and reclaimed water led to the formulation of an ambitious project adjacent the Parafield ASR scheme aimed at producing drinking water supplies from the same source of treated stormwater. 'Aquifer Storage Transfer and Recovery' (ASTR) the concept to be tested will use separate injection and recovery wells to extend the residence time of the injected stormwater in the aquifer and to allow for additional natural treatment through the aquifer. This will produce more predictable levels of chemical and microbial contaminant attenuation, essential for the provision of water of potable quality, than can be produced by ASR 'aquifer storage and recovery' (which uses the same well for injection and recovery). In this case the ambient groundwater is brackish and will require freshening with stormwater before recovered water can be harvested at an acceptable salinity. A Hazard Analysis and Critical Control Points (HACCP) approach has been adopted to provide multiple barriers for protection of water quality in the urban catchment, in selecting water for harvesting and in treating the water, and will verify effectiveness of treatment and assist in transferring this methodology to other catchments. This paper describes the existing Parafield ASR scheme and outlines the planning for the ASTR project.

Keywords

ASTR; ASR; stormwater reuse; modelling, HACCP, wetlands.

INTRODUCTION

The Parafield Stormwater Harvesting Facility is a unique project of this kind initiated by the City of Salisbury Council in converting stormwater from an urban nuisance and coastal pollution threat into a valuable resource for industry and the community. Stormwater from the local catchment is diverted into a series of uniquely designed capture, holding and wetland treatment basins. Treated stormwater in excess of the immediate needs of local industry is stored in an aquifer via two ASR wells for recovery at times when detained runoff is inadequate to meet demand.

The concept for the ASTR project builds on the Parafield ASR project, more than 20 operational ASR projects conducted over 10 years with stormwater, reclaimed water and potable water in South Australia (Gerges et al, 2002; Hodgkin, 2004) and on the results of a recent research project where water quality improvements have been documented at ten sites in USA, Australia and the Netherlands, as summarised in Dillon et al (2005). To date stormwater ASR in South Australia has been for use in irrigation, industry, or to replenish aesthetic urban lakes. However, this relies on finding convergence of the location of a suitable source of stormwater, and land to harvest it, with the localised high demand for water of irrigation quality. If water could be produced that may be demonstrated to continually meet drinking water guidelines, this could enable the existing system of water mains to be used for distribution, and allow better use of urban brackish aquifers to assist in securing water supplies for the water-stressed city of Adelaide.

This paper describes some of the planning for an internationally unique type of project that will inject urban stormwater into a brackish aquifer with the intention of recovering water fit for continuous sustainable supply at potable quality. This process is referred to as 'ASTR' 'Aquifer Storage Transfer and Recovery' where separate injection and recovery wells are used to extend the residence time of the injectant in the aquifer. This allows for greater natural treatment through the aquifer than can be assured by ASR 'Aquifer Storage and Recovery' where the same well is used for injection & recovery (Figure 1). The enhanced residence time and travel distance in the aquifer is expected to provide more predictable levels of chemical and microbial attenuation of contaminants essential for the provision of water of potable quality.
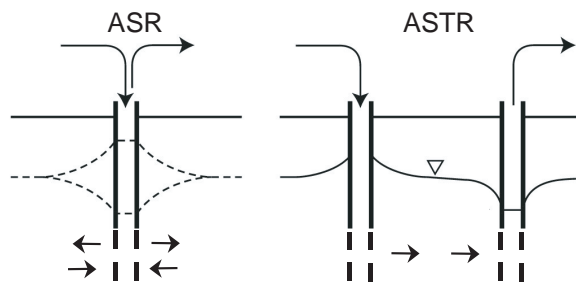
Figure 1. Schematic representation of the differences between ASR and ASTR (after Dillon 2005)

PARAFIELD ASR SCHEME

Design and operation

The Parafield stormwater project involves diversion of stormwater via a weir in the main Parafield drain to a 50,000 m^3 capacity capture basin. From there, it is pumped to a similar capacity holding basin, from which it gravitates to a two hectare reed bed. There nutrient, metal and organic pollutant loads are typically reduced by up to 90 per cent. The system is designed to hold stormwater for around 10 days to ensure optimal treatment efficiency, prior to ASR or use. The current supply capacity of the scheme is 1.1 Mm^{3}y^{-1}. The second stage would add other catchments and boost the supply to 2.1 Mm^{3}y^{-1}.

The wetland-treated stormwater typically has a salinity of 200 mg/litre and the salinity of the groundwater into which it is injected is from 1,900 to 2,000 mg/litre. The injected stormwater forms a plume or lens of low salinity water. The aim of the ASR scheme is to build up a reserve storage of injected stormwater in the aquifer over the first few years of harvesting, which will provide a buffer against a series of dry seasons. This will also enable extraction of up to 100% of the quantity injected during the previous season and ensure continuity of supply.

The ASR system consists of two ASR wells to a depth of 180m in the T2 aquifer, a confined Tertiary limestone aquifer which is regionally extensive. Injection rates are typically 30 L/s and recharge water quality has to meet the

Environment Protection Authority (EPA) requirements. The ASR wells are equipped with extraction pumps and recharge (injection) facilities. There are five observation wells used for monitoring the effects of the ASR operation. Three are located around the Parafield facility and two are placed 900 m to the south one in the overlying T1 aquifer and one in the T2 aquifer.

The extraction pumps are variable speed drive submersible units each with a nominal capacity of 4.3 ML/d, giving a total ASR delivery capacity of 8.6 ML/d. The recharge capacity of each well is nominally 3.0 ML/d, giving a total of 6.0 ML/d recharge capacity. The well pumps and recharge facilities are operated automatically by the control system which can be accessed by telemetry from the Salisbury Council offices. Both wells operate together either in the recharge mode or in delivery mode or are inactive at any one time. Individual wells can be taken out of service. All cleansed water flows in excess of the pipeline demand rate are recharged to the ASR wells. When there is no water flow from the reedbed the full demand is supplied from the ASR wellfield.

It is infeasible to extract 100% of the volume of water that has recharged the aquifer and maintain acceptable salinity. This is because of mixing of the recharged water and the native groundwater around the edges of the recharge plume. Hence it is assumed that only 80% of the recharged water can be extracted at an acceptable salinity. Flow controllers are located on the recharge connections to avoid over pressurising the aquifer during recharge. The flow controller setting at the well head is based on the hydraulic characteristics of the completed well. At times the wells suffer clogging from particulates and biomass build-up. Each well has a purging system for clearing of the clogging material. This is discharged to the in-stream basin via 150 mm diameter purge water pipes. Flow, level and salinity data are collected from the ASR operating and observation wells by the control system as part of the ASR Management Plan required by the Department of Water Land Biodiversity and Conservation (DWLBC) and the Northern Adelaide – Barossa Catchment Water Management Board (NABCWMB). Due to this project Australia's largest wool processing company (G.H. Michell & Sons) receives water with a salinity (TDS) of less than 250 mg/L, which is significantly lower than the salinity of mains water derived in dry years mostly from the River Murray (which frequently exceeds 400 mg/L).

Operational results

Stormwater harvesting and wetland

Stormwater could be selected for harvesting and poorer quality water was bypassed from the ASR system. Table 1 indicates the quality of water in the drain and in the discharge from the wetland. The data indicate no parameters exceed the national guideline for irrigation water quality, and most parameters for wetland-treated stormwater compare favourably with existing drinking water supplies.

Table 1. Contaminant reduction through selection of water from Parafield Drain for harvesting and treatment in the Parafield Wetlands*

PARAMETER

Suite Taken	Parafield drain median	Reedbed median	Percent reduction
General			
Conductivity (µS/cm)	457	191	58[#]
Total Dissolved Solids	250	100	60[#]
pH	7.70	7.00	9
suspended solids	13.00	3.00	77
total organic carbon	8.90	4.40	51
turbidity	8.10	1.70	79

Table 1. (contd.)

PARAMETER

Suite Taken	Parafield drain median	Reedbed median	Percent reduction
Major ions			
calcium	29.9	15.70	47
magnesium	5.4	3.50	35
potassium	5.05	2.05	59[#]
sodium	57.35	18.85	67[#]
bicarbonate	83	59	29
chloride	78	25	68[#]
fluoride	0.55	0.20	64[#]
sulphate	47.6	10.60	78[#]
Nutrients			
ammonia as N	0.033	0.0125	62
phosphorus – Total P	0.067	0.024	64
TKN as Nitrogen	0.77	0.26	66
total nitrogen as N	0.74	0.295	60
nitrate + nitrite as N	0.0945	0.021	78
Microbiological			
E. Coli (coliforms/100 mL)	83	29	65
Metals			
Aluminum	3.02	0.010	99
Arsenic	0.0005	0.0005	0
Cadmium	0.00025	0.00025	0
Copper	0.0080	0.002	75
Iron	0.275	0.16	42
Lead	0.0034	0.0006	82
Manganese	0.0127	0.0041	68
Zinc	0.056	0.029	48

* All data reported in mg/L, unless otherwise noted. Data obtained from City of Salisbury Database
\# Water selected for harvesting avoids higher salinity runoff.

Aquifer injection

Injection commenced into Production Well 1 on the 17 June 2003 and Production Well 2 on 17 July 2003. Commissioning of the injection system continued through into mid August with refinements being made to the operating logic. A total of 193 ML and 166 ML was injected into Well 1 and Well 2 respectively, giving a total injection volume of 359 ML for the period (Figure 2).

Under normal operating conditions given sufficient water in storage, one well would be given an injection rate set point of 27 l/s, with the other well operating with a variable injection rate between 4 and 27 l/s. This regime was set up to maximise the injection volume available, taking into account water demand which varied from 6 to 30 l/s, pump performance, and well head pressures. During June 2004, this methodology was revised based on reduced demand by the wool processing industry, and injection rates at both wells were varied between 4 and 35 l/s.

Results from the ASR scheme at Parafield indicate that during the injection season 2003/2004, the groundwater

levels in the aquifer rose 10 to 15 metres at a distance of up to 2 km from the injection point. This slowly returned to its original level after injection ceased.
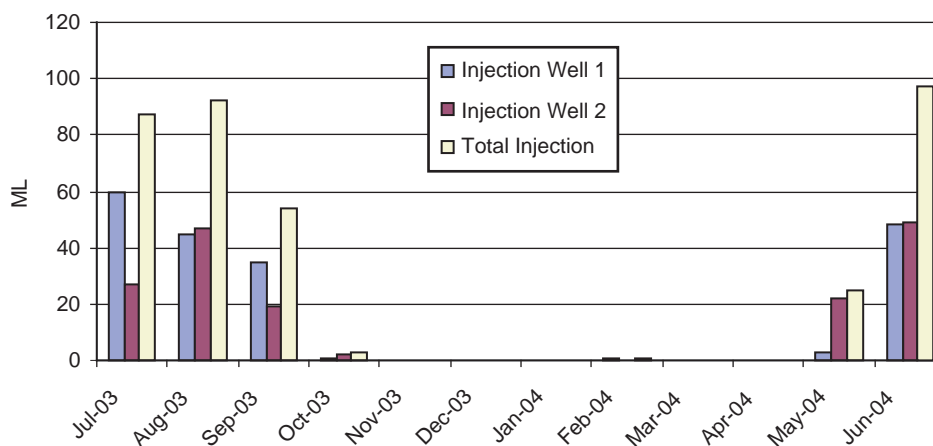
Figure 2. Monthly injection volumes 2003/2004

Aquifer extraction

Extraction from the aquifer occurred through Production Well 1 during January, February and March 2004 via a temporary pump installed during this period. Water was extracted from the well into the wetland via a temporary lay-flat hose, for distribution to wool scouring 2. A total of 36 ML was extracted at that time.

The quality of extracted water was extremely good, with a slight increase in iron content from the aquifer at one site and some sand and silt during the first few hours of extraction at most sites.

PROPOSED GREENFIELD ASTR PROJECT

Proposed ASTR scheme description

Success with the Parafield ASR scheme and experience from other research on ASR (e.g. Dillon et al, 2005) led to a proposal to undertake a major demonstration project at Greenfield, adjacent to Parafield. This would inject a further 400,000 $m^3 y^{-1}$ of wetland-treated stormwater into the brackish aquifer and recover water fit for continuous sustainable supply at potable quality (Figure 3).

The key objectives of the project are:

1. to demonstrate that stormwater from regulated urban catchments can be stored in aquifers and recovered to produce sustainable supplies of high quality drinking water;
2. to demonstrate that the multiple barrier methods used for managing water quality, including in-aquifer treatment and monitoring methods, are economic, robust, sustainable and transferable within Adelaide and to developed and developing countries;
3. to validate and document a Hazard Analysis and Critical Control Points (HACCP) methodology for drinking water supplies from waters of impaired quality, and to identify the range of likely successful applications and any constraints.

The Parafield ASR scheme has existing infrastructure for monitoring and collecting urban stormwater in the Parafield wetland, treating it in a wetland and delivering it by pipeline to the proposed ASTR site and subsequently

delivering recovered water to a mixing tank. There it will be blended with reclaimed water to reduce the salinity of the product water and distributed to a new subdivision via a dedicated recycled water distribution system for use in toilet flushing and garden watering. Space has been allocated for a six well ASTR system where the aquifer is suitable and there are existing monitoring wells. A preliminary HACCP plan has been produced and an existing stormwater pollution control program is already in place (Swierc et al., 2005).
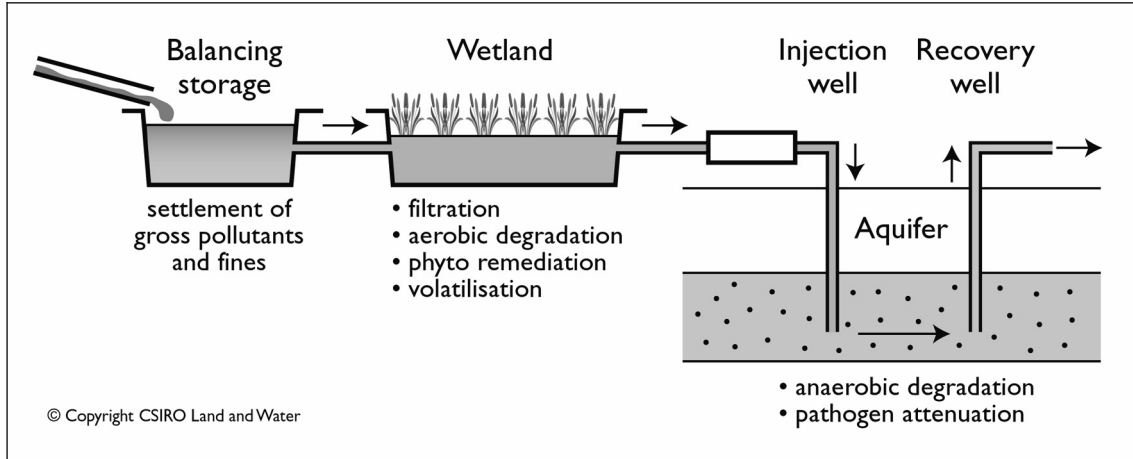
Figure 3. Schematic representation of the ASTR Process

The study site at Greenfields is 900m south of the Parafield ASR scheme on parcels of land largely administered by the City of Salisbury. This site was chosen ahead of two nearby sites (Parafield and Greenfields North) after weighing up considerations such as land use and availability, aquifer storage potential, groundwater salinity, the potential for hydraulic interaction with existing ASR operations and the number of existing monitoring wells (Rinck-Pfeiffer, 2004).

The same aquifer as at Parafield is to be recharged. This is the 'T2' aquifer, the second of several Tertiary marine-deposited formations continuous across the Adelaide Plains and is composed of fossiliferous and marly limestones through to siliceous calcarenite (Gerges, 1999). Locally, the T2 aquifer is 52 m thick and encountered at depths of between 154 and 206 m below ground surface. The transmissivity is moderate (125 m^2/day), with available evidence suggesting that flow is porous rather than through fissures or karst. However the aquifer is known to be heterogeneous with respect to depth, although the major layers may be continuous over the distances of tens to hundreds of metres (Pavelic et al., 2001). The effective porosity of the aquifer is assumed to be 0.25. The ambient groundwater is brackish, with a salinity of ~1,900 mg L^{-1} TDS measured at the existing well.

Defining well positions

One of the issues in establishing the project was to identify the optimal number and arrangement of injection and recovery wells that meet the various hydrogeological, operational and regulatory constraints at the site selected for investigation. Preliminary groundwater modelling was undertaken to test a range of possible ASTR well-field designs. An established numerical flow and solute transport model (FEFLOW) was used. Also a new semi-analytical model was developed specifically for this project to validate the FEFLOW results. The semi-analytical model involves tracking of movement of injected and recovered water fronts. Some key results from these two independent methods are reported (Pavelic et al., 2005) in these proceedings.

The proposed ASTR trial was simulated at an operational scale of 400,000 – 600,000 m^3/yr. Numerical simulations for a six-well system were found to meet the salinity and travel time constraints for the assumed conditions over the long-term with inter-well separation distances of 75 and 100 m (values of 50 and 150 m were also tested). The 75 m separation met the criteria above with the lowest annual volume of injectant for the assumed aquifer parameters at

the site. The analysis also took account of the quantity and sequence of flushing, orientation and magnitude of hydraulic gradients, uncertainties in aquifer properties, the proportion of water available for recovery, time-lag between injection and recovery, and the relative efficiency of two-, four- and alternative six- well arrangements. The results revealed that both the salinity and travel time constraints could be met with a 6-well system arranged within a quadrilateral domain with a uniform inter-well separation distance of 75 m (Pavelic et al., 2005).

HACCP Plan

The success of the ASTR project relies on several factors of which water quality is one of the most critical ones as it will have to meet Australian Drinking Water Quality Guidelines and community and government acceptance. A preliminary HACCP plan has been drafted for the purpose of assisting in the design of the ASTR project and the monitoring and control elements (Swierc et al., 2005).

The planned system for production and supply of potable water can be divided into four conceptual parts, as shown in Figure 4. These are summarized as follows.

- *Catchment* – The area where stormwater is captured and enters the stormwater drainage system. It is comprised of two subcatchments, the Parafield Drain and Ayfield Drain catchments which include urban and industrial zones. In the industrial zone, pollution control awareness is part of council and catchment water management board activities. This can be viewed at a barrier to pollution of stormwater at its source.
- *Stormwater Harvesting Facility* – Water is diverted from the stormwater drain into a pond, pumped to a holding pond and then gravity fed through a reedbed pond. The reedbed pond provides most of the treatment of the stormwater prior to ASR/ASTR.

- *Aquifer Storage Transfer and Recovery (ASTR)* – The ASTR comprises four injection wells surrounding two recovery wells. Water will initially be injected into the recovery wells to rapidly develop a fresh water plume surrounding them. After development of the fresh water plume, water will be pumped into the injection wells and pass through at least 75m of aquifer before emerging in the recovery wells. During passage the water quality improves, particularly with respect to pathogens or organics that may be present in injectant.

- *Potable Water Supply* – Initially the recovered water will be used to dilute the salinity in a recycled water supply to Mawson Lakes subdivision. If it can be demonstrated over a sustained period of time that the water complies with the Australian Drinking Water Guideline it will, subject to consents and community consultation, be supplement the existing drinking water supply through the established mains water distribution system.

Possible modelling approaches to assess the effectiveness of water treatment through the wetlands. However, for the working HACCP plan, models will need to be verified for their suitability in predicting pollutant attenuation and meeting water quality that complies with the Australian Drinking Water Guidelines. A monitoring plan to verify water quality at critical control points of the entire ASTR system will need to be developed. This plan will need to account for sampling and analysis during system startup and ongoing system operation.
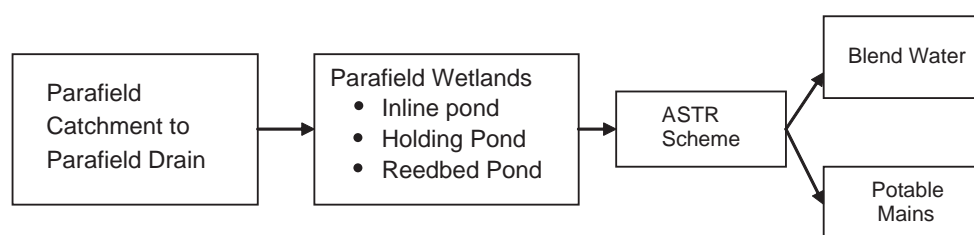
Figure 4. Project system flowchart

CONCLUSIONS

Parafield ASR scheme

During the first year of injection and recovery, the water quality objectives of the ASR scheme were met. Although some of the water quality parameters of the recovered water (extracted water from the aquifer) were slightly greater than the corresponding ambient groundwater parameters, they were still within the guideline values for Freshwater Aquatic Ecosystems and Irrigation. There was no evidence of leakage of injectant from the T2 aquifer into the T1 aquifer.

Proposed ASTR Scheme

By undertaking the attendant research on water quality changes, catchment water quality management, and development and implementation of the HACCP plans to provide water quality assurance, it is expected that a robust package of methods will be developed that will allow ASTR to be successfully adapted to other sites in Adelaide, and elsewhere in the developed and developing world. For country towns with expensive or low yielding supplies this approach could reduce costs for water utilities to between 10% and 50% of costs of current water systems.

If successful this would provide the evidence required for licensing these operations for water supply and provide guidelines for adaptation in developing countries to facilitate achievement of UN Millennium goals with reliance on natural treatment processes.

This project is intended to:

- demonstrate an economic new way of managing urban water resources,
- quantify disinfection and attenuation of contaminants in aquifers,
- create new expertise and knowledge,
- open an international opportunity for inexpensive water treatment and storage,
- save 400,000 m^3/yr of River Murray and Mt Lofty catchment water, and
- reduce stormwater discharge to Barker inlet.

The project can be undertaken under existing environmental and health regulations in South Australia, and it is anticipated that success could help to facilitate changes in environmental regulations in other states and countries to enable such projects to proceed.

While the idea of transforming urban stormwater into drinking water may sound far-fetched and subject to a high level of risk, there are a number of similar arrangements which indicate the concept is not only feasible but may provide a cost effective solution. Towns in the Mt Lofty Ranges discharge their stormwater to creeks and rivers in the catchment of the reservoirs feeding Adelaide. In Mount Gambier, urban stormwater has recharged an aquifer used as a drinking water supply for more than 120 years with no evidence of water quality decline (Vanderzalm et al 2004). In Europe where water supply is extracted from rivers, upstream towns discharge both their stormwater and treated wastewater into rivers. The city of Berlin doesn't extract water directly from rivers but via bores located close to the river bank. In doing so they use the natural filtration process of the bank. This concept is not dissimilar to that being proposed in the ASTR project.

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Review of effects of drilling
and completion methods of ASR wells
in unconsolidated aquifers

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Peter Dillon and Dominic McCann

Abstract

The influence of alternate drilling techniques, well completion designs, and redevelopment methods on the rate of clogging of conventional and ASR wells in unconsolidated formations is explored through a review of the literature. The few comparative case studies that were found revealed that: wells drilled with cable tool significantly outperformed reverse circulation rotary; using biodegradable mud gave rise to less clogging than when bentonite-based mud was used; residual mud on or in the vicinity of the borehole wall severely limited recharge capacity; and completion with wire wrapped screens and natural gravel pack gave significantly higher performance than wells with slotted casing and emplaced gravel pack. These conclusions demonstrate the importance of the choice of drilling technique, the quality of the drilling, well completion and design, in very significantly enhancing ASR well performance.

Research is needed to determine: geological conditions under which reverse-circulation rotary is more effective than rotary drilling; the effect of reaming following rotary drilling with biodegradable mud; the effectiveness of dual-wall reverse-circulation drilling; and to assess the performance and maintenance of horizontally drilled ASR wells in unconfined aquifers. There appears to be inadequate data to reliably predict life-cycle costs of ASR wells.

Keywords

ASR; completion; drilling; unconsolidated aquifers.

INTRODUCTION

It is generally known that clogging problems can be more severe for injection or ASR wells in unconsolidated aquifers than in consolidated aquifers. This is because the screen and gravel pack that are typically required in unconsolidated aquifers restrict the ability to redevelop the well once clogging has occurred.

The aim of this research presented here is to evaluate what is currently known about the effects of different drilling techniques, well completion designs, and redevelopment techniques on the performance of ASR wells in unconsolidated formations. The main issue is to avoid excessive clogging that leads to reduced injection rates and increased need to implement unclogging procedures. By understanding the impact of the various factors that affect the recharge capacity of ASR wells over the long term, the volumes of water stored and recovered can be maximized, and costs minimized. Owing to the sparse information on the effects of alternative completion arrangements for ASR wells, conventional water production wells were included in the review. Case studies were sought on: operational problems and how these have been resolved; comparative studies where different drilling methods or well completion techniques have been used and rates of clogging or well yield observed; and innovative borehole design techniques. Several researchers were contacted and invited to share their information and experience. This paper summarizes a recent report by Segalen et al. (2005).

COMPARATIVE STUDIES ON WELL CONSTRUCTION AND COMPLETION

Eight case studies where the effects of drilling and completion methods in unconsolidated formations on well performance during injection or extraction have been investigated. These cover an ASR well in Mannheim, Ontario (Wooton et al. 1997); 45 recharge wells and 25 recovery wells in the Berkheide dunes, Netherlands (Kortleve, 1998); 4 injection wells in the Amsterdam Dune Area, Netherlands (Van Duijvenbode and Olsthoorn, 2002); 35 abstraction wells in Bergambacht, Netherlands (Timmer et al. 2003); 20 infiltration wells and 12 abstraction wells in Castrican, Netherlands (Stakelbeek et al.1996); an injection well and a pumping well, DIZON pilot, Netherlands (Stuyfzand et al. 2002), 4 ASR wells and 4 recharge wells in Glendale, Arizona (Chaves et al. 2003); and an injection well and an extraction well in the Palo Alto Baylands, California (Sheahan, 1977).

Surprisingly few case studies were found to allow direct comparisons of drilling and completion methods. These are briefly described below.

Comparison of reverse-circulation rotary and cable tool drilling at Bergambacht, Netherlands

A study was conducted by Timmer et al. (2003) on the clogging of conventional wells that pump anaerobic groundwater from an aquifer composed of sandy, fluvial sediments. The well field is arranged in a straight line along a distance of 500–1,000 m adjacent to the River Rhine. The first set of wells was drilled using cable tool in 1968 and the second set was drilled using reverse-circulation rotary drilling in between the first ones. This study shows that the rate of decline in specific capacity for the 7 wells drilled using the reverse-circulation rotary technique is twice that of the 15 wells drilled using the cable tool technique. Although the initial specific capacity for the cable tool was only 3% higher than for rotary-drilled wells, it is thought that rotary drilling left a greater thickness of mud in the near well zone which acted to more efficiently trap colloids mobilized in the aquifer by pumping, thus reducing specific capacity more quickly.

Comparison of reamed and unreamed rotary drilled wells at Amsterdam, Netherlands

Amsterdam Water Supply and De Ruiter developed a new drilling method in 1993 for the construction of injection wells (Kortleve, 1998). This method consists in drilling to full-depth using standard rotary drilling and then introducing a slightly larger bit into the hole to scrape off the layer of sludge (remnants of the drilling mud used) from the wall. At the same time clean water is run down the hole. The water containing the scrapings in suspension is then recovered. This method was shown to be effective in drilling the 'Berkheide well recharge system' in the Netherlands. During the scraping, tThe recharge rate of the drill hole increased by a factor of 16 (from 25 m^3/h to 400 m^3/h) for an unchanged pressure differential (Kortleve, 1998). Therefore reaming the borehole enhanced the recharge rate by a factor of 16. This study shows that rotary drilling followed by borehole reaming may be a valuable technique to minimize clogging in ASR wells.

Comparison of well completions at Glendale Arrowhead Ranch Recharge Facility (Glendale, Arizona)

The Glendale project compares the performance of two ASR wells, situated closeby and targeting the same aquifer, drilled using the same method ('flooded' reverse-circulation), and completed with different well designs (Chaves et al. 2003). A 350 mm diameter stainless steel, wire-wrap, natural gravel-packed designed ASR well (type A) is compared to a 350 mm stainless steel, slotted casing, artificial gravel-packed designed well (type B). The study shows that the type A well outperforms the type B well with a four-fold higher specific capacity and a 30% higher recharge rate. The combination of a wire-wrap screen and a natural gravel pack (type A) seems to permit a higher recharge

rate and to maintain a higher recharge capacity after one season, than the combination of slotted casings and artificial gravel pack (type B).

Table 1 summarizes these comparative study results. Further studies are reported in Segalen et al. (2005) that suggest larger screen apertures result in less clogging, and wells drilled with polymer muds perform better than those drilled with bentonite muds.

Table 1. Relative effect on specific capacity of three comparative studies on drilling and completion methods

Methods compared	Rel. spec. capacity ratio	ASR wells	Reference
Reamed vs unreamed rotary drilled wells	16:1	Yes	Kortleve et al. (1998)
Wire screen and natural gravel pack vs slotted casing and emplaced pack	4:1	Yes	Chaves et al. (2003)
Cable tool vs reverse circulation rotary	2:1	No	Timmer et al. (2003)

CONCLUSIONS AND RECOMMENDATIONS

The influence of different drilling techniques, well completion designs and redevelopment methods to reduce operational problems for conventional and ASR wells completed in unconsolidated aquifers was studied. Only a limited number of case studies were found, however they provided valuable information that allowed the following conclusions to be drawn:

- well performance is strongly affected by both the drilling technique and the procedures employed during drilling;
- removal of residual mud from the borehole is required to minimize clogging problems and to maximize well efficiency – biodegradable muds are preferred over bentonite muds for this reason;
- ASR well performance can be significantly increased by careful consideration of the well completion and design, particularly with respect to the type of screen.

The knowledge gaps identified by this review suggest that field-based research may be useful to:

- compare reverse-circulation rotary and standard rotary drilling;
- determine the effect of reaming using rotary drilling with biodegradable mud;
- determine the effectiveness of dual-wall reverse-circulation drilling;
- assess the effect of the aquifer characteristics (particularly hydraulic conductivity) on the effectiveness of the drilling technique and the completion of the well;
- determine the feasibility and effectiveness of horizontal drilling for ASR;
- evaluate protocols for well redevelopment to optimise cost effectiveness of ASR well performance.

Without further investigations, reliable estimation of the final costs and life expectancy of ASR wells cannot be made.

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Water quality changes during Aquifer Storage and Recovery (ASR): results from pilot Herten (Netherlands), and their implications for modeling

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Abstract

ASR (Aquifer Storage and Recovery) is a water management technique to store surplus water in an aquifer, for use during future needs, using dual purpose wells (that both inject and recollect). The quality changes of drinking water during a recent ASR test in a deep anoxic, sandy aquifer in the Southern Netherlands were dictated by (redox) reactions with organic material, pyrite and manganous siderite. These reactions decreased during successive ASR-cycles by leaching, coating with iron(hydr)oxides and increasing pH.

Tests with adding O_2 and $NaNO_3$ to the injection water, to speed up aquifer inactivation, failed probably by not adding also a pH buffer. An anoxic zone developed around the ASR bore hole wall during storage, by decay of microbes grown during injection.

Iron and manganese, which dissolved more remote from the ASR well, immobilized in the most aerobic zone during recovery. This is explained by sorption to iron(hydr)oxides that precipitated during previous injection and storage phases. As a result, with increasing ASR cycles the recovery efficiency (dictated by iron removal) improved.

The water quality changes have been modelled with an ASR version of the Easy-Leacher model, incorporating the crucial changes in the ASR-proximal zone and the effects of sorption to iron(hydr)oxides.

Keywords

Aquifer storage and recovery, ASR, hydrochemistry, pyrite oxidation, siderite dissolution, modeling.

INTRODUCTION

Aquifer Storage and Recovery (ASR) is a water management technique, in which water is stored in an aquifer during periods of water excess, and recovered by the same (injection) well from the groundwater reservoir during periods of water shortage (Pyne, 1995). ASR may also yield financial benefits by reducing the peak factor in water production (and water treatment) and by raising the security of water delivery.

ASR is an increasingly popular technique with a rapidly expanding number of operational sites in the world. In the Netherlands, however, notwithstanding large-scale application of artificial recharge since 1955, no ASR-system is currently working. This failure relates to the following: (a) decreased water demands in the past years do not urge for increasing subsurface storage of drinking water in the 'wet' Netherlands; and (b) it is feared that a quality deterioration of (drinking) water during subsurface storage, especially regarding Fe and Mn, will necessitate a costly post-treatment of the water recovered.

Recent bench marks of drinking water prices in the Netherlands push water supply companies to lower their costs. ASR, if operating without quality deterioration, is thus becoming an interesting option also in the Netherlands, mainly to reduce water treatment costs by peak shaving.

In 1999 Water Supply Company Limburg (WML) therefore started with an ASR-experiment in Herten (Limburg), to test the technical feasibility of ASR, not only for that site but also for a larger scale application in the province of Limburg (notably the Ruhr Valley graben). The main questions to be answered were: (1) are there hydrological constraints associated with ASR in this region? and (2) are the water quality changes in the target aquifer, of drinking water from production plant Herten, such that they really necessitate a second treatment? This paper is based on a report by Wakker et al. (2003).

MATERIAL AND METHODS

The ASR test well and 2 observation wells are shown in cross section in Fig. 1. All available well screens (10) have been sampled on a bidaily to biweekly basis for inorganic and microbiological chemical analysis, using conventional purging and analytical procedures. In total 10 aquifer cores of 1 m length were taken from the whole target aquifer (159–170 m below land surface) and were analysed on their geochemical reactivity (Buijs and Van der Grift, 2001). The infiltration and pumping rate was on average 45 m^3/h.
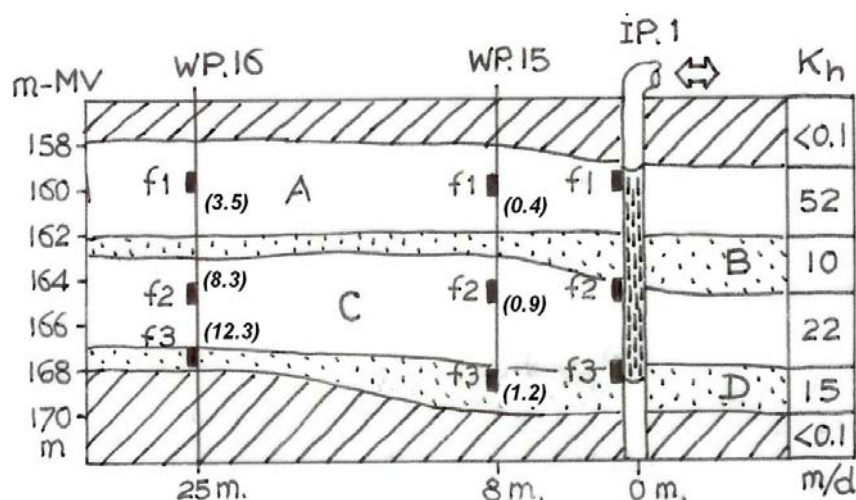
Figure 1. Position of the ASR-well IP1 and the 2 observation wells WP15 and WP16 in cross section, with the hydrogeological structure, coding of aquifer layers (A-D), position of well screens (f1-f3= observation well screens), and mean travel time from/to IP1 (within brackets, in days). K_h = horizontal hydraulic conductivity.

The ASR-experiment in Herten lasted from October 2000 till February 2003. It consisted of 4 ASR cycles (Fig. 2), of which 2 cycles were carried out with additional dosage of oxidant in order to inactivate reactive aquifer phases (notably minerals containing iron and manganese) with the purpose to recover drinking water that would not require any post-treatment. The ASR-cycles were not in normal order (being injection, storage, recovery etc.), for various testing purposes. Results of the test have been simulated and extrapolated to future ASR cycles on a production plant, by using the expert model EL-ASR of Kiwa Water Research, a new variant of Easy-Leacher (Stuyfzand, 1998).
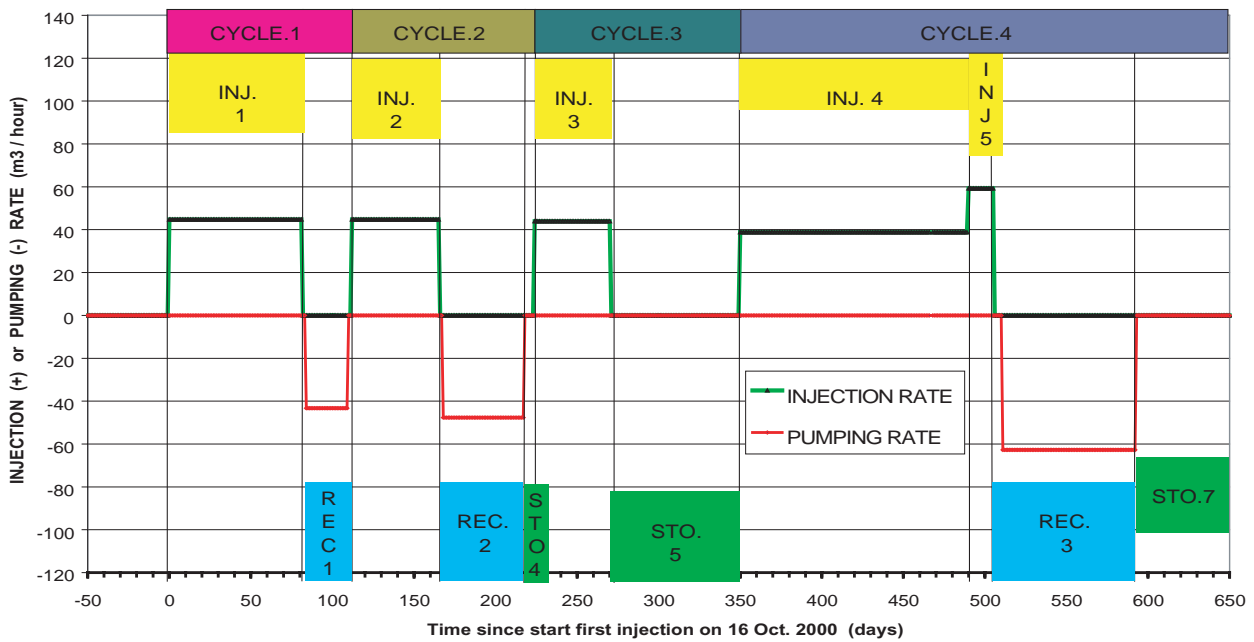
Figure 2. Variation in injection and pumping rate during the discerned periods and ASR cycles. Oxygen was added during injection period 3 (34 mg/L; dissolved thanks to >17 bar water pressure at depth of injection screen), and $NaNO_3$ during injection period 5 (14.6 mg NO_3/L). Storage periods 1-3 too short for marking.

THE TARGET AQUIFER

The target aquifer is composed of fluvial sands of Pliocene age. The grain size distribution, geochemical analyses and reactivity experiments (in an O_2 and CO_2 controled reaction chamber) revealed the presence of 4 aquifer layers (Fig. 1). All layers are deep anoxic, practically without $CaCO_3$, and contain small but significant amounts of reactive BOM (bulk organic material; 0.02–8%) and pyrite (210–1,050 mg/kg d.w.). Only layer D contains a significant amount of reactive manganous siderite (<0.25%).

The native groundwater composition (Table 1) can be characterized as pH-neutral, calcareous (slightly calcite undersaturated), deep anoxic (however without methane), oligohaline-fresh, high in iron, unpolluted, and of the $Ca(HCO_3)_2$– type.

CLOGGING OF THE ASR-WELL

The ASR-well in Herten did not clog (Fig. 3). Especially during the additional oxygen dosage there was a temporary increase in resistance at the well screen (0.2–0.8 m; probably due to gas bubbles), but this disappeared again during the next storage phase. During storage and recovery the ASR-well regenerated spontaneously. The excellent infiltrability of Herten drinking water was to be expected, because it satisfies all general quality guidelines of infiltration water for ASR application to sandy aquifers (see for instance Perez-Paricio and Carrera, 1998; Hijnen et al., 1998).

WATER QUALITY CHANGES IN THE TARGET AQUIFER
Mean changes

The test results demonstrate that the quality of the injected drinking water changed indeed (Table 1). There were (small) increases for sulphate, iron, manganese, arsenic, barium, cobalt, nickel, zinc and colony counts 22ºC, and

Table 1. Mean water quality during the 3 ASR cycles in the period 16 October 2000 till 11 June 2002.
       IP.1-f0 = ASR injection/pumping well screen.

SAMPLE POINT	t50 or daynr. day	pH field	O2-field	Cl	SO4	HCO3	NO3	Ca	Fe	Mn	NH4	As	Co
						mg/L						g/L	
INJECTION (mean input, for each monitoring well samples with comparable number of pore flushes)													
INPUT	0	7.47	12.0	22.5	26.0	220	1.1	73.0	0.01	0.01		0.5	0.5
WP 15-f1	0.4	7.31	6.7	21.0	28.8	213	1.0	71.3	0.01	0.03	0.03	2.5	6.0
WP 15-f2	0.8	7.38	8.1	21.3	31.8	224	1.2	74.0	0.01	0.01	0.03	2.8	8.5
WP 15-f3	1.2	7.22	5.2	18.3	26.3	210	10.0	70.0	0.07	0.02	0.03	1.5	0.5
WP 16-f1	3.5	7.11	0.5	21.0	36.0	201	0.3	70.0	1.27	0.04	0.03	2.8	0.5
WP 16-f2	8.3	7.11	2.5	21.3	34.4	209	0.4	70.9	0.46	0.07	0.03	1.8	5.0
WP 16-f3	12.1	7.05	1.0	21.0	28.0	205	1.7	64.0	2.30	0.11	0.03	1.5	0.5
STORAGE (mean all periods)													
IP.1-f1	58	7.11	0.6	21.8	40.7	215	0.5	77.0	0.64	0.07	0.07	7.0	4.1
IP.1-f2	98	7.10	0.8	21.5	45.0	220	0.4	84.5	0.94	0.17	0.10	3.0	14.0
IP.1-f3	58	7.02	0.8	21.5	40.1	226	0.4	84.1	0.11	0.14	0.04	1.9	1.3
WP 15-f1	59	7.09	0.9	20.8	38.0	208	0.4	71.4	0.05	0.06	0.03	2.9	6.1
WP 15-f2	29	7.00	0.5	22.5	39.8	216	0.4	74.1	0.01	0.09	0.03	3.4	26.3
WP 15-f3	59	6.88	0.3	21.7	39.7	225	0.3	76.0	1.03	0.40	0.03	1.9	7.2
WP 16-f1	58	7.00	0.5	21.9	44.4	200	0.3	71.2	0.81	0.08	0.03	4.8	5.8
WP 16-f2	29	6.82	0.8	21.5	41.2	206	0.3	70.0	1.55	0.20	0.03	4.8	8.3
WP 16-f3	58	6.91	0.3	22.1	39.4	205	0.3	67.0	4.21	0.22	0.07	3.1	0.6
RECOVERY (mean all periods)													
IP.1-f0	20	7.16	0.8	20.7	33.0				0.01	0.02			
IP.1-f1	18	7.18	1.4	20.9	37.5	208	1.1	70.7	0.02	0.01	0.03		
IP.1-f2	21	7.18	1.2	20.8	31.4	211	0.8	69.4	0.02	0.02	0.03		
IP.1-f3	18	7.11	1.9	21.4	31.5	221	0.5	71.2	0.01	0.02	0.03		
WP 15-f1	18	7.18	1.1	20.7	35.1	211	1.4	70.6	0.02	0.03	0.03		
WP 15-f2	17	7.24	2.0	20.9	31.3	212	1.9	69.1	0.01	0.02	0.03		
WP 15-f3	18	7.07	0.6	20.9	29.9	213	0.5	68.6	0.57	0.10	0.03		
WP 16-f1	22	7.16	0.4	19.7	30.9	204	0.6	69.0	1.04	0.05	0.03		
WP 16-f2	17	7.14	0.5	19.6	25.5	210	0.3	67.3	2.25	0.10	0.04		
WP 16-f3	22	7.02	0.4	19.7	27.1	202	0.3	64.0	3.36	0.16	0.06		
NATIVE GROUNDWATER													
IP.1-f0	0	7.00	<0.5	7	<5	300	<0.5	72.0	5.60	0.20	0.14	12.0	4.0
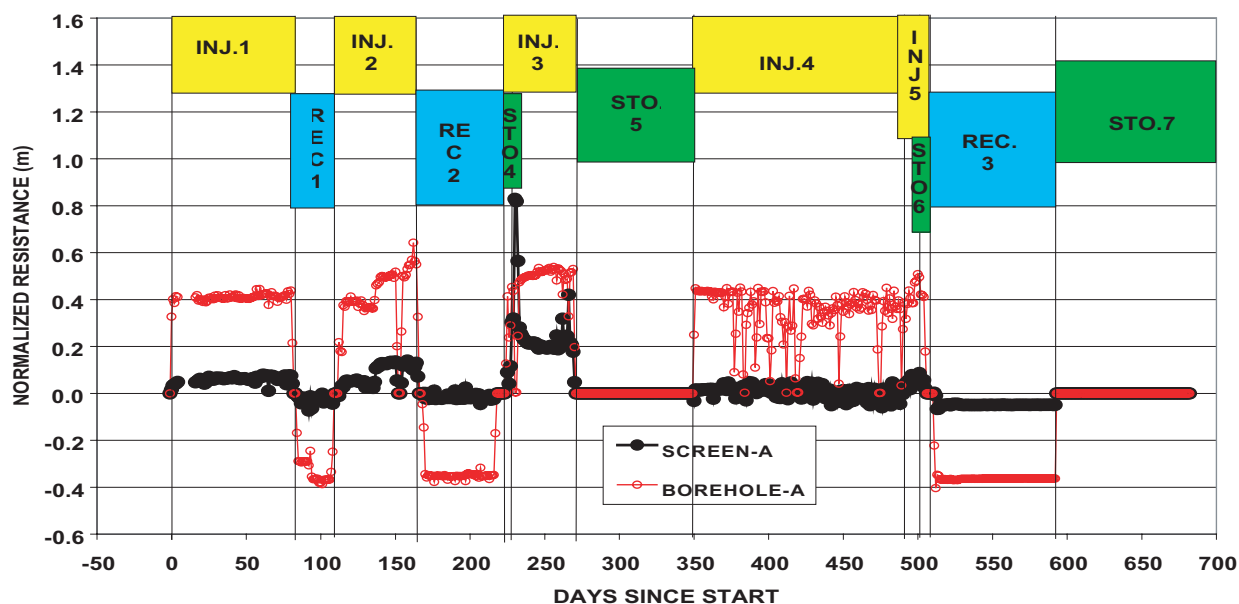
Figure 3. Time plot for the normalized resistance by the ASR well screen (Screen) and the borehole wall (Borehole)
in aquifer layer A. The curves do not show a structural head build-up during injection (INJ) and backpumping (REC),
thus showing that the ASR well did not clog. Both storage (STO) and recovery contribute to the reduction
of the hydraulic resistance formed during injection.

decreases for oxygen, nitrate, silicate, pH, hydrogencarbonate and the calcite saturation index. Ammonia and methane did not show significant changes (< 0.05 mg/L), nor did organic micropollutants (all below detection) and indicators of faecal contamination (not found).

The observed changes are mainly caused by the oxidation of BOM and pyrite (FeS_2), and the oxidation or dissolution of a manganous siderite ($Fe_{0.90-0.95}Mn_{0.05-0.1}CO_3$). In the upper aquifer layer (A) pyrite oxidation is the maior reaction (largest increases of sulphate, arsenic, cobalt, nickel and zinc), and in the lowest layer (D) the weathering of siderite is dominating (iron, manganese and TIC increasing most).

The results in Table 1 reveal that the mean quality changes are stronger during storage than during recovery. This is curious, but can be explained as follows. First, the longer storage phases 4 and 7 were preceded by recovery periods, and thus reflect water that remained in the aquifer for a longer time than the water previously recovered. And second, the other long storage phase 5 followed upon the exceptional injection phase 3 with additional O_2 inputs (34 instead of 9.5 mg/L), and was not followed by a recovery phase.

Development of an anoxic zone around the borehole wall

A remarkable phenomenon was the development of a more anoxic zone of several meters around the ASR well during storage (Fig. 4), probably indicating the die-off of a microbiological community that formed there under
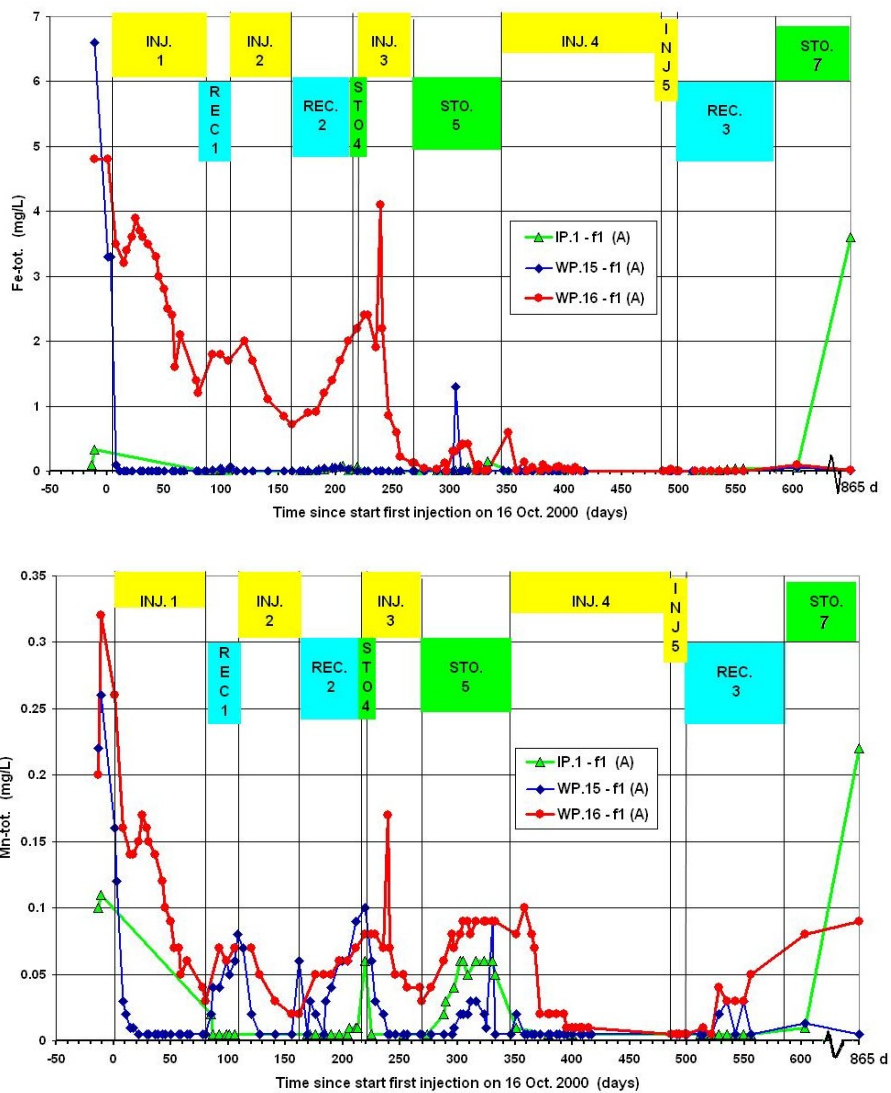
Figure 4. Plot of the iron and manganese concentrations in upper aquifer zone A, in observation screen f1 of the ASR-well and WP.15 (8 m, 0.4–1.2 days) and WP.16 (25 m, 3.5–12 days). Periods with injection (INJ), storage (STO) and recovery (REC) are numbered as in Fig. 2

(sub)oxic conditions during injection (Stuyfzand et al., 2002; Vanderzalm et al., 2002). This is deduced amongst others from the stronger Mn and Fe mobilisation in the observation screens within the ASR-well (amongst others IP.1-f1) during storage phase 5 and especially 7, as compared to observation wells WP.15 and 16 at resp. 8 and 25 m distance.

Subsurface Iron Removal (SIR)

Problems with iron and manganese in the water recovered arose during low O_2–concentrations (<0.5 mg/L) and low pH (<7.1) in the aquifer, especially during the first few ASR cycles. During the recovery phase, after starting up the system, the so-called Subsurface Iron Removal process (Van Beek, 1983; Appelo et al., 1999) strongly reduced the concentrations of iron (Fig. 5) and, to a less extent, those of manganese and ammonia.
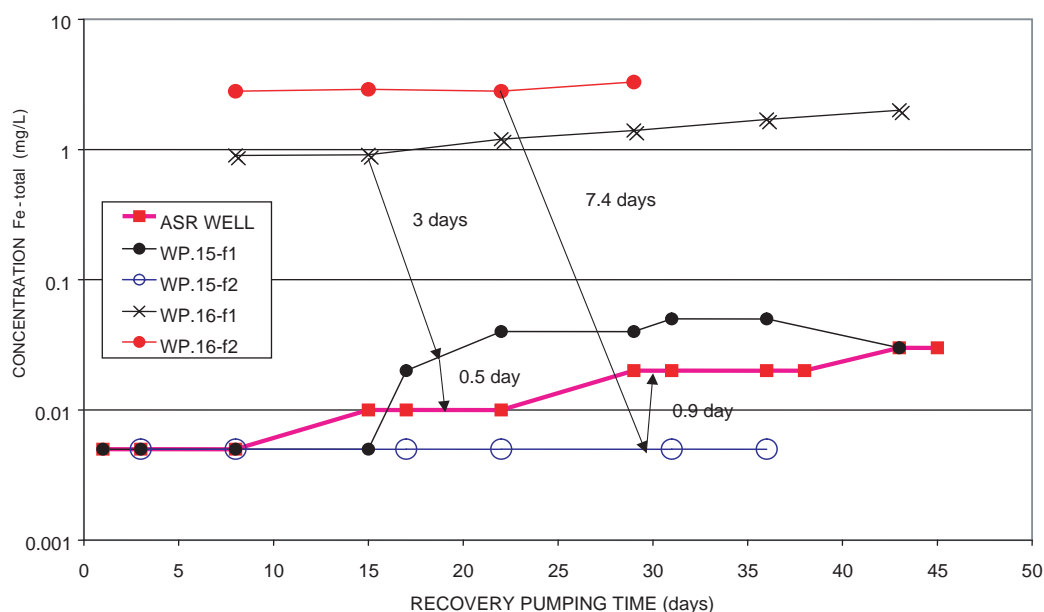
Figure 5. Developments in the iron concentration during recovery period 2 (after 2 days of storage following injection period 2), in the ASR-well (the water recovered), and on 4 sites in the aquifer. The arrows indicate how the iron concentration is changing, within the indicated time (by sorption), during backflow from WP16 (25m) via WP15 (8m) to the ASR well

Extinction of water quality changes

In accordance with experiences elsewhere (Pyne, 1995), also here the reactions of the infiltration water with reactive aquifer components gradually diminished (with increasing cumulative injection volume). Consequently the iron and manganese concentrations decreased (Figs. 4 and 6). This is explained by: (1) leaching of pyrite and siderite; (2) coating of them by iron(hydr)oxides, which reduces their reactivity; and (3) a pH increase in the aquifer thanks to the extinction of the above mentioned oxidation reactions.

During recovery, in addition, the efficiency of the SIR process increases (Van Beek, 1983; Appelo et al., 1999), which leads to further declining concentrations of iron, manganese and ammonia.
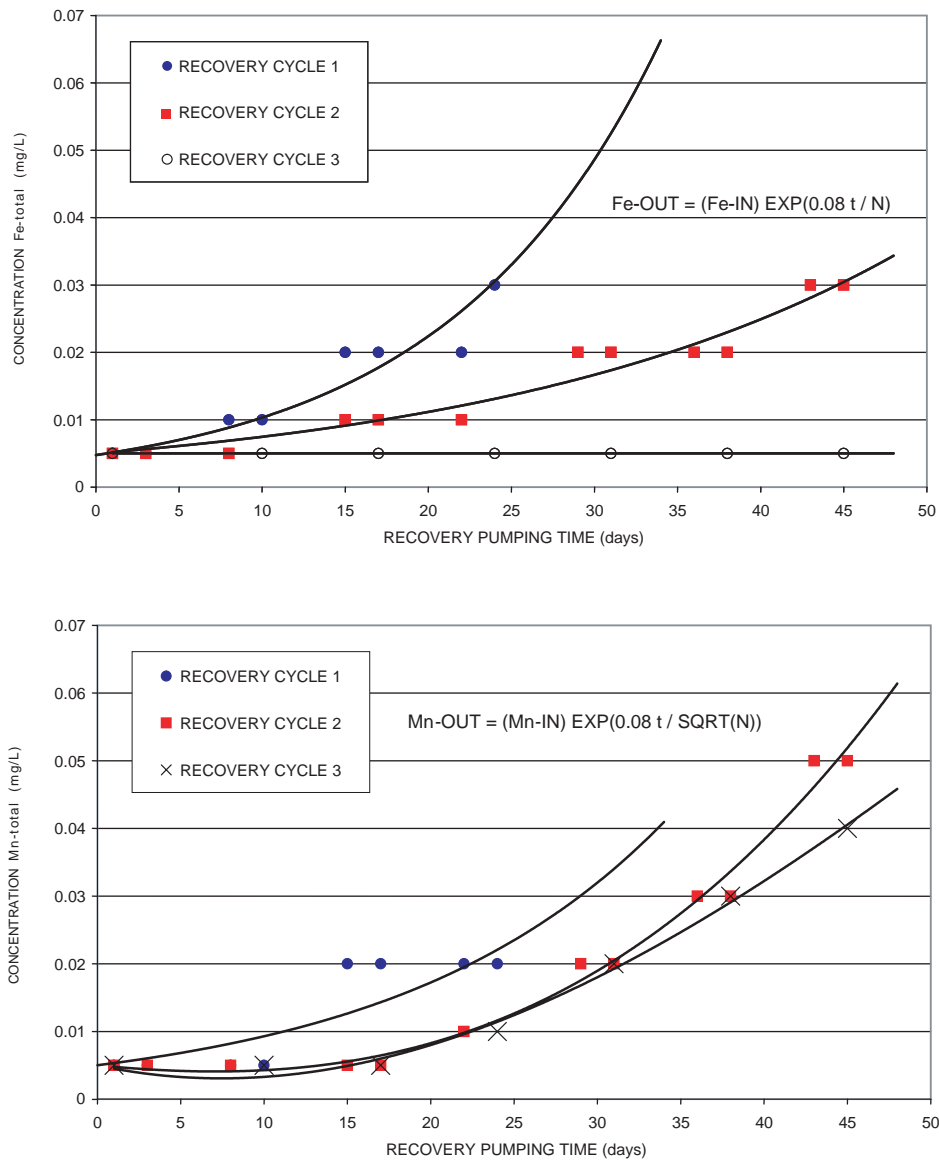
Figure 6. Plot of concentrations of iron (above) and manganese (below) in the backpumped water during recovery cycles N = 1-3 of the Herten ASR test

EFFECTS OF DOSING CHEMICALS

Current experiences

In current ASR practice oxidizing, desinfecting and pH-raising chemicals are dosed to the infiltration water, if needed. The success of ASR in the USA can, as a matter of fact, be partly attributed to the dosage of sodium hydroxide and a desinfectant.

These additives are useful in: (1) preventing exorbitant growth of bacteria; (2) coating the reactive iron minerals with iron(hydr)oxides, (3) keeping the aquifer (sub)oxic for a longer time, and (4) buffering the acidifying action of oxidation reactions. All this may indeed prevent the dissolution of Fe^{2+}, Mn^{2+} and sometimes arsenic, as well as microbiological problems.

Oxygen (tested in Herten)

The positive effects of O_2-dosage (on average 25 mg/L) to nearly saturated infiltration water (9.5 mg/L), during injection period 3, were overshadowed during storage phase 5 by a strong pH decline due to enhanced oxidation processes. This pH drop caused manganese problems during the subsequent storage period. The pH should be prevented to drop below 7-7.2, by adding NaOH.

Nitrate (tested in Herten)

The dosage of nitrate (0.22 mmol $NaNO_3$/L) to the infiltration water (with only 0.02 mmol NO_3/L), during injection period 5, contributed to an accelerated oxidation of BOM and pyrite, with 75% less production of acid (advantageous) and a slower consumption rate (also advantageous because nitrate that way keeps the water (sub)oxic for a longer time than oxygen does). However, from an esthetic point of view the addition of oxygen is to be strongly preferred.

Sodium hydroxide (not tested in Herten)

The acidifying effects, in consequence of adding oxygen, can be easily buffered by also addin NaOH or another pH raising agent.

Desinfectant (not tested in Herten)

In the USA chlorine or hypochlorite is dosed for preventing the development of a dense, microbial community around the borehole wall, because microbes both clog the well and consume O_2 and NO_3. In the Netherlands, these oxidants have been largely abandoned due to the formation of hazardous byproducts. Therefore alternatives are needed, for instance consisting of intermittent (1) backpumping at a high flow rate after juttering with compressed air, or (2) injection of water containing ozone or H_2O_2, followed by the actions mentioned under 1.

MODELING

The ASR testing did not reflect the normal ASR operation in a year, with for instance 150 days of injection, followed by 60 days of storage, 120 days of recovery and 35 days of stand-still. This means that simulations and extrapolations with a transport model were needed, in order to predict the performance of a true ASR plant.

Various models are nowadays available (amongst others PHT3D of Prommer et al., 2003; SWIFT-PHREEQC of Gauss et al., 2000) that could do the job, however, after various extensions to account for: (a) development of an anoxic zone around the borehole wall; (b) the process of subsurface iron removal; and (c) extinction of water quality changes.

A less laborious route was followed by transforming the expert model Easy-Leacher (Stuyfzand, 1998) into a transport code fit for ASR (EL-ASR 2.0). It is beyond the scope of this paper to describe this model, which accounts for the above mentioned problems a-c.

It follows from from Fig. 7 and 8 that, without adding oxygen and NaOH, iron problems (Fe > 0.2 mg/L) disappear after 8 cycles, and that the first few m^3 of water recovered always need to be disposed of.
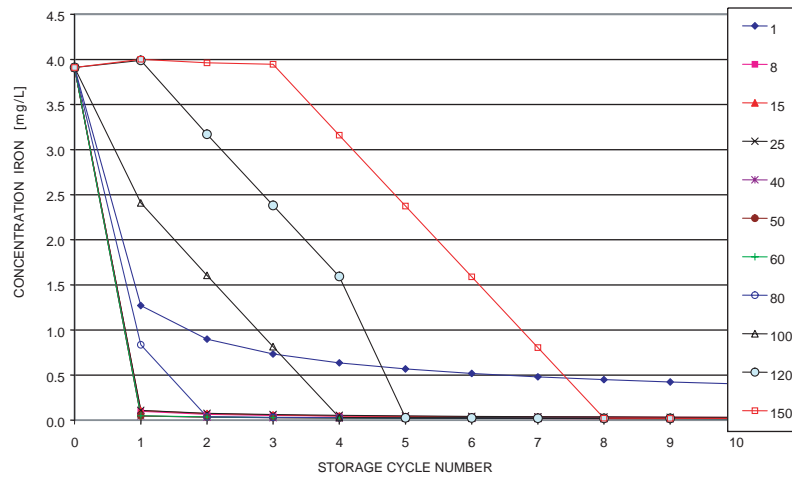
Figure 7. The with EL-ASR 2.0 calculated Fe-concentration at 1-150 m distance from the ASR well in aquifer zone A at the end of each storage period of ASR-cycles 1-10. Each cycle consists of 150 days of injection, 60 days of storage, 120 days of recovery and 35 days of stand-still.
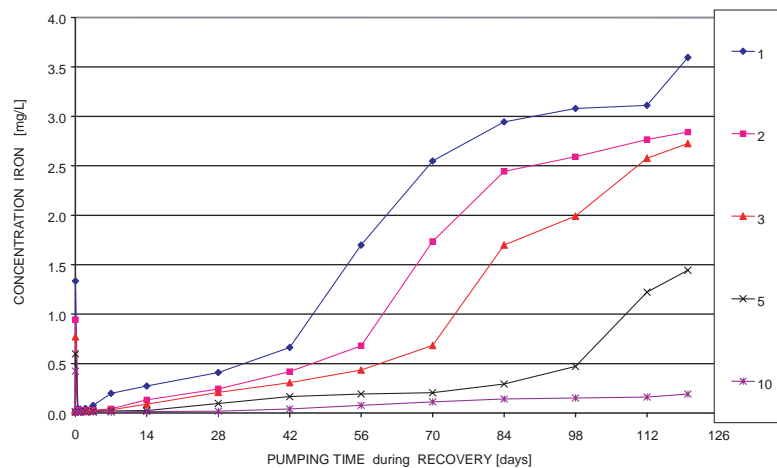
Figure 8. The with EL-ASR 2.0 calculated Fe-concentration in the water recovered by the ASR well during cycles 1-3, 5 and 10. Same cycle program as in Fig. 7.

CONCLUSIONS

ASR on the Herten site proved to be hydrologically feasible, because the water could be injected and recovered with a sufficiently high rate without clogging problems, and there was no evidence of a cumbersome mixing with native groundwater (using Cl as a tracer).

The relatively small quality changes of Herten drinking water in the aquifer studied, pose a problem especially regarding the increases for iron and manganese (drinking water guidelines resp. 0.2 and 0.05 mg/L). The reason is that these changes may require a simple post-treatment. However, the mentioned deviations from the drinking water standards are small, and can be easily cured by (a) dilution with conventionally treated groundwater, or (b) a short aeration followed by rapid sand filtration.

On the other hand, the rather small quality problems can be easily prevented by 2 measures: (i) the addition of oxygen and NaOH during the initial injection periods; and (ii) periodical dosage of a desinfectant like O_3 and/or H_2O_2 and backpumping of it after juttering.

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Proposed scheme for a natural soil treatment system in Kuwait

Mohamed Al-Senafy and Mohsen Sherif

Abstract

A pilot scale project using soil aquifer treatment (SAT) was constructed in the Sulaibyia area southwest of Kuwait City to assess the treatment efficiency and the suitability of the site. The study included seven recharge basins with a depth of 2 m and a total infiltration area of 900 m^2. Seventeen monitoring wells were installed in the vicinity of the recharge ponds to monitor the water levels below the recharged basins and to collect water samples for chemical and biological analysis. Several difficulties were encountered during the operation of the pilot study. These were mainly attributed to the impermeable nature of the soil and the high evaporation rates. Based on this, it was recommended to implement a large scale SAT project in northern Kuwait taking into account the maximization of the recharge/recovery efficiency and minimizing the loss of water due to evaporation. The recommended location to implement the first stage of this large-scale SAT should have a suitable subsurface lithology for basin recharge and enough unsaturated thickness for proper soil aquifer treatment in addition to the continuous source of treated wastewater. The proposed scheme should include 12 recharge basins with a side slope 1:1 and a total base area of 4,800 m^2. The basins will be classified into four groups' base on the difference in their bed elevations. Each 4 basins will be lowered by 30 cm from the other group allowing the flow of the water under gravity conditions. The system will be operated in short alternative cycles of wetting and drying (three days), and proceed only during the period of October–May.

Keywords

Basins; recharge; recovery; wastewater.

INTRODUCTION

In an arid country like Kuwait with no natural storages and limited groundwater, artificial recharge can play a major role in the management of water resources. According to previous studies conducted in Kuwait, three different types of storage were identified. These are seasonal, long term and crisis storage. Artificial recharge and storage in deep aquifers is ideal for seasonal storage. For long term and crisis conditions, storage as fresh water lenses will be ideal (Viswanathan and Al-Senafy, 1998). Given the limited available water resources and the increasing demand, municipal wastewater can play a special role in maintaining and increasing the water resources of Kuwait, especially since the wastewater effluent is estimated to be 70 to 80% of the freshwater consumption per capita (Al-Awadi et al. 1992). Treated wastewater effluents are free from health hazards and must be considered as valuable water resources for irrigation of certain crops, greening enhancement, landscaping, land reclamation, car washing, industrial process, and toilet flushing. This can be especially effective and feasible where the supply depends on desalinatedwater, as the resultant wastewater generally tends to be low in dissolved solids. Considerable amounts of treated wastewater are produced annually in Kuwait. Most of this water is treated at the tertiary level. The remaining portion of the produced wastewater is treated at the secondary level only due to the limited capacity of the tertiary treatment facilities. The total treated wastewater in 1999 was estimated at 404,060 m^3/d (147.5 million m^3/y). By the year 2015 the available wastewater is expected to reach 230 million m^3/y (MPW, 1999). The treated wastewater after SAT would contribute to the national water resources and alleviate the burden on other resources like groundwater and desalination water (Al-Senafy and Al-Otaibi, 2002). The aim of this paper is to fulfill this recommendation.

The treated wastewater is more suitable than brackish groundwater for cultivation. This is attributed to its' lower salt content. It also contains some important fertilizers that are essential for vegetation such as nitrogen, phosphorous and potassium. This wastewater is also a good source for the different elements needed by vegetation and are not available in freshwater. Therefore, the use of treated wastewater could be more suitable than both brackish water and freshwater for irrigation practices.

The total current consumption of wastewater is estimated at 58.6 million m^3/yr (MPW, 1999) and therefore, the excess treated wastewater that is available for utilization is estimated as 88.9 million m^3/yr. On an average, only 32.5% of the available wastewater is utilized and the rest is discharged into the sea.

In principle, the tertiary treated wastewater should be used directly wherever and whenever possible for irrigation of all green areas in the public domain and along highways. It can also be used for restricted irrigation of trees, some crops and vegetables that are not eaten uncooked. This will save considerable amounts of fresh or brackish water that would have been used otherwise. However, wastewater consumption in farm irrigation and other greenery projects differs considerably between summer and winter. Based on the available data (MPW, 1999), wastewater consumption is maximum during June through September. For the rest of the year (October through May), the consumption is reduced by about 30%.

For ultimate benefit from the management point of view, it is proposed to fully utilize the wastewater during peak time (June to September) directly from irrigation application to reduce the demands on desalinated water and groundwater. Otherwise (October through May), the excess wastewater can be used for a artificial recharge of depleted aquifers.

A PILOT SAT STUDY IN KUWAIT

Water resources in Kuwait are confined to three main sources: desalination water, groundwater and wastewater. These three sources are bounded by the high costs of the desalination plants and their vulnerability to damage, limited availability of fresh groundwater and depletion of aquifers, low quality of wastewater and unsuitability for direct use. The quality of the groundwater in Kuwait varies from brackish to saline except two freshwater lenses. Increased use of this brackish groundwater for irrigation, cause the deterioration of groundwater quality and the lowering of water table.

The full utilization of wastewater represents a promising option to alleviate the water shortage problem and sustain the depleting aquifers in Kuwait. Due to the expected increase in population along with the increase in the per capita water consumption, wastewater will become more available with time.

Soil aquifer treatment SAT is a modern approach to an old method of wastewater treatment and reuse. Besides its reliability with respect to the effluent purification, it can also fulfill the function of seasonal and multi-annual water storage. The SAT system uses complex physico-chemical and biological processes occurring in the unsaturated zone and in the aquifer to improve the wastewater quality (Viswanathan, 1997).

The artificial recharge of wastewater into aquifers using SAT technique should be widely implemented to ensure the full utilization of all the available wastewater. SAT has been applied in many countries around the globe. The quality of the wastewater can be improves significantly to meet the requirements of potable water uses and unrestricted irrigation of all kinds of crops, vegetables and fruits. Meanwhile, the reuse of wastewater would reduce the demands on both desalinated water and fresh groundwater. It will also help to restore the water levels in the depleted aquifers and conserve the natural water resources of the country.

A pilot scale SAT project has been constructed in the Sulaibyia area located about 50 km southwest of Kuwait City in Kuwait to assess the suitability of the site, the implemented recharge system and the adapted techniques. The project included seven recharge basins with a depth of 2 m and a total infiltration area of 900 m^2. Seventeen monitoring wells were installed in the vicinity of the recharge ponds to monitor the water mounds below the basins and collect water samples for chemical analysis. Three of the monitoring wells have a depth of 100 m, whereas all others have a nominal depth of about 50 m. The recharge basins were connected by a pipe network to convey the treated wastewater to the different basins according to the operation schedule. Measuring and controlling devices including flow meters, valves and evaporation pan were fixed to record the flow to the basins and measure the evaporation rates throughout the year. The project has been in operation all the days for more than three years with the exception of stopping periods for drying and maintenance. The water table rise was measured on a monthly basis and the evaporation rates were measured daily. The total flow to the entire system and the discharge of the tertiary treated wastewater to the different basins were measured weekly. Water samples were collected from the 17 monitoring wells every month and the samples were analyzed to determine the variation in the quality of the recharged water after SAT. A total of 16,790 m^3 of tertiary treated wastewater was released to the basins. An unsymmetrical water mound with a height of 1.9 m was formed below the infiltration basins. It is estimated that a total of about 8,000 m^3 of water has reached the groundwater. The quality of the recharged water has improved significantly and hence the pumped water can be used unrestrictedly in irrigation and other potable uses.

RECOMMENDATIONS OF THE PILOT STUDY

Many problems were encountered during the operation of the pilot scale SAT project. These problems were mainly attributed to the unsuitable lithology of the selected sites, operation schedule, and big depth of the pond and high evaporation rates.

Based on the gained experiences from the implementation and operation of the first SAT project in Kuwait this pilot project, it is recommended to implement a large scale SAT project in two phases to ensure the efficiency of both the recharged and recovery process and minimize the loss of water. In this context, the following recommendations were made:

1. The recharge basins should be shallow and designed in such a manner to allow for the transference of the recharged water from one basin to the other under gravity and without pumps, as much as possible, to reduce the operation costs.
2. Shorter operation cycles (flooding/drying) should be implemented to allow for the proper restoration of the infiltration rates and, hence, increase the average annual infiltration rate (cycles should not exceed one week or so).
3. The operation of the entire system should be terminated during the summer season (June, July, August, and September). During this period, the evaporation rates are very high and more than 16 mm/d (Al-Senafy, 2001).
4. The recovery of the freshwater lens can be practiced during the summer (period of peak demand). The pumped water can, therefore, be integrated into the water distribution system and supplement the water shortage.
5. The implementation of a large-scale wastewater recharge/SAT project to recharge a total of about 30000 m3/yr should be implemented in two phases.

SITE SELECTION CRITERIA FOR A LARGE SCALE SAT PROJECT

A new site should be selected in northern Kuwait to implement the first stage of the large-scale SAT project. The selected sites should satisfy the following criteria.

1. Availability of land to construct a large-scale Sat project. The total area required for the recharge of about 30 million m^3/yr is about 1.8 km^2, including the area required for pipe lines, other facilities and maintenance works. This area is estimated based on an average infiltration rate of about 16 cm/d (MEW, 1977) and assuming that each

pond will be operated only 50% of the time. Based on the actual infiltration rate that will be measured during the first stage of the project, the total area required can be revised and reduced accordingly.

2. The subsurface lithology should be suitable for basin recharge, i.e., no clay layers or any other low permeable strata are encountered between the basin bed and the subsurface water table.

3. The thickness of the unsaturated layer should be enough to allow for the proper function of Sat processes. On the other hand, this thickness should also be enough to accommodate the expected rise in the water table due to water mounding below the recharge basins. It is therefore, recommended to select a site with a water table depth of about 20 m or more below the ground surface.

4. Availability of continuous source of treated wastewater.

5. It is also recommended that the selected site be located near an area where water demands during the summer period (June through September) can be supplemented by the covered groundwater.

DESIGN OF THE LARGE SCALE SAT RECHARGE FACILITY

In the first stage of the large-scale SAT project, it is proposed to construct 12 recharge basins, each with a base area of 20 x 20 m and side slope of 1:1. The recharge basins are classified into four different groups: A, B, C, and D. The bed level of the basins are lowered by 30 cm from one group to the other, allowing for the dull drainage of the basins of group A to the basins of group B. Likewise, the basins of group B can be drained into the basins of group C and those in group C can be drained into the basins of group D. The difference in the bed elevation between the basins of group A and group D is 90 cm. The drainage of one basin to the other is controlled by valves installed on pipes connecting the different basins. The flow will be under gravity and no pumps are needed for water transformation from one basin to the other.

The depth of all the basins in group A is 30 cm. This depth increases from one group to the other by the same value. Therefore, the applied water depth can be increased in group D up to 1.2 m. This will allow for proper testing by changing the water depth to define the optimum hydraulic depth that would result in maximum infiltration rate.

The system will be operated in short cycles (two or three days) to allow for proper drying and full restoration of the infiltration rate. The three basins in group a will be filled with the treated wastewater first for three days, after which, the valves between the basins of group A and group B will be opened to drain the A basins and fill the B basins. However, for drying of the D type basins, pumps should be used and the water could be directed to any of the other groups according to the operation schedule.

Basins of groups A and C can be operated simultaneously for three days, and then the water in these basins (in group A and C) is transferred to the other basins in group B and D, respectively under gravity. These basins will be active for the same period as for the other two groups (A and C). On day 7, the water in the basins of group B will be transferred to the basins of group C and the water in the basins of group D will be pumped to basins of group A. Under this operation schedule, all the basins will be operated 50 % of the time and inactive otherwise to allow for drying and decomposition of the organic matter that may clog the bottom of the basins. Other operation schedules can also be considered and tested to define the optimum one.

A set of pumping and monitoring wells should be designed based on the characteristics of the selected site and the depth to the water table. The recharge by treated wastewater should be conducted for a total period of 8 months every year (October through May). The pumping wells should be operated during the high water demand (June through September). The pumped water should be integrated into the water distribution system to supplement any water shortage during this high stress period.

Based on the result of the first stage, the full scale SAT project to recharge an approximate amount of 30 million m^3 of treated wastewater should be implemented. In this full scale project, the dimensions of the basins can be

increased to about 250 x 250 m. Assuming that a total of 24 basins will be operated for 8 months (50% of the time), then the expected volume of recharged water is 120,000 m^3/d and 28.8 million m^3/yr. The net area of the bottom of the recharge basin required to allow for this recharge is estimated at 1.5 km^2. The total area of the site, including the banks and all other facilities is estimated at 1.8 km^2.

CONCLUSION

Based on the pilot study conducted at the Sulaibyia area 50 km southwest of Kuwait City in Kuwait, soil aquifer treatment (SAT) is considered as a suitable technique in Kuwait for the quality improvement for the secondary or tertiary treated wastewater. Moreover, it can solve the problem of storage of secondary or tertiary treated wastewater. It can act as a buffer storage which is useful for varying supply and demand of treated wastewater and this aquifer storage is a cost effective method. This method or treatment and storage offers a very important psychological advantage that it breaks the direct, pipe to pipe connection of the reuse system, because water after SAT comes out of a well as groundwater and has lost its stigma and identity of municipal wastewater. This artificial recharge can be considered as ideal for seasonal storages. Since this is proposed scheme is in northern Kuwait as freshwater lenses it is ideal for long term storage.

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Groundwater recharge from a percolation tank to a Deccan basalt aquifer: a case study from western Maharashtra, India

Uma Badarayani, Himanshu Kulkarni and Vinit Phadnis

Abstract

Small check dams in the Deccan volcanic province are constructed to artificially recharge groundwater in the Deccan basalts of central-west India. These check dams are a part of watershed development projects and few attempts are made to understand the exact dynamics of infiltration from such structures. The Chikhalga on study in Kolwan valley of Pune district revealed that only micro-scale studies are able to pick up artificial recharge impacts in a microwatershed from the Deccan basalts. The quantification of natural and artificially induced infiltration in the Deccan basalt is a complicated exercise, requiring multifaceted measurements. Water balances on the scale of a single structure such as the check dam CD3 in Chikhalgaon, are useful in understanding the amount and nature of infiltration processes, giving a good understanding of the underlying shallow unconfined aquifer. Impacts of livelihoods are difficult to relate to artificial recharge from individual structures because other livelihood diversification trends may mask these effects which, in themselves, are quite small as compared to the volumes of rainfall and surface runoff.

Keywords

Deccan basalt, check dam, shallow aquifer, infiltration, water balance.

INTRODUCTION

Artificial recharge to augment groundwater resources is gaining momentum across India and other parts of the developing world. The more traditional methods of artificial recharge in India are the spreading methods that include putting checks across streams to create tanks or ponds through which infiltration occurs over a period of time, subsequently recharging underlying aquifers. The hydrogeology of hard-rock regimes and highly variable rainfall patterns pose great complexities to fully understand the recharge processes as well as the implications of augmenting groundwater resources through such processes. These implications clearly include two aspects:

1. The physical dynamics of how recharge occurs
2. The effects of artificial recharge on the livelihoods of communities

This case study from Chikhalgaon village, located within a completely basaltic setting, formed a part of the DFID-funded, British Geological Survey-led project called Augmenting Groundwater Resources by Artificial Recharge (AGRAR). The study attempted to understand artificial recharge processes on a micro scale, within basalt aquifers, and their implications on the livelihoods of the rural population.

A watershed development project was implemented by a Pune based NGO, called GOMUKH, in Chikhalgaon watershed. A detailed monitoring of the site was carried out using a monitoring network set up to collect hydrological, hydrogeological and meteorological information (Kulkarni et al, 2003). This paper brings together some of the significant information with the aim of developing a water balance for the check dam and for the aquifer underlying it.

Check dam CD3

The check dam CD3 is a masonry structure with dimensions as given in Table 1. Intensive monitoring was undertaken in the vicinity of CD3, with respect to hydrological and hydrogeological information, presented in the next section.

Table 1. Details of the check dam (CD3)

Elevation of CD3 (above msl)	615 m
Top width	22 m
Height of CD3	2.05 m
Total capacity of structure during bank-full stage	1,500 m^3
The total catchment area of CD3	0.7861 km^2

METHODOLOGY

A systematic groundwater monitoring process was set up in the Kolwan valley on the basis of the AGRAR guidelines for fieldwork (Gale et al, 2003), with practical considerations and standards provided by Gunston (1998). The actual monitoring was conducted in three stages. In the first stage, an automatic weather station was established at Chikhalgaon to obtain uninterrupted meteorological data. In the second stage, a network of observation boreholes was drilled around CD3 to measure water levels, and v-notch, staff gauges and a stilling well with the automatic water level recorder were installed to measure inflows and overflows. In the third stage, pumping tests were carried out on the boreholes (after an uninterrupted year of water level monitoring) to assess the properties of the underlying shallow aquifer.

Weather data and measurements at CD3

Six weather parameters were monitored at the AWS at an hourly interval. These parameters were air temperature, relative humidity, solar radiation, rainfall, wind velocity and rate of evaporation. This data was used in the calculations to estimate the water balance of the check dam. A detailed topographic survey of the main structure (CD3) around which intensive monitoring was undertaken, was carried out to enable calculation of volume changes and stage levels, relative to other monitoring points like dug wells and observation boreholes. Inflow and overflow discharges from the check dams were calculated using the discharge estimation tables and formulae based on the Herschey's equation (1995).

Groundwater levels

Nine observation boreholes were drilled near CD3. The sites for these boreholes were selected primarily to ascertain the influence of the check dams on the water levels in the adjoining aquifer, in different directions and at different distances. Resistivity data, borehole logs and detailed geological observations of surface exposures in existing dug wells helped obtain an idea about the shallow basalt aquifer and the variation therein. A conceptual model of the Chikhalgaon aquifer was developed on the basis of this information (Figure 2(a)). The Chikhalgaon aquifer is a well bounded aquifer, with an impermeable base in the form of the unjointed compact basalt that is overlaid by vertically jointed, compact basalt. The main storage in the shallow aquifer is within the sub-horizontally jointed/fractured compound basalt unit, an observation that is in line with studies from other areas of the Deccan basalt region (Kulkarni and Deolankar, 1995; Kulkarni et al, 2000).
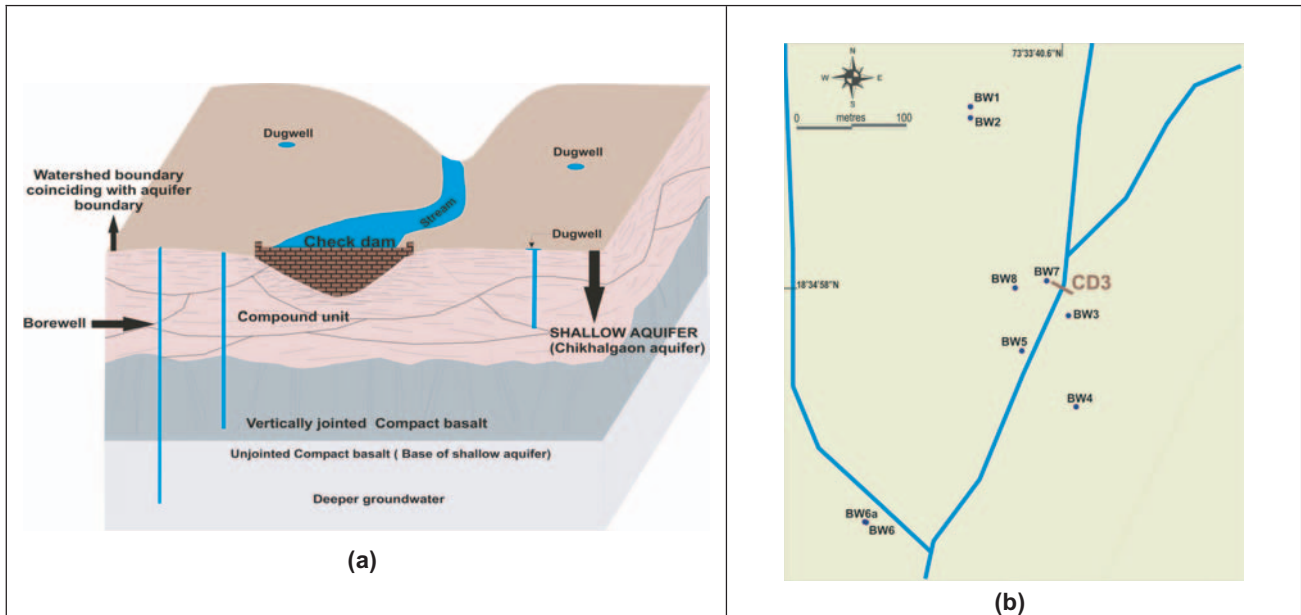
Figure 2. Conceptual model of Chikhalgaon aquifer and location of boreholes around CD3

Of the eight observation boreholes drilled in the vicinity of CD3, seven were completed within the shallow aquifer only, whereas one bore well (BH1) was drilled into deeper basalt aquifers. All observation boreholes were 152 mm in diameter and were cased to a depth of about 3 m from the ground surface to prevent caving in of the topsoil and weathered mantle. Figure 2(b) is a plan showing the location of boreholes around CD3.

Pumping tests

Pumping tests were conducted on some of the observation boreholes during January 2005, after the check dam CD3 ran dry. These tests were intended to give an idea about the transmission and storage properties of the Chikhalgaon aquifer. Groundwater abstraction from the Chikhalgaon aquifer in general, and in the vicinity of the check dam CD3, in particular, is quite limited (there is virtually no abstraction from near the observation boreholes because of the absence of pumping dug wells). The analysis of pumping test data was carried out by several methods, including the suite of methods provided in AQUIFERTEST PRO 3.5. Data was also analysed using the BGSPT package developed by Barker, 1989.

Livelihood surveys

A socio economic survey was conducted to assess the impact of watershed development on the livelihoods of the people. More than 50% population of the Chikhalgaon village was covered under the 'household-based' survey. The main emphasis of the study was to identify any changes in the source of irrigation, cropping pattern and the overall socio-economic status of the population due to availability of water. Both the scenarios viz 'before and after the check dam' and 'with versus without the check dam' were considered while formulating the questionnaire for this survey (Kulkarni et al, 2003; Kulkarni et al., *in preparation*).

RESULTS

Initial plots of ground water contours in the vicinity of CD3 were generated using water level data obtained from observation boreholes and the adjoining dug wells, on weekly basis. The mainstream, on which the structures are constructed, acts as a discharging boundary at most places. There is little flow across the stream from one side to

the other. Considering these factors, water table contours were plotted for three seasons (Figure 3). This data represents two dry and one wet seasons.

Additionally, a water table contour map was generated for August, 2004 where only dug well data was used (Figure 3a). This plot did not pick up the change in groundwater flow pattern in the area close to CD3, clearly indicating that groundwater flow followed the surface topography. Including observation borehole data, revealed the local effects of infiltration from the check dam CD3. This effect was evident in the form of a mound around CD3 (Figures 3b, 3c & 3d). The mound was most prominent during the dry season (February 2005). It was not so pronounced during the wet season, because natural infiltration in 'recharge' areas upstream of CD3 (in the vicinity of BH2) masks the local effect created from the infiltration from CD3. Pumping test data from boreholes indicated that the Chikhalgaon aquifer is a low transmissivity and low storativity aquifer. Limited transmissivity implies a slow development and dissipation of the mound created just downstream of CD3. Pumping test data yielded transmissivity values of the order of 1–5 m^2/day, whereas the storativity (specific yield) varied over several orders of magnitude. However, comparison of the specific yield values with those obtained for similar settings in the Deccan basalt (Kulkarni et al, 2000) imply that the Chikhalgaon aquifer has a specific yield varying between 0.0001 and 0.01, with an average value of about 0.005.

Notwithstanding the values of T and S obtained from pumping test data, it was interesting to note the relative trends in drawdown and recovery between these wells. Borehole BH6, which took more than two days to recover, implied that boreholes close to CD3 (BH8, BH5 and BH3, which recovered over a matter of a few hours) were drawing water from close by (? from the mound just downstream of CD3).

[what is "(?"]
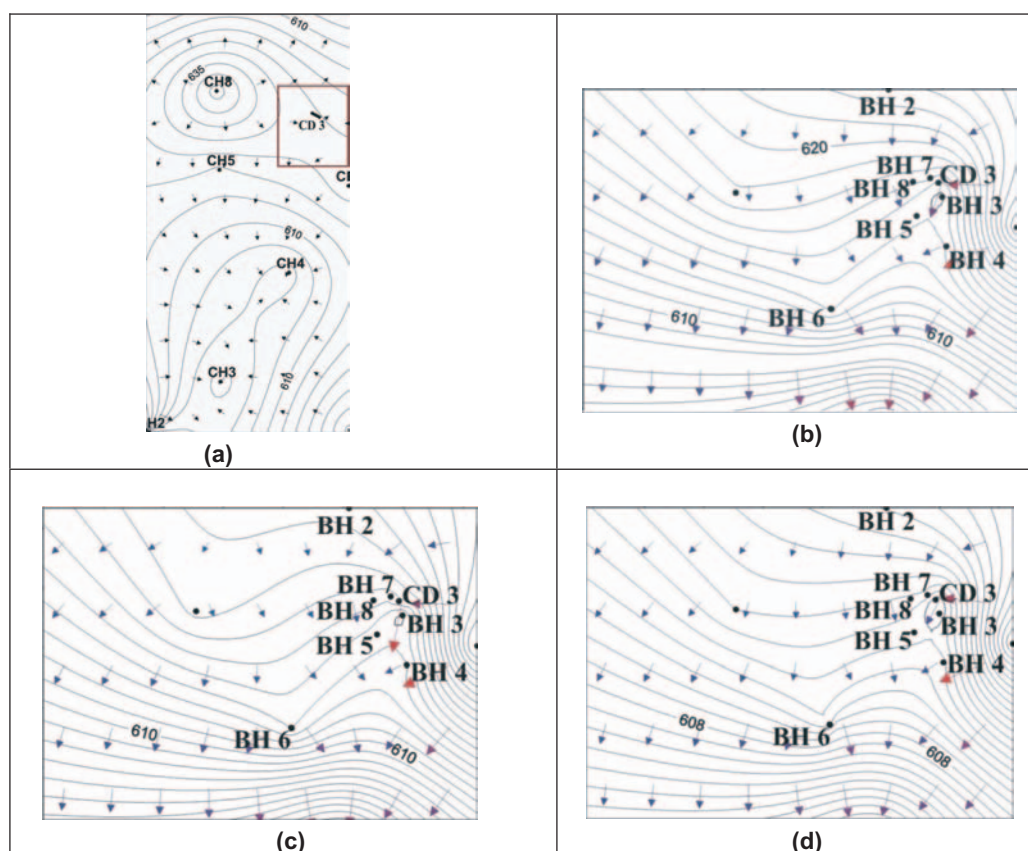
Figure 3. Groundwater contour map near CD3. The figure includes water table contours using only dug well data (a) and using borehole data for three seasons (b, c and d)

Water balance in CD3

To quantify losses/gains to CD3 during an annual cycle, the water balance for the structure was split into two major components. The first component of the water balance has been estimated for the period of overflow in the check dam, which was 108 days. This period corresponded to the time during which the water level in the structure declined, essentially as a response to evaporation and infiltration. The period taken for the structure to completely dry up was 63 days after the overflow stopped, over which the 'dry season' (second component of the water balance) water balance was estimated. Using water level fluctuations in BH2 during the monsoon period, with respect to events of rainfall, the cumulative 'natural infiltration' was calculated to be about 105 mm (Kulkarni et al, in preparation).
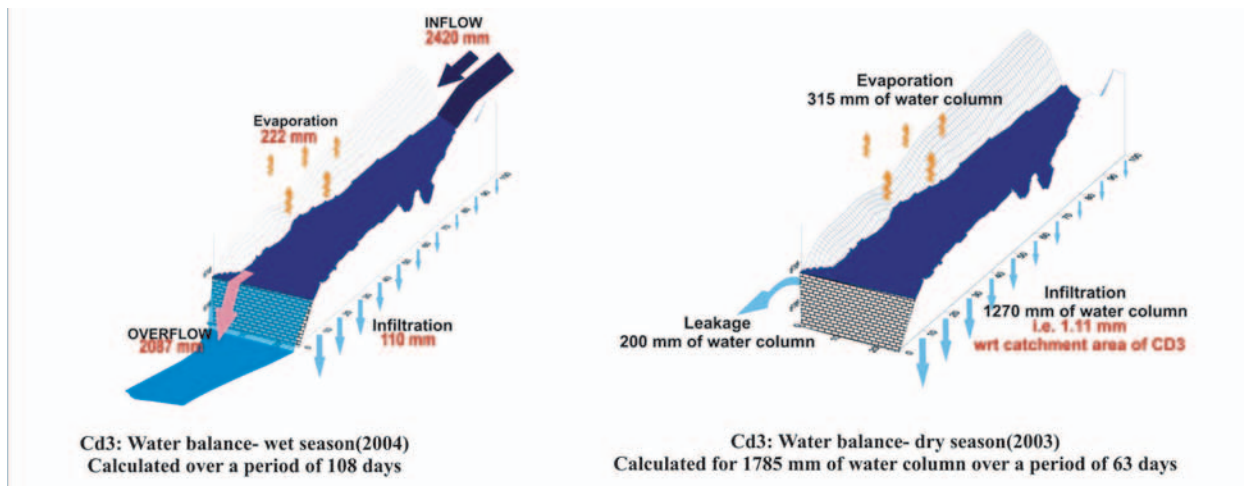
Figure 4. CD3: Water balance – wet season (2004) and dry season (2003)

Impacts on livelihood

The actual impacts of the check dam on the livelihoods of the people may be quite small as compared to the other 'push and pull' factors, but it is something that cannot be neglected, at least not on the scale of four or five watersheds where GOMUKH is scaling up this activity. Groundwater abstraction for irrigation is still quite limited but there are signs of increasing abstraction, after the check dam was constructed. This increase can be attributed to a perceivable improvement in groundwater conditions, artificial recharge being one of the drivers to this effect.

CONCLUSIONS

The basalt aquifer in Chikhalgaon is a low storage, variable transmissivity aquifer. Natural infiltration and artificial recharge (through the check dam) contribute to the net recharge to the shallow aquifer in Chikhalgaon. The water table in the vicinity of the check dam is influenced by the mound created from the infiltration from CD3. The response of the shallow basalt aquifer to natural and artificial infiltration is quick and takes a few weeks. Most significantly, studies on the micro scale are necessary to pick up the nature and extent of impacts from artificial recharge initiatives to underlying basalt aquifers, given the complexity in the hydrogeological properties of the Deccan basalts.

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Study on sustainability of irrigation agriculture by diversion in the lower reach of the Yellow River

Jianyao Chen, Yoshihiro Fukushima and Makoto Taniguchi

Abstract

The lower reach of the Yellow River is referred to the range starting from Huayuankou of Henan Province to Lijin of Shandong Province. Since groundwater table was shallow and high in salinity, most part of this area was not developed until the 1970s when the Yellow River was diverted to leach out salt and irrigate crops. With the decrease of flow in the Yellow River, the sustainability of irrigation agriculture is becoming an issue to be dealt with since 1997 when no water flowed in the reach from the river mouth up to about 780 km within a total period of 226 days. The aim of this study is to analyze the sustainability of agriculture after about 30 years of irrigation practice under the situation of water shortage. It is concluded that agriculture in the lower reach depends greatly on the Yellow River, which is low in flow rate in last twenty years, and 50-60% of the groundwater in the shallow aquifer comes from the Yellow River; saline water does no harm to crops as before and agriculture is sustainable as long as the Yellow River could supply water for irrigation.

INTRODUCTION

The lower reach of the Yellow River is defined as the zone from Huayuankou (about 100 m a.m.s.l.) to Lijin (about 12 m a.m.s.l.) with an administration area of about 4.43×10^4 km^2 and a population of about 2.5×10^7 (Ruan, 1997). This is an important agricultural area with an irrigation area of about 1.93×10^4 km^2 in 1990 by using the water diverted from the Yellow River (Xi, 1999; CDCID, 2002), accounting for about one-third of total irrigation area of the whole basin. Since much of sediment precipitates in the lower reach, raising the river bed, the Yellow River is higher than the riparian zone, i.e., the Yellow River is a suspended river in the lower reach and recharges the local aquifer along the river. Water table fluctuates due to precipitation and water diversion ranging from 1 to 4 m depth.

Water shortage is serious in the lower reach due to decreasing inflow at Huayuankou station and high demand for domestic, industrial and agricultural water use, especially in the North China Plain. The main objective of this study is to analyze the sustainability of irrigation agriculture in the lower reach under the situation of salt accumulation and water shortage.

METHODS

Piezometers with depths of 4, 5, 6, 8, 10 m, respectively was set up at Yucheng experimental station of Chinese Academy of Sciences. Water samples were collected from the piezometers monthly, together with soil water samples by means of vacuum method (porous cup) from 30, 50, 70, 90, 120, 150, 200 cm depth. Tensiometers were set up in the unsaturated zone at the same depth as the porous cups, and tension was measured manually at 3 days intervals. Chemical composition of NO_3^-, Cl^-, and SO_4^{2-} were analyzed by ion chromatography, Na^+, K^+, Mg^{2+}, Ca^{2+},

SiO_2 by Inductively Coupled Plasma, HCO_3^- by titration, and stable isotopes (^{18}O and deuterium=D) by a Delta S mass spectrometer. Water table data of 26 observed wells during the period of 1990–2000 in the lower reach was obtained from the Bureau of Water Conservation of Shandong Province. Monthly discharge data at Huayuankou and Lijin was collected from the Yellow River Conservation Committee. Actual evapotranspiration at Yucheng station was measured daily by lysimeter. Related data, such as land use, were collected from published papers, reports, together with rainfall, pan evaporation data from Meteorological Bureau of Shandong Province.

RESULTS AND DISCUSSIONS

The Yellow River flows to the northeast in the lower reach with an average topographic gradient of about 1/8200. The regional groundwater flows in the same direction as the surface water (Fig. 1), though the Yellow River recharges the riparian zone with a higher gradient, e.g., 1/4651 at Yucheng City of Shandong Province (Chen et al., 2002). The aquifer transmissivity obtained by a pumping test is about 93–746 m^2/day (Zhang et al., 1988), annual recharge directly from the Yellow River for the river length of about 650 km in the lower reach is estimated to be $4.7 \times 10^6 - 3.8 \times 10^7$ m^3 by using the gradient at Yucheng City. Groundwater table changes but with a narrow range of about 2 m due to the recharge from the river. When the drying up occurs in the main channel, recharge to the groundwater ceases and water table drops down obviously, e.g., in 1997, there was no flow in the lower reach for a total period of 226 days, water table at one of the observed wells (indicated by the arrow in Fig. 1) dropped down 6.15 m (Fig. 2).
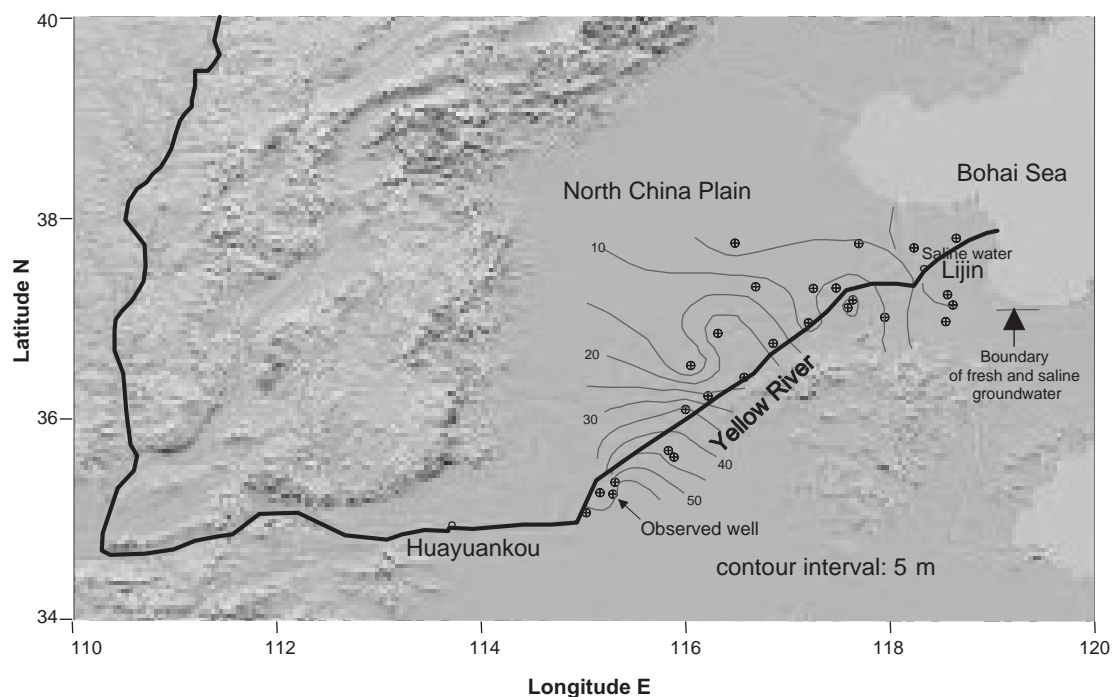
Figure 1. Groundwater table contour in the lower reach based on data in Dec. 2000

The impact zone of the Yellow River on the aquifer was indicated by the groundwater table variation along the distance perpendicular to the Yellow River (Fig. 3). Average water table variation in 1997 was 2.2 m, higher than the years in 1999 and 2000, when there was no drying up due to water supply from the Xiaolangdi Reservoir. Water table variation (WTV) shows no difference along the distance from the Yellow River in 1997, while WTV in 1999 and 2000 is high near the river and decreases as it is far way to a distance of about 40 km, and then increases again. Thus, a zone with a distance of about 40 km from the Yellow River is regarded as the impact area of the river in terms of recharge. This result is similar to that indicated by the evidences of electrical conductivity (EC), Cl^- concentration and isotopic features (Chen et al., 2001).
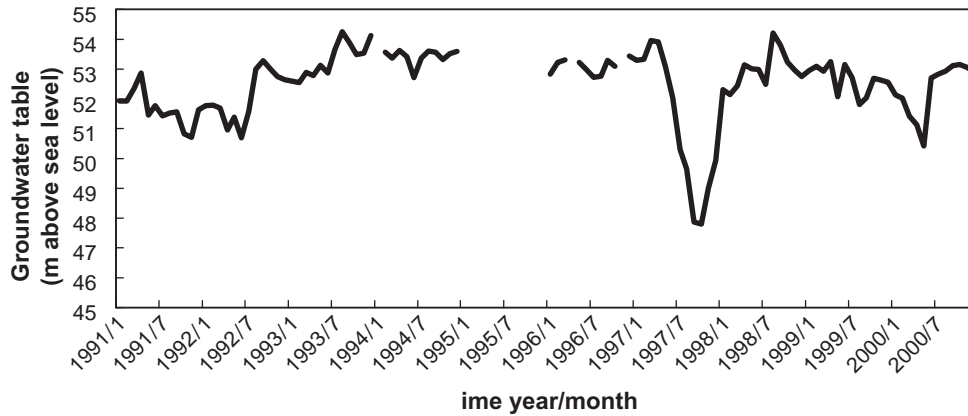
Figure 2. Water table change at one of the observed wells (indicated by the arrow in Fig.1) from 1991 to 2000
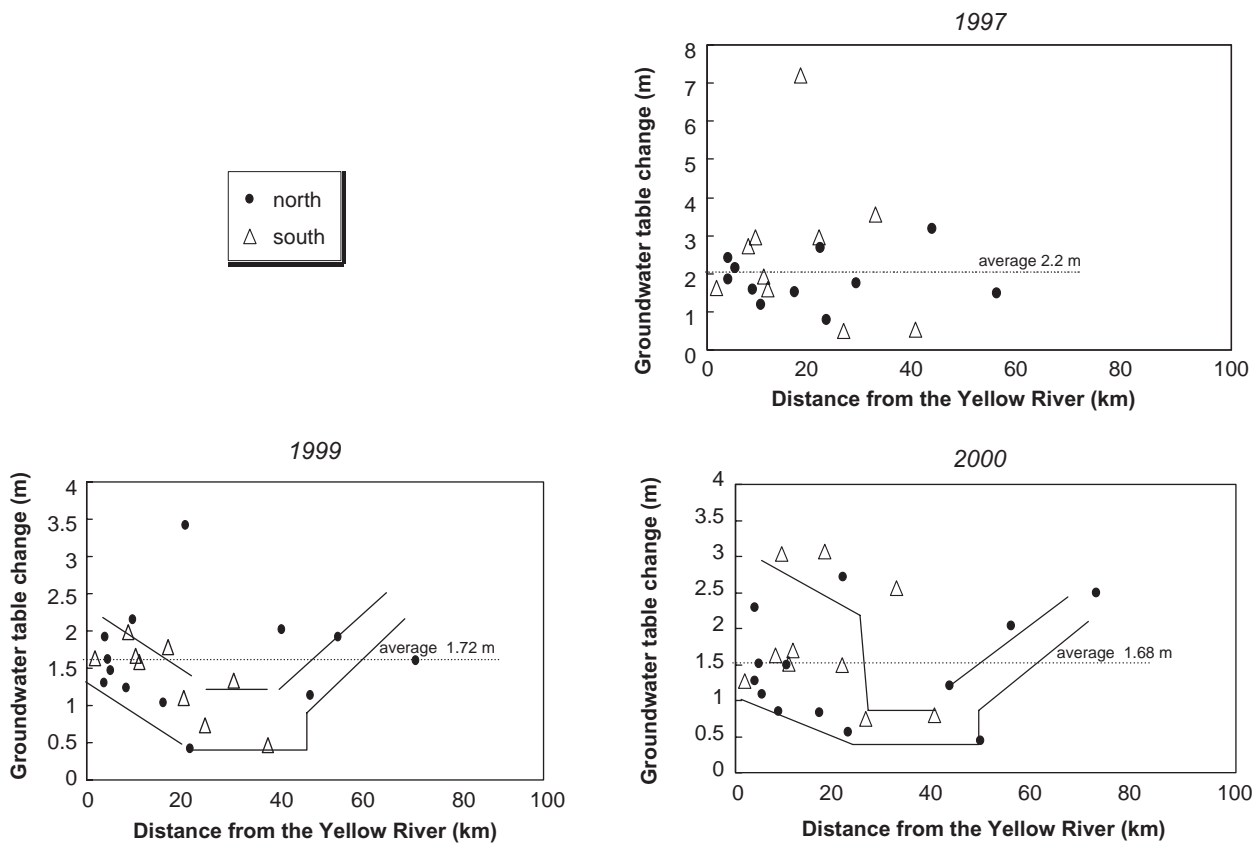
Figure 3. Groundwater table variation along the distance perpendicular to the Yellow River for the observed wells in the southern and northern parts of the river

Annual discharges at Huayuankou and Lijin decrease in the last 50 years (Fig. 4), especially in the last 20 years, while water diverted, defined as the difference of discharge between Huayuankou and Lijin, for irrigation, industry and domestic use in the reach increases. More than half of water in the lower reach has been diverted in the 1990s. Calculated water use and change of river bed height are given in Table 1. In the end of the 1950s, many irrigation projects were built for land reclamation. Unfortunately, drainage canal from the agricultural land was not well planned and constructed, and most of the projects were abandoned due to salinization problem occurred thereafter. Thus, only 6% of the river water was used in the 1960s.
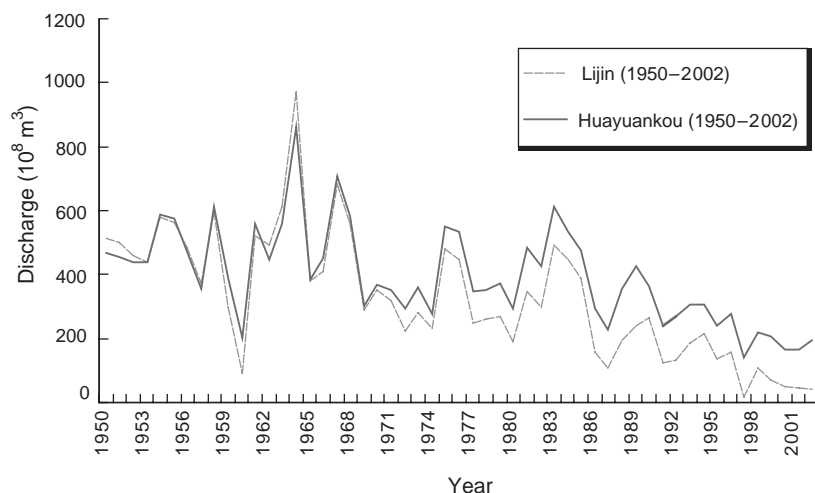
Figure 4. Annual discharge at Huayuankou and Lijin during the period of 1950–2002

Table 1. Water use calculated and height of river bed in the lower reach

Decade	1950s	1960s	1970s	1980s	1990s
(A) Water diverted, 10^{10} m^3	42.3	31.3	88.9	136.7	121.9
(B) Observed discharge at Huayuankou, 10^{10} m^3	458.7	521.8	372.8	420.2	236.9
Ratio= A/B, %	9.2	6.0	23.8	32.5	51.5
Height of river bed at Huayuankou*, m	92.65	92.25	93.24	93.29	93.35
Height of river bed at Lijin*, m	9.44	9.74	11.05	10.83	11.7

As we know, average sediment load of the Yellow River is as high as 35 kg/m^3, and it is affected greatly by the flow rate and the construction of reservoir. Fig. 5 shows a linear relationship between annual discharge and sediment at Lijin. Annual sediment is generally less than 5×10^8 tons in the 1990s as the discharge reduces and much more
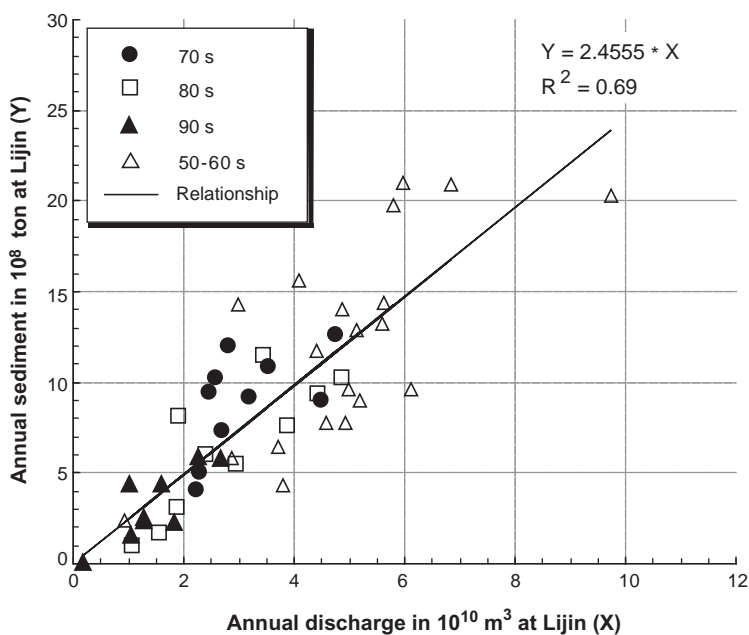
Figure 5. Relationship between annual sediment and discharge at Lijin station

sediment load is kept in the channel, raising the river bed accordingly. Height of river bed at Huayuankou dropped down in the 1960s due to the construction of Sanmenxia Reservoir. The heights at Huayuankou, Jiahetang, Gaocun, Sunkou, Luokou, and Lijin increase 0.7, 0.86, 1.6, 0.88, 0.94, 2.26 m respectively from the 1950s to the 1990s. The average height in the lower reach is thus estimated to increases 1.2 m in the last 50 years with an increase of 2.3% in topographic gradient given an average altitude of 51.6 m for the lower reach. Supposed hydraulic conductivity remains constant, recharge to the riparian zone is estimated to increase 0.0468% annually.

Annual average evapotranspiration measured by lysimeter at Yucheng experimental station is about 927 mm (Chen et al., 2004), while annual average precipitation at Jinan of Shandong Province during the period of 1951–2001 is 670 mm. The deficit of 257 mm is supplied by the water from the Yellow River. According to the isotopic signature of ^{18}O and ^{2}H, 50–60% of the groundwater in the shallow aquifer to the depth of 20 m comes from the Yellow River (Chen et al., 2002). Most area of the lower reach was alkaline land before the reclamation, and high salt content still remains in the unsaturated zone even after irrigation of more than 30 years. The highest concentration was found at a depth of 120 cm in the vertical profile from 0.3 m to 10 m depth at Yucheng Station (Chen et al., 2002; Chen et al., 2004). Main reason for this concentration distribution is the occurrence of zero flux plain (ZFP) at 120 cm depth (Fig. 6).
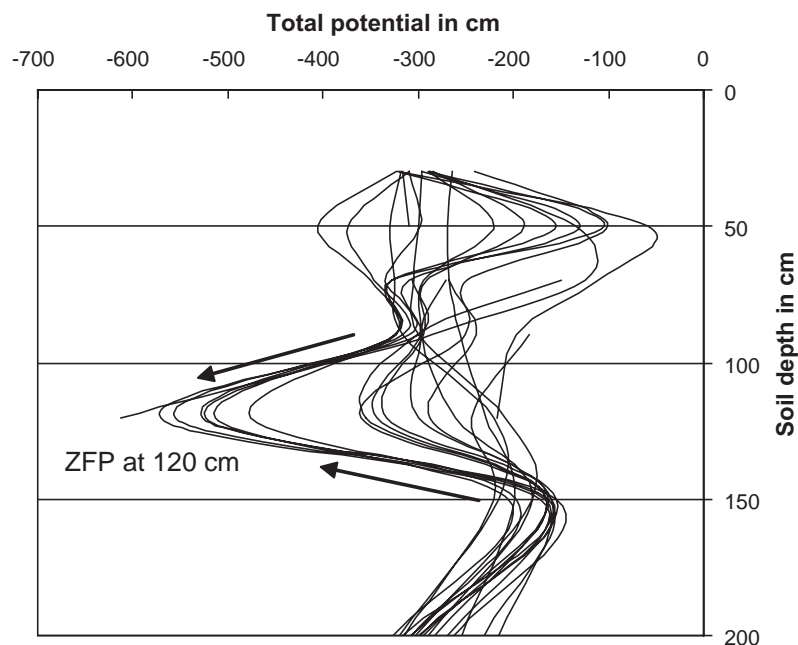
Figure 6. Change of total potential in the unsaturated zone at Yucheng Station during the period of Sept. 2000–Nov. 2001

CONCLUSION

The irrigation agriculture has been sustained for more than 30 years by water diversion from the Yellow River though water available for the diversion in the lower reach has decreased. This practice has not only redistributed spatial water resources, but affected recharge and groundwater quality by leaching out surface salt. The diverted water accounts for about 50–60% of local groundwater resources. The highest salt concentration occurs at the depth of 120 cm, which is deeper than most of root depth of wheat, corn and bean, reducing the possible harm of salt to crop. The impact zone of the Yellow River in terms of recharge in the riparian zone is estimated to be around 40 km away from the river. Recharge rate is estimated to be $4.7 \times 10^{6} - 3.8 \times 10^{7}$ m^3/y and increase 0.0468% annually due to the rising of river bed in the last 50 years. Water diversion, recharge from the suspended river, and the occurrence of ZFP are regarded as the main reasons for the sustainability of the irrigation agriculture in the

lower reach. Agriculture in the lower reach is sustainable as long as the Yellow River could supply water for irrigation. In case of the drying up, given a specific yield of 0.1, the shallow aquifer of 20 m depth could provide the water deficit of 257 mm for at least 2–3 years. This is the reason that the crop production did not decease in 1997.

ACKNOWLEDGEMENT

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Surface runoff and groundwater recharge measured on semi-permeable surfaces

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Abstract

Since the year 2000, a rainwater fee is charged in Berlin for each square meter and year, which is dependent on the runoff into the public sewer. At the TU Berlin, the surface runoff and groundwater recharge during precipitation events is monitored on different semi-permeable surfaces. In general, semi-permeable surfaces show an increased groundwater recharge in comparison to native landscapes. The surface runoff is directly connected to the intensity of the rainfall and varies from 0 to 18% of yearly precipitation depending on the surface material, runoff increases up to 80% of the precipitation during one stormwater event.

Keywords

Semi permeable surfaces; surface runoff; groundwater recharge, urbanization; evapotranspiration.

INTRODUCTION

Urbanization is increasing world wide. Urban centers can have negative environmental impacts including escalated flood risks, polluted surface waters and increased groundwater depletion. Rainwater retention measures successfully reduce these effects.

Since 1998, thirteen semi-permeable surfaces for the surface runoff and groundwater recharge are monitored in the installation described below. Some of them infiltrate only through the gaps (there are three variants with greened gaps), some, in addition infiltrate through the material itself. They represent the types currently used in Germany.

A rainwater fee replacing the common wastewater fee based on the drinking water consumption was introduced in the year 2000. The rainwater fee is charged for each sealed square meter and year, depending on the stormwater runoff into the public sewer.

This new fee influences possible implementations of decentralized measures of rainwater retention, such as greened roofs, rainwater retention and infiltration systems. Quantitative information for designing such fee systems are needed.

At the department of Applied Hydrology (www.tu-berlin.de/~Wasserhaushalt) investigations of all aspects of urban hydrology are carried out.

METHODS

Figs. 1 and 2 show some aspects of the experimental installations of which the results are discussed here. The testing field was originally constructed at the waterworks station in Berlin-Jungfernheide in 1981. Some results have already been published. In 1998, this installation was brought to the Dahlem experimental fields and integrated into existing stations of hydrologic measurements, a lysimeter station, greened roof plots and a climatological station.

Fig. 1 and 2. Testing field of semi-permeable surfaces at the TU Berlin

The testing field was modified to measure the precipitation and runoff in a high time resolution of one second, thus allowing a detailed investigation of single precipitation events.

The equipment used are 26 tipping buckets of 100 ml volume, connected to a computer system for online-measurements. From tipping buckets quality samples can also be taken.

The surface of each plot has a size of 4 square meters each. With the resolution of 100 ml, the surface runoff and the groundwater recharge can be expressed as 0.025 mm.

The intensity of the runoff is recorded based in seconds between of two runoff-volumes of 100 ml accumulated in the tipping buckets, runoff intensities up to 0.025 mm per second can be measured. Mistakes in measurements on high intensities can be recognized, mainly the mechanical behavior of tipping buckets under different runoff intensities. Losses in water quantity, typical for measurements with tipping buckets for precipitation and runoff, have been corrected.

In contrast to real hard surfaces in urban areas the surfaces are generally unused. On the other hand they are nearly 25 years old and soil clogging took place.

RESULTS AND DISCUSSION

The surface runoff is directly connected to the intensity of the rainfall and varies from 0 to 18% of the year's precipitation. Stormwater runoff increases up to 80%, depending on the surface material and the precipitation intensity. The annual groundwater recharge is up to 4 times increased, compared to native landscapes. Missing hydrological component of urban areas and semi permeable surfaces is always the evapotranspiration.

Fig. 3 shows the relation between precipitation, surface runoff and groundwater recharge from july to december 2000 as a mean of 8 common semi-permeable surfaces. Greened surfaces are not included in this figure, they mostly don't show any surface runoff at all.

The surface runoff during the year is lower than expected. The reason is that the precipitation intensity is lower than 25 Liters per second and hectar for 90% of the precipitation in the year. This is equivalent to 9 Millimeters per hour. Stormwater runoff of a high intensity is representative for several events in the summer month (see Fig. 3).
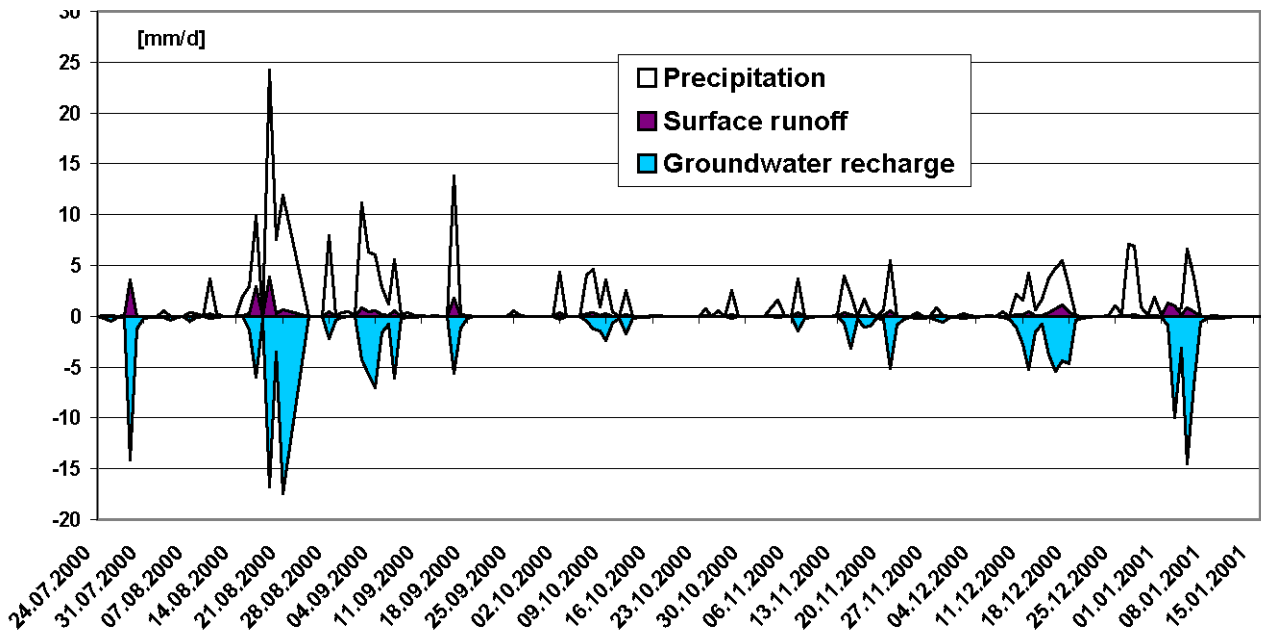
*Fig. 3. Precipitation, surface runoff and groundwater recharge in Millimeter/day
as a mean of 8 semi permeable surfaces*

Highest surface runoff rates are measured on semi permeable surfaces which infiltrate only through the material itself. Our example here is porous asphalt (see Fig. 4). The impermeabilization increases the first years after implementation up to a certain level, due to an input of fine depositions. On the other hand the infiltration rate can be considered as being high for the whole year; the groundwater recharge under these surfaces is increased compared to vegetated landscapes. Half of the surface runoff during a year is directly connected to precipitation in a high intensity in the summer months. A large proportion of runoff is due to the affect of freezing in the wintertime.
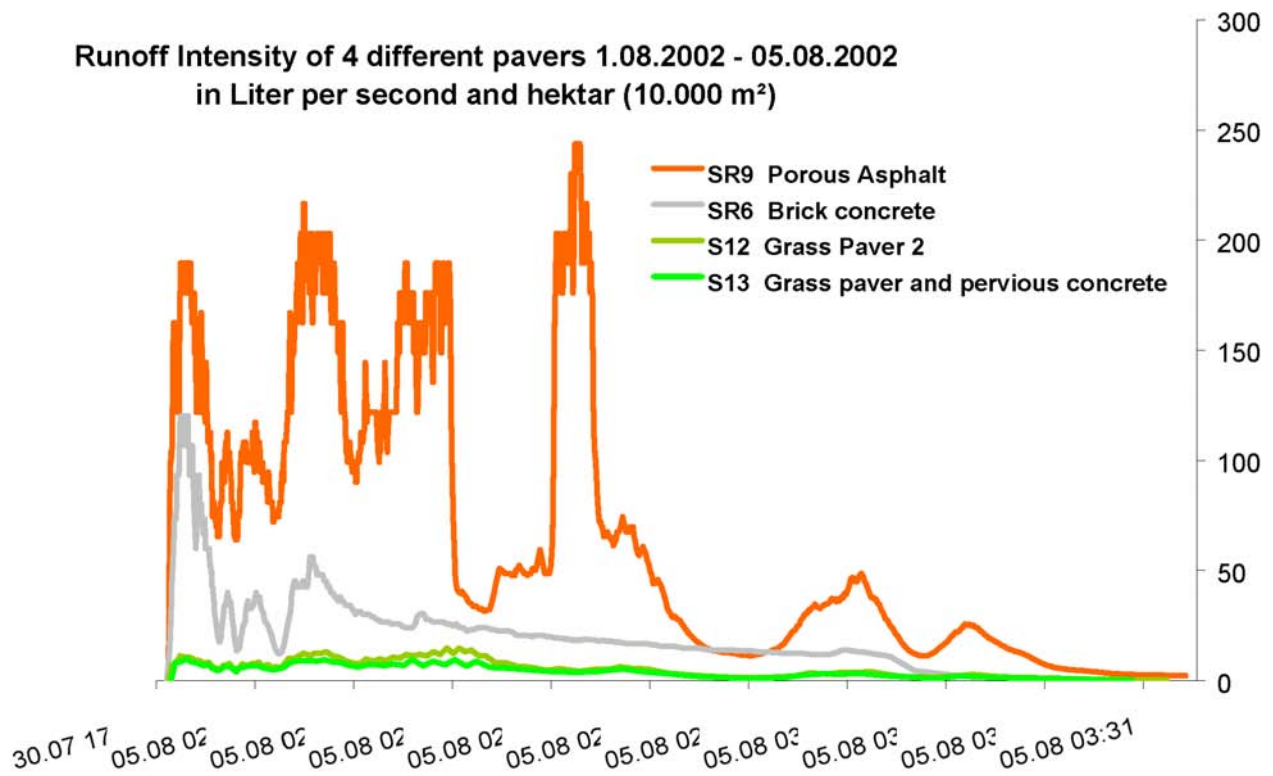
Fig. 4: Runoff intensity of 4 different pavers

CONCLUSIONS

In general, semi-permeable surfaces show an increased groundwater recharge, compared to native landscapes. This might compensate for the impermeability of streets and buildings in urban areas. The hydrologic balance in urban areas is strongly modified due to the low evapotranspiration rates and to the different nature of the infiltration processes which take place. The unexpectedly high groundwater recharge rates in cities are one consequence of this. These measurements described are intended to help understand this.

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Traditional rainwater harvesting technologies: key to drinking water security for desert comunities in arid regions of India

N.R. Grey and O.P. Sharma

Abstract

The Indian part of the Thar Desert is a hot arid region of 200,000 sq km with a population of 20million. It has the highest desert population density in the world. Communities live in scattered hamlets where shifting sands make water transport extremely difficult and metalled roads are few and far between. Drinking water security at a family level can be achieved using traditional water harvesting practices, with appropriate technology added, so as to improve water catchment efficiency and water quality. In particular the provision for each family in a village of a *taanka* - underground water storage tank with a surrounding apron (acting as a micro water catchment area) – has proven a low cost means to provide drinking water security for the family over the 10 months between the monsoon seasons. The paper gives the design detail, evaluates effectiveness for differing rainfall patterns, water quality questions and the social implications of drinking water security on the lives of women, children and the family unit. It discusses the other water needs of the family, of their animals and of other livelihood needs. It addresses questions of sustainability and the replication of the model across the Thar Desert.

INTRODUCTION

Traditional water harvesting systems that have been practiced since the birth of civilisation in the Indus valley have, over the past 60 years rapidly decayed as water 'on tap' has been promised to the desert people. The Thar Desert (Figure 1) has a monsoonal rainfall varying from 100 to 300 mm; the rain falls with great intensity (26 mm on average) on 5 to 20 days in a year. The mean onset of the monsoon season is 15th July and withdrawal 1st September (Khan, 1998). The study of rainfall of a particular village can show no rain falls in a village or in some cases rain falls on one part of a village but none on the other part. Four out of 10 years are on average drought years making agriculture in the region extremely precarious.

The people live in scattered hamlets grouped together in villages with typical populations of 1,000 to 1,500. There is a mixture of caste with 15 to 20% classified as schedule caste and about 5% as tribal. The population is largely Hindu but with 10 to 20% Muslim. Over the last 100 years population density has increased strongly. The sex ratio, is very low (913 females per 1000 males). (Resource Atlas of Rajasthan, 1994)

The people rely on animals for their livelihood. Their desert cattle are extremely hardy and until recently Rajasthan was known as the milk pail of India. Nearly half of India's wool is produced in Rajasthan. The Gujar caste keep flocks of sheep, the men migrating with their sheep many hundreds of miles to the Punjab – to places where food and green pasture is available. The typical family will have a cow and a few goats. Almost all families have some land with average holdings of 3 to 5 Ha, on which they may hope to grow some fodder and food crops for their family.
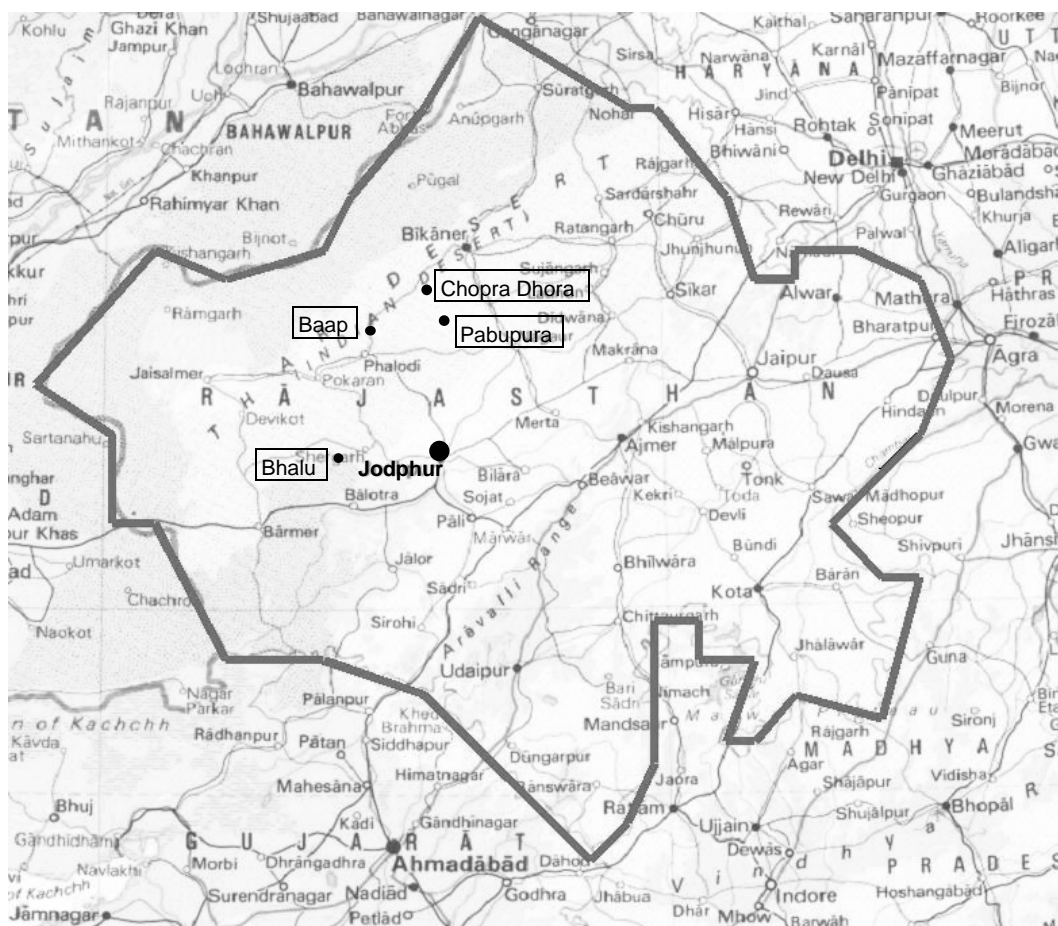
Figure 1. Rajasthan State, North West India
Population 56.47 million (2001 India Census). The Indian Thar Desert occupies
the northwest half of Rajasthan and extends into Gujarat to the south and Haryana to the north.

The plough, overgrazing and the breakdown of the feudal system, which meted out severe punishment for cutting down trees, has reduced the biomass of the desert. Tree, shrub and grass cover have been much reduced over the past 50 years and the fierce hot May and June winds whip up sandstorms of greater and greater intensity.

Most of the able bodied men migrate to seek work in October after the crops have been harvested and return in June to prepare for the monsoon rains. Life is hard for the women and children. Maternal mortality is amongst the highest in the world (UNICEF 1992-93), child marriage commonplace and female literacy 1 to 2% in the villages. Most diseases are water related and include leukorrhea due to poor hygiene.

Water is the cry of the people in every part of Rajasthan and the critical issue for Rajasthan and the Thar Desert in particular. It is the natural entry point for gaining the trust of the villagers. It is the key to poverty alleviation. Over the past 15 years there have been a number of high profile water harvesting success stories. These transformed the lives of the village people and also enabled them to survive the severe drought years 1999–2003. The challenge now is to develop and demonstrate models that can be replicated across Rajasthan.

The objective of this paper is to show how one particular water harvesting structure is used and has transformed the lives of rural communities in the desert areas of the Thar.

HISTORICAL CONTEXT

The people in the Indus Valley Civilisation (3000 BC –1500 BC) depended on good water management. The exca-

vations of Harappa in the 1920's revealed a town laid out so that every drop of water falling on the roofs and pavements was harvested and collected in reservoirs on the perimeter of the villages (National Museum, Delhi). The people in the Thar Desert have had a long history developing a multitude of systems to harvest and store water. This was combined with a cultural heritage, which deeply respected the value of water. This was linked (up till the breakdown of the feudal system) with strong community values to support the construction and maintenance of water harvesting structures and the catchment areas around them were sacred with strong punishment for those doing damage, cutting trees or allowing animals to roam prior to the monsoon rains. In the last 50 years an increasing number of deep boreholes have extracted groundwater and politicians have promised water on tap. (Agarwal, 2001). The years of severe drought 1985–87 and 1999–2003 demonstrated the dependence of the people of the Thar Desert on the traditional systems (Agarwal, 2001). During the last 10 years there has been serious effort by Government and NGOs to revitalise, repair and improve traditional water-harvesting systems.

TAANKAS AND RAINFALL HARVESTING

A simple calculation shows that 100 mm of rain falling on I Ha of land is 1 million litres of water enough to give drinking and cooking water annually to 273 people at 10 litres per day. Field trials in the Negev Desert demonstrate the importance of using small water catchment areas to maximise the efficiency of rainwater collection. (Evani, 1971). The average Indian village needs 1.12 Ha of land to capture the 6.57 million litres of water it will use in a year for cooking and drinking (Agarwal, (2000)). In the Thar Desert the land area required varies from 1 to 5 Ha depending on rainfall and size of village.

Taankas are underground storage tanks usually constructed of concrete and stone (Figures 2 and 3), harvesting rainwater generally providing drinking water for a family or group of families. It is a system where rainwater from roof top, courtyard or natural or artificially prepared catchment is directed to an underground tank.

In the Thar desert, since ancient times water was collected in underground water storage tanks for human drinking water purposes. In the rich family homes water was collected from the roof whilst families without a tile or concrete roof resorted to collecting water from the land. (Vangani et al., 1988). The traditional taanka suffered from seepage, evaporative loss and pollution. At the Central Arid Zone Research Institute (CAZRI) in Jodhpur, the design of taanka was improved. (Ahuja et al., 2000). Their design of a 21,000 litre capacity tank was to provide drinking and cooking water for a single family of 6. They calculated that, for a dependable annual rainfall of 130 to 250 mm (at 60% probability), a catchment area of 420 to 780 sq.m was required.

Figure 2. Photograph showing family taanka with apron for harvesting water

Figure 3. Photograph showing close up of taanka and extraction hatch

Wells for India working with the NGO GRAVIS, Jodhpur have been helping the poorest families to build taankas with the ultimate aim that every family in a village should have drinking and cooking water. A typical village has 200 families and population of 1000. The target is to provide the 100 poorest families with taankas .

Wells for India built the first taanka in Bhalu village during the years 1996 to 2001. 40 taankas were built giving water to 70 families. Then in January 2002 a project involving a cluster of 7 villages was started in Baap Block in the north of Jodhpur District. A field programme was then established in the remote area of Pabupura (395 sq. km), comprising 105 scattered hamlets and 1,139 households with a population of 7775 and 17,346 livestock. In the summer months April – June, when all the ponds are dry, it is 40 to 60 Km to Baap where the nearest potable drinking water is available.

In autumn 2003 a new project was started in Chopra Dhora, an even more remote and socially deprived community with the aim of improving village ponds and providing taankas especially for the benefit of women and children. The target cluster of 6 hamlets covers 151 sq. km., 531 households with a population of 3,186.

Up until mid 2005 the following taankas have been built with associated benefits in the Pabapura cluster of seven villages:

- 28 community taankas supporting 219 families, including 300 school children
- 300 household taankas (85% belonging to the poorer schedule caste and Scheduled tribe section of society.)
- 80 household taankas repaired
- A group of 16 family owned taankas built for the landless families on community land.
- Roof water harvesting for primary schools, 4 taankas for 4 schools.
- Experimental work to harvest water from courtyards into taankas for use for animals and horticulture
- In 4 villages, 20 families have planted 5 different varieties of fruit trees. 50% of the trees: desert plum, citrus fruit, pomegranate, lemon and plum are surviving more than 2 years.

SOCIAL BENEFITS OF TAANKAS

A case study serves to illustrate the changes in people's livelihoods brought by the taankas.

Case Study of Mrs Dhudi Devi

The Dhudi family are a typical poor family living in Cackhoo village near Pabupura. 25 years old Mrs Dhudi has a daughter and two sons. The husband owns the 3.2 Ha of land of which more than half is sandy. They own one cow and three goats. Mr Dhudi migrates in October when the food and fodder crops have been harvested. He returns early July in anticipation of monsoon rains. A 25,000 litre taanka was constructed near their house; Mr Dhudi provided his labour free. The family daily water consumption is 174 litres and the taanka in a typical year harvests rainwater sufficient for xx months. Rather than just use the taanka for family drinking water and watch their animals die, they have decided to rely on the taanka to meet the daily needs of themselves and the animals. For five months of the year there is also water in a nearby pond suitable for the animals and domestic use. In addition 2 Km away is a small ground water tank which is filled by government pipeline; it has water in it once in 7 days. In the summer the filling is less frequent. When the harvested rainwater is exhausted the family refill the taanka by purchase of water at a cost of Rupees 4,000 (£50 or $90). Before having a taanka they relied on a dilapidated small 250 litre tank and spent Rupees 10,000 each year on water.

Field experience is that families are using the taankas for much more than the drinking water needs of the family. Their dilemma is that besides water for drinking and cooking there are domestic water requirements and sources of water for the animals may be distant. Drinking water for the family may be the priority but the animals provide livelihood and without livelihood they cannot remain in the village. When the local sources of water are dry they would depend on the camel and tractor drivers to collect water. Having a taanka to collect and store water does mean they pay less and can guarantee to have water for their use every day of the year.

Case studies show that when families have taankas, the woman's day is transformed. She no longer has the weary search for water. Because she knows the water is there, the psychological transformation is remarkable. (Grey, 2003). Poor drinking water quality is a major reason for infant and child mortality and without taankas, women are forced to walk long distances to collect polluted water. The shortage of water causes personal hygiene problems resulting in skin diseases and amongst women widespread leukorrhea.

The very low female literacy in the villages of the Pabupura and Chopra Dhora clusters of 1 to 2% (4.06% for the whole of the Baap Block) is due to the heavy responsibilities put on the girl child and the low importance placed on education of girl children. She may attend school for a year or two but soon drops out. Provision of water is proving an important factor to increase numbers of girls going to school.

Education of children about the importance of water, how to maintain water structures and how to use water to improve health and hygiene are proving important.

Water provision is proving an important entry point or catalyst for growing community involvement to better lives in the villages. The formation of a Village Development Committee drawn from all sections and the formation of Women Self Help Groups is the direct result of starting community water projects. This organisation helps the process of holistic development, ensuring sustainability of the water based projects as well as other parallel initiatives.

DISCUSSION

The erratic nature of the rainfall means that in some years no rainwater enters the taankas. In such cases they are filled by tractor tankers or camel carts in the period immediately after the monsoon season when water is more accessible and cheaper to acquire. Where and when there is a pond nearby with water, the women may carry the water to augment the filling of the taanka.

Field tests of rainwater collected during the monsoon season suggest that about 60% of the rain enters the taankas. This compares with data from CAZRI, Jodhpur that show average run off during a four-year period varying from 29% for an untreated apron catchment surface up to 65% for the best-sealed surfaces (personal communication). In Bhalu the total rainfall for 2001 was recorded as 109.5 mm compared with average rainfall close to 300 mm. The 22,000 litre (22 m^3) taankas with an apron of 120 m^2 on average collected 6.1 m^3 water. A decision was made to increase the size of the aprons.

The Pabupura cluster of 7 villages has an average rainfall of 270 mm of which 230 mm falls during the monsoon season. The aprons are 337 m^2 and in an average year should collect 46 m^3. However, during years of drought the taankas will not fill sufficiently to fully meet the needs of the family for drinking water. In the Pabupura cluster of villages it is found that after rainfall efforts are made to top up the taankas from close-by sources of water while they last.

Each household taanka costs approximately £200. A typical village has a population of 1000 incorporating 200 families of which 100 are too poor to finance their own taanka's construction. A typical village requires 5 taankas for schools, 5 community taankas, 50 household taankas to be built and 15 taankas to be repaired at a total capital cost of around £13,000. Add to this £3,000 costs for training, capacity building and education before during and after construction to give a total of around £15,000 per village.

Where there are no locally available materials for constructing the taanka aprons, materials need to be brought in by tractor and wagon at higher cost. It is estimated that across the Thar Desert there are 20 million people of whom 17 million live in villages. Two thirds of the villages could benefit from taankas at a total cost between £200 and £250 million.

CONCLUSIONS

The provision of a taanka to a family transforms their lives. In years of poor rainfall it will not fill completely. It then does provide storage and allows the family to fill the taanka with water when it is more plentiful.

Further work is required monitoring over a long period the efficiency of the rainwater catchment for different apron surfaces. Work is required on the cost and benefits for different parts of the desert with lower and higher rainfall. Further experience is required in order to develop a model that can be replicated across the Thar Desert. Future priorities for R and D may be identified:

- NGOs with ability and interest to initiate water projects have proven essential to generate the community involvement, thus creating sustainable village-based water management. There is a shortage of quality NGOs prepared to work in the harsh environment of the Thar Desert.

- Government money has been used to build taankas for government schools. The employment of contractors and the low involvement of the community has lead to poor quality of construction and poor maintenance thereafter.

Ways have to be found to involve the village community in all stages of Government financed community water projects.

• The availability of local materials for construction of taanka aprons and further research into the efficiencies of different surfaces are required.

• Families need water for their animal and for their domestic use. Developments of courtyard water harvesting and the researching and optimisation of other means to harvest and store water is required.
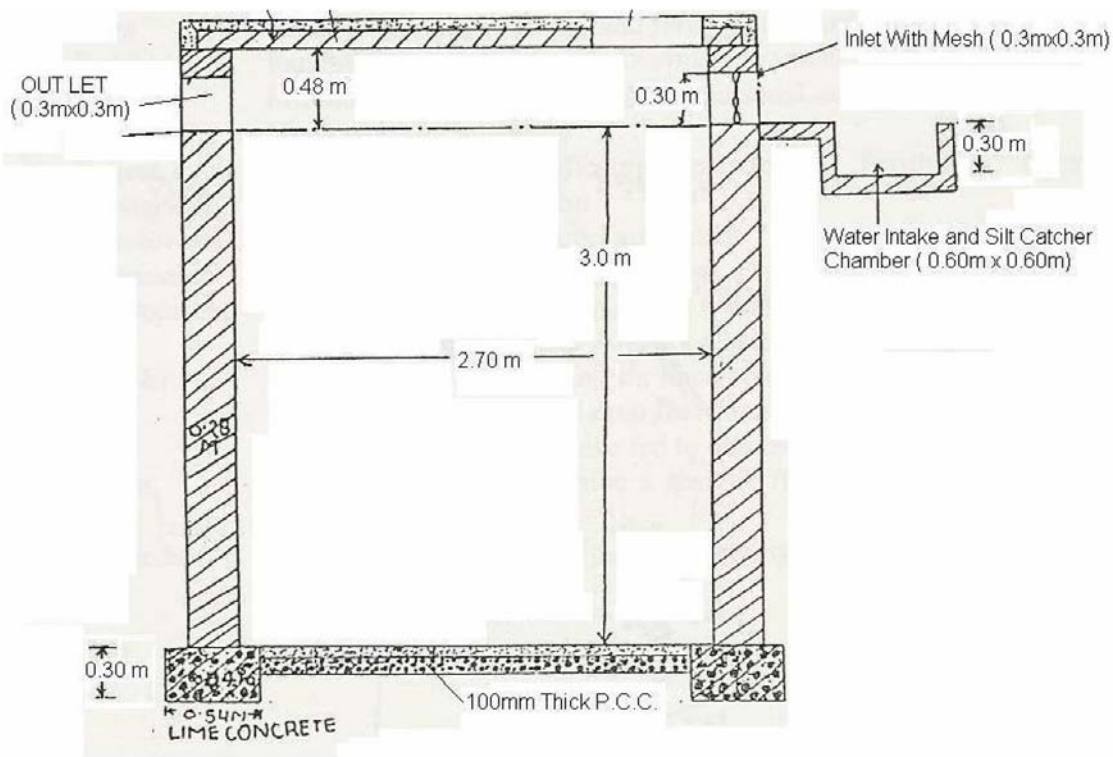
Figure 4. Cross section of typical taanka.
3 metres in diameter, 3–4 metres deep, the major part below ground level Typical water volume 20,000 litres
with a water harvesting artificial catchment apron of 15–25m diameter, 3% slope

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Approximating technical effectiveness of low technology aquifer recharge structures using simple numerical solutions

Ilka Neumann, John Barker, David Macdonald and Ian Gale

Abstract

This study describes numerical approaches to quantify technical effectiveness of low-technology artificial recharge structures e.g. small check dams and spreading basins, as seen commonly in rural environments in arid and semi-arid developing countries. The described methodologies enable effectiveness of artificial recharge structures, i.e. their ability to replenish the aquifer, to be assessed. Technical effectiveness of recharge facilities is thereby evaluated on two scales: impact at recharge structure scale and impact of artificial recharge from structures on aquifers. Emphasis is placed on simple numerical solutions to evaluate technical effectiveness, which can be applied locally without detailed knowledge of artificial recharge processes. The methodologies presented have been used to assess the impact of artiificial recharge structures at three case study sites across India.

Keywords

Analytical solution, numerical modelling, effectiveness, low-technology recharge structures.

INTRODUCTION

In recent years India has seen a substantial rise in groundwater abstraction to meet the rising demand for water for domestic supplies and irrigation. Concerns are being raised about the sustainability of groundwater resources and the livelihoods it supports. One of several measures adopted in India to address problems of groundwater depletion is the promotion of the use of water harvesting structures to increase recharge. A range of facilities from traditional to sophisticated are employed; for a detailed description of recharge structure types the reader is referred to United Nations (1975), Huisman and Olsthoorn (1983), O'Hare et al. (1982), Gale et al. (2002) and others. Considerable investment and effort are required to restore and maintain such facilities in India, however no systematic evaluation of their technical performance has been carried out. The impact of such structures are often anecdotal and their overall performance is currently not established.

While the overall effectiveness of artificial recharge schemes is governed by a variety of factors such as climate, hydrogeology, source water availability and quality, operational and management issues and socio-economic considerations, the study reported on here focuses on the technical aspects of the effectiveness of low-technology recharge facilities, i.e. the ability of the structure to recharge the aquifer. It is part of a larger study on the 'augmentation of groundwater by artificial recharge' (AGRAR), in which the impact of recharge structures in three locations in India is assessed: Coimbatore in Tamil Nadu, the Kolwan Valley near Pune, and the Aravalli Hills in Gujarat. The sites are located in different hydrogeological environments. Recharge structures in Coimbatore are situated on alluvial deposits on basement rocks, structures in the Aravalli Hills are on weathered basement rocks while the Kolwan Valley is situated within Deccan basalt.

METHODOLOGY

The technical effectiveness of the recharge structures is evaluated at two scales:

1. Impact at recharge structure scale;
2. Impact on the groundwater resources in the local aquifer.

Emphasis is thereby placed on the development of simple numerical solutions to help evaluate technical effectiveness, which can be applied locally without detailed knowledge of artificial recharge processes.

Impact at recharge structure scale

At the recharge structure scale, the decline in water level in the structure is an indicator of the performance of a recharge structure. The water balance for the structure can be simplified during periods of no precipitation and when surface inflow and outflow can be neglected. A further simplification can be made if losses due to leakage, direct abstraction etc. can be neglected, and if the structure is under effluent conditions in relation to the aquifer. With these simplifications the water balance can be summarized as:

Infiltration = Change of volume of water in the structure - Evaporation

Under such conditions, the balance between evaporation and infiltration will determine the effectiveness of the artificial recharge scheme.

The change of volume in the reservoirs at the Indian case study sites is monitored by recording the change in reservoir water level with time. For periods without direct abstraction and rainfall, this is translated into infiltration rates after subtracting estimated open water evaporation rates. Assuming the overall change in unsaturated zone storage is negligible, this infiltration is equal to groundwater recharge.

Impact of artificial recharge on the groundwater resources in the local aquifer

Besides estimating groundwater recharge from structures, the distribution of this additional water recharged is important in determining its availability to the user. This was assessed by developing both analytical and numerical models. Setting up of data intensive detailed numerical models to evaluate site-specific recharge schemes is not always feasible, especially for low-technology schemes, e.g. small check dams, in developing countries. Emphasis was therefore placed on the development of a simple numerical tool to allow the order-of-magnitude impact of artificial recharge to be examined and visualized for aquifers of various hydraulic properties. The aim of developing an analytical model was to provide a tool for assessing the impact of artificial recharge structures for users not familiar with numerical modelling codes, such as MODFLOW (McDonald and Harbaugh, 1988). It allows users to investigate homogeneous aquifer systems and their responses to recharge using Excel spreadsheets. The numerical model was used to verify the analytical code by simulating responses of homogenous aquifers to recharge events and comparing model results. Additionally, it was used to simulate the impact of artificial recharge on non-homogenous aquifers and to incorporate abstraction from boreholes. The analytical and numerical models allow:

*   Estimation of expected water level rise following recharge events within various hydrogeological settings.
*   Assessment of the likely impact zone of a recharge structure, i.e. the area in which a rise in water level is experienced.
*   Simulation of the dissipation of the groundwater mound with time.

The analytical solution and numerical model

To facilitate the estimation of effects of radially-symmetric patterns of pumping from a homogeneous aquifer, an analytical solution was introduced by John Barker of University College London (Macdonald et al. 1995, 1998). Subsequently this has been further developed. The solution forms the basis for the spreadsheet model used within this study to investigate the impacts of recharge to a homogenous aquifer, determining water level rises and volumes, respectively. The model assumes an isotropic, homogenous aquifer, with a recharge structure of radius R1 situated at the centre. A schematic diagram of this set-up is shown in Figure 1.

Recharge is distributed uniformly within the recharge structure at rates specified in $m^3/d/m^2$. The recharge rate is held constant over specified time periods. The model then calculates water levels in the aquifer and volume balances. Water levels are calculated at user-specified distances from the centre of the recharge structure. The aquifer can be treated as infinitive or a no-flow boundary can be defined at a radius equivalent to the extent of the aquifer. A numerical model was set up using MODFLOW, which is able to simulate radial flow outwards from the recharge structure similarly to the analytical model. The model is bounded by general-head boundaries with the same groundwater head all round and recharge is applied using a constant flux boundary condition.
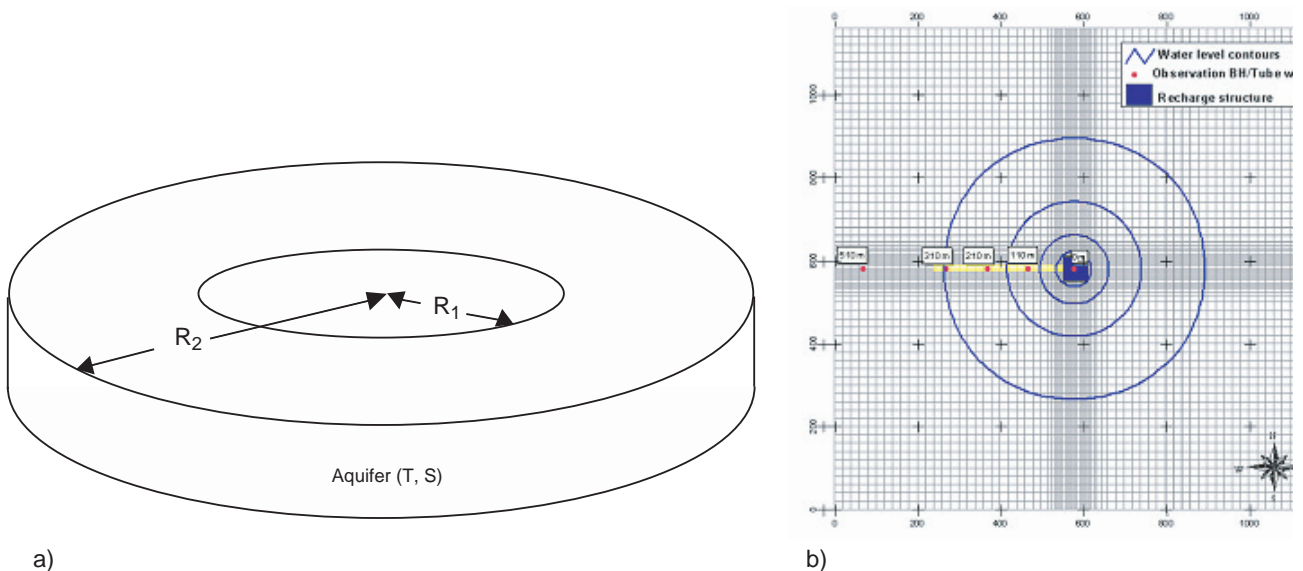
Figure 1. a) Schematic diagram of the analytical model set up. R_1 = radius of recharge structure, R_2 = radius of aquifer; b) schematic diagram of the set-up of the numerical MODFLOW model

RESULTS

Impact at recharge structure scale

Water levels in recharge structures were measured over a period of nearly 3 years at the three Indian case study locations. In addition, surface water inflow and outflow and evaporation from standing water in structures and direct abstraction was recorded. Preliminary results show water level decline rates in the structures vary widely. Decline rates as low as 3.5 mm/d suggest some reservoirs to be highly inefficient, acting basically as evaporation pans. Rates as high as 51mm/d for other reservoirs suggest considerable infiltration (Figure 2).

The measurements of water declines in the case study areas show, that these can be useful indicators for the reservoirs efficiency in recharging the aquifer. Generally, the faster the water level declines, the less water is lost to evaporation and the more efficient the reservoir is in recharging the aquifer. If the unsaturated zone is not limiting the rate of infiltration, a steady downward flux to the water table occurs and a linear decline in reservoir levels is
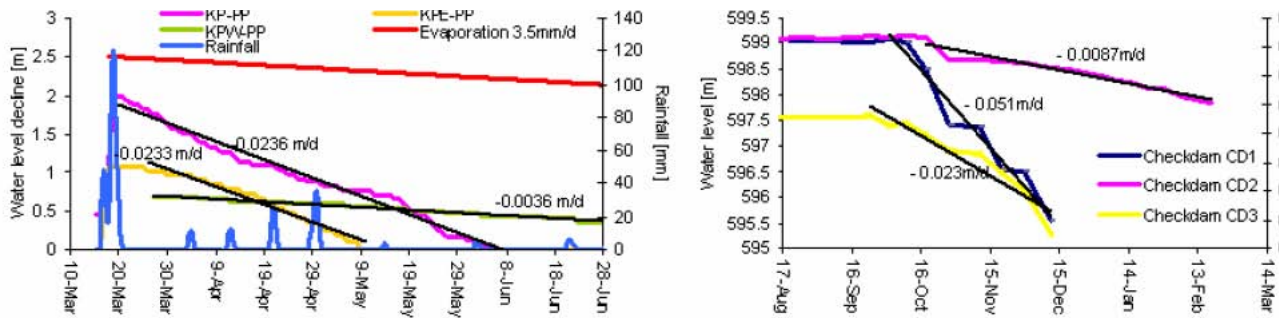
Figure 2. Examples of reservoir stage declines in three reservoirs in the Kodangipalayam watershed in Tamil Nadu (Palanisami et al. 2004) and in three reservoirs in the Kolwan Valley near Pune (Kulkarni et al. 2004)

observed. Factors, such as moisture content in the unsaturated zone, clogging of the reservoir bed over time, decrease in the reservoir ponding depth or a rise in groundwater table influence the observed stage declines. A summary of decline rates over time and their causes is shown in Figure 3.
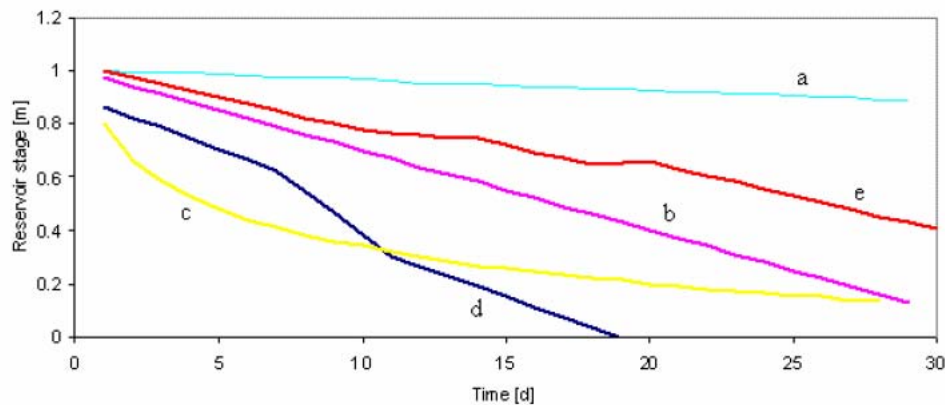
Figure 3. Summary of reservoir stage decline rates

Impact of artificial recharge on the groundwater resources in the local aquifer

Various analytical and numerical simulations were carried out to establish the likely impact of recharge events on aquifers with different properties. In the examples shown here, the size of the recharge structure and its recharge rate have been chosen to be within the range observed typically in the Indian case studies. The structure simulated is 55 m by 55 m in size and recharges at a rate of 48 mm/day for 30.4 days (1 month). The transmissivity and storativity of the aquifer is varied in the model to cover a range of hydrogeologically plausible aquifer settings from T = 15 m^2/d to T = 1,500 m^2/d and specific yield Sy = 0.05 to Sy = 0.2 (scenarios Sc1 to Sc4). The effect of inhomogenties, such as fracture zones or high permeable weathered zones, on water levels over time was investigated by introducing a high permeable zone (K: 50 m/d) of 15 metres width into aquifers with transmissivities of 30 m^2/d (Sc6) and 15 m^2/d (Sc5). Model results are presented in Figures 4a and 4b.

The water level in the aquifer rises to its maximum underneath the recharge structure at 30.4 days, when recharge ceases (Figure 4a). The maximum is thereby dependent on the permeability of the ground, with highest water levels being achieved for the lowest conductivities and smallest effective porosities. The maximum water level built-up does not exceed 2.6 m underneath the recharge structure for any of the modelled scenarios. This decreases under homogenous aquifer conditions to maxima of below 0.4 m at a distance of 110 metres from the centre of the recharge structure (Sc1 to Sc4). For high permeable aquifers (scenario Sc2) the maximum height of the recharge mound remains in the order of a few centimetres.
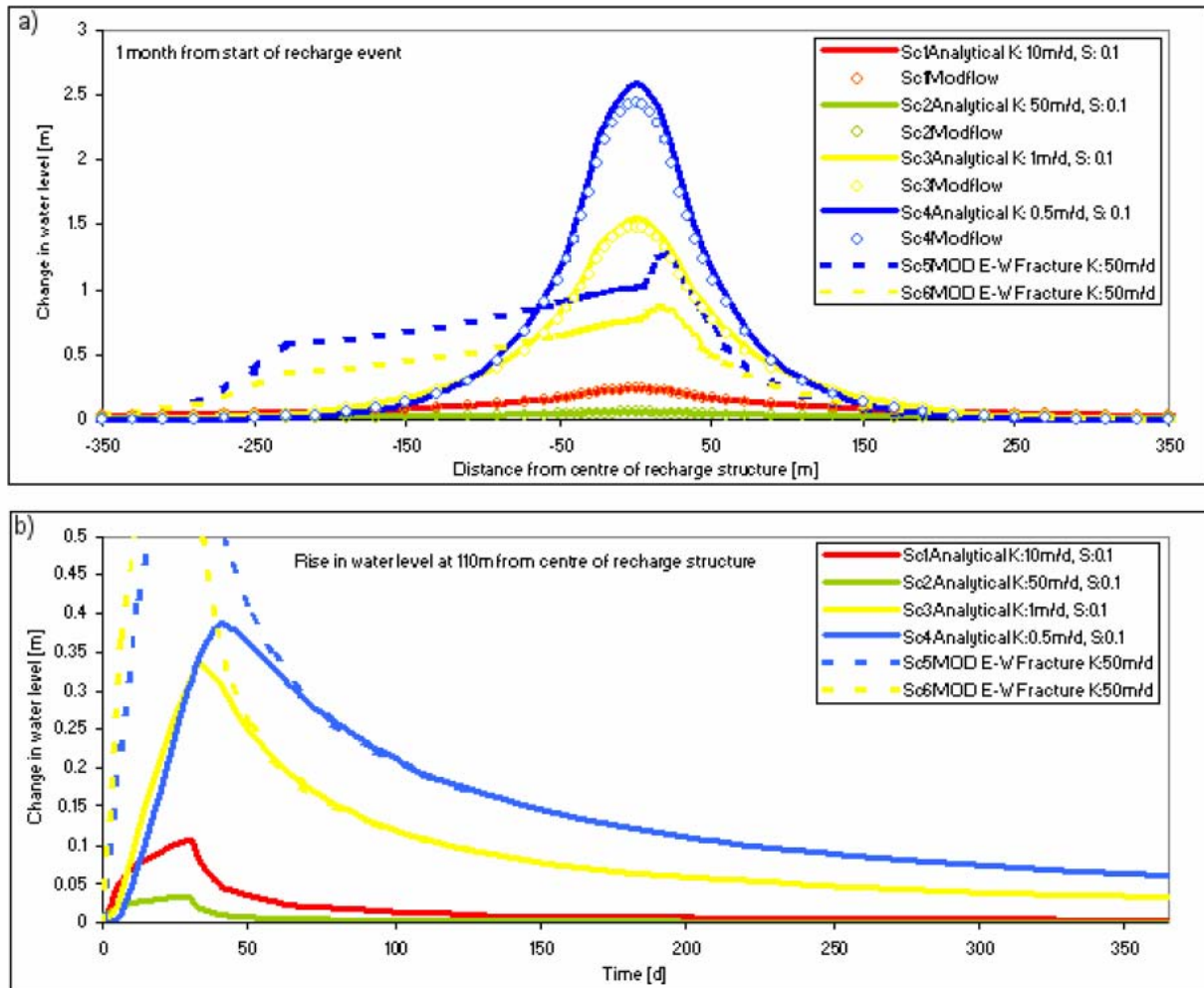
Figure 4. Change in water level due to a one month recharge event with time, a) Water level with distance from the recharge structure, b) levels are shown at 110 m distance from the centre of the recharge structure with time. Data labeled prefix.*Analytical and* prefixMOD *are generated using the analytical and MODFLOW model respectively.*

The recharge mound subsides quickly underneath the recharge structure by spreading radially outwards to occupy an ever increasing area of the aquifer (Figure 4b). Subsequently, water levels fall sharply in the vicinity of the recharge structure over time but increase radially outwards in the aquifer. This increase however is small in all modelled homogenous aquifer scenarios, due to the recharge volume being small compared to the volume of the aquifer the water occupies over time. Maximum water level rises outside the immediate area of the recharge structure remain in the order of centimetres for all modelled scenarios.

The effect of inhomogentities in the aquifer is to channel recharge water instead of spreading it evenly over a large aquifer volume as in the case of the previously discussed homogeneous radial flow scenarios. The result in the simulated high permeable weathered zone (Sc5 and Sc6) is an increase in water levels to several times the height observed under radial flow conditions in a homogenous aquifer, with elevated levels sustained over longer periods.

CONCLUSIONS

Technical effectiveness of artificial recharge structures was evaluated on two levels. On a recharge structure scale, the rate of infiltration in relation to evaporation was established. This determines whether the structure is fit for the purpose and can be approximated by measurements of water level declines in reservoirs during periods of no inflow

and outflow except for recharge and evaporation loss. In a second step, the area of benefit, i.e. the zone of impact of the artificial recharge structure was approximated, which is dependant on time scale and hydrogeological conditions at the site. This establishes the likely beneficiaries of the scheme. Conclusions drawn can be summarised as follows:

Recharge structure scale

- Decline rates serve as a good indicator for the reservoir's efficiency in recharging the aquifer. Generally, the faster the water level declines, the less water is lost to evaporation and the more efficiently the reservoir recharges the aquifer.
- Data from Indian case sites suggest decline rates vary widely. Decline rates as low as 3.5 mm/d suggest some reservoirs to be highly inefficient, acting basically as evaporation pans. Rates as high as 51mm/d for other reservoirs suggest considerable recharge.

Aquifer scale

- Within the range of parameters used in the modelling study, the impact of the simulated recharge structure on water levels is minimal for all but the immediate vicinity of the structure itself if aquifers are isotropic and homogenous. Water level rises remain in the order of centimetres, due to the recharge volume being small compared to the volume of the aquifer the recharge water occupies over time.
- The impact of a recharge structure increases the larger the ratio between recharge water volume and receiving aquifer body volume, i.e. the smaller the aquifer being recharged, the larger the effect. Preferential flow paths in the form of discrete zones of higher permeability can act as such zones, where recharge water is channelled resulting in a substantial rise in water levels, sustained over longer periods.
- The rate at which water levels subside after recharge events depends on the aquifer parameters with levels sustained longer, the lower the aquifer permeability.

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Impact of meteorological elements on groundwater level in Rostov region

Olesya Nazarenko

Abstract

Influence of the meteorological factors on formation of groundwater of the Don – Donetsky basin (south western Russia) is investigated. Correlation of meteorological elements and groundwater level is established. Influence of meteorological factors on a level of subsoil waters is appreciated.

Keywords

Groundwater, meteorological elements, recharge, Russia.

INTRODUCTION

Rostov region faces water shortage due to climate, geographical location and human activity. That is why water and especially groundwater is the subject of special interest. In recent years the significance of this problem has increased. Groundwater is the main source of fresh water in the region. It is used for not only domestic water supply, but also for irrigation.

The interaction between the land surface and atmosphere has been identified as one of the most important processes in climate studies. Long-term change of a groundwater level in different conditions is defined by climatic factors. Meteorological factors have the most effective influence on their formation and recharge. An air temperature, an atmospheric precipitation, evaporation, air humidity and also atmospheric pressure has an appreciable influence on subterranean waters here. Hydrological should be taken into the mind in the river valleys.

Many investigators have addressed their study to the problem of climate change and its influence on groundwater level. This problem has a leading role in the study of Kovalevsky et al. (1998). He gives a detail description of the interconnections of a groundwater level with some meteorological elements, such as precipitation, evaporation, temperature, and humidity.

Study area

The study site is the Don – Donetsky basin, located in the southwest part of Russia. It was used as a case study area for modeling the sensitivity of an aquifer to changes in recharge and influence of climate on it. This area situated in the north part of the Rostov region. There is no highly developed industry in this part of the region. The topography of the area is flat.

The main groundwater system in north part of Rostov region is composed of the sands, the depth of them is from 0,8 up to 35 m. The recharge changes widely. The main source of groundwater feed is precipitation and infiltration of Paleogene and Cretaceous aquifer. The practical meaning of the alluvial deposits is wide enough. They are used for domestic water supply and irrigation. Groundwater is of high quality, fresh, and both soft and of high hardness. Temperature of water is 8–10 degrees.

The climate is a continental with insufficient precipitation, high evaporation and annual temperature, and low humidity during the spring-and-summer period. The mean annual precipitation is 508 mm and the mean annual temperature is 7,4 º C.

The exploration of the area was carried out over the period 1954–2000 by the Rostov expedition. The investigation included geological mapping, estimation of groundwater level, and chemical analysis of water. Information about groundwater level was taken every third day, chemical analyses of water was made once a month.

RESULTS AND DISCUSSION

We analyzed the precipitation, temperature and groundwater level data from 1954 to 1993. Table 1 shows the variations of meteorological variables.

Table 1. Trends of temperature and precipitation in Rostov region

Observation (years)	Continuation of the observation	Trend	
		Temperature ($º C$ / year)	Precipitation (mm / year)
1954–1993	40	0, 21	11, 21
1954–1959	6	−0, 10	10, 50
1960–1969	10	−0, 08	10, 75
1970–1979	10	0	−0, 53
1980–1989	10	0, 12	20, 73
1990–1993	4	1, 98	7, 25

After examination of the average annual precipitation, we found out that the precipitation over average was marked in 1958, 1966, 1970, 1977, 1988, 1989, 1990, and 1993. The maximum annual mean air temperature during the measurements period was in 1955, 1957, 1962, 1966, 1971, 1975, 1979, 1981, 1983, 1989, 1990, and 1992. The exploration showed that increasing of precipitation leads to the groundwater raising (Figure 1).
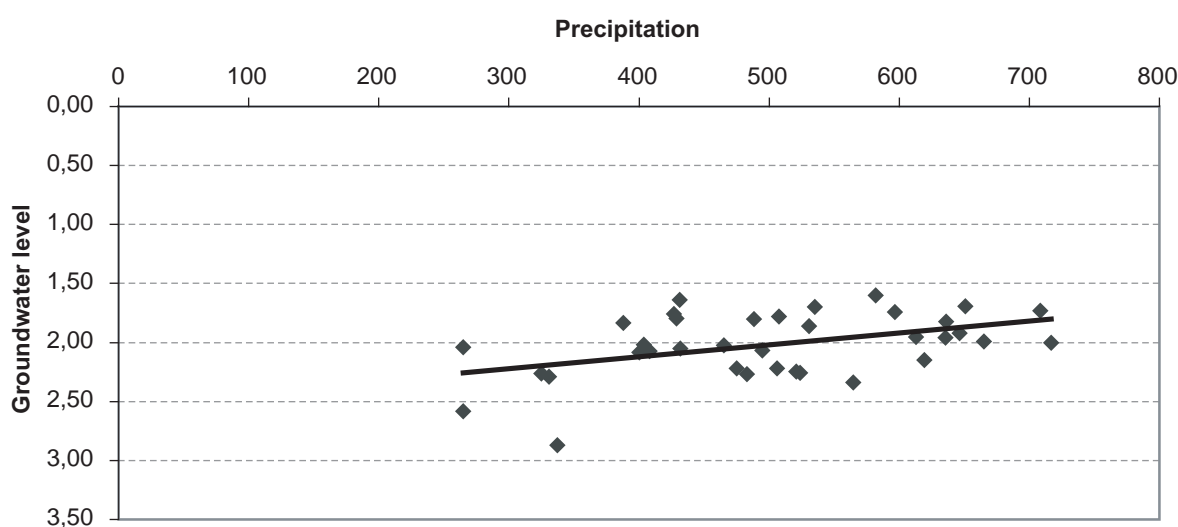
Figure 4. CD3: Water balance – wet season (2004) & dry season (2003)

In order to define the climate influence on the groundwater level, the analyze of precipitation, temperature, evaporation and humidity was held. Special attention was given to the experimental observation of three areas 1, 4 and 8. The difference between these areas is shown in the Table 2. In the table we can find data that shows groundwater level in the beginning and in the end of observation. It was noticed the groundwater rising from 0,15 m for 22 years of observation (low flood plain) up to 0,99 m for 41 years (1st river terrace).

Table 2. Attributives relative to the area of study

Area	Height (m)	Geomorphologic zone	Age	Aquifer	Time of observation	Groundwater level depth (m)	
						Beginning	End
1	77.46	First river terrace	aQ_{III}	Sand	1954-1994	18.28	17.29
4	61.72	High flood plain	aQ_{IV}	Sand	1954-2000	2.87	2.02
8	58.14	Low flood plain	aQ_{IV}	Sand, clay	1979-2000	1.32	1.17

Table 3 shows the results of correlation of the meteorological elements and groundwater.

Table 3. Estimated correlation

N°	Correlation	N° 1 Seasonal	N° 1 Annual	N° 4 Seasonal	N° 4 Annual	N° 8 Seasonal	N° 8 Annual
	P – L	– 0,28	– 0,09	0,33	– 0,44	0,38	– 0,20
	T – L	– 0,64	– 0,21	– 0,35	– 0,23	0,27	– 0,11
	H – L	0,80	0,16	0,78	– 0,21	0,31	– 0,23
	L– E	– 0,79	0,3	– 0,55	0,05	0,1	0,4
	L – P – T	0,42	0,14	0,66	0,52	0,52	0,23
	L – P – H	0,89	0,21	0,91	0,46	0,58	0,31
	L – T – H	0,88	0,32	0,84	0,26	0,94	0,1
	L – E – P	0,86	0,38	0,74	0,41	0,26	0,5
	L – E – P – T	0,88	0,41	0,81	0,51	0,75	0,8
	L – H – E	0,94	0,44	0,86	0,19	0,75	0,5
	L – H – E – P	0,90	0,44	0,86	0,42	0,83	0,66

(L – groundwater level, P – precipitation, T – temperature, E – evaporation, H – humidity).

The examination showed strong correlation between groundwater level and humidity. The plural correlation is higher and is equal 0, 94 for group level – humidity –- evaporation. Precipitation plays an essential role in groundwater recharge not only straightly but also through humidity. For the average long-term data factors correlation shows absence or insignificant interrelation. The low correlation of groundwater level with meteorological elements does not mean absence of the influence, and shows the influence of surface water and other factors on the water level. Meteorological elements influence on the groundwater level and give from 25% (seasonal) up to 65% (annual) of the recharge.

CONCLUSIONS

The region is situated in the best recharge conditions. Main source of groundwater feed is precipitation. Increase of the mid-annual temperatures (warming of winters) and precipitation resulted in increasing of the groundwater recharge. Climate plays a significant role in the groundwater recharge and precipitation is the main source of increasing groundwater level. The role of meteorological factors can be appreciated with the help of statistical programs. Groundwater recharge, caused by climate, differs from 25% (seasonal) up to 65% (annual).

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Conceptual approach of recharge estimation at the West Bank aquifers - Palestine

N. Salim and W. Wildi

Abstract

Understanding the processes and mechanisms under which groundwater reservoirs get recharged is one of the keys to water availability and water distribution. A relatively simple estimate of recharge as the difference between rainfall, evapotranspiration and runoff is not feasible in arid to semi-arid environments, because of the high evapotranspiration, runoff and low rates of recharge. Detailed studies are therefore required to improve quantification and a better understanding of the recharge processes. This fact is compiled with many complexities of recharge in the study area like:

- Rainfall is very unevenly distributed in time and space
- Geology is relatively complex
- The interaction between precipitation and runoff, over impermeable strata will influence recharge at the contact between the impermeable strata and the aquifer
- Steep hill slopes in many areas will allow rapid runoff generation during heavy rainstorms, so creating zones of intense recharge from wadi floors.

These complexity leads to significant localised inaccuracies in the estimation of recharge. This paper will suggest a new method for recharge estimation, which adopts remote sensing and GIS as tools for further analyses.

Keywords

GIS, Groundwater recharge, Palestine, Remote sensing, West Bank.

INTRODUCTION

The West Bank is highly influenced by the Mediterranean climate which is characterised by long, hot, dry summers and short, cool, rainy winters. Rainfall is limited to the winter and spring months. Climate within the relatively small area is affected by the general topographic trends which divide the West Bank into four main climatic regions: the Jordan Valley, the Eastern Slopes, the Central Highlands and the Western Slopes.

Rain and sometimes snow are the main sources of groundwater recharge. Groundwater in shallow, intermediate and deep-seated aquifers ranges in depth from tens to several hundreds of meters in lithological units ranging from Albian- lower Cenomanian to the Holocene.–Six aquifers in the West Bank are distributed within three main basins (Western, Eastern and Northeastern) lined by structural limits. The surface water is mainly represented by wadi-flow and structural springs, which seep their water from limestone or dolomitic limestone formations.

BACKGROUND AND RESULTS OF RECHARGE ESTIMATES FROM PREVIOUS STUDIES AND METHODS

Many authors have quoted the recharge to the aquifers of the West Bank, either as part of geological, hydrogeological or socio-economic studies and discussions, as a main input into aquifers. Most of the estimates have been based

on catchment scale water balances and/or empirical relationships between rainfall and recharge. The previous recharge estimations and studies of the West Bank are presented in Table 1 (McKenzie et al., 2001). These different studies simplified the case and considered recharge as a specific percentage of rainfall.

Table 1. Summary of previous recharge estimations of the West Bank (McKenzie et al, 2001).

Source	Area	Date	Annual recharge $(Mm^3 \ a^{-1}))$	Details of calculation
Rofe and Raffety (1965)	Northern (?) West Bank	1963/64	821	25.7% of rainfall
Rofe and Raffety (1965)	Northern (?) West Bank	1964/65	836	25.7% of rainfall
Israeli Hydrological Service (1997)	West Bank	Not specified	836	Based on Goldschmidt and Jacobs (1958); empirical model
Guttman (1995)	West Bank	Not specified	800	Not known

Other studies have been carried out at the basin scale and did not cover the whole West Bank. These studies considered the following aspects:

- The catchment scale water balance (Goldschmidt, 1955; Goldschmidt and Jacob, 1958; Rofe and Raffety, 1965; Guttmann, 1998).
- Groundwater models (Bachmat, 1995; Guttman and Zukerman, 1995; Camp Dresser Morganti (CDM), 1997; 1998; Ghanem, 1999).
- Hydrograph regression curves (Ba'ba, 1996). However spring hydrograph regressions on 3 spring systems within Eastern Aquifer Basin, do not produce a recharge estimate for the whole Eastern Aquifer Basin.

All these studies are based on methods to assess and calculate the recharge using the empirical models undertaken by earlier authors. The output of these methods does not reflect the reality and have a high uncertainty, given the lack of data or the type of data available.

RECHARGE PROCESSES

The main components of recharge model have been defined as follows:

- Rainfall recharge (direct recharge through soil and bare rock),
- Runoff and subsequent infiltration via wadi floors,
- Urban recharge processes,
- Agricultural processes.

NEW METHODOLOGY

The new model of recharge will be based on theoretical hypotheses and analyses, which mainly depend on understanding the hydrological cycle in the region. The model will be developed in the context of understanding recharge

processes acquired by other investigations, personal experience, and field surveys. This knowledge will be compiled and reform in a GIS database for further analyses.

The factors of recharge will be integrated and spatially distributed within a GIS model allowing for refinement and development of the model (Figure 1). This allows for calculating the recharge in different scenarios (climate, anthropogenic and land use) and with different contributions from leakage, waste water and irrigation returns to reflect the changing management of West Bank water resources. To integrate all these factors, the recharge model has to be broken down by three stages, the first of which is–concentrated on information level where the basic data within the GIS system is held as vectors; points, lines or polygonal regions as appropriate. The second stage includes gridding the maps and converting them into raster layers, the data are transformed to a rectangular grid for spatial distribution of recharge parameters. This allows for a simple implementation of arithmetical relationships between topography, geology, climate and recharge. Analysing all the geomatics information and reclassify them in a hydrological concept, such as transferring the geology into hydrogeological concepts and maps; Elaborating of digi-
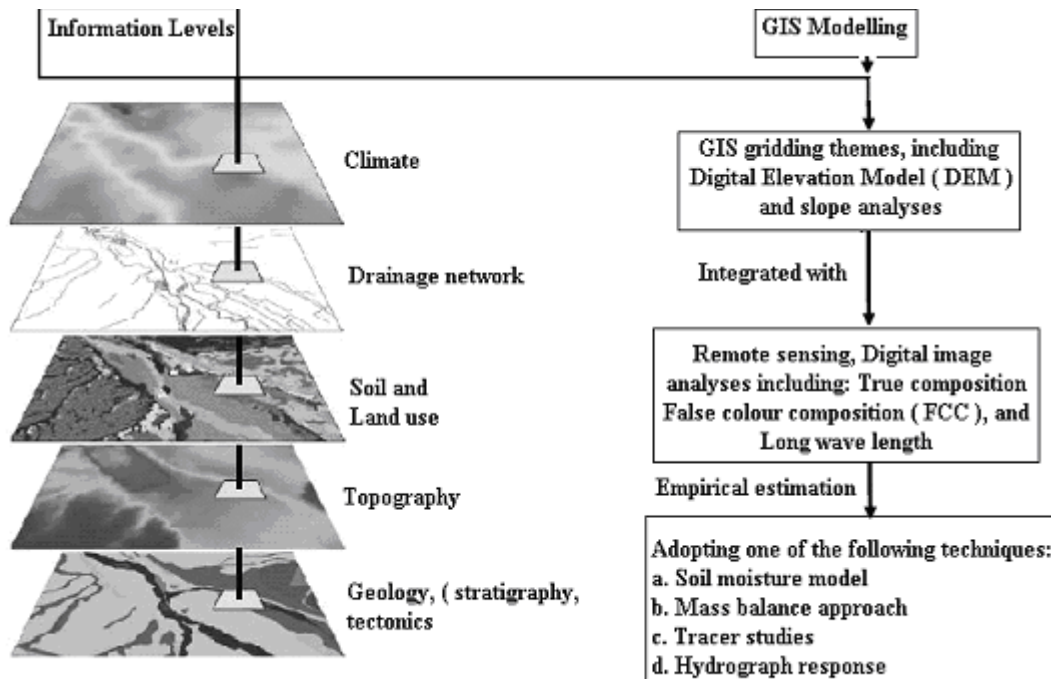
Figure 1. Schematic diagram of recharge modelling applying GIS and remote sensing.

tal elevation models (DEM); hill slope and Combination of information levels (e.g. land use, topography, drainage network, etc). The third step will adopted the GIS multi-criteria analyses base on recalculation of different factors.

The overall approach will start on a simple basis and increase the complexity as data and understanding allow. As data and understanding increases, the complexity of the calculation of recharge can be increased within each object. The initial selections of objects are as follows:

- Rainfall and evapotranspiration generation objects,
- Digital elevation model generation object allows slope analyses
- Grid object allows:
- Runoff routing to wadis,
  - Sub-surface routing; sewers to wadis, interflow, karstic features,
  - Recharge nodes,
  - Soil recharge processes, including irrigation losses,

- Urban recharge,
- Wadi channel infiltration,
• Unsaturated zone object.

The recharge will be calculated at a node object; each node object is held within a grid, which will determine the routing of runoff. As a consequence of the development in geomatics, (GIS, GIS-modelling and remote sensing) recharge estimation may now consider all the available information levels and their spatial distribution. The implementation of this approach may allow an improved estimation of quantitative values of recharge in different zones and point locations. These different themes will be gathered, treated, analysed and integrated with scenario analyses, mainly on climate variations and anthropogenic factors.

DISCUSSION

The (West Bank) aquifer basins boundary is not well defined. Despite the fact of information we got from water level hydrographs, structural geology and the interpretation of the relation between the aquifers. The degree of interconnection between aquifers remains far from well established. Clearly, a better understanding of these boundaries is required for both physical and chemical flow modelling. This interconnection between the aquifer units gives a high uncertainty for recharge assessment. To be able to overcome this uncertainty a detailed further study is still needed. The main factors in the regional context are the following:

a. Distribution of major karst and tectonic recharge pathways,
b. Distribution of land uses and land use change,
c. Spatial and time distribution of rainfall in relative with topography,
d. Distribution of bare rock and soil cover.

Covering these numerous information levels and parameters requires a wide variety of data, which is not available. Satellite image analyses and GIS could be used as tools to reduce the gap in data.

Relations between surface water and groundwater will be difficult to assess in some areas because of the complicated interaction between the pathways. Tracer studies might give a good value for this specific problem, but it is still very limited.

The new geomatic based recharge model suggests dividing the area of the West Bank into different zones of recharge (Figure 2). This is a result of the multicriteria analyses where the area is divided naturally by structural geology into grabens and forces the groundwater to follow a specific path, in addition to the climatic parameters which is taken into consideration.

CONCLUSIONS

1. All previous studies have given a non certain image of recharge, and in particular the methods based on groundwater flow models, given the lack of data and the insufficient knowledge of the groundwater flow in the area that is highly influenced and oriented by structural geology. The GIS model suggests solving this problem by subdividing the area into local units with specific infiltration rates and recharge characteristics. This method might appear complicated, but it could give better results as far as we deal with highly fractured area.

2. Furthermore, the data obtained from regional monitoring programs do not always reflect the real hydrological and hydrogeological situation. The implementation of a groundwater recharge modelling based on GIS-techniques needs a wide variety and good quality of data, which are not always available. Some of these data could be obtained through satellite image analyses such as structural geology and land use.
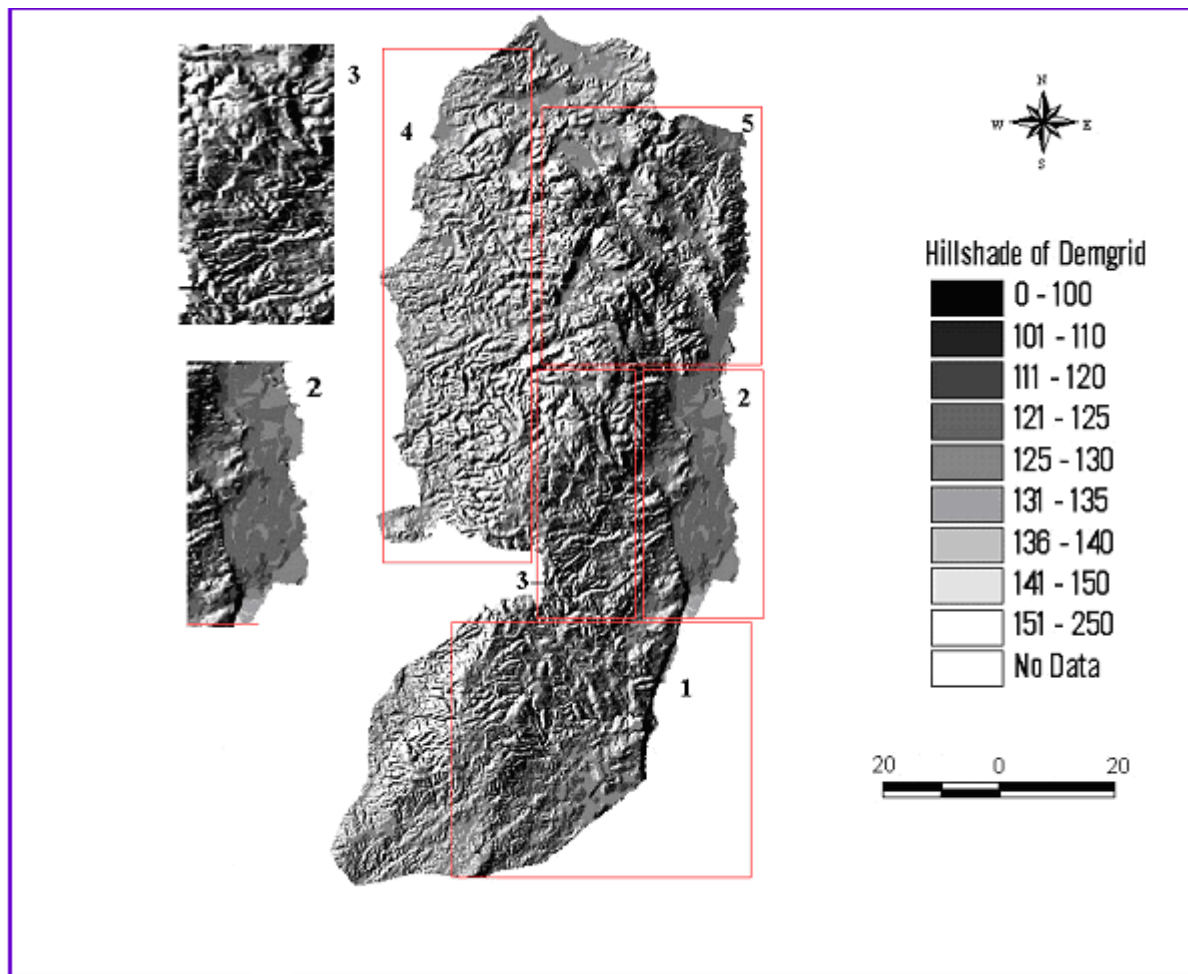
*Figure 2. Example of information level "topography", hill slope analyses of West Bank
from digital elevation model, subdivision into recharge analysis main zones*

3. The relation between the wadi flow and the groundwater is not well established. The former is considered as one of the main sources of recharge and it might be considered as the main source of recharge in summer. Hence, there is no rainfall in summer and all recharge comes from the wastewater flowing through these wadis. This type of recharge is the most important, as, on one hand it compensates for the lack of water quantity and, on the other hand, it aggravates the quality of water by adding toxic substances to the aquifers.

4. Differences in topography and their interaction with the different geological outcrops have not been taken into consideration. Slope analyses and hills shade analyses would contribute effectively in spatial calculation of recharge.

5. The time variant and space data sets that are available, or could be generated from satellite images and GIS include topography, geology, soil cover, land use, precipitation, other climate parameters, and water allocations will thus be necessary to elaborate a reasonably robust model to link recharge estimates to all important related factors as well as to the available data.

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A case in traditional ingenuity: 'Khatri' system of water management

Ashok Sharma

Abstract

Objective

This paper seeks to examine the ingenuity, efficacy, utility and current relevance of a traditional knowledge based water harvesting and aquifer recharge system used by the mountain communities in northern hills of India for centuries and demonstrate its potential for transplantation in other similar parts of the globe.

Methods

Khatri is a unique time-tested combinatorial system of community based rain water harvesting and recharge based on indigenous knowledge, self regulation and community based management that has sustained rural communities scattered across dry calcareous conglomerates of peripheral Shivalik hills adjoining the Dhauladhar Himalayas in India for centuries. Even today, the system exists as lifeline for people in typically water starved Hamirpur hills of Himachal Pradesh with first recorded implementation of modern piped water supply dating as recent as late seventies. The system is entirely based on traditional ingenuity with strong tradition of property rights and role of women in the management process. System manifests high-caliber traditional knowledge of design & construction assuring adequate water quantity and quality with least-cost ideally suited for mountain communities.

Results

Innovative aspects include rainwater harvesting based on scientific utilization of natural bouldery strata for retention and percolation of rainwater seeping into manmade rock-cut cavities in sandy layers for recharge of aquifers and storage of pure drinking water. Sandy layers are used for filtration using natural properties of calcareous formations for water quality preservation while covered storage ensures prevention of evaporation losses and protection against contamination. Upstream pollution is prevented through community regulation of land-use in the catchments and human activity is restricted to downstream areas. The system can be created locally with indigenous tools employing locally available unskilled workforce at a very nominal cost and serves perennially without need for special maintenance skills other than periodic cleaning and repairs.

Conclusions

The system has served the needs of the mountain communities over centuries meeting satisfying the water quality and public health norms at least cost and is particularly suitable to offer decentralized solutions for land locked mountain communities across the developing world very reliably and cost effectively.

Traditional ingenuity based indigenous systems though fast disappearing with onslaught of modern technologies, often surge back during times of distress, emergencies and system failure. A case study of age old traditional system of water harvesting to meet the sustenance needs of rural communities in land locked mid hills of Himachal Pradesh in India reaffirms this hypothesis. Considered to be a traditional system of rainwater harvesting and storage in the sub-Himalayan region of India, these 'horizontal wells' in the Kangra region of Himachal Pradesh are a unique example of indigenous knowledge and innovative skill. A typically local response to prevailing needs and conditions, these wells stand firm in their promise to provide drinking water while water taps run dry for most part of the hot summer months. Interestingly, the qualitative aspect of water from this traditional system has a definite edge over the public distribution system.

A careful examination reveals that Khatri system is unique to pockets in Kangra and Hamirpur districts in the mountainous state of Himachal Pradesh where most hills are made up of coarse-grained sandstone and quarto feldspathic rocks. Although annual rainfall in this region is as good as 2200 mm, most of it occurs during the short season of monsoon months and water scarcity is an acute problem for remaining part of the year. Khatri has stood out as an innovative and sustainable answer to these conditions and Khatris situated at the foothills store water that filters down through the near porous rocks to accumulate in the hand carved reservoirs.

Constructionally, Khatri is a cave-like structure dug horizontally along the hill located after a careful survey based on experience of the local skilled workers. The mouth of the cave is wide enough for human entry, with limited operational space inside. The basin of the well is often a couple of feet deep to facilitate storage of water that trickles from all sides. A typical khatri can store as much as 30 to 50 kilolitre of clear water. In local parlance, Khatris are known to mean two different types of systems. One belongs to the drinking water regime for personal use of the communities while the other is meant for use of cattle and household utilities. The latter is often based on roof top collection of rainwater and subsequent storage into hand carved cave like storage structures. This paper seeks to investigate in details the system in vogue for drinking water storage and its scientific rationale and relevance.

Khatri is a product of unique ingenious socio-engineering both in terms of concept as well as constructional features. It is interesting to note that Khatris are traditionally typical to a specified and isolated zone located at heights ranging 2000 to 4000 feet above the mean sea level. It may also be seen that their location is typical to the Dhauladhar formations that comprise calcareous sandstone and are located along a clearly defined flow pattern that further seems to guide the process of human settlement along these isolated hamlets. Not only do communities know the exact location in the hills where a khatri could be dug, they precisely regulate its size and design. On a particular hill, the khatris are dug at different points but at the same elevation. A new khatri cannot be dug at a level below the existing ones because the water in the khatri above will trickle down to the lower level. Similarly, the locations of Khatris are maintained above the levels where cattle are maintained so as to avoid contamination of water upstream of the harvesting systems and structures. No lining is done along the walls of the khatri as it may prevent the inward seepage of clear water.

Technologically, Khatri is a unique design endeavor that utilizes the geo-climatic conditions available in the region to create a low cost facility for not just harvesting and storage of rainwater but also utilize the geo formations as natural filter for ultra-pure water quality typical of ultra-filtration process. It also utilizes the basic character of the formation for maintaining softness of water and zero bacterial contamination on account of micro-porosity of formations. Same is true of other characteristics like odor and off-taste typical of other rural water supply systems in the third world. And all this is perennial with no costs associated.

Khatri is a unique catchment-based system with its own governance system, featuring individual as well as community ownership. From the inception, location, positioning, maintenance and ownership, everything is well defined with housewives at the helm of affairs. Over the years, modernization in favour of large engineering

structures and metallic pipes have failed to bypasses this system that has stood the test of time particularly in times of distress on account of water scarcity as well as sporadic incidences of water-borne epidemics.

Himachal Pradesh is the first State in India to enact legislation for mandatory harvesting of rooftop rainwater for all new buildings since the mid-90s. However, the focus of such legislation has a strong urban bias with focus on large

commercial and government buildings. Such initiative could have yielded positive results if the focus remained on revival of this traditional system through individual and community development schemes and financial incentives coupled with participative catchment-based governance at the village or community levels.

The *khatri* system has stood the test of time and as not yet lost its relevance to the people of Kangra, Mandi and Hamirpur districts who still maintain them in accordance with a well defined regulatory mechanism. Not only are old *khatris* still working, but communities continue to carve-out new ones. While community *khatris* are left open, family *khatris* usually have a small gate with a lock. Although most communities are supplied piped water through Government schemes operated by the Irrigation and Public Health department, a *khatri* is maintained as a strategic reserve that could steer them through a possible crises that is a never to be forgotten probability. Water crisis is common in this region where altitudes vary from 2,000 to 4,000 feet and scatter of population across isolated hamlets makes it distinctly difficult to ensure alternative supply and distribution of water. This is particularly so during the lean seasons or periods of acute shortage that might spread over five to seven months in a year. The system offers low cost sustainable alternative to groundwater reserves, as skilled and locally trained masons may be able to construct the self-maintained decentralized structures at an average cost of about Rs 15,000–20,000 each.

Roof-top rain water harvesting to recharge groundwater reservoir of a group of houses

R.P. Singh, Bhavna Pathak, Rukma Parthvi and Padma Parthvi

Abstract

Roof-top rain-water harvesting (RRWH) requires connecting the outlet pipe from the roof top to divert collected water to existing well, tube well, bore well or a specially designed well. Urban housing complexes and institutional building having large roof area or a group of residential buildings can be utilised for the purpose. The Central Groundwater Board, India has suggested several methods of RRWH for individual houses and multistorey buildings. We have designed RRWH techniques for housing complexes that are useful for a group of residential buildings. Rainwater is the purest form of natural water. It can be used as recharge water without any treatment. The water used for this purpose can be taken from any source but it must be pure like drinking water or rain water. It should not have even soil, sand and silt in it. Recharge water should be surely silt free. If germs are present, it should be disinfected by UV irradiation, ozonization or chlorination before using as recharge water. Strict regulations regarding aquifer recharge are required.

INTRODUCTION

Harvest the rain. Wrangle water from the sky for watering, washing and even drinking by using a roof-top rainwater harvesting (RRWH) system that requires connecting the outlet pipe from the roof top to divert collected water to existing well, tube well, bore well or a specially designed well (CGWB, 2000; Vashistha et al., 1980). Its components are catchment area (roof); conveyance system (guttering, downspouts, and piping); storage (cistern); and filtration and distribution. The primary expense for a rainwater collection system is the storage tank (cistern). It can be as simple as directing gutters to a lidded garbage can or as complex as a concrete cistern, roof washer and filtration system. Retrieving water from the cistern can be done by gravity, if the cistern is high enough, or by pumping, similar to the method used to withdraw and pressurize water from a well. Taking the process further, runoff could be collected and stored for use in the immediate facilities, thereby reducing city-supplied volumes, or for sending water back to the municipal facilities. If the infrastructure could be created or modified to direct runoff to city storage facilities, that water could be treated and added to the potable supply. Cities that currently have underground drainage systems might continue those systems to direct the flow to wherever they can store the high volumes associated with hard surface runoff. Urban housing complexes and institutional buildings having large roof area or a group of residential buildings can be utilised for the purpose. The Central Groundwater Board of India has suggested several methods of RRWH for individual houses and multistory buildings only. The present study aims to depict RRWH techniques for housing complexes that are useful for a group of residential buildings or a colony. RRWH gives pure water at very low cost.

RAINWATER ENDOWMENT AND HARVESTING POTENTIAL

Urban housing complexes and institutional buildings having large roof area, a group of residential buildings or any building can be utilised for RRWH. The total amount of water that is received in the form of rainfall over an area is called the rainwater endowment of that area. Out of this, the amount that can be effectively harvested is called water harvesting potential.

Water harvesting potential = Rainfall (mm) × Collection efficiency

Factors affecting collection efficiency include amount of rain (three-hundredths to one-tenth of an inch is needed to wet the roof and fill the roof washer, some of the rain overshoots the gutters or spill out of the gutters during heavy downpours), capacity of storage tanks, runoff coefficient, the first flush wastage etc. Efficiency is usually presumed to be 75% to 90% depending on system design and capacity.

Volume of rainfall over the rooftop = Area of rooftop × Average annual rainfall × Runoff coefficient

DECONTAMINATION AND DISINFECTION

Rainwater is the purest form of natural water. It can be used for aquifer recharge without any treatment. The water used for this purpose can be taken from any source but it must be pure like drinking water or rainwater. It should not have even soil, sand and silt in it. Recharge water should surely be silt free. If germs are present, it should be UV irradiated, ozonized or chlorinated before using as recharge water.

There are many filtration methods and roof-wash systems for decontamination that vary according to intended use. One method uses a floating filter connected to a flexible water line inside the cistern. This method withdraws water from approximately 1 foot below the surface, considered to be the cleanest water 30 cm in any body of water. The water to be used for human consumption can pass through an inline purification system or point-of-use water purification system such as an ultraviolet filter. Uses other than for human consumption do not require any purification.

Preliminary filtration and a roof-wash system provide the first line of defense against contamination. Rainwater harvesting systems supplying potable water also should include measures to treat water before use (Hartsung, 2002). Several treatment options including microfiltration, UV sterilization and ozonation, are available. Many experts agree that filtration and UV treatment provide adequate protection, making ozonation unnecessary.

Most systems use a combination of physical filters which remove particulates, and a UV-light chamber which kills bacteria and other organisms by exposing them to high-energy ultraviolet light.

A less expensive option is to treat water with chlorine or iodine, as is typically done with municipal water. The most common chemical added is chlorine in the form of sodium hypochlorite which is available in liquid form. Household bleach which is 5% sodium hypochlorite, is commonly used. The downsides to chlorination are the taste of the treated water and health concerns about harmful chemicals that could result from the added chlorine. In the presence of organic matter, chlorinated hydrocarbons may be formed which are suspected carcinogens.

ROOF WASHER

Between rainstorms, various pollutants can settle out of the air and onto the roof. Many rainwater harvesting systems incorporate a roof washer to keep these contaminants from entering the cistern. Roof washers capture and discard the first several gallons of rainwater during a storm before conveying the rest to the cistern. A very simple roof-wash system can be made out of a 15 or 20-cm vertical PVC or polyethylene pipe installed beneath the gutter with an inlet just above each downspout to the cistern. Commercial roof washers range in price from $100 for a water diverter (available by mail order only from SafeRain, an Australian company) to $600 for a separate roof washer.

AWARENESS

Awareness campaign are essential for RRWH (Anon, 1998a; Anon, 1998b). This needs:

- Educational materials from government agencies, NGOs and the aqiufer communities;
- Aquifer issues to be brought into primary and secondary school classrooms, and to environmental educators at museums, state parks, market places, and soil and water conservation districts;
- Numerous presentations and workshops;
- Education to landowners, local government officials, utility managers, citizens groups, and the public about source water issues;
- Printed materials, TV documentaries and a website for wider dissemination.

MONITORING

Regular monitoring of rainwater, harvesting system particularly water in the storage tank and groundwater in the aquifer is required (Agarwal, 1998a). The following points should be kept in mind:

- *Water parameters:* COD, bacteria, nitrate, fluoride, arsenic, heavy metals, pesticide and excess nutrients;
- *Techniques:* Isotopic analysis, antibiotic resistance analysis and general water quality testing;
- *Detailed evaluation:* The extent and land use of the recharge area, and surface and groundwater movement in the basin including mobility, persistence and distribution of toxics;
- *Environmental compartments:* Water, soil, food, air and biota to obtain baseline values and trends over time;
- *Data use:* Validating predictive models like Aquifer Vulnerability Assessment (AVA).

ROOF AS THE MOST COMMON RAINWATER CATCHMENT SYSTEM

The roof of a building is the most common rainwater catchment system, though a separate building designed especially for rainwater harvesting (a 'water barn') may be used.

The best roofing material for rainwater catchment is uncoated stainless steel or factory-enameled galvanized steel with a baked-enamel, certified lead-free finish. With any metal coating, the manufacturer must be asked whether the coating contains heavy metals. Red paint used on metal often contained lead in the past. Any existing metal roof being used for a potable water catchment system should be tested for lead.

Wood shakes, concrete or clay tiles, and asphalt shingles are more likely than other materials to support the growth of mold, algae, bacteria and moss which can potentially contaminate water supplies. Treated wood shingles may leach toxic preservatives, and asphalt shingles may leach small amounts of petroleum compounds. In addition to the health concerns, a porous or rough roof surface holds back some of the water that would otherwise make it into the cistern. Asphalt roofing has a 'collection efficiency' of about 85% while enameled steel has a collection efficiency of more than 95%. With asphalt roofing, more of the rainwater stays on the roof in a typical rainstorm (i.e., the roof stays wet), though the actual percentage depends on the duration of the storm.

To be most effective, the roof should be fully exposed and away from overhanging tree branches. This reduces the risk of contamination from rotting leaves or droppings from birds and insects in the trees. If possible, avoid using roofs of buildings that rely on wood heat, as the smoke particles and soot deposited on the roof may contain PAHs and other hazardous materials.

RRWH METHODS FOR AQUIFER RECHARGE

RRWH for an individual house or a goroup of houses requires connecting the outlet pipe from roof-top to divert collected water to existing well, tubewell, borewell or a specially designed well. Roof-top rain- water collected may be used to recharge groundwater reservoir through any one of the following proven techniques:

1.   Abandoned or running well
This method can be used for a colony provided hydro-geological parameters permit. For a group of houses the recharge well should be so deep that it can saturate water of the entire area.

2. Settlement tank
It is used to remove silt and other impurities from rainwater. It can have an unpaved bottom surface to allow standing water to percolate into the soil. It has provision for inflow, outflow and overflow. It holds excess of water till it is soaked up by the recharge structure. Any container masonry or concrete or underground tank, old disused tanks may be used as settlement tanks.

3. Recharge pit
Recharge pits are constructed for recharging shallow aquifers. 1 to 2 m wide and 2 to 3 m deep recharge pit is generally constructed. After excavation, the pit is refilled with boulders and pebbles at the bottom followed by gravel and then course sand at the top. The collected water from the rooftop is diverted to the pit through a drain pipe. Recharge water is filtered through the pit.

4. Soakaway
It is a bored hole of upto 15 cm in diameter, drilled in the ground to a depth of 6 to 10 m. It can be left unlined if the soil is stable and clayey. It is filled with filter matter like brickbats and is lined with PVC pipe to prevent collapse. A small sump is made at the top of the soakpit where the runoff can be retained before it infiltrates through the soakaway.

5. Recharge Trench
A recharge trench is constructed when permeable strata of adequate thickness is available at shallow depth. It is constructed across the land slope along the boundary walls. It may be 0.5 to 1 m wide, 1 to 1.5 m deep and 10 to 20 m long depending on the availability of land and roof-top area. It is filled with boulders at the bottom followed by pebbles and by course sand at the top. The collected water from the roof is diverted through the drain pipe to the trench. The trench should be periodically cleaned. The method is suitable for a building having the roof-top area of 200 to 300 m^2. For a much bigger area a bigger trench is required or several trenches may be required.

6. Recharge Shaft
A recharge shaft is dug manually or drilled by the reverse/direct rotary drilling method. Its diameter varies from 0.5 to 3 m depending on the availability of water to be recharged and its depth varies from 10–15m below ground-water level. It is back filled with boulders, gravel and coarse sand. The bottom of the shaft should end in permeable strata, i.e., course sand. It should be constructed to 10 to 15 metres away from the buildings for their safety. It should be cleaned regularly by scraping the top sand layer and refilling it periodically. It is constructed where shallow aquifer occurs below clay layer.

7. Defunct Borewell
A defunct borewell can be used for recharging the collected water for a group of houses (Fig. 1). A circular pit of 1 m diameter for a depth of 0.6 m below ground level is dug around the borewell. The bore and the pit are filled with broken bricks. The top 0.3 m portion of the pit is filled with sand. The circular pit is covered with perforated slab at the top. The slab requires regular cleaning so as to keep its holes open to receive water.

Rainwater

PVC pipe

Artificial Recharging

Gate of colony
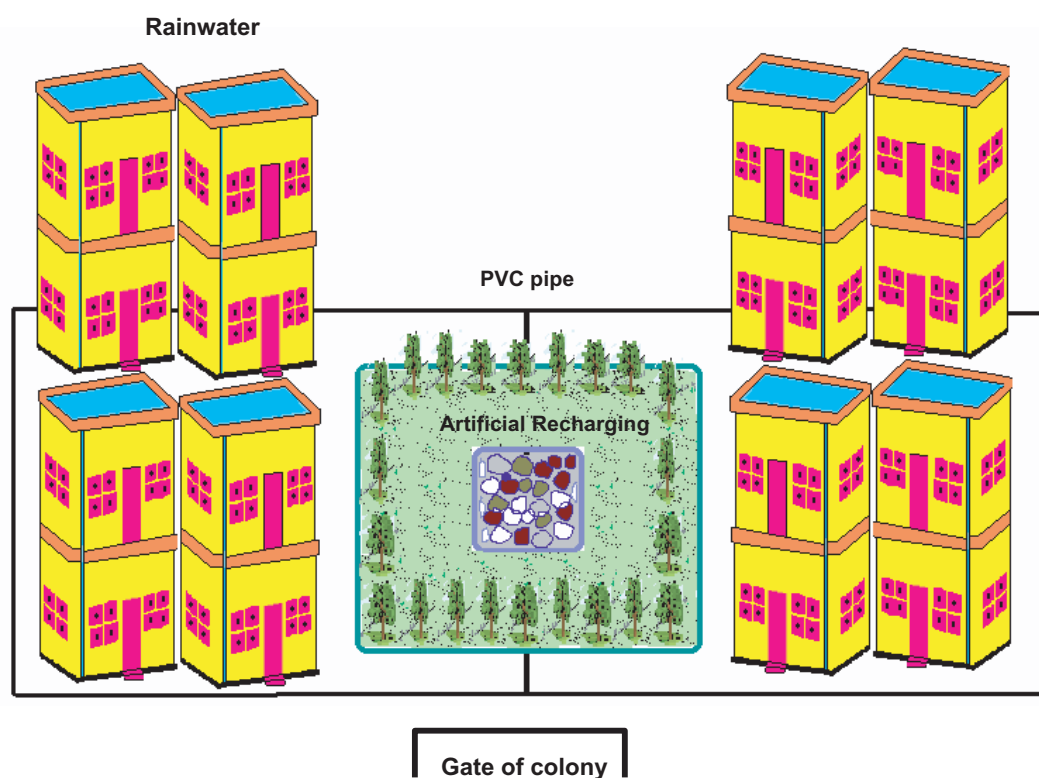
Figure 1. Recharge through borewell for a group of houses or a colony (Rainwater from the roof top is directed to the recharge borewell or pit in the centre of the park through PVC pipe)

BENEFITS OF RRWH

Collecting and using rainwater has numerous benefits, ranging from improved water quality to reduced stress on underground aquifers (Table 1). RRWH is a good option for aquifer recharge in urban areas where natural recharge is very low due to compaction, and not much land is available for implementing any other artificial recharge measure. Rainwater can be used for potable water (drinking, cooking, bathing) or nonpotable uses such as landscape irrigation, livestock watering and washing. Rainwater typically has very low hardness levels. High hardness reduces the use of soaps and detergents, and eliminates the need for a water softener. Fewer minerals also save wear and tear on plumbing fixtures. Stored rainwater also is a good standby in times of emergencies such as power outages or during periods of extreme drought when wells dry up; or where water supplies are not available, dependable or cheap. Rainwater harvesting reduces the impact on aquifers, lessening the demand on ecologically sensitive or threatened aquifers. Collecting some of the rainwater falling on impervious surfaces also minimizes erosion and flooding. Rainwater is bacteriologically pure, free from organic matter and is soft in nature. Because it doesn't have to be treated, pumped or distributed through a complex network, harvested rainwater saves energy and the use of chemicals. Some municipal water users sometimes switch to harvested rainwater as a way to avoid chlorination and fluoridation treatments. Rain runoff which otherwise flows into sewers and storm drains and is wasted, is harvested and utilised. RRWH reduces strain on municipal water supply. It improves groundwater level, yield and quality.

Table 1. Assessment of quantity of rainwater harvested

	Individual houses	*Multistoried building*	*Group of houses/ Colony*
Roof top area	100 m^2	500 m^2	2,500 m^2
Total quantity available for recharge per annum	50 m^3	250 m^3	1,250 m^3
Water available for a 5- member family	100 days	500 days	2,500 days

For example, the water level in the Panchsheel Park area, considered one of Delhi's posh colonies, was declining rapidly. Keeping in view the growing water problems, the Resident Welfare Association established a rainwater harvesting system for the entire colony, which improved groundwater level and quality.

Direct capturing of rainwater significantly reduces our reliance and pressure on water sources such as rivers, dams, canals etc. It helps in reducing the flood hazard by improvement in infiltration and reduction in run-off. It improves groundwater quality through dilution, specially for hardness, fluoride and nitrate. It can reduce groundwater pollution by 80-90%. Rainwater may be harnessed at place of need and may be utilised at the time of need. The structures required for harvesting rainwater are simple, economical and eco-friendly.

LEGISLATIONS

Currently, there are no defined regulations regarding aquifer recharge in developing countries like India except RRWH system is mandatory for new building constructions at some places, but certain rules do exist in developed nations such as USA. For example, Texas does not regulate rainwater collection for indoor or outdoor household use unless the system is backed up by publicly supplied waterlines. If there is a backup system, there must be an air gap between the public water lines and the rainwater lines. This air gap must exceed two diameters of the city line in width. The only other regulation regarding rainwater collection relates to physical maintenance of cisterns. This regulation is directed at public safety. To date, the primary governmental interest in rain and related runoff has been in pollution prevention. As more and more people begin to look to rain as a source of drinking water, the need for adequate regulation grows. The necessary regulations, whatever they may be, need to be started now while public implementation is still minimal. Legislations should seriously consider the following items: Distance from cistern to any source of contamination such as a septic tank, cistern height in relation to level of roof washer to ensure proper function, requirement of roof washer to capture the first flush, inclusion of overflow pipe directed to non-flooding area, sanitization minimum standards, filtration standards, application of drinking water standards to cisterns intended for potable supplies (regardless of number of people supplied), general health and quality standards for operation (prevention of mosquito breeding grounds), and structural standards to ensure proper design and installation.

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Assessment of water harvesting and groundwater recharge through continuous contour trenches

Mukund Shinde, Ian Smout and Sunil Gorantiwar

Abstract

A simulation model is presented for assessment of the water harvesting and groundwater recharge through continuous contour trenches (CCT) in the semi-arid watersheds. The model is based on the simulation of field and trench water balance. The model operates on daily basis. The water balance parameters such as runoff, infiltration and evaporation are estimated by SCS-CN, the Green Ampt and Penman methods, respectively. Daily rainfall and other climatic data series are the input requirements of the model. In addition to assess the groundwater recharge, the model can also be used to decide the dimensions of a trench system to achieve certain recharge by analysing simulation runs of the model for different trench strategies for the given conditions. The application of the model is described with one case study on micro watersheds from the semi-arid region of Maharashtra, India. The case study results showed that for the existing trench system the average contribution to groundwater recharge through trench and field are 53 and 47% respectively. Model under predicted the groundwater rise in the micro watershed.

Keywords

Groundwater recharge; trench; semi-arid areas; water harvesting; simulation model.

INTRODUCTION

Water harvesting is essential to sustain crop growth in semi arid and sub humid regions. In a watershed development project different water-harvesting techniques ranging from small in-situ land alteration techniques to big tanks are integrated in a manner such that disadvantages of one technique are offset by the advantages of the other. Continuous contour trench (CCT) is one such technique extensively used in the semi-arid regions of India. CCTs are constructed on the non-arable lands in the watershed. These trenches are rectangular excavations made in the soil on sloping lands. Their primary objective is to harvest the runoff flowing down the slope and recharge the groundwater. The trenches are called continuous contour trenches (CCT) since they run continuously along the contour with lengths covering the entire width of the field. A typical layout of the CCT is shown in Fig. 1 and Fig. 2. A rough stone spill is provided at the end of the CCT, where it joins the gully (or field drain). This facilitates storage of water in the CCT and safe disposal into the drain during heavy rains.

The trench system can be defined as a series of continuous contour trenches at some interval running along the contour and covering the entire field or micro-catchment to accomplish water harvesting and groundwater recharge. Currently the design parameters of this trench system are decided empirically, and hence there is a need to develop analytical approach for designing the trench system and assessing its performance. Previously Guo (1998) and Akan (2002) developed the performance indicators in different forms and proposed methodologies based on simulation model for infiltration trenches (IT) in the USA. However CCTs differ from ITs in that ITs are bigger in size and are filled with gravel and the CCTs are smaller in size and kept empty to take full advantage of their storage capacity. This paper presents a simulation model and indicators for the continuous contour trenches and discuses their utility with one example case study.
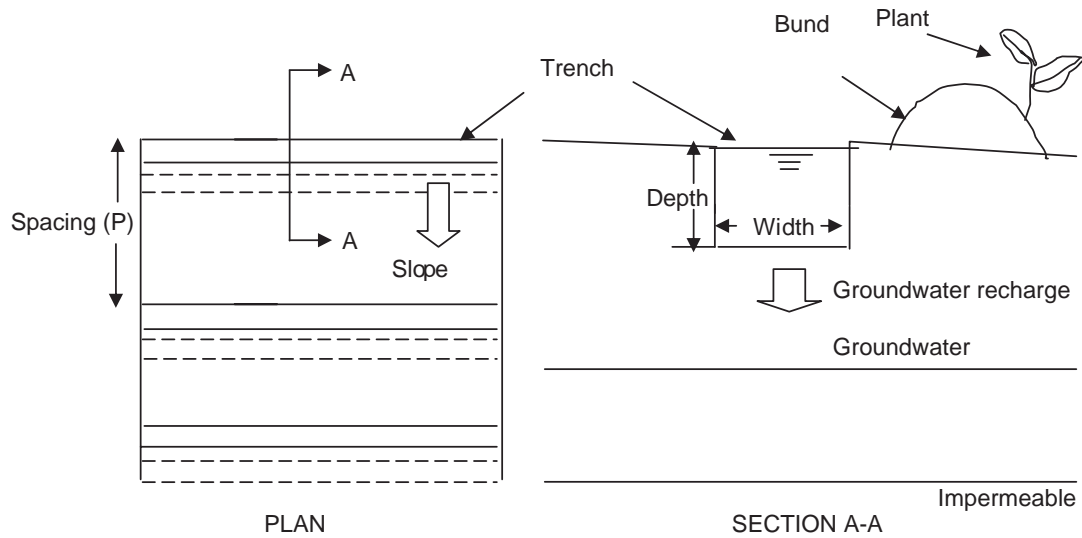
Figure 1. Typical layout of the continuous contour trenches

Figure 2. Layout of continuous contour trenches on the field

METHODOLOGY

In this paper a term 'water harvesting potential' (WHP) is introduced as an indicator that describes the ability of the trench system to harvest the rainfall and runoff water and recharge the groundwater. Gross WHP (equation 1) is proposed to indicate the total water intercepted by the trenches and net WHP (equation 2) to indicate the total water infiltrated from the trenches. WHP would be a useful indicator to know the performance of different trench systems in water harvesting.

$$GrossWHP = \frac{Qif - Qof}{Qif} \, x \, 100 \qquad (1)$$

$$NetWHP = \frac{Qif - (Qof + Evap)}{Qif} \, x \, 100 \qquad (2)$$

Where,
Q_{if} = Inflow to the trench, Q_{of} = Overflow from the trench and *Evap* = Evaporation from the trench.

Simulation model

Simulation model is based on the field and trench water balance. Inflows to the trench system are runoff and rain falling over the trench and outflows are infiltration, evaporation and overflow from the trench. Model operates on daily time step. The different processes simulated in the water balance model of the trenches are: runoff, infiltration and evaporation. Runoff is simulated with the help of SCS Curve Number method (USDA 1986). The choice of this method was influenced by the easy availability of the parameters of the CN method and the wide applicability of the method for Indian conditions (Bhatnagar et al 1996, Srivastava 2001, Panigrahi and Panda 2003). Infiltration from the trenches is modelled with the Green and Ampt (1911) method for ponded conditions. Infiltration is assumed to occur through the bottom of the trench only. For simplicity it is assumed that the small soil bund on the down slope side of the trench and the vegetation on bund if any do not affect the infiltration. Evaporation is estimated on a daily basis by Penman (1948) method.

Main assumptions the model are: i) soil deposition is considered as negligible and hence it does not influence the infiltration in the trench. This assumption is particularly valid in non-arable lands with shallow soils, which is a case when trench systems are adopted, ii) groundwater table does not interfere with the infiltration process and iii) inflow into and outflow from the trench are instantaneous.

Field water balance

Field water balance estimates the water balance parameters for the field between the two trenches. For simulation, the soil profile is divided into two zones i.e. soil zone and vadose zone. Water balance is carried out for the soil zone on daily basis. Inflow to the soil zone is through rain only and outflows are through runoff, evaporation and deep percolation. Inflow is assumed to take place at the beginning of computation time step (day) and outflows are assumed to take place at the end of time step. Water in excess of field capacity of the soil zone goes as deep percolation to the vadose zone. The total volume of deep percolation from the field and total volume of water infiltrated from the trench (described below) termed together as recharge volume, contributes to the groundwater storage.

Trench water balance

Inflow (Qif) to the trench includes runoff from the field between the two trenches and the rain falling over the trench. This inflow instantaneously fills the trench. If the inflow volume is more than the available trench capacity, the excess water moves along the trench and joins the gully over the spillway of the trench as overflow (Qof). But for simplicity in the model, it is assumed that the overflow is also instantaneous. The volume of water retained in the trench is then subjected to infiltration and evaporation losses.

Flowchart of the model (coded in C language) is given in Fig. 3. For assessing the groundwater recharge from a trench system, simulation needs to be performed for number of years for which rainfall data is available.

THE CASE STUDY DESCRIPTION

One micro-watershed (Fig. 4) at Akola in semi arid region of Maharashtra, India was selected as case study for describing the applicability of the developed model. The micro-watershed consists of two fields with field No. 1 of 4 ha and field No. 2 of 3.5 ha areas. Both the fields are laid with CCTs. There are two tanks of capacity 487.18 m³ at the end of field-1 and 917.47 m³ at the end field-2. Excess water from the fields is collected in these tanks. One piezometer is located in field No. 2 to observe the groundwater levels. The input parameters of the model are:

i) Climate Data: 1976–77 to 2003-04 (28 years), ii) Trench sizes: S1= 0.3 x 0.3 m (half of the existing size),

S2 = 0.6 x 0.3 m (existing size with 0.6 m width and 0.3 m depth), S3 = 0.6 x 0.6 m (twice of the existing size). iii) Trench spacing: P1 = 5 m (existing spacing), P2 = 10 m (two times of existing spacing), P3 = 20 m (four times of the existing spacing), iv) Soils: The micro-watershed has two fields with different soils. The average soil properties are presented in Table 1.
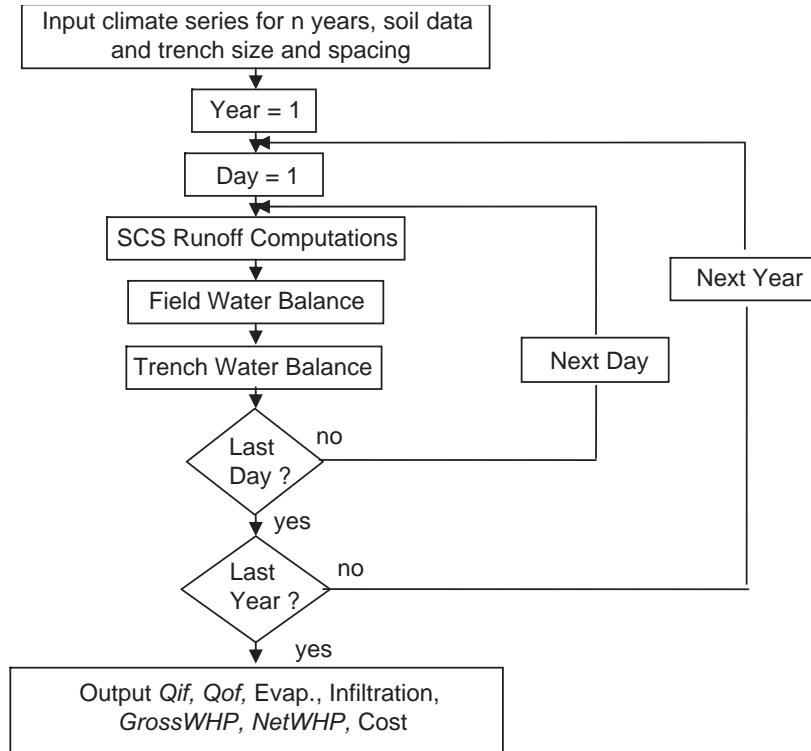
Figure 3. Flowchart of the field and trench water balance simulation model
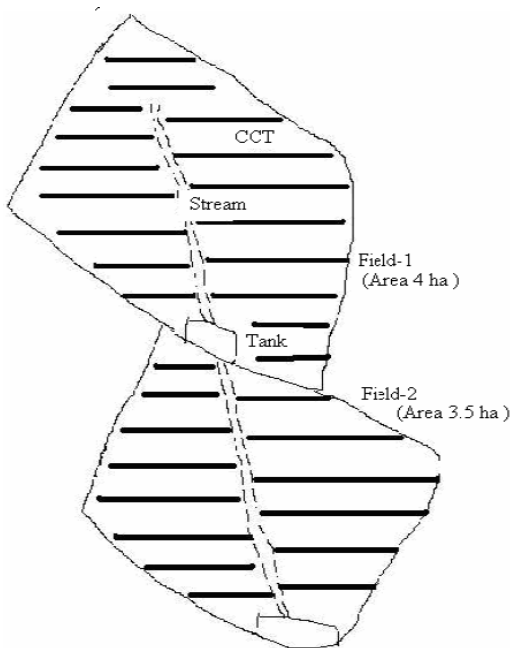
Figure 4: Sketch of micro-watershed (not to scale)

Table 1. Average soil properties of experimental field

Property	Field-1	Field-2
Soil type	Sandy clay	Sandy loam
Field Capacity, %	36.39	28.55
Wilting Point, %	10.78	13.68
Bulk Density, gm/cm^3	1.40	1.40
Porosity	0.45	0.47
Saturated Conductivity, mm/hr	1.92	6.68
Saturation Moisture Content, %	44.53	46.61
Suction Head*, mm	218.5	110.1

(* Rawls et al 1983)

RESULTS AND DISCUSSION

The simulation model was run for 28 years. Model generates daily output of runoff, inflow to the trench, infiltration, evaporation and overflow from the trench. The effects of rainfall and trench size and spacing on the trench performance in terms of WHP are evaluated and discussed.

Effect of rainfall on WHP

The model was run for 28 years i.e. from 1976–77 to 2003–04, however outputs for only last 10 years are presented in Table 2 for brevity. From the table it is observed that gross WHP was almost 100 per cent in low to medium rainfall years (1991–92, 96–97, 98–99, 2000–01, 2001–02 and 2003–04). This suggests that trenches are more effective in water harvesting in low to medium rainfall years. The observed data of inflows in both the tanks showed that there was no inflow in these tanks during the years 1996–97, 98–-99, 2000–01 and 2001–02. Model also predicted 100% gross WHP during these years. Since model simulates groundwater recharge through field and trench, only those years during which there was no inflow in the tanks were selected for studying the groundwater recharge through the trench system. The results are discussed under groundwater recharge section.

Table 2. Water harvesting potential for P_1S_2 trench combination

Year	Rainfall (mm)	No. of rainy days	Runoff (mm)	No. of runoff days	Gross WHP (%)	Net WHP (%)
1994–95	1011.2	62	118.5	17	99.0	88.4
1995–96	562.4	33	87.5	9	95.9	86.3
1996–97	710.4	42	22.9	13	100	80.9
1997–98	827.8	46	88.5	14	91.3	80.2
1998–99	870.2	51	50.6	11	100.0	85.1
1999–00	976.5	49	120.3	15	91.6	82.7
2000–01	646.4	36	35.2	10	100.0	87.0
2001–02	634.1	33	72.8	13	100.0	89.9
2002–03	639.1	36	84.6	10	85.1	76.4
2003–04	380.8	34	4.9	3	100.0	67.3

Water balance of the trench system

From the water balance components of the trench system for different years it was found that, overflow from the trench varied from zero in low rainfall years to 40% in high rainfall years. On an average infiltration and evaporation contributions to the total water outflow from the trench were 80 and 10 per cent respectively with remaining 10% as overflow from the trench.

Effect of trench size and spacing on the WHP

From the results it is observed that, WHP tends to increase with increasing trench size for given spacing and decrease with increasing trench spacing for given size. This is depicted in Fig. 5. With such analysis optimum trench spacing and size combination can be arrived to attain desired WHP. For example in the present case around 78% gross WHP can be attained by adopting any of the following trench spacing and size combination:5 m spacing with 0.3 x 0.3 m section (P1-S1), 10 m spacing with 0.6 x 0.3 m section (P2-S2), 20 m spacing with 0.6 x 0.6 m section (P3-S3).
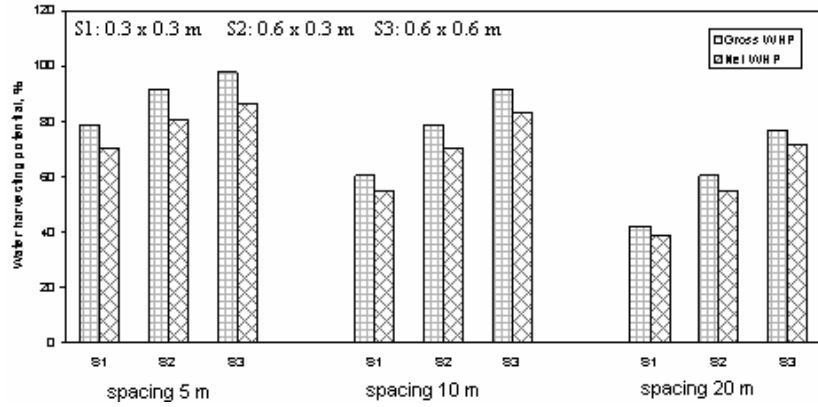
Figure 5. Water harvesting potential as a function of trench spacing and size

Groundwater recharge

Table 3 presents observed and model predicted groundwater table rise. From the results of 28 years of simulation, it is observed that average contribution to the groundwater recharge through trench and field are respectively 53% and 47% for trenches of 0.6 x 0.3 m size spaced at 5 m. From the table it is also seen that the model tends to under predict the rise in groundwater level. This reflects the simplicity of the water balance process in the model and need for model calibration. Nevertheless the results are useful to give indicative predictions of the groundwater recharge in micro-watersheds treated with continuous contour trenches.

Table 3. Groundwater recharge through trench system

Year	Rainfall (mm)	Groundwater recharge (volume m³)	Rise in groundwater table height	
			Observed	*Computed*
1996–97	710.4	13,469.57	2.55	1.80
1998–99	870.2	17,290.746	4.08	2.30
2000–01	646.4	10,167.27	1.93	1.36
2001–02	634.1	10,367.29	1.75	1.38

Cost and WHP

The construction costs at Rs 35/m³ (1 USD = Rs 44.55) of different trench system configurations were worked out. Fig. 6 shows the variation of cost with WHP for different trench configurations. The figure is useful to know the WHP for a given investment in the trench system. Such analyses are useful in the watershed development project preparations.
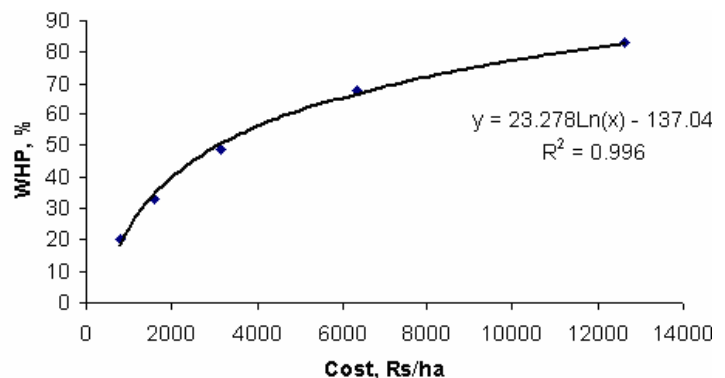
Figure 6. WHP as a function of cost

CONCLUSION

This paper introduced definitive term 'Water Harvesting Potential' (WHP) and related formulae for assessing performance of CCTs. In addition, paper also presented the simulation model based on water balance for estimating

different components of the formulae. The paper presented the strength of the model in knowing the performance of CCTs and understanding the influence of different governing parameters such as rainfall and trench geometry on performance of CCTs. Such type of the analysis is useful for the agricultural planning of the watersheds.

Comparison of observed groundwater levels with those predicated by the model indicated that the predicted levels do not deviate greatly from the observed levels and overall model under predicted the rise in groundwater level. Nevertheless the results are useful in assessing the usefulness of CCTs. As expected the cost of CCTs increased with WHP. Though the scope of the paper was limited to present the methodology for assessing the performance of CCTs, the developed model can also be used to optimise the CCTs system by repeatedly running the model for different designs of CCT systems and comparing WHP of these systems for the watershed.

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Artificial recharge of treated wastewater effluent enables sustainable groundwater management of a dune aquifer in Flanders, Belgium

Emmanuel Van Houtte and Johan Verbauwhede

Abstract

In July 2002 the Intermunicipal Water Company of the Veurne region (IWVA) started artificial recharge of an unconfined aquifer in its dune water catchment St-André. Wastewater effluent was used as the source for the production of infiltration water. This plant, called 'Torreele', with a production capacity of 2,500,000 m³/year, combined membrane filtration techniques to achieve the stringent standards set for the quality of the infiltration water.

The whole project was developed to create a sustainable groundwater management; the natural groundwater extraction was reduced from 3,700,000 m3/year to 2,700,000 m3/year. By 2010 another 500,000 m3/year will be saved.

Keywords

Reuse, membranes, artificial recharge, groundwater management.

INTRODUCTION

The Intermunicipal Water Company of the Veurne Region (IWVA) is responsible for the distribution of drinking-water in the western part of the Flemish coastal plain. At the beginning of the nineteen-nineties of the previous century the IWVA could no longer increase the groundwater extraction from its dune water catchments of St-André (123 ha) and the Westhoek (87 ha) to fulfill the increasing drinking-water demand (Fig. 1). In the latter salinity increased dramatically since the beginning of the nineteen-eighties of the previous century. *Sustainable groundwater management* was needed.
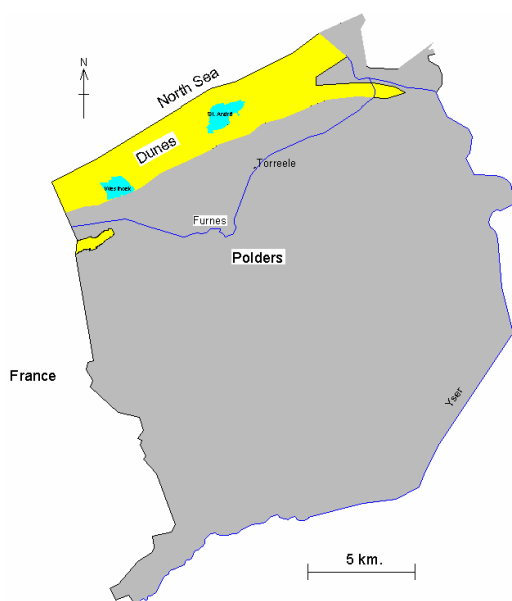
Figure 1. Localisation of the IWVA-area

Artificial recharge of the unconfined aquifer of the dune water catchment St-André was the selected solution. It should restore the groundwater quality and enhance the ecological values of the dune areas as the extraction of natural groundwater reduced. Effluent from a nearby wastewater treatment plant was selected as the source for the production of infiltration water.

DESCRIPTION OF THE WESTERN FLEMISH COASTAL PLAIN

Topography and geology

In the western Flemish coastal plain different landscapes appear (Fig. 1). The dune ridge protects the inland from the sea and is situated in the northern part, with a width varying between 2 and 2,5 km and a ground level from +6 to +35 m.

The unconfined aquifer under the dune belt was formed by 25 to 35m thick Quarternary sandy deposits, but in some dune areas thin layers of fine grained sediments occur.

South of the dunes, in the polder area, two types of landscapes could be distinguished, these being the more elevated sandy creek ridges (+3,5 to +5m) and the lower marsh basins (+2,5 to +4 m). The creek ridges have sandy Quarternary sediments whereas, under the marsh basins fine grained sediments (clay, silt and peat) occur. The polder area is drained by canals that discharge large amounts of surface water to the sea, especially during winter.

The Quarternary sediments lie upon Tertiary clay. This Ypresian clay, 110 m thick, separates the upper unconfined aquifer from the confined aquifer. This confined aquifer was formed by very fine silty green sands of Landenian age with a thickness of 20 m and lay on green clays.

Hydrogeology

Under the dunes a fresh water lens formed by infiltrating rainwater. North of the dunes salt water is present under the shore. In the polder area, fresh water filled the upper and relict salt water the lower part of the aquifer. A small transition zone of brackish water lay in between. Under the creek ridges more fresh water occurs because of the higher infiltration rate of rainwater here compared with the marsh basins, where fine sediments reduce infiltration rates.

In natural conditions, groundwater flows from the dunes towards the polder area and the shore, preventing ingress of saline water. The groundwater extraction diminished the fresh water outflow from the dune area. Overexploitation, causing the inflow of salt water from the aquifer under the beach and polder areas, threatens the quality of the dune aquifer and should be prevented. As St-André and Westhoek are situated in the dunes, sustainable groundwater management of both exploitations was inevitable.

THE CHALLENGE OF WATER-REUSE

Since wastewater treatment improved over the last decades, the effluent quality is of such a nature that it should be put to beneficial use, especially in regions where water is scarce. The IWVA-region could be considered as such an area since fresh water is only available under the small dune ridge along the coast and with a limited capacity. Deeper aquifers are no alternative and surface water resources, e.g. drainage water from the polder area, are limited, often brackish, and only available during the winter period. However, due to tourism, most water is needed in the summer period. A project to integrate wastewater effluent into the existing drinking-water production was a logical choice for the IWVA.

The high ecological value of the dunes necessitated constant consideration and it was especially so when developing the infiltration project. A shallow depth, gentle slopes, bendy banks, an island in the middle and some wide bays increased the natural interest of the infiltration pond. The extraction wells were integrated within the existing infrastructure, minimizing the impact on the environment. The quality of infiltration water was subject to stringent standards. Both the salt and nutrient content had to be low and together with the hygienic safety of the water, this was the biggest challenge when integrating wastewater effluent into the drinking-water production. Based on the experience at Orange County, where clarified secondary effluent was treated for groundwater injection as a barrier against saline water intrusion and where new pilot tests were performed (Leslie et al., 1996), a combination of *membrane filtration techniques* was chosen to produce infiltration water:

- ultrafiltration (UF) is the first treatment step, removing suspended solids and bacteria from the effluent;
- reverse osmosis (RO), the final treatment, not only removed salts, nutrients and viruses, but also small organics (e.g. pesticides), endocrine disrupting compounds and pharmaceuticals are reduced for over 90%.

DEVELOPMENT OF THE PROJECT

Ten years of research preceded the start of the project. The Flemish Institute of Nature Conservation performed an ecological study of the water catchment of St-André resulting in a delineation of the infiltration area, the indication of ecological conditions (Kuijken et al., 1993) and an ecological management plan for this area. Two infiltration tests, using groundwater from nearby sewage works, gave valuable information about the hydrogeological para-meters of the dune aquifer in St-André (Lebbe et al., 1995). The impact of artificial recharge in St-André was simu-lated using MODFLOW (Van Houtte and Vanlerberghe, 1998). The model represented the unconfined aquifer in these dunes with the northern and southern borders of the model area, presenting the shore and the polder area respectively, introduced as constant head boundaries. The horizontal conductivity of the sands is 14 m/d with a mean natural infiltration rate of 280 mm/year. The mean annual hydraulic heads were calculated; seasonal effects were not taken into account. The calculation showed that the groundwater flow in St-André would not change very much. However, although the rise in hydraulic head would be small, the outflow of groundwater to the shore would increase with 10 %.

From 1997 until 1999 pilot tests were performed on the effluent from the waste water treatment plant (WWTP) at Wulpen, which is operated by Aquafin. Different micro- and ultrafiltration systems were used for the pre-treatment (Van Houtte et al., 1998 and 2000) and two types of reverse osmosis membranes were tested for the desalination (Van Houtte and Vanlerberghe, 2001).

INFILTRATION WATER PRODUCED AT THE 'TORREELE' PLANT

Production of infiltration water

The treatment plant, built besides the WWTP Wulpen and called 'Torreele', came into operation the 8th of July, 2002. The production capacity is 2,500,000 m^3/year. Based on the experience of the pilot tests the IWVA has cho-sen the following treatment steps (Van Houtte and Verbauwhede, 2003):

- a 1 mm prescreen to remove particles from the effluent before entering the buffer reservoir;
- a submerged UF system using ZeeWeed$^®$ element cassettes containing chlorine resistant membranes with pore sizes of maximum 0,1 µm performing outside-in filtration;
- a cartridge filter;
- two identical RO skids having each 36 pressure vessels of 8 inch diameter and 6 m length of which 30 contain 6 high surface area, low-energy, brackish water RO membrane elements (8" BW 30LE-440 DOW) each, in a 2 stage (20 –10) configuration;
- UV disinfection.
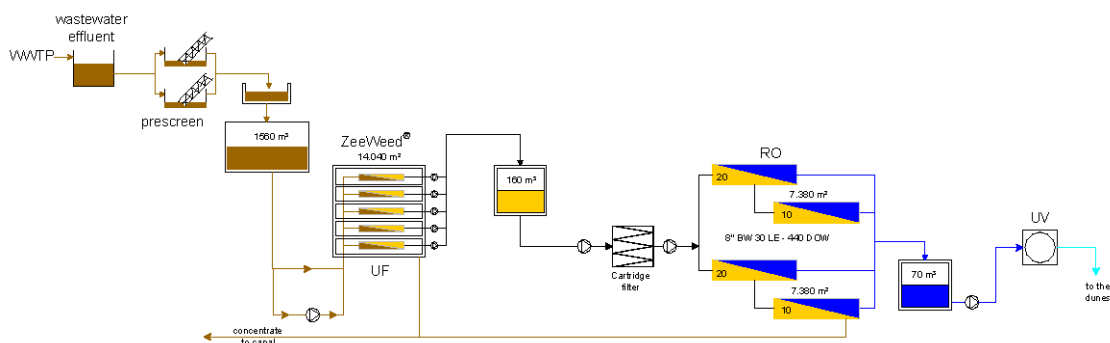
Figure 2. Process scheme of the 'Torreele' plant

The UF and RO system work at a recovery of respectively 87 and 75%. The concentrate is drained in the nearby canal together with the part of effluent that has not been treated at 'Torreele'. To prevent scaling and biofouling on the RO membranes, scale-inhibitor and monochloramines are dosed to the UF filtrate prior to the RO process.

Quality of infiltration water

Initially the infiltration water was composed of 90 % RO filtrate and 10 % MF filtrate. Re-mineralising RO filtrate with UF filtrate resulted in a quality that matched the natural dune water (Table 1). During the first two years of operation, from time to time atrazine, simazine and diuron were detected in the effluent; other pesticides were never found. If present, the level of atrazine in the effluent was so low (maximum of 0,1 µg/l) that it could never be detected in the infiltration water. However as they were not removed by UF, simazine and diuron could be found in the infiltration water. The levels were low, respectively maximum 0,12 and 0,06 µg/l, because comparative tests showed that they were removed between 97,6 and 98,6% by RO. These results confirmed tests performed during the pilot tests on removal of estrogenic activity. Based on the response of the recombinant yeast estrogen assay to the unaltered samples, the estrogenic activity of the effluent was moderate to low and in the filtrates it was lower compared with the effluent (Van Houtte, 2000). Pauwels (2003) confirmed those results on the full-scale plant, but no estrogenic activity could now be detected on the filtrates. Recent research showed that the removal of ionic pharmaceutical residues and pesticides by the RO membranes used at the 'Torreele' plant exceeded 95 % (Drewes et al., 2005).

The IWVA decided in May 2004 to use no longer UF filtrate as part of the infiltration water. The RO filtrate was re-mineralised using sodium hydroxide. Since then the infiltration contained less organics and the nutrient content was lower, which should benefit the infiltration. On one exception for diuron (0,016 µg/l), pesticides were no longer detected in the infiltration water. As the RO removed all bacteria and viruses, the UV installation could be stopped.

Table 1. Quality of infiltration water produced at the 'Torreele' plant

	Infiltration water (July 2002 - May 2004)	Infiltration water (May 2004 - February 2005)	Infiltration water standard
pH	5,70 – 7,67	6,05 – 7,16	>6,5 and <9,2
Conductivity (µS/cm)	150 (35 – 262)	49 (22 – 73)	1000
Chloride (mg Cl/l)	22 (2 – 36)	2,5 (1 – 4)	250
Sulphate (mg SO$_4$/l)	10 (6 – 17)	<1	250
Sodium (mg Na/l)	18 (5 – 30)	11 (5 – 16)	150
Total hardness (°F)	3,6 (1,8 – 5,8)	<1	40
Nitrate (mg NO$_3$/l)	6,9 (1 – 16)	2 (<1 – 3,6)	15
Ammonia (mg NH$_4$/l)	0,31 (<0,05 – 0,84)	0,21 (<0,05 – 0,61)	1,5
Total phosphorous (mg P/l)	0,1 (<0,1 – 0,3)	<0,1	0,4
Total Organic Carbon	0,9 (0,5 – 2)	<0,2	–

Mean value with minimum and maximum between parentheses.

ARTIFICIAL RECHARGE AT ST-ANDRE

Concept

After recharging the unconfined aquifer in St-André, the water is recaptured using 112 new wells with filter elements between 8 and 12 m depth. The residence time into the soil had to be minimum 40 days with a maximum of variation to level different qualities of infiltration water (Table 2). This underground passage of minimum 40 days ensured elimination of all pathogens.

Early 2004, after checking all distances, the infiltration pond was locally recalibrated to increase the distance with the nearest wells.

Table 2. Repartition of distance between wells and infiltration pond

Distance	Number of wells
< 40 m	10
40 – 49 m	44
50 – 59 m	23
60 – 69 m	17
70 – 79 m	3
80 – 89 m	2
> 90 m	13

The surface area of the infiltration pond is 18,200 m^2; the infiltration capacity is 2,500,000 m^3/year. Yearly 3,500,000 m^3/year should be extracted, or 1,4 the infiltration rate, to ensure that the infiltration water is totally recaptured. The groundwater is treated with the existing plant: aeration and rapid sand filtration.

Monitoring wells were installed; three are controlled online, as is the level in the infiltration pond.

Results of artificial recharge

Infiltration started the 8th of July, 2002 and apart from half September until half December 2002, and 10 days at the end of January 2004, Torreele produced normally (Fig. 3). The salinity of the water extracted out of the infiltration area gradually decreased, from 800 µS/cm before infiltration started to around 400 µS/cm (Fig. 3). When half May 2004, UF filtrate no longer was used for the infiltration water, salinity declined again. Now the salinity stabilized around 300 µS/cm.
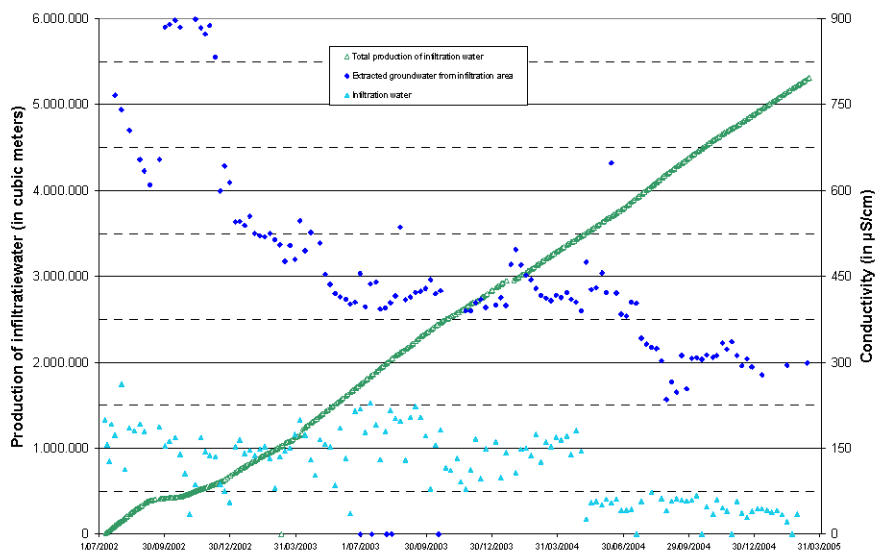
Figure 3. Production of infiltration water and evolution of salinity (left)

Monitoring between the infiltration pond and the extraction wells showed that:
- near the infiltration pond there is a great vertical flow to the lower layers; this flow reduced significantly with depth and was locally hindered by clayey sediments;
- more distant from the infiltration pond, the vertical flow reduced and the water flowed mainly horizontally through the aquifer;

- the chloride, sulphate, sodium, potassium and nutrient content did not increase during its soil passage;
- the pH rapidly increased during its first meters of soil passage but when the water infiltrated to deeper layers the pH decreased somewhat again;
- this change of pH is related to a change of hardness and proved that calcium carbonate dissolves into the infiltrating water; total alkalinity increased in the same way and the concentration of carbon dioxide decreased.

Other observations are the gradual decrease of the organic content, as well as of iron and manganese. It means that the quality of the recaptured water, extracted at 1,4 the volume of infiltration, is of better quality compared to what it was before infiltration started (Table 3). This benefits to the drinking-water produced at St-André:

- the salinity, nutrient and organic content is much lower;
- the hardness stabilized around 15 °F which increased the comfort for the customers and benefits the environment as less soap should be used;
- the iron content gradually decreased from around 5 mg/l to 1 mg/l which means that the sand filters are less frequently backwashed saving groundwater.

Despite the presence of small doses of pesticides into the infiltration water, they have never been detected in the recaptured groundwater.

A negative impact of the artificial recharge by reuse of wastewater effluent, is the temperature of the water. The pumped groundwater in St-André had a constant temperature of 11 to 12 °C, but since infiltration water varied from 9 to 23 °C, the recaptured groundwater gets warmer (up to 18 °C) during most of the year (Fig. 4). The drinking-water produced is more susceptible to re-growth and as a prevention the IWVA had to chlorinate its water. Since September 2003 prior to distribution, the drinking-water in St-André is treated using UV-irradiation.
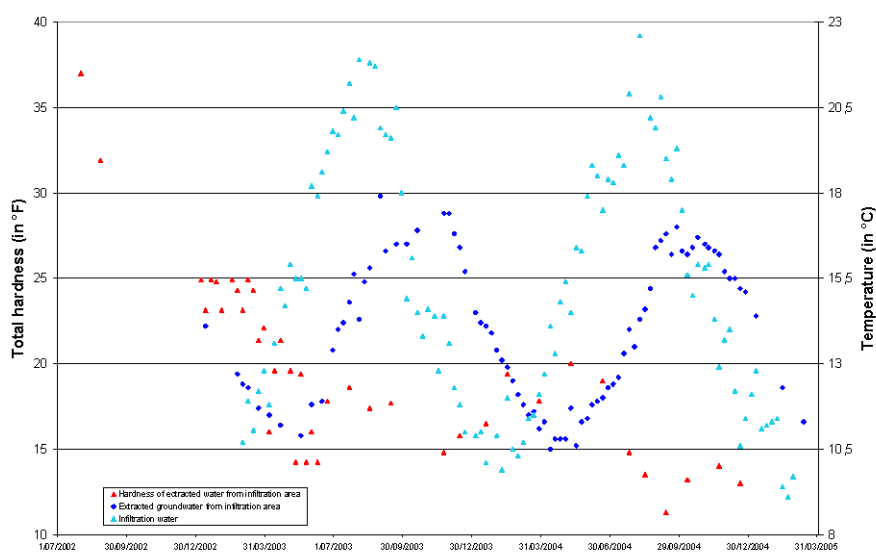
Figure 4. Variation of temperature in the infiltration water and the re-extracted water

Before infiltration started, sodium and calcium were the dominant cations respectively in the lower and upper part of the unconfined aquifer under the infiltration area, wheras bicarbonate was the dominant negative ion. Since infiltration started, this did not change and it was noticed that the groundwater quality of the lower part of the aquifer and at the edges of the infiltration area remained quite stable, indicating that the infiltrated water is very well recaptured by the installed wells.

Table 3. Quality of groundwater recaptured after infiltration

	May 2002	May 2003	May 2004	February 2005
pH	7,27	7,41	7,53	7,61
Conductivity (μS/cm)	814	436	475	295
Chloride (mg Cl/l)	53	33	39	15
Sulphate (mg SO_4/l)	103	41	47	23
Sodium (mg Na/l)	39	23	26	18
Total hardness (°F)	38	19	20	12
Nitrate (mg NO_3/l)	7,6	1,7	2,2	<1
Ammonia (mg NH_4/l)	0,72	0,38	0,28	0,21
Total alkalinity (mmol)	28,9	15,3	16,2	11,7
Total Organic Carbon	6,1	2,3	2,3	1,9

SUSTAINABLE AND ECOLOGICAL MANAGEMENT

Thanks to artificial recharge, the extraction of natural groundwater could be reduced substantially, and this bene-fited most for the dune water catchment in the Westhoek. In that area, since 10 years the IWVA gradually reduced the groundwater extraction. The salinity in the wells nearest to the polders declined from approximately 1,500 μS/cm in 1992 to 1,000 μS/cm in 2004.

In St-André the groundwater quality extracted outside the infiltration area did not change very much, but here problems with increasing salinity never occurred. There is a relative rise in groundwater levels resulting in an increased outflow of fresh water from the dunes. In the vicinity of the infiltration pond, where the water level is close to the surface, some valuable plants have been monitored in 2003 and 2004. In St-André there is eco-logical management of the dunes since 1994; around the infiltration pond this management was recently adjusted to enhance the positive development that was observed. Ecological management of the dunes of the Westhoek recently started.

CONCLUSION

Artificial recharge, reusing wastewater, enabled a substantial reduction of the natural groundwater extraction in the dune aquifers of the Flemish coastal plain. Membrane techniques were successful in producing infiltration water with low salt and nutrient content. Due to this project the quality of the extracted water improved and the ground-water levels have risen, enhancing the ecological value of the area and increasing the outflow of water from the dunes to the sea and the polders. This is a valuable protection against saline intrusion indicating the extraction gets sustainable.

Reuse of wastewater combined to artificial recharge opens opportunities in many regions. It could be a solution for inadequate water supply and for deteriorating water quality. Membrane techniques offer a safe solution when drink-ing-water is at stake; infiltration afterwards ensures a multiple barrier.

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Geochemistry during infiltration and flow

A comparison of three methods
for determining travel times
near a large artificial recharge facility

Dror Avisar, Jordan F. Clark, Jeni McDermott and G. Bryant Hudson

Abstract

Hydrogeologic calculations, transient tracers and deliberate tracer experiments are established methods of determining ground water travel times. A two-year long deliberate tracer experiment using sulfur hexafluoride (SF_6) was conducted at the Montebello Forebay (LA County, CA) artificial recharge site to determine travel time to wells within 150 m of spreading ponds. SF_6 was detected at seven monitoring wells, all screened within 50 m of the ground surface and at nine of the eighteen production wells. Travel time was best correlated with screen depth. At four of the nine wells with SF_6 detections, hydrogeologic travel times were less than 0.3 years (16 weeks) and are in basic agreement with the SF_6 results. However, at the other five wells, the hydrogeologic travel times were estimated to be more than 4 years, significantly longer than indicated by the tracer data. Transport through low permeability layers due to gaps or leakage must be occurring. At all wells where SF_6 was not detected, the hydrogeologic travel times were greater than 3 years. At the production wells, tritium/^{3}He apparent ages were greater than 10 years, indicating mixing of young and old ground water caused by the long well screens.

Keywords

Artificial recharge, geochemical tracer, ground water travel time, SF_6, Tritium/^{3}He dating.

INTRODUCTION

Quantification of subsurface residence times and flow paths are important criteria for managing artificial recharge sites. This basic hydrologic information is needed for evaluating subsurface water quality changes that may result from *in situ* biogeochemical reactions and mixing of water from different sources. It is also needed for validation of numerical models of flow (e.g., Thomson et al., 1999). Ground water travel time near artificial recharge sites is not simple to define as flow paths to wells can be numerous and convoluted. This especially true at facilities, such as the Montebello Forebay in Los Angeles, California, where percolation from large spreading basins is the principle method of recharge. Ground water travel times can be determined using hydrogeologic calculations or modeling, transient tracers, and deliberate tracer experiments. Each of these methods determines the flow of ground water differently and thus, provides complimentary information.

Ground water travel times can be estimated using Darcy's law and inferred flow paths. These flow paths are determined based on the depth of well perforation, aquifer stratigraphy, pump capacity, and distance to recharge location. Probable travel times are estimated using local hydrogeologic data, these inferred flow paths, and known hydraulic gradients (Bookman-Edmonston, 1994).

Determining residence time for shallow ground water is also possible using environmental tracers such as tritium/^{3}He (Schlosser et al., 1989; Cook and Solomon, 1997). Tritium, a radioactive isotope of hydrogen, was released in large quantities in the late 1950s and early 1960s during above ground nuclear bomb testing. Although there is a natural (cosmogenic) source, most tritium in the environment was produced during these tests. The tritium concentration in precipitation reached a maximum in the mid-1960s, 2 to 3 orders of magnitude higher than

prior to the tests. Since then, its concentration has decreased quasi-exponentially. The age of the ground water can be calculated using the radioactive decay relationship between tritium and its daughter isotope, ^{3}He:

$$ t = \frac{t_{1/2}}{\ln(2)}\left(1 + \frac{[^3\text{He}]_{tri}}{[\text{T}]}\right) \tag{1} $$

where t is time (apparent ground water age), $t_{1/2}$ is the half-life of tritium (12.43 years), $[\text{T}]$ is the measured tritium content, and $[^3\text{He}]_{tri}$ is the concentration of the ^{3}He derived from the decay of tritium. $[^3\text{He}]_{tri}$ is calculated from a helium mass balance that considers other helium sources (Schlosser et al., 1989; Cook and Solomon, 1997). The typical analytical uncertainty of this method is ±2 years. However, the uncertainty of the apparent age can be much larger if mixing between flow paths of different ages occurs.

With deliberate tracer experiments, tracer is added to the recharge water prior to percolation and its arrival at wells is determined directly by periodic sampling. The tracer breakthrough curves are characterized by a number of times including the initial, peak, and mean arrivals. In a homogenous aquifer, the initial and mean arrival times represent, respectively, the fastest and mean flow paths in the aquifer. The former cannot be determined with a high degree of certainty from either geochemical dating techniques or numerical flow models and is the most important time scale when evaluating the potential transport of reactive contaminants near artificial recharge facilities. In aquifers with preferential flow, the tracer breakthrough curves are more complicated, often showing multiple peaks (Clark et al., 2004). The first breakthrough will be representative of transport through the fastest flow path.

Sulfur hexafluoride (SF$_6$), a non-toxic, odorless, colorless, non-reactive gas tracer, is commonly used during deliberate tracer experiments at artificial recharge sites (Gamlin et al., 2001; Clark et al., 2004; 2005). SF$_6$ is less expensive than fluorescent dyes and ionized substances but has similar properties, making it possible and cost-effective to tag large bodies of water (>10^6 m^3) without the interference of density-induced flow (Istok and Humphrey, 1995). Laboratory and field experiments have shown that its transport is not retarded within porous media (Wilson and Mackay, 1996; Gamlin et al., 2001). The loss of SF$_6$ at the air-water interface from gas exchange can be problematic in recharge facilities that rely on infiltration from spreading basins or rivers. In order to define the input function of tracer to the ground water at these settings, careful monitoring of the surface water is required.

This paper presents an application and comparison of these three methods for determining travel times near the Montebello Forebay recharge site in North Los Angeles County, California (Figure 1). This artificial recharge site is
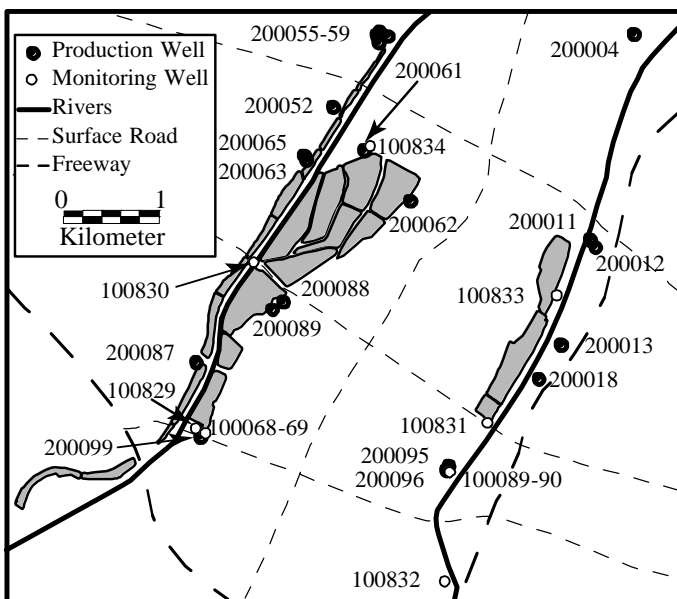
Figure 1. Map of the MontebelloForebay spreading basins and wells sampled during the study. The Rio Hono basins and river lie to the west of the San Gabriel basins and river. The light and heavy dashed lines are, respectively, major surface roads and freeways(I-5 and I-605).

within the principle recharge area for the Los Angeles ground water basin. This alluvial basin is filled with layers of sands, gravels, silts and clays occurring in discrete aquifer or aquitard units, however the spatial extent of any given layer is poorly known (Bookman-Edmonston, 1994). During the last decade, approximately 1.6 x 10^8 m^3/yr (129,000 acre-ft/yr) of surface water were recharged annually from the Montebello Forebay recharge site. About 40% of this water is natural storm runoff, 25% is imported water, and 35% is reclaimed (Johnson, personal communication).

The recharge site contains 23 shallow (<4 m deep) spreading basins adjacent to the Rio Hondo and San Gabriel River with a total wetted area of about 200 hectares. Additional recharge occurs through unlined portions of the San Gabriel River; the Rio Hondo is lined with concrete throughout the study area. Travel times to ten monitoring and eighteen production wells were examined. With the exception of only two, the wells studied are within 150 m of the spreading basins. Depth of production varies considerably. Tops of well screens are as shallow 8 m and bottoms are as deep as 270 m.

METHODS

SF$_6$ tracer experiment

Beginning on February 14, 2003 and continuing for a period of seven days, 99.99% pure SF$_6$ gas was injected into spreading basins by bubbling every one to two days. Surface water samples (5–8 samples per pond) were collected from a small boat during this time. Personnel from the Water Replenishment District of Southern California collected well samples every two to ten weeks after the injection period from ten monitoring and eighteen production wells. All the samples were collected in pre-weighed Vacutainers. All samples were analyzed using a gas chromatograph equipped with an electron capture detector following the method developed by Clark et al. (2004). The precision and detection limit were ± 3% and 0.04 pmol l^{-1}, respectively.

Tritium/^{3}He groundwater age dating

Tritium/^{3}He samples were collected independent of the SF$_6$ experiment as part of the State of California's Groundwater Ambient Monitoring and Assessment (GAMA) program. All tritium and ^{3}He samples were collected in 500 mL glass bottles and 10 mL copper tubes sealed with pinch-off clamps, respectively. At Lawrence Livermore National Laboratory, the ^{3}He/^{4}He isotope ratio and concentrations of ^{4}He and the heavier noble gases were determined using a noble gas mass spectrometer (Radamacher et al., 2001). Tritium was measured by a ^{3}He accumulation method (Surano et al. 1992).

Hydrogeologic travel times

Geochemical travel times will be compared to hydrogeologic travel times calculated by Bookman-Edmonston Engineering, Inc of Glendale, CA (Bookman-Edmonston, 1994). The calculations used inferred flow paths between the spreading basins and wells, known hydraulic gradients, an assumed uniform porosity of 20%, and mean hydraulic conductivities calculated from well test data for the highly permeable layers and estimates based on grain size for the impermeable layers.

RESULTS AND DISCUSSION

SF$_6$ was not detected in any background samples collected six and two months prior to the start of the tracer experiment. After injection, concentrations of SF$_6$ within the spreading basins ranged from 10 – 80 pmol/L with an

average concentration of 30 pmol/L The temporal and spatial variability is due to a variety of factors including wetted area, percolation rate, basin mean depth, the amount of SF_6 injected, and time since the last injection. Tracer was detected in the wet basins for about two weeks after the end of the injection period.

SF_6 arrived at seven of the ten monitoring wells sampled during the two year long experiment, indicating that this gas tracer was successfully transferred through the unsaturated zone to the water table (Figure 2). SF_6 was not detected at one shallow well (#100832) which was more 1 km from the spreading basins. The maximum ground water SF_6 concentration was 4.8 pmol/L, 15% of the mean concentration in the surface water, and 20% of the concentration in the nearest spreading basin. All monitoring wells with tracer detection have depths of less than 45 m below the nearest spreading basin. In addition to #1000832, SF_6 was not detected at monitoring wells #100068 and 100089. Both of these wells have screen depths greater than 70 m below the closest spreading basin. There is a strong correlation between screen depth and travel time for all monitoring wells sampled (Figure 3). There is no relationship between horizontal distance and travel time. The lack of a correlation may reflect the proximity of the wells to the spreading basins (distance < 150 m).
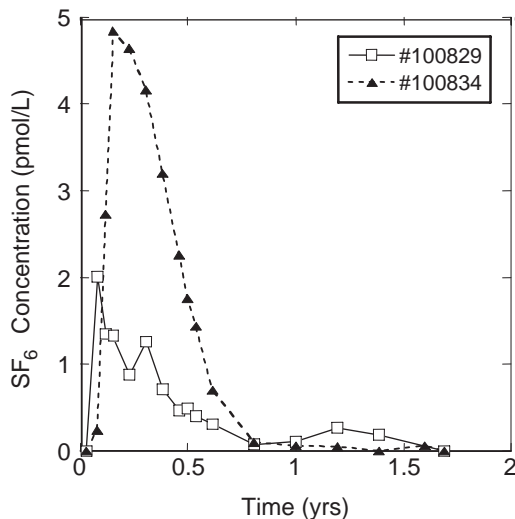
Figure 2. SF_6 breakthrough curves at two monitoring wells. See Figure 1 for well locations.
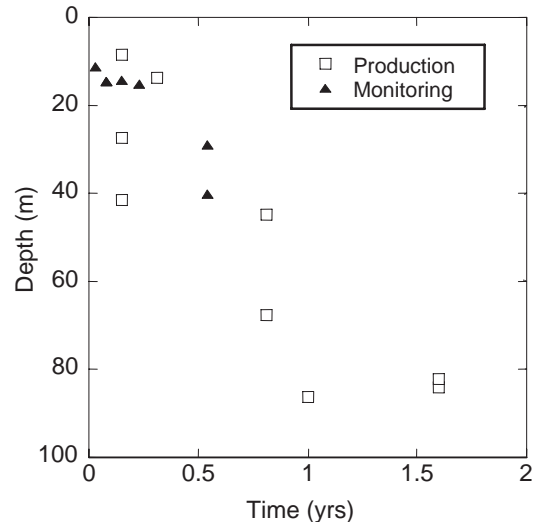
Figure 3. Vertical travel times to production and monitoring wells located within 150 m of the Montebello spreading basins during the SF_6 tracer experiment

SF_6 was detected at nine of the eighteen production wells. The maximum concentrations observed at the production wells were much lower than at the monitoring wells. At seven out of the nine production wells with SF_6 detections, the maximum concentrations were less than 0.4 pmol/L or an order of magnitude less than at the monitoring wells and two orders of magnitude less than the spreading basins. With the exception of two wells (#200061 and #200065), SF_6 peaks were difficult to resolve as tracer concentrations were near the limits of detection and the breakthrough curves were not smooth. The lower concentrations and complex breakthrough curves are probably due to dilution of the tagged water with untagged ground water caused by the long screened intervals. Similar to the monitoring wells, there is a strong relationship between tracer arrival time and depth to the top of the perforation (Figure 3) and no correlation between distance and travel time. Depth may be the most important factor influencing travel time because the deeper the well perforation is, the more likely is it for the screen to be situated below layers with low hydraulic conductivity.

The hydrogeologic analysis conducted by Bookman-Edmonston (1994) divided the production wells into two main categories. The first consisted of four shallow wells (#200061, 200055, 200058, 200065) screened in the unconfined aquifer. For all of these wells, the hydrogeologic and SF_6 travel times are consistent, both indicating travel

times of 0.3 years (16 weeks) or less. The hydrogeologic study identifies all other wells as having perforations in a confined aquifer with little direct contact with surface water. At these wells, estimated travel times range between 2.5 to 14 years. The longer travel times were caused by slow flow through the confining clay layers. Within the time constraints of the two-year tracer experiment, a complete analysis of the accuracy of the hydrogeologic travel times cannot be completed. However, four of the nine production wells near the Rio Hondo spreading grounds (#200062, 200088, 200089, 200099) that were determined to be in the confined aquifer, received tracer. The five wells that did not receive tracer are all located on the western side of the Rio Hondo River while the four wells that did see tracer are located on the eastern side. There is no discernable difference in the well log geology between the eastern and western sides. Nevertheless, the comparison between the tracer and hydrogeologic travel times suggests that discontinuities (gaps, fractures, or inter-bedded layers of course material) within the clay layers exist on the eastern side. Of the five production wells studied near the San Gabriel spreading basins, only one well (#200012) screened in the lower confined aquifer received tracer in less than a year. All other deep wells studied near this river did not receive tracer during the study.

The T/^{3}He apparent ages at production wells were between 7 and 40 years, all significantly older than travel times determined by the other two methods. The older apparent ages reflect mixing of young and old ground water within the well. The old component must have a residence time of decades and could make up the largest fraction at some wells. Because T/^{3}He ages do not mix linearly (the mixed age is weighted by each flow path's initial tritium content) this technique leads to over-estimates of the mean travel time. Good agreement between travel times determined with deliberate tracer experiments and T/^{3}He ages were found at three of the four monitoring wells where both techniques were used. Good agreement has also been found at other artificial recharge sites (Clark et al., 2004).

IMPLICATIONS

Each method of determining the travel time between the recharge basins and wells used during this study is based on different principles and assumptions. A conceptually better model of the flow and travel times can be achieved by comparing the results of the three methods. With deliberate tracer experiments, such as the one performed with SF$_6$ here, travel along the fastest flow paths is quantified. Results of deliberate tracer experiments are limited by their duration and by dilution with untagged water which can lower tracer concentrations below the detect limit. Hydrogeologic calculations require local geologic information usually extrapolated from well bore measurements to estimate the flow along specific paths. Because details about preferential flow paths are usually lacking, these calculations estimate mean travel times and rates. The use of a deliberate tracer experiment concurrently with hydrogeologic calculations can either validate or lead to better models of the hydro-stratigraphy. When significant mixing of young and old ground waters occurs as in most production wells, T/^{3}He apparent ages are the most difficult to interpret. If linear mixing occurs, then these ages would be equivalent to the mean age. However, because the initial tritium content of recharge water varied by orders of magnitude during the last few decades, T/^{3}He apparent ages do not mix linearly. Nevertheless, the T/^{3}He apparent ages from this study indicate that productions wells are drawing in an old component and that this component has a travel time greater than a decade.

The California State Department of Health recommends that ground water containing reclaimed water reside in the subsurface for a minimum of six months to allow for virus inactivation (Bookman-Edmonston, 1994). In order to ensure the six-month residence time, all wells must be farther than 150 m from the point of infiltration. This study suggests that using horizontal distances does not ensure a specified travel time within the ground water system; the depth of production is a better criterion to ensure desired residence time close to spreading ponds.

ACKNOWLEDGEMENTS

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Use of geochemical and isotope plots to determine recharge to alluvial aquifers: Lockyer Valley, Queensland, Australia

Malcolm E. Cox and Andrew S. Wilson

Abstract

The source of groundwater recharge to alluvial aquifers of the Lockyer Valley is difficult to confirm, especially as water chemistry of bores is highly variable. To better define recharge over 100 bores have been analysed for major ions and stable isotopic ratios δ^2H and $\delta^{18}O$. Preliminary identification of hydrological processes was by ionic ratios (e.g. Cl/HCO3) and isotope plots, however, although these methods can indicate the source of water, they cannot show the degree of mixing. A plot of log TDS (mg/L) versus δ^2H ‰ was used with some success. Within the plot four 'end members' can be established, with typical values: A. sandstone bedrock; B. stream storm flow; C. deep artesian basin; and D. strongly evaporated alluvial groundwater. Within the ABC triangle relative percentages of each member can be determined; within the ABD triangle relative percentages of stream recharge and sandstone water can be established, and the degree of evaporation is indicated. The plot confirms the primary process of recharge to alluvial aquifers is directly as storm flow in streams, and indirectly from surrounding basalts. The contribution to alluvium of bedrock sandstone water is also shown to be significant during periods of low stream flow and low groundwater levels.

Keywords

Alluvial aquifers, Australia, evaporation, groundwater, irrigation, recharge, stable isotopes.

INTRODUCTION

The Lockyer Valley of southeast Queensland is an important area of crop production based on intensive groundwater irrigation. The agricultural development has placed excessive demands on groundwater and there is a trend of increasing salinity in many parts of the valley. As a consequence various studies have reported aspects of water chemistry and isotope composition of groundwater in the Lockyer Valley (e.g. Talbot and Dickson, 1969; Stevens, 1990; Dixon and Chiswell, 1992; Dixon and Chiswell, 1994).

The Lockyer Valley supplies around 35% of the state's irrigated vegetables as both winter and spring crops, plus fodder and small crops. Prior to 1936 irrigation in the Lockyer was limited, but by 1956 the area under irrigation had increased to 3,500 ha, and by 1969 to 12,000 ha. By the mid-1970s the area irrigated had stabilised at around 13,000 ha; an estimate is that on the order of 16,800 ha of the valley has potential for irrigation. More recently there has been rural subdivision and the establishment of hobby farms. The supply of irrigation water in the catchment is closely related to rainfall. During the period 1990 to 1995 the severe drought affecting much of eastern Australia had a marked impact on irrigation in the valley. This drought broke in November, 1995, and in January, 1996 the area was subject to flooding, but dry conditions have subsequently reoccurred. There are substantial variations in water quality in the Lockyer, both spatial and temporal, and these can be difficult to assess with the irregular data on file. Results of studies to date show that hydrological processes are complex, notably due to the mixing of various waters.

The source and character of recharge to the groundwaters of the alluvial aquifers have been difficult to confirm, especially as water chemistry of bores within both the alluvium and the underlying sandstone is highly variable throughout the valley (Cox et al., 1996). Many alluvial bores also have a component of bedrock input, which

typically increases during dry period irrigation. Also of significance is natural evaporation in streams of poor flowand of irrigation runoff, plus the discharge of deep artesian basin CO_2–bearing waters. Groundwater in the basalt aquifers of the surrounding ranges is also chemically different.

PHYSICAL SETTING

Climatic factors

The Lockyer Valley is a roughly circular basin of approximately 2,500 km^2 and is formed of a sequence of sedimentary rocks which are dominantly sandstone. Central to the valley is the meandering Lockyer Creek, which drains eastward to the Brisbane River. The town of Gatton, population of 4,600, is the largest in the valley and is 90 km west of the state capital Brisbane (Fig. 1). The climate of the area is sub-tropical and humid, with relatively long hot summers and winters that are short and mild with occasional frosts. Rainfall central to the valley has an annual average of 720 mm, while the annual rainfall increases on the adjacent ranges to the west and south to around 1,200 mm. Stream flow for much of the drainage system is ephemeral in nature and related to summer storms (November-March). There is a marked rainfall deficit and the average annual rate of evaporation is estimated to be 1,900 mm (Queensland Bureau of Meteorology, unpub. data). Alluvial groundwater levels respond rapidly to storm stream flow, and are strongly influenced by prolonged dry periods as well as the associated increase in irrigation extraction; water tables are now commonly below stream bed levels.
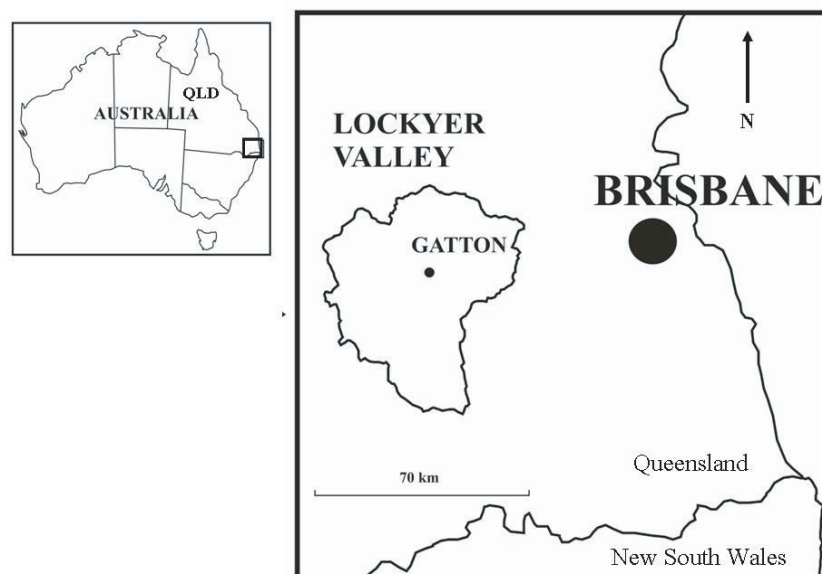
Figure 1. Location of the Lockyer Valley in southeast Queensland and in relation to Brisbane

Morphology and hydrogeology

The valley is bordered on the south and west by the Great Dividing Range which is typically 250–300 m higher than the valley floor. This range is flat-topped and covered by Tertiary age basalt flows which weather to form dark fertile soils. The headwaters of the larger tributaries of the Lockyer drainage system abutt the basaltic ranges and escarpments in the south and west. To the north, the drainage divide is less well defined, and streams flowing south to Lockyer Creek are smaller and more irregular in nature. In that area, the rock types are predominantly coarse sandstone and granite, and of undulating topography. These northern subcatchments have limited development of alluvium and as a consequence less irrigation.

The sedimentary formations of the valley consist of a conformable sequence of Triassic and Jurassic sediments and are described in detail by Cranfield (1981) and Wells and O'Brien (1993). These sequences predominantly consist of flat-lying fluvial sandstones, siltstones and shales, with minor conglomerate units (Fig. 2). Here we use a generalised description for the sedimentary formations of, lower: Gatton Sandstone (exposed in central valley floor along Lockyer Creek), middle: Winwill Conglomerate, and upper: Koukandowie Formation (exposed in sides of valley and upper slopes).
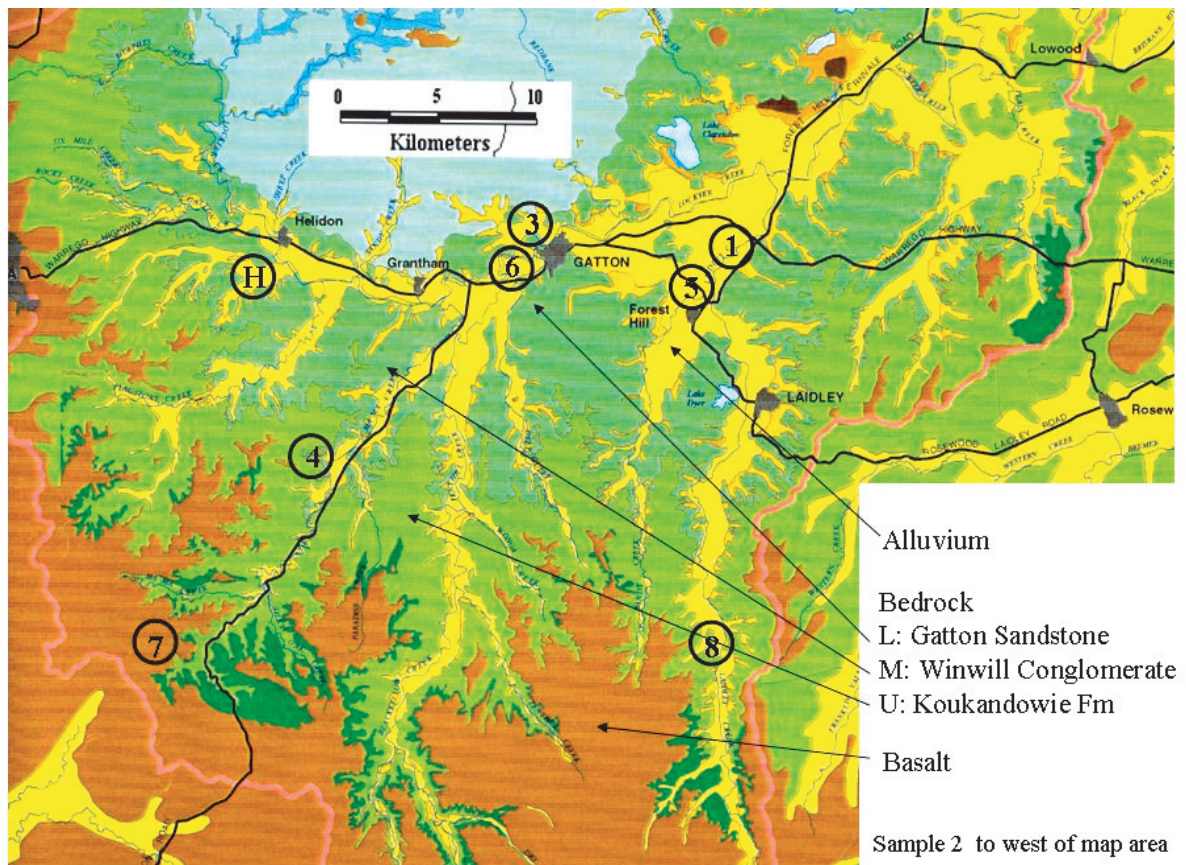
Figure 2. Location and geology map of the Lockyer Valley (after Department of Natural Resources and Mines). Lockyer Creek runs west to east past Gatton in the central valley. Locations of selected groundwater bores are numbered; H is Helidon, a spa and spring area.

The groundwater supplies used for irrigation in the area are largely obtained from the Quaternary alluvium, which within the Lockyer drainage system covers approximately 28,000 ha. Water bores show the alluvium to be between 20 and 30 m thick in the central drainage systems, and to directly overlie the sedimentary formations. Alluvial deposits are composed of well-graded gravel, sand and silt with a clayey matrix; cobbles also occur in channel deposits. Clay-rich layers are common in the broad alluvium of the middle sections of Lockyer Creek. Typically, the lower section of the alluvium is composed of coarser sands and is the main water-bearing layer; commonly this layer can be semi-confined by silt-clay layers. Hydraulic conductivities (K) of these coarse sand-gravel layers are typically 50 to 80 m/day (Wilson, 2004).

In Lockyer Creek and its larger tributaries the stream channels tend to be incised into the alluvium, commonly 3–5 m, are often narrow and without distinct alluvial terraces. Over time these channels have meandered back and forth across valley floors and have formed floodplains of rich soil which can be 2 to 6 km in width. In dry periods, very little water is seen in the drainage system, but after heavy storms, stream channels rapidly receive and transmit flow.

Surface and groundwater supply

Stream water is the main source of recharge to the alluvial aquifers and the response of groundwater levels to flood flows in the stream is rapid. During dry periods, however, such stream flow may only be in upper parts of the catchment. Water tables typically rise 7 to 10 days after several days of heavy rain in the headwaters or on the escarpment. Available records confirm that in dry periods the water table is below the creek beds and that hydraulic gradients slope towards the edges of the valleys. Small weirs to increase alluvial recharge have been built on most tributaries and although subject to some siltation most of these weirs are achieving an annual recharge of around 700 ML (DPI-WR, 1993). There are now 16 artificial recharge weirs on streams throughout the valley.

State government licensing for use of surface water supplies has been a requirement for many years. However, except for the central section of the valley around Gatton the use of groundwater for irrigation, and other purposes, has not been subject to the same statutory controls. Most of the expansion of irrigation in the 1960s and 1970s was from groundwater, which now accounts for over 80% of irrigation supply, and corresponds to an average annual groundwater withdrawal of some 45,000 ML. Groundwater storage in the alluvial aquifers was estimated at a safe annual yield of around 25,000 ML (e.g. DPI-WR, 1993). These amounts indicate that the alluvial aquifers are being over-exploited, which is clearly evident during dry periods when many bores become dry. The demand for irrigation water therefore exceeds supply, except in the wettest seasons. A preliminary valley-wide assessment of groundwater data over a 30 year period (Tien et al., 2004) indicates that the long-term trend is of increasing salinity, which based on the sites tested is 60%, or approximately 2% per year. The overall trend during this period is for decreasing groundwater levels, particularly over the last 15 years.
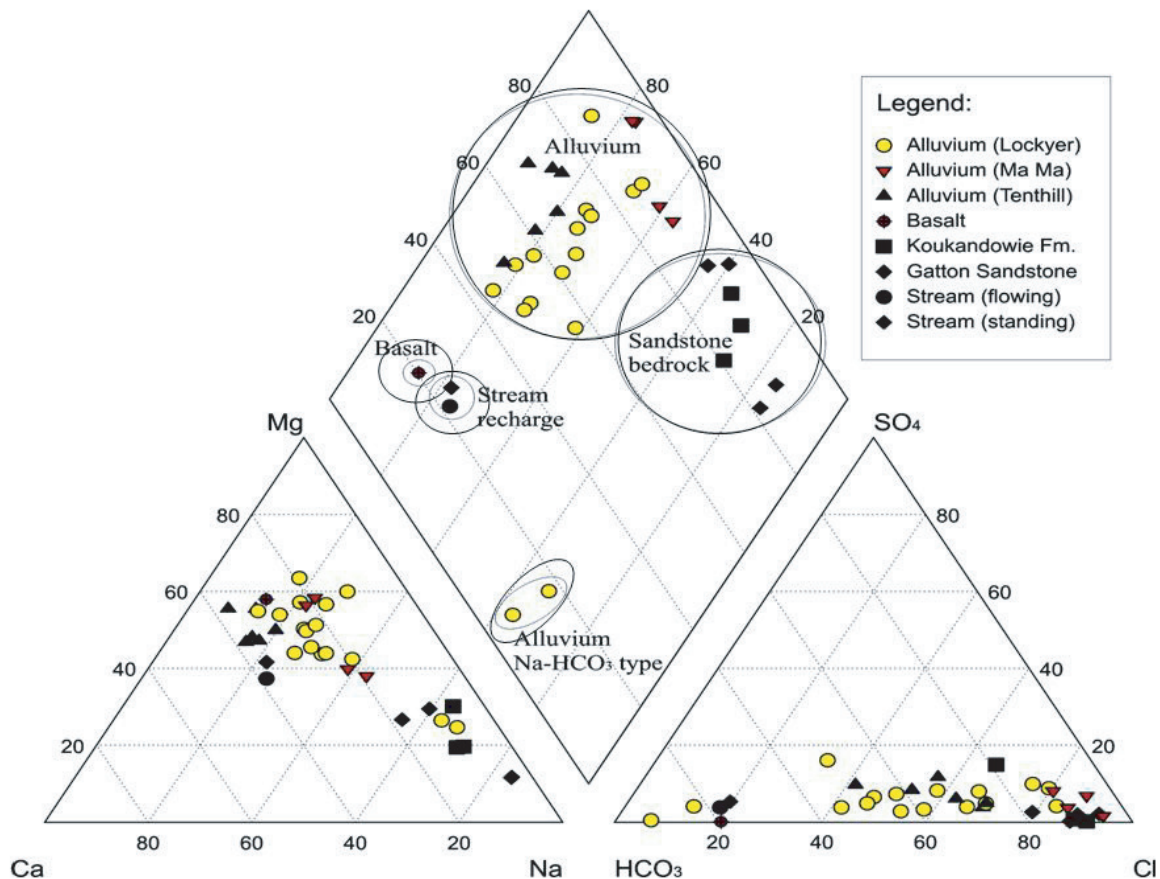
Figure 3. Piper diagram of groundwaters in the southwestern Lockyer Valley collected during 2003 displaying chemical difference between different groundwaters.

RECHARGE TO ALLUVIAL AQUIFERS

Background

The variation of surface and groundwater chemical composition throughout the valley reflects a variety of sources and mixing of waters. Although it is generally accepted that recharge is primarily derived from the surrounding ranges, the alternative sources and possibly proportions to the alluvial aquifers have not been determined.

The use of hydrogeochemistry and of stable isotope analyses have provided some insight to recharge and mixing, at least in a qualitative sense. Studies by Dixon and Chiswell (1992; 1994) in the southwest of the valley confirmed that bedrock groundwaters are identifiable and do discharge into the alluvial aquifers. Studies in Sandy Creek in the southeast (McMahon, 1995; McMahon and Cox, 1996) confirmed that for groundwater within the sedimentary profile there is both a variation of salinity and chemical composition for different formations. The study by Mcleod (1998) of Ma Ma Creek in the southwest confirmed that with continued irrigation under dry conditions a greater proportion of bedrock groundwater is drawn into the alluvial bores. All studies show that overall, groundwater within the lower bedrock formations has a higher total dissolved solids. A recent investigation (Picarel, 2004) in the lower (eastern) Lockyer Creek confirms that some very high saline groundwaters are present in bedrock depressions. That study also demonstrated that 2004 water levels have been drawndown into the lower sections of the alluvium.
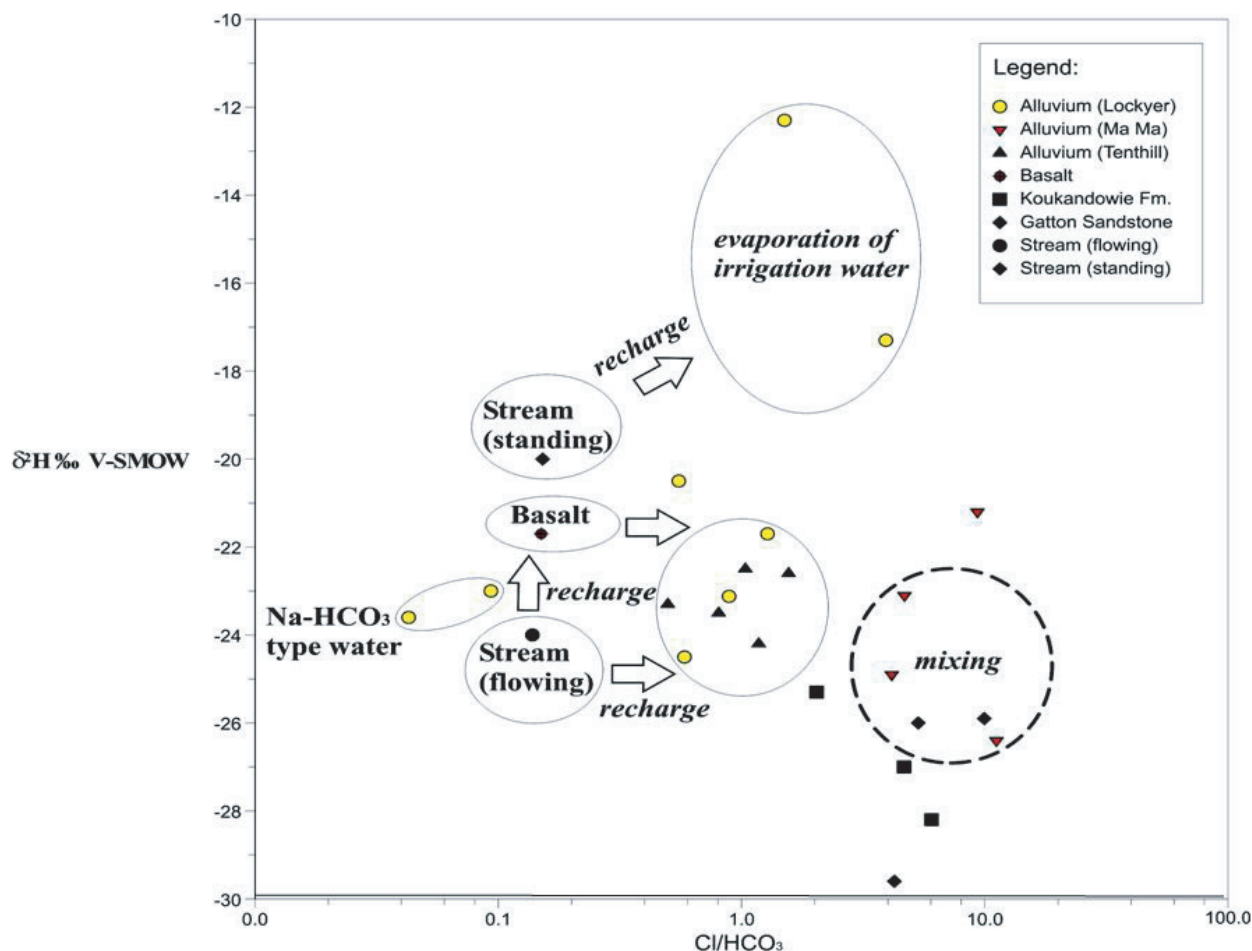
Figure 4. Plot of $\delta^2 H$‰ versus Cl/HCO_3 for groundwaters in the southwestern Lockyer Valley collected during 2003 displaying processes such as mixing and evaporation.

Chemical composition of groundwaters

Although most bores extract from alluvium some also extract a component of water from the underlying sandstones. Most bores in the valley are drilled 30–50 cm into bedrock, and are screened in the lower 1–2 m of the alluvium. The proportion of this latter groundwater can be also seen as baseflow and increases during periods of low river flow, and also downstream in many drainages. For example, the mainstream Lockyer is mostly Na,Mg,Ca-HCO_3 water, but at low flows a Na,Mg-Cl water dominates. The highest salinity alluvial groundwaters are mainly in the centre of the valley, concentrated in the floodplains that support the most intensive agriculture. There are, however, conflicting views as to whether the high salinity levels in certain areas are a natural occurrence, or whether they have been worsened by poor quality water entering the alluvial aquifers.

Recent investigations in the Tenthill-Ma Ma catchments in the southwest and the alluvial plain at their conflence with Lockyer Creek (Wilson, 2004; Wilson et al., 2004) were effective in identifying groundwaters from different aquifers. Alluvial aquifers are dominated by waters of Mg,Na(Ca)-Cl(HCO_3) composition (Fig. 3). It can be seen in the Piper plot that in the central valley groundwater in the alluvium has a greater proportion of HCO_3; although this can reflect a rainfall recharge influence, it can also be related to high HCO_3 deep waters. This plot clearly demonstrates the Na-Cl nature of groundwater in the sandstone bedrock; basalt groundwater from the surrounding ranges is commonly of a Mg-HCO_3 type, and similar to flowing streams (i.e. recharging waters).

Salinity variations also exist between these various waters and typically depend on, (a) bedrock formation, and (b) irrigation practice, or both. Typical salinities (mg/L) are, alluvium: Lockyer (800–3,700), Tenthill (1,000–2,500) and Ma Ma (5,400–7,700); sandstone: Gatton, lower (5,700–7,500) and Koukandowie, middle-upper (3,000–6,800).
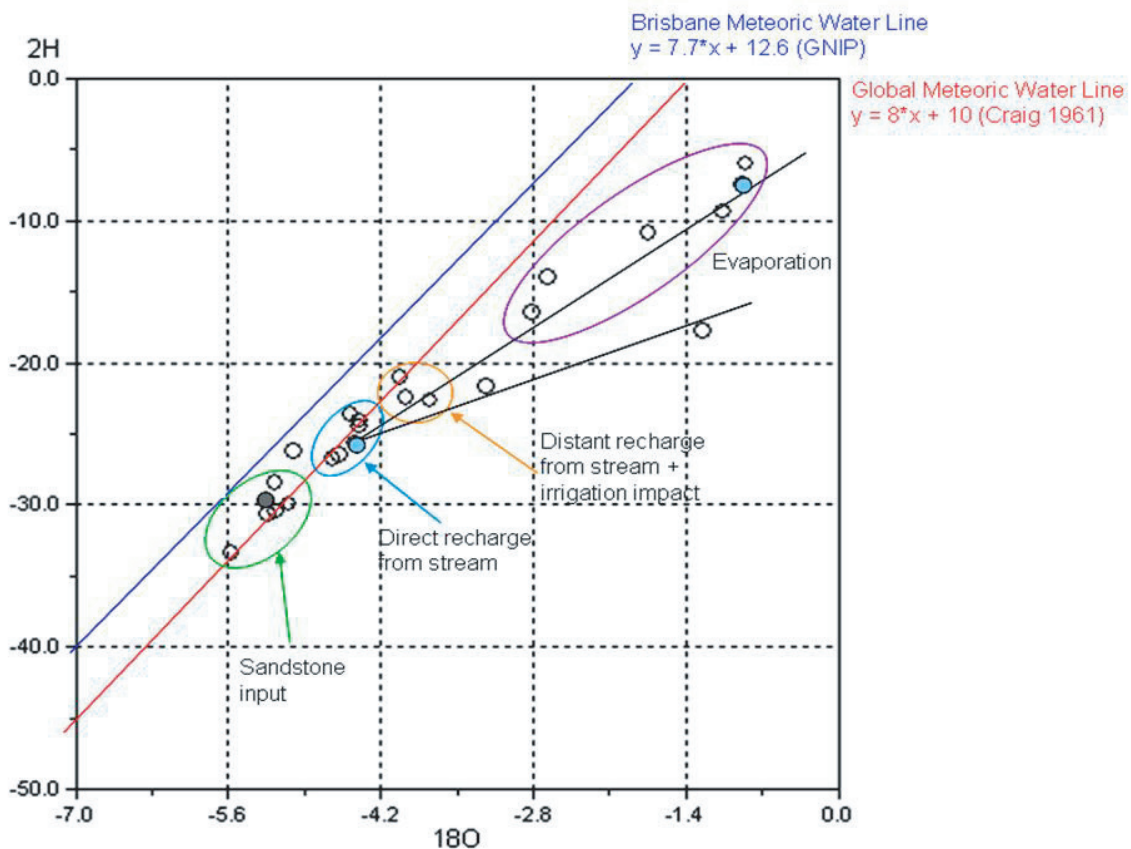
Figure 5. Plot of $\delta^2 H$ versus $\delta^{18}O$ for groundwaters from the eastern section of the Lockyer Valley (in northeast of Figure 2). Samples were collected in 2004 (after Picarel, 2004).

Both hydrogeochemical and isotope methods provide some understanding of the hydrological processes occuring. The Piper plot indicates some degree of mixing between waters. These processes are displyed well in a plot of δ^2H‰ versus Cl/HCO$_3$ (Fig. 4), for which both parameters indicate the source of the water. Flowing streams represent recharging water and runoff or throughflow from basalt aquifers; alluvial and bedrock groundwaters are evident lower in the plot, as is some mixing between them. Typically, groundwater from bedrock aquifers has the most depleted values of δ^2H‰ and higher Cl/HCO$_3$ ratios. Evaporative processes in standing streams and of irrigation water are evident in the upper part of the figure.

As an additional method of determining hydrological processes in the Lockyer Valley, groundwater samples from the lower (eastern) alluvial irrigation area were analysed for stable isotopes (Fig. 5). In this part of the valley alluvium is largely overlying the lower bedrock formation, the Gatton Sandstone. The isotopic compositions for most samples tend to fall along the Global Meteoric Water Line (Craig, 1961). Samples of flowing streams and alluvial groundwater immediate to streams are grouped on this MWL; groups with differing degrees of surface or near-surface evaporative concentration fall along a line with a steep slope. The shaded samples in the 'stream recharge' and 'evaporation' groups are stream waters; the latter is from a non-flowing pool. The lower trend in δ^{18}O enrichment is groundwaters in alluvial bores that are distant from the stream, and contain re-circulated irrigation water. Alluvial bores with significant bedrock input (plus bores into sandstone) are shown to be the most depleted; this supports the idea of recharge to sandstone bedrock aquifers as being distant (further to the west or south).
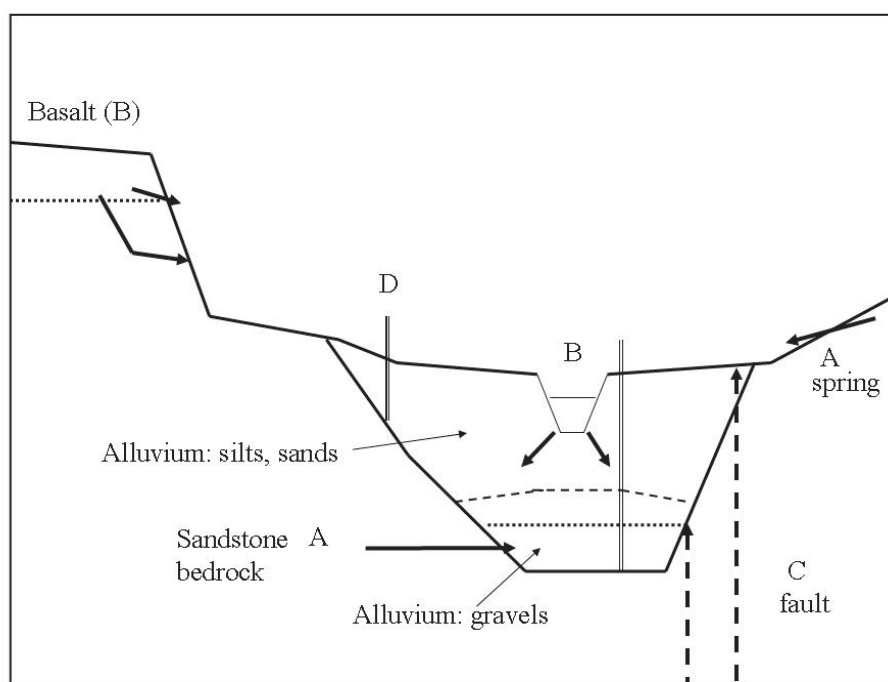
Figure 6. Main processes of the hydrological systems related to recharge of groundwater within alluvial aquifers of Lockyer Valley. Two examples of groundwater irrigation bores are shown; the deeper type is the most productive (e.g. 5–8 L/sec) and produces the most drawdown.

DETERMINATION OF SOURCE OF RECHARGE

Hydrological processes

The plots above demonstrate the type of hydrological processes that occur within the valley, and the obvious connection between groundwater and surface waters. The main forms of recharge are also indicated. These processes are summarised in Figure 6, in which features A to D also represent endpoints in the ternary diagrams of Figure 7.

These processes are as follow:

A: discharge from sandstone bedrock as (a) seepage into permeable sands and gravels at base of the alluvium, (b) surface discharge from springs in outcropping bedrock, and (c) seepage along upper sections of stream courses;

B: recharge of low salinity water from (a) streams during periods of flow, and (b) basalts on surrounding ranges as throughflow to stream headwaters;

C: discharge of deep basin waters (Great Artesian Basin) from faults at base of alluvium, or from exposed bedrock. These waters are cool, but contain abundant CO_2;

D: alluvial groundwaters that have experienced evaporative concentration, usually towards the edges of alluvial plains or at shallow depth.

Of note is the overall low permeability of the shalower profile of the alluvium, due to high silt content. Low infiltration rates were determined for surface alluvial material in the central parts of the valley (e.g. Ellis and Dharmasiri, 1998). This finding supports the conclusion that recharge to the alluvium is dominantly along the drainage system. Flow of groundwater through the sandstones is relatively slow, and recharge to these foramtions is distant to the immediate valley (e.g. Dharmasiri et al., 1997).

Definition of proportion of recharge

To assist in the future management of the groundwater resources better definition of recharge processes will be of value. Although the forms of recharge are relatively well understood, no quantification of amount has been reported. As shown above water chemistry and isotope character can reflect many processes. Further consideration of over 100 samples from throughout the valley indicated some merit in a plot of log TDS (total dissolved solids, mg/L) and δ^2H (‰).

Figure 7 (upper) displays a diamond-shape field consisting of four end points and composed of two ternary plots. The end points are the representative values for different water types and are based on 2–4 samples. The left triangle ABC is the field for sandstone-stream-deep basin sources; the right triangle ADC is the field for sandstone-stream-evaporation. The two triangles are divided by lines of 10%; the position within each triangle enables the relative proportion of each source to be calculated. For the ADC triangle the endpoints are stream waters, sandstone waters and groundwaters influenced by evaporation; as the latter parameter is relative fields of low, medium and high are identified.

In Figure 7 (lower) eight examples of groundwaters from a variety of bores throughout the valley have been assessed to test the method.

Results are summarised in Table 1, showing the indicated percent recharge from each primary source, as well as TDS (mg/L). It is clear that bores with a larger component of sandstone groundwater input are more saline. These results are compared to Cl/HCO_3: bores with a deep basin component typically have very low values, however, low Cl/HCO_3 values can also indicate samples more proximal to rainfall recharge areas near to stream headwaters (e.g. sites 7 and 8).
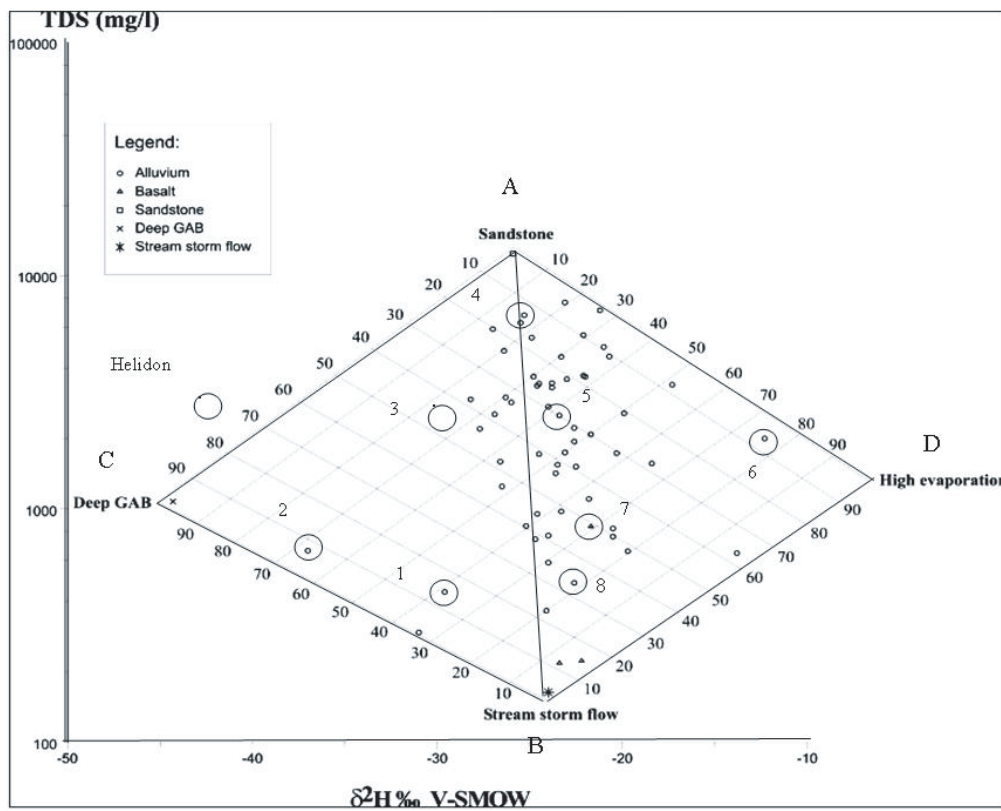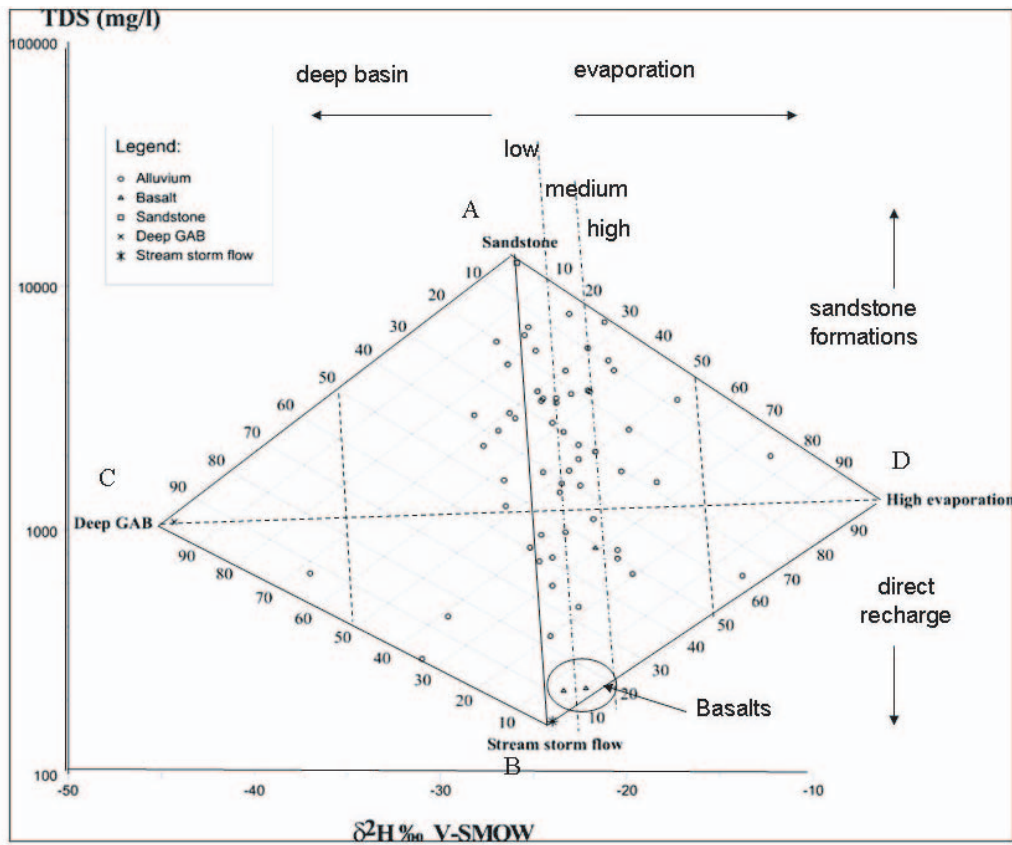
Figure 7. Plots of log TDS (mg/L) and δ^2H (‰) showing (upper) various fields,
and (lower) examples of water samples summarised in Table 1.

Table 1. Parameters of selected groundwaters

Location and type	Water origin (%)			TDS (mg/L)	δ^2H ‰	Cl/HCO₃
1. Bore: alluvium (755/LV20) (to lower sandstones)	A. B. C.	Sandstone Recharge Basin	14 60 26	620	-29.6	0.08
2. Bore: deep lower sandstones (LV22; Oakey to west)	A. B. C.	Sandstone Recharge Basin	8 32 60	650	-37.0	0.44
3. Bore: alluvium (822) (to lower sandstones)	A. B. C.	Sandstone Recharge Basin	55 25 20	3350	-29.6	0.16
4. Spring: from upper sandstones (GW 27)	A. B. D.	Sandstone Recharge Evapn low	84 13 (~3)	9750	-25.3	2.04
5. Bore: alluvium (GMC) (to lower sandstones)	A. B. D.	Sandstone Recharge Evapn low	57 30 (~13)	3300	-24.0	2.59
6. Bore: alluvium (502) (to lower sandstones)	A. B. D.	Sandstone Recharge Evapn very high	24 6 (~70)	2300	-12.3	1.50
7. Bore: basalts on ranges (LV16) (to upper sandstones)	A. B. D.	Sandstone Recharge Evapn medium	30 50 (~20)	830	-21.7	0.15
8. Bore: alluvium (LV4) (to upper sandstones)	A. B. D.	Sandstone Recharge Evapn low	21 67 (~12)	450	-22.6	0.26

CONCLUSIONS

In summary, the log TDS (mg/L) and δ^2H (‰) plot shows the major forms of recharge are directly from streams and indirectly from basalts, and from sandstone bedrock. A wide range of degress of mixing occur, and temporal variation of rainfall is also a factor. The latter is especially the case during extended dry periods. In the ABC field, some bores are shown to have a component of deep basin water; although this is the case for some deep bores, a number of bores in the valley floor alluvium have a high HCO_3 content, suggesting shallow dispersion of such water. For exmple, bore 1 near Laidley may have such an input from a concealed fault. Of note, the Helidon spa water falls outside the field, but is indicated to be 80% deep basin water; these waters spread over 100 metres along the drainage system. In the ADC field most alluvial bores display minor or medium evaporative concentration, but a number (e.g. 6) are shown to have experienced marked evaporation.

ACKNOWLEDGEMENTS

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Effect of sprinkling infiltration on soil acidity and fertility properties on a forested esker in Central Finland

J. Derome, A.-J. Lindroos and H.-S. Helmisaari

Abstract

The aim of the study was to determine the acidity and nutrient status of the humus and uppermost mineral soil layers of a forest soil after two years of sprinkling infiltration, and one and two years after cessation of the treatment. Sprinkling infiltration was carried out in a Scots pine stand located on an esker in Central Finland. Humus and mineral soil samples were taken from two control plots and two plots subjected to sprinkling infiltration. As a result of sprinkling infiltration, the base saturation of the humus layer was close to 100%, and the pH above 6. The cation exchange capacity of the humus layer had increased considerably owing to the increase in pH. The humus layer, as well as to a lesser extent the uppermost mineral soil layers, had retained large amounts of Ca and Mg from the lake water used in infiltration, but there was no corresponding increase in exchangeable K concentrations. The treatment strongly reduced the concentrations of extractable P, presumably due to immobilization at elevated pH values. Sprinkling infiltration appears to have a relatively long-lasting, positive effect on the acidity and nutrient status of the surface soil on the forested esker.

Keywords

Base saturation; cation exchange capacity; forest soil; macronutrient; pH; sprinkling infiltration.

INTRODUCTION

Artificial recharge of groundwater is an important method for producing household water in Finland. One possible way to recharge groundwater artificially is to infiltrate surface water through the forest soil and unsaturated percolation water zone down into the saturated groundwater zone. This kind of infiltration method is called sprinkling infiltration. Special attention has to be paid to environmental aspects when planning artificial recharge groundwater plants in forested areas. In sprinkling infiltration, the forest ecosystem is subjected to the addition of extremely large amounts of water, the chemical composition of which differs considerably from that of natural precipitation. Appreciable changes in the pH and exchangeable Ca and Mg concentrations of the forest soil have been reported in a study carried out on a forested esker subjected to sprinkling infiltration in southern Finland (Lindroos et al., 2001). Sprinkling infiltration has also caused marked changes in nitrogen transformation in the same study area (Lindroos et al., 1998; Paavolainen et al., 2000).

Although information is already available about possible changes in the chemical properties of the forest soil due to surface water infiltration at some sites in Finland, more data are needed about the behaviour of these processes in different kinds of boreal forest stand and podzolic soils. It is also important to obtain more information about the duration of the changes in the chemical properties of the forest soil after surface water infiltration has ceased, and about the possible recovery of the soil to its original condition. The aim of this study was to determine the acidity and nutrient status of the humus layer of a forest soil at the end of the treatment, and one and two years after the cessation of sprinkling infiltration. The studied artificial recharge (AR) groundwater area is located in central Finland, and the AR groundwater plant produces household water for the city of Jyväskylä.

METHODS

The study site is located in a 110 year-old Scots pine stand growing on an esker near to the city of Jyväskylä in Central Finland. The sub-xeric site is of the *Vaccinium vitis-idaea* forest site type (Cajander, 1949), and the soil consists of relatively coarse, stratified sand deposits. The soil type is a humic podzol. Soil samples were taken in autumn 2001, 2002 and in 2003 from two control plots and from two plots that had been subjected to sprinkling infiltration during 1999–2001. The organic layer and mineral soil (0–5, 5–10, 10–20 and 20–40 cm depth layers) were sampled at 20 systematically located points on each plot (30 x 30 m in size). The samples were combined to give 2 samples/layer/plot. The samples were dried at 60 °C, and the organic layer samples then milled to pass through a 1 mm sieve and the mineral soil samples passed through a 2 mm sieve to remove stones and large roots. pH was determined in a soil/distilled water slurry (15 ml soil/25 ml H_2O). Exchangeable Ca, Mg, K and Na were determined by extracting the samples with 1 M $BaCl_2$ (3.75 g organic layer sample or 15 g mineral soil/150 ml extractant), and exchangeable acidity (EA) by titrating the extract to pH 7. The element concentrations were determined by inductively coupled plasma atomic emission spectrometry (ICP/AES). Cation exchange capacity (CEC) was determined as the sum of equilavent concentrations of Ca, Mg, K, Na and EA. Base saturation (BS) was calculated as the proportion of base cations (Ca, Mg, K, Na) out of CEC.

RESULTS AND DISCUSSION

Acidity

During the two-year sprinkling irrigation treatment, the pH of the organic and mineral soil layers (Fig. 1) had increased to ca. 6, which is relatively close to that of the lake water (mean pH 6.9) used for AR. These pH values are very high compared to the values normally measured in the Finnish forest soils (Tamminen, 2000). During the two-year period after the cessation of the treatment the pH decreased to some extent (ca. 1.5 pH-units), but was still clearly above the level on the control plots. This increase is considerable considering the fact that the lake water was not sprinkled evenly over the whole plot (30 x 30 m). A similar, rather strong decrease in soil acidity has been observed in studies carried out at other sprinkling infiltration plants in Finland, and at these sites the pH has not returned to its original level even 5 years after the cessation of sprinkling infiltration (Helmisaari et al., 2003).
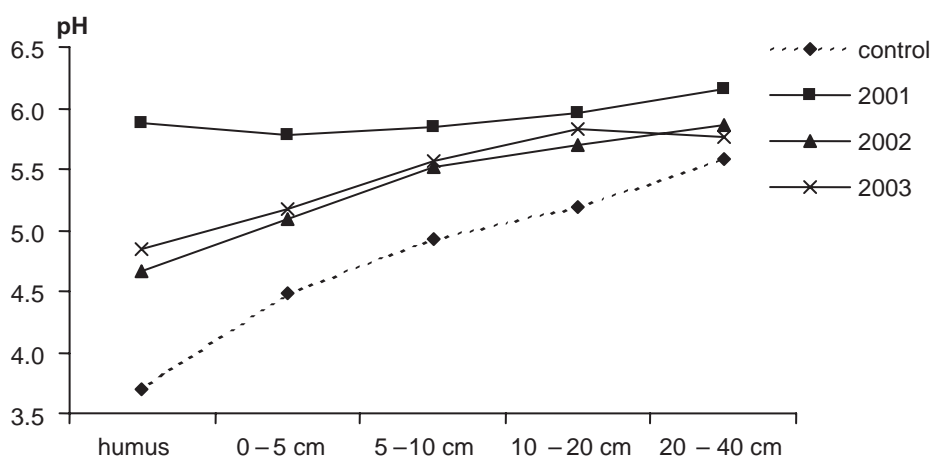
Figure 1. Mean pH in the humus layer and at different depths in the mineral soil on the control plots, and on the treated plots at the cessation of sprinkling infiltration (2001), and one year (2002) and two years (2003) after cessation of the treatment.

Cation exchange capacity and base saturation

Cation exchange capacity (CEC) is a measure of the number of negatively charged cation exchange sites in the soil, i.e. the soil's capacity to bind cations, and hence CEC plays an important role in maintaining site fertility. The cations bound on the cation exchange sites include important plant nutrients such as Mg^{2+} and Ca^{2+}, as well as the acidic cations Al^{3+} and H^+. In forest soils, which are usually relatively coarse textured and have a low content of fine material (e.g. silt and clay) in Finland, the majority of the cation exchange sites are associated with the organic matter in the humus layer and underlying mineral soil. As cation exchange sites are formed through the dissociation of so-called functional groups (e.g. carboxyls) in the organic matter, CEC in forest soils is strongly dependent on soil pH: the higher the pH, the higher the proportion of dissociated functional groups, and the greater the CEC value. Sprinkling infiltration considerably increased the CEC of the humus layer, and it remained at approximately the same level for two years after the cessation of the treatment (Fig. 2). Although the increase in pH (Fig. 1) undoubtedly explains part of the increase in CEC, changes will most probably also have occurred in the quality of the organic matter in the humus layer. The increase in available nitrogen (data not shown) and the high concentrations of both Ca and Mg, have probably stimulated microbial degradation of the relatively acidic, slowly decomposing coniferous litter layer characteristic of forest soils. The CEC in the mineral soil layers appeared to have been relatively unaffected by the treatment..
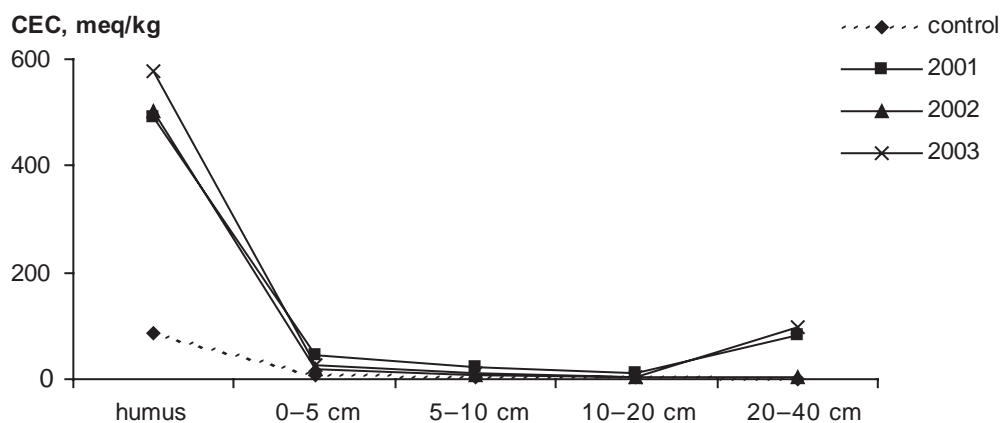
Figure 2. Mean cation exchange capacity (meq/kg dm) in the humus layer and at different depths in the mineral soil on the control plots, and on the treated plots at the cessation of sprinkling infiltration (2001), and one year (2002) and two years (2003) after cessation of the treatment. Note: owing to the large difference in the bulk density of the humus layer and mineral soil, the CEC values in the humus and mineral soil are not directly comparable.

Base saturation is a measure used to depict the proportion of cation exchange sites occupied by so-called base cations (Ca^{2+}, Mg^{2+}, K^+, Na^+). The maximum value of BS, i.e. 100%, means that all the cation exchange sites are occupied by base cations. Sprinkling infiltration strongly increased BS in the humus layer to over 95% during the treatment, and there was no decrease following cessation of the treatment (Fig. 3). There was also an extremely strong increase in the mineral soil, and BS continued to increase following cessation of the treatment. This is undoubtedly due to the gradual movement of e.g. Ca and Mg cations down the soil profile, following their release in the overlying humus layer. The base cations originate from the lake water and, although the base cation concentrations in the lake water were relatively low, the total amount of Ca and Mg added in the lake water during the two-year treatment is extremely large due to the large amounts of infiltrated water. These base cations have obviously displaced the other cations (e.g. H^+ and Al^{3+}, data not shown) from cation exchange sites in the humus layer and mineral soil, thus resulting in an increase in soil pH (Fig. 1).

Exchangeable calcium, magnesium and potassium concentrations

The concentrations of exchangeable Ca and Mg in the humus layer increased 3-fold as a result of the sprinkling infiltration treatment (Table 1). As well as playing an important role in acid buffering processes in the humus layer, Ca and Mg are important macronutrients. It is clear that the increase in exchangeable Ca and Mg, together with the increased availability of nitrogen (data not shown), represents an improvement in the fertility of the site. Despite the fact that the soil is relatively coarse-textured, the increase in site fertility is likely to persist for a considerable length of time because the increase in CEC, and the migration of organic matter down into the underlying mineral soil (data not shown), will reduce the losses of Ca and Mg through leaching.
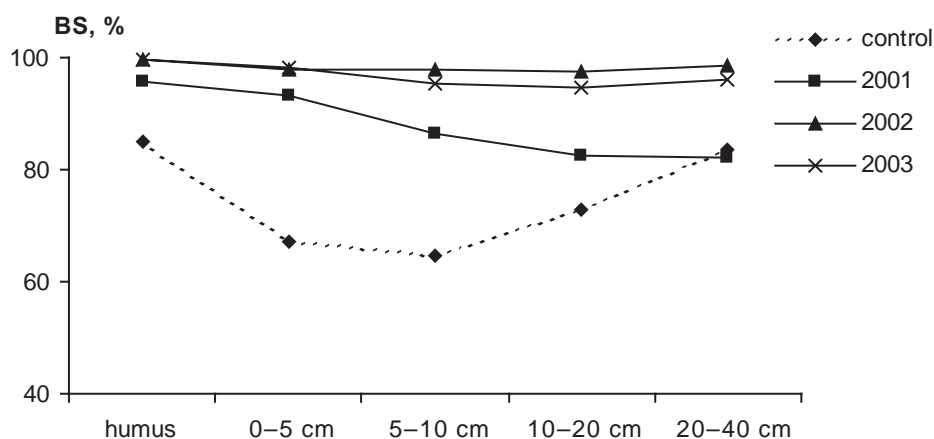
Figure 3. Mean base saturation (%) in the humus layer and at different depths in the mineral soil on the control plots, and on the treated plots at the cessation of sprinkling infiltration (2001), and one year (2002) and two years (2003) after cessation of the treatment.

Despite the input of K in the lake water, the treatment has not significantly affected the exchangeable K concentrations in the humus layer (Table 1). This is not a surprising result because K^+ has a relatively low affinity for cation exchange sites (it is a monovalent cation) and it is not able to compete with the elevated Ca^{2+} and Mg^{2+} concentrations (Bohn et al., 1985).

Table 1. Mean exchangeable Ca, Mg and K and extractable P concentrations in the humus layer on the control plots, and on the treated plots at the cessation of sprinkling infiltration (2001), and one year (2002) and two years (2003) after cessation of the treatment

	Control	Year		
		2001	2002	2003
Ca, mg/kg	2516	7313	8033	7605
Mg, mg/kg	308	1003	1030	2198
K, mg/kg	694	638	625	506
P, mg/kg	143	ND	53	55

ND = not determined.

Sprinkling infiltration has strongly reduced the concentrations of extractable P in the humus layer. Despite the fact that there was also an input of P in the lake water, it would appear that the relatively high pH values (above pH 7) in the humus layer have resulted in the immobilization of P (Hartikainen, 1979).

CONCLUSIONS

Sprinkling infiltration had a relatively strong effect on the acidity and fertility status of the humus and uppermost mineral soil layers, and it appears that there will not be a very rapid return to the soil conditions prevailing before the treatment. However, the treatment considerably increased the capacity of the soil to withstand acid loads, and increased the concentrations of the important macronutrients Ca and Mg. The only negative effect of sprinkling infiltration appears to be the decrease in the important plant nutrient P, the availability of which decreased owing to the increase in pH. Whether or not this represents a threat to site fertility requires further study.

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Anaerobic ammonia oxidation during sub-surface transport

Peter Fox and Shivani Shah

Abstract

Nitrogen removal has been sustained in numerous Soil Aquifer Treatment (SAT) systems where ammonia was the primary form of nitrogen in the reclaimed water. During SAT, groundwater is recharged using percolation basins and treatment may occur during percolation in the vadose zone and subsequent transport under saturated conditions in the aquifer. Since there is insufficient organic carbon to support heterotrophic denitrification, autotrophic nitrogen removal mechanisms capable of sustaining nitrogen removal were examined. In particular, anaerobic ammonia oxidation (ANAMMOX) was evaluated using column studies and batch studies. Nitrogen removal was sustained in column studies for over 600 days without the addition of supplemental carbon. When nitrate alone was fed to the columns, ammonia pre-adsorbed on the soil was bioavailable and sustained nitrogen removal. When a combination of nitrate and ammonia were fed, nitrate removal rates were similar to the soil column with pre-adsorbed ammonia. When nitrite and ammonia were fed to the soil columns, nitrogen removal rates increased by an order of magnitude as compared to a mixture of nitrate and ammonia. This suggests that the conversion of nitrate to nitrite is the rate-limiting step and soil systems have the capability of converting nitrate to nitrite.

Keywords

Anaerobic ammonia oxidation, nitrate, nitrite.

INTRODUCTION

Soil Aquifer Treatment has been used extensively for the recharge of groundwater using reclaimed water in arid areas. During Soil Aquifer Treatment (SAT), reclaimed water is percolated through the vadose zone and biological removal mechanisms remove contaminants of concern including organic carbon and nitrogen. This study focuses on how nitrogen removal may be sustained in SAT systems. Nitrogen removal efficiencies of greater than 70% have been sustained using secondary effluent with ammonia nitrogen concentrations in excess of 20 mg-N/L. There is insufficient organic carbon in secondary effluent to sustain the observed nitrogen removal efficiencies. Furthermore, the majority of organic carbon is removed near the soil surface where aerobic conditions exist and conditions are inappropriate for denitrification. During SAT, ammonia is primarily removed by adsorption when water is applied since there is insufficient oxygen to support nitrification. When water is not applied, air enters the soil column and nitrification of adsorbed ammonia will occur. During subsequent additions of reclaimed water, ammonia and nitrate are present in the soil column under anoxic conditions thus creating conditions appropriate for ANAMMOX.

Anaerobic ammonium oxidation may be the oxidation of ammonium with nitrate in the absence of oxygen and organic carbon (Mulder et al. 1995).

$$5NH_4^+ + 3NO_3^- \rightarrow 4N_2 + 9H_2O + 2H^+ \qquad (1)$$
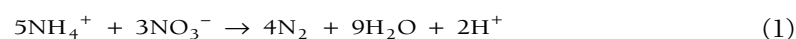
Further study with isotopically labeled nitrogen compounds indicated that instead of nitrate, nitrite is the most probable electron acceptor (van de Graaf et al. 1995) and equation 2 is the accepted stoichiometry for ANAMMOX.

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O \qquad\qquad (2)$$

Inhibition studies showed that this process is a biologically mediated process (van de Graaf et al. 1995). Using isolation techniques, ANAMMOX bacter were identified to be *Planctomycetes*. Several anaerobic ammonium oxidizing *Planctomycetes* have been identified including '*Candidatus Brocadia anammoxidans*' (Strous et al. 1999). *Candidatus* B. anammoxidans is a very slowly growing obligatory anaerobic bacterium with doubling times of 10–14 days (Kuenen & Jetten et al. 2001). During application of ANAMMOX in a fluidized bed reactor (FBR) using ammonium and nitrite as the electron donor and electron acceptor, respectively, the molar conversion ratio of ammonium (NH_4^+) to nitrite (NO_2^-) observed was 1:1.3 ± 0.06 with 0.22 ± 0.02 moles of nitrate production per mole of ammonium. Results showed that the main product was nitrogen gas with 10% of nitrate (NO_3^-) production of total nitrogen feed (van de Graaf et al. 1996). Gable (2002) studied ANAMMOX in soil systems. Experiments were carried out using soil columns, which were fed with ammonia, nitrate and bicarbonate. Results showed removal efficiencies of 60–87%. Genetic analysis of bacaters showed 97–98% similarity to *Candidatus Brocadia anammoxidans* (Gable et al. 2002).

METHODS

Four soil columns (Columns I, II, III, and IV) containing soil from a recharge basin at the Sweetwater Underground Storage and Recovery facility in Tucson, Arizona were used in this study. This facility has been removing nitrogen for over 15 years. The results presented here will focus on Columns III and IV. Columns III and IV were acrylic columns of 7.5 cm diameter and 93 cm in length. These columns were packed with 6.4 kg of moist soil. During the initial study period of the first 260 days, Columns IV was fed tap water with 20 mg-N/L ammonia, 30 mg-N/L nitrate and 1mM of bicarbonate while Column III was fed tap water with 30 mg-N/L of nitrate only and 1mM of bicarbonate. Column III was the control column to determine the effects of ammonia pre-adsorbed on the soil. From day 486 to day 505, Column III was allowed to aerate to convert adsorbed ammonia on the soil to nitrate and eliminate the effect of adsorbed, while Column IV was maintained under saturated anoxic conditions. After 20 days the columns were restarted and Column III was fed tap water with 30 mg-N/L nitrate and 1 mM of bicarbonate. Column IV was fed 30 mg-N/L ammonia, 30 mg-N/L nitrate and 1 mM bicarbonate to determine the impact of higher ammonia concentrations. From day 590, the electron acceptor was changed from nitrate to nitrite to observe effects on nitrogen removal rates. Both columns were fed with the same concentration of nitrite instead of nitrate. On day 670, the flowrate to Column IV was increased 3 fold and the flow was increased 3 fold again on day 720.

In addition to the soil columns, an acrylic column identical to Columns III and IV was used with a non-woven cloth media. The column was operated identical to Column IV in terms of feed composition changes after it was seeded for over 100 days with effluents from Column IV.

RESULTS

Nitrate removal occurred in both Columns III and IV with similar nitrate removal rates for the first 500 days. In both cases, the nitrate removal rate was slowly decreasing over time even and this effect was independent of the ammonia concentrations since the ammonia concentration was increased in Column IV (Figure 1) while ammonia was never added to Column III. Ammonia concentrations were below detection limits in the effluent of Column III while Column IV effluent ammonia concentrations were approximately 10 mg-N/L. After aeration of Column III, the rate of nitrogen removal continued to decrease in Column III. When the feed composition was changed to

nitrite, negligible removal of nitrite was observed in Column III and ANAMMOX activity apparently ceased. However, excellent removal of nitrite and improved removal of ammonia was observed in Column IV when the feed composition was switched from nitrate to nitrite (Figure 2). Therefore, the switch from nitrate to nitrite resulted in over a two fold increase in total nitrogen removal in the soil system. When the flowrate was tripled on day 670, there was a temporary increase in effluent nitrite concentration as the nitrogen loading was increased but the nitrite concentration decreased rapidly with over 95% nitrogen removal efficiency. The rapid response to the increase in loading suggests that microbial growth was occurring with the soil system and the increased microbial population was capable of removing the nitrogen at high efficiency. When the nitrogen loading rate was increased, the effluent nitrate concentration from Column IV also increased consistent with the studies reported by van der Graaf et al. (1996) where approximately 10% of the influent nitrogen was converted to nitrate. While the data is not shown, the loading was increased by another factor of 3 by increasing the flowrate on day 720 and the increased microbial activity was capable of maintaining 95% nitrogen removal efficiency. Therefore, a composition of nitrite and ammonia resulted in over an order of magnitude nitrogen mass removal rate as compared to a mixture of nitrate and ammonia.
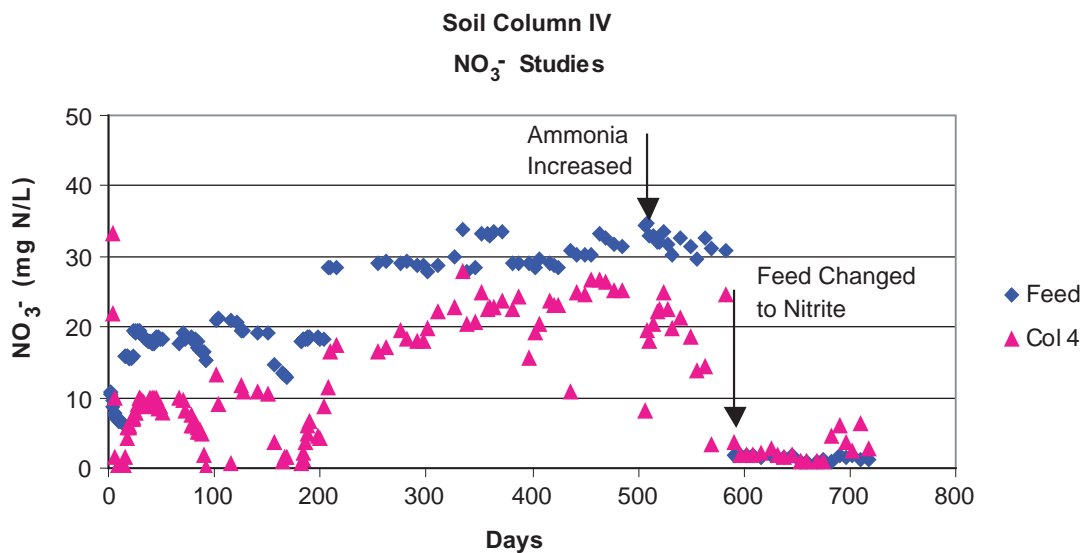
Figure 1. Nitrate Concentrations for Column IV
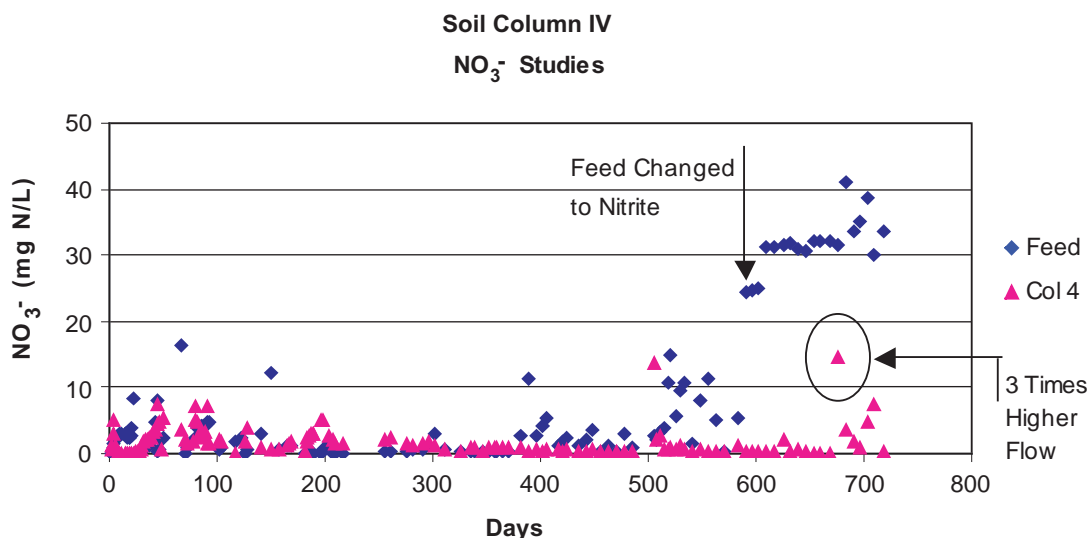
Figure 2. Nitrite Concentrations for Column IV

These results suggest that the conversion of nitrate to nitrite is a potential rate-limiting step in soil systems. There are two factors that lead to this observation. The first factor is that although nitrogen removal was sustained with a mixture of nitrate and ammonia, the nitrogen mass removal rate decreased over time (Table 1). The second factor is that nitrogen mass removal rates increased by over an order of magnitude when a mixture of nitrite and ammonia was used. It is possible that a component on the soil surface is involved in the conversion of nitrate to nitrite and this unknown component was becoming less available for the reaction as time proceeded under the isolated laboratory conditions. Furthermore, the column that was using non-woven fiber as an attachment media and was seeded with microorganisms from Column IV never removed nitrogen when a mixture of ammonia and nitrate was fed to the system. However, when a mixture of nitrite and ammonia was fed to the column containing the non-woven fiber, removal steadily improved over a 100 day period until over 95% removal of influent nitrogen was observed. The lack of nitrogen removal with nitrate and ammonia in the absence of the soil further enhances the argument that a soil component is involved in the transformation of nitrate to nitrite. One possible soil component that could be involved in nitrogen transformations is manganese. Based on thermodynamic calculations, it was observed that N_2 formation is possible at standard conditions through many reactions between manganese and nitrogen species (Luther et al. 1996). It has been reported that under oxic conditions, the oxidation of NH_4^+ occurs by MnO_2 in marine sediments and NO_3^- reduction by Mn^{2+} was observed under anoxic conditions (Luther et al. 1997). This possible interaction is presently under investigation to determine how it might sustain nitrogen transformations during sub-surface transport.

Table 1. The average nitrate removal rates for Column III and Column IV

	Nitrate Removal Rates (mg-N/d)	
Days	*Column III*	*Column IV*
0 to 216	7.5 (Gable et al. 2002)	7.8 (Gable et al. 2002)
260 to 350	6.9 ± 1.0	5.6 ± 1.4
350 to 450	6.4 ± 3.2	5.5 ± 1.7
450 to 590	6.3 ± 2.3	6.9 ± 2.5

CONCLUSIONS

Anaerobic ammonia oxidation is a potential sustainable removal mechanism for nitrogen compounds during SAT. In the laboratory, nitrogen removal was sustained for a period of greater than 600 days. However, the nitrogen mass removal rates appeared to decrease in the laboratory when a mixture of nitrate and ammonia was fed to soil columns. When a mixture of nitrite and ammonia was fed to the columns, nitrogen mass removal rates were increased by an order of magnitude. Under laboratory conditions, it is not clear if nitrogen removal with a mixture of ammonia and nitrate could be sustained indefinitely. However, under field conditions the primary forms of nitrogen observed are nitrate and ammonia and nitrogen removal has been sustained in the field. Cyclic aerobic/anoxic conditions occur in the field were not simulated in the laboratory and these conditions might sustain transformations that were not observed in the laboratory. The conversion of nitrate to nitrite appears to be a rate-limiting step for anaerobic ammonia oxidation in soil systems and the exact mechanism for this step has yet to be determined.

The nitrogen removal rates observed in soil columns and batch tests yield a first-order decay constant for nitrogen of approximately 0.08 d^{-1}. Tracer tests have estimated the travel time in the vadose zone to be approximately 14 days at the field site where the soils used in this study were obtained. Based on the first order decay constant

measured in the laboratory, a nitrogen removal efficiency of 64% is predicted which is similar to the 70% removal that is typically observed under field conditions.

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Geochemical evaluation of artificial recharge to intermediate flow systems in a carbonate aquifer from northeast Mexico

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Abstract

Artificial recharge of groundwater has been implemented to replenish extracted groundwater in a coal mining area of northeast Mexico, a semi-arid area where alternative water sources are scarce. In this region, the main aquifer is overlying a coal seam and surface coal mining requires dewatering the aquifer around the open-pit mining areas. Extracted groundwater (29×10^6 m^3/year) is mainly injected into two sites, the geochemical evaluation of the artificial recharge was made after the determination of a geochemical conceptual model, delineated from the interpretation of the water chemistry analysed in 70 samples. Results indicate saturation of calcite, dolomite, barite and chalcedony, no relation with salinity or groundwater flow was identified. Supersaturation of carbonates and goethite was detected in some samples, especially in recharge water and lagoons. Subsaturation of gypsum, strontianite, celestite, rhodochrosite and siderite is also a common characteristic. The chemical changes derived from the mixing between original aquifer water and recharge water were simulated indicating that some minerals can be precipitated from the mixture. The modeling exercise shows that calcite, goethite, barite and chalcedony are precipitated from the mixture, however calcite shows the largest amount of precipitation and represent the major cause of permeability reduction around the injection sites.

Keywords

Arid regions, groundwater recharge, hydrochemical modeling, hydrochemistry, injection wells.

INTRODUCTION

Successful groundwater management requires the understanding of the different properties within a groundwater flow system. Artificial recharge is becoming an important component in groundwater management, especially in semi-arid areas where alternative water sources are limited. Although almost 65% of Mexico has a semi-arid climate, only few artificial recharge projects have been implemented within a sustainable groundwater management scheme in the country.

Artificial recharge of groundwater has been implemented to replenish extracted groundwater in a coal mining area of northeast Mexico. In this region within the Coahuila state, included in the Fuentes-Rio Escondido Carboniferous Basin, coal mining is one of the main industrial activities. Surface coal mining requires overburden excavation using walking dragline procedures, the main aquifer (Tertiary age) in the area is intercepted by the open cut because it is located above the coal seam (Cretaceous age). Several wells have to be used to pump out groundwater (29×10^6 m^3/year) and dewater the aquifer around the open-pit mining areas, producing environmental impacts on water quantity and modification of local and intermediate groundwater flow systems.

As a component of an integral environmental friendly strategy, Minera Carbonífera Rio Escondido (MICARE) has designed, constructed and nowadays operates one of the most important and successful artificial recharge projects in Mexico, injecting the total amount of extracted groundwater from the dewatering process into two recharge sites

through several recharge wells. These actions have been applied together with additional mitigation factors to minimize environmental impacts related with groundwater quantity.

During infiltration water chemistry generally is not in chemical equilibrium with the minerals in the soil or aquifer, chemical reactions will develop changing both mineral and water compositions while the system tends towards a new equilibrium state. This is especially true for most of the artificial recharge projects; furthermore, injected water can be chemically incompatible with the native groundwater under the recharge conditions related with the project.

After extracting groundwater from the aquifer above the coal seam, the artificial recharge projects implemented by MICARE inject that water to the same aquifer but in a different place, not too far from the open-pit mining area. From the extraction area to the recharge sites, water is distributed along a series of open unlined channels, allowing infiltration of a minor amount of water and interaction with the atmosphere. The latter action produces some modifications in temperature and gas exchange with the atmosphere generating changes and inducing a new equilibrium in the water before it is injected. These modifications generate a different water, which might not be in equilibrium with the aquifer.

The main purpose of this paper is to show the use of geochemical modeling into assist on the evaluation of the consequences of the artificial recharge actions in the two recharge sites implemented by MICARE. The idea was to determine whether the mixing between natural groundwater and recharge water will cause an adverse impact on the aquifer or native groundwater quality.

The study area

The mining operations for coal extraction are located in the northeast part of Mexico, the region involved in this selected case study is located in the municipality of Nava, Coahuila, some 30 km south from Piedras Negras city placed just in the international border with the United States of America.

Climate is dry with an average precipitation of 490 mm/year, rainy season ($\approx$ 80% of total precipitation) is between April and October, potential annual evaporation (1,830 mm) exceeds precipitation and mean annual temperature is 17.8°C.

The study area consist of a gently topography that slopes (0.003) towards the NE, maximum altitude in the western is about 400 masl (meters above sea level) and 200 masl close to the Rio Bravo, some minor elevations (30 m above plain level at the most) are detected around Piedras Negras city and along principal river courses.

The zone belongs to the Hydrologic Region 24 (Rio Bravo-Conchos), Rios Escondido-San Antonio-Castaños sub-basin (Figure 1). These streams are ephemeral and only the Rio Bravo shows base flow during the year.
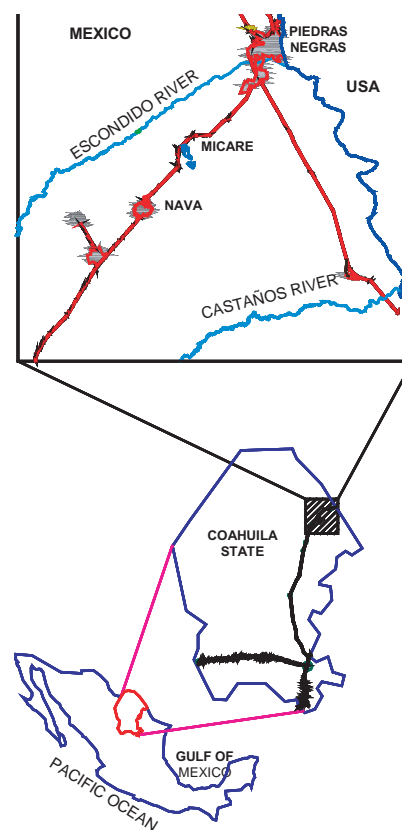
Figure 1. Location map showing the study area

Geology

Quaternary deposits represented by unconsolidated sediments and chalk are covering older geological units in most of the plain. Cretaceous formations (Upson, San Miguel and Olmos) outcrop in the southwest region, they are mainly composed of carbonate, sandstone and shale. Below the plain, a Tertiary locally unit called Sabinas-Reynosa Conglomerate (2–6 m calcareous conglomerate lower unit and a ≈ 25 m sand-clayey chalk unit locally known as 'caliche' and some gypsum) discordantly overlies the Olmos Formation. At least 9 coal layers have been identified within the Olmos Formation, 5 of them have economic value and are exploited using surface and subsurface coal mining methods.

Hydrogeology

The Sabinas-Reynosa Conglomerate comprises the main aquifer in the area; local and intermediate flow systems have been recognized throughout this hydrogeological unit. The intermediate groundwater flow system is mainly controlled by the gently slope towards the northeast and the calcareous conglomerate, in some parts of this unit a karst-conduit network has been identified from direct observation and core sampling. The catchment area of the intermediate flow system has a recharge zone in the west part of the region and a general discharge zone adjacent to the Rio Bravo, close to the international border with the United States of America (IMTA, 2004).

Considering that the aquifer is above the Olmos Formation, mining operations require its complete depletion around the open-pit (40–70 m depth) for coal production. A number of groundwater extraction wells are used to capture groundwater flow towards the open pit, modifying the natural groundwater flow in the aquifer. Total groundwater extraction for dewatering purposes accounts ≈ 0.91 m^3/s; 0.49 m^3/s are distributed in a channel network in the direction of the Santo Domingo artificial recharge project, and 0.42 m^3/s towards a second facility called the Aqueduct artificial recharge project. As soon as extracted groundwater is on surface, a gravity driven system of open unlined channels diverts it to recharge sites. Each of these two recharge sites consists of several injection wells, gravity induced injections rates are typically $0.04-0.06$ m^3/s in every well within each of the artificial recharge sites.

METHODS

The general hydrogeochemical conceptual model was determined from 70 groundwater samples taken in different parts of the aquifer. Samples were obtained from active wells, springs, lagoons in abandoned open pits and water used for artificial recharge just before its injection.

Temperature, specific electrical conductivity (SEC), Eh (Pt-electrode), pH and dissolved oxygen (DO) were measured in the field using an in-line flow cell. Alkalinity was also measured in the field by Gran tritation. Samples for major anions were taken in 500 ml bottles, for major cations, silica and trace elements were filtered with a 0.45 μm membrane filter and acidified with ultra-pure nitric acid (1 ml acid per 100 ml sample) and stored at 4°C until analyzed.

Analytical methods used for major anions are compatible with those reported in APHA (1995), major cations measurements were done by atomic absorption, calculated ionic balance was within 5% in 100% of the samples. A complete suite of trace elements was analyzed by inductively coupled plasma mass spectrometry (ICP-MS), although only a few (Ba, Fe, Mn, Sr) were used in the present investigation; accuracy of ICP-MS analyses was controlled using duplicates, blanks and appropriate laboratory standards, it was also checked during each batch using reference standards including SLRS-4 and NIST 1640.

RESULTS AND DISCUSSION

The geochemical evaluation of the artificial recharge was done for each of the recharge sites. Original aquifer water chemistry in both recharge facilities is missing, therefore it was determined from samples taken from wells around the recharge sites, but far enough to guarantee they are not altered by the recharge operations. Recharge water chemistry was evaluated from chemical analyses of samples taken just before each of the injection sites. The chemical changes derived from the mixing between original aquifer water and recharge water were simulated accordingly with the geochemical conceptual model using PHREEQC (Parkhurst, 1995).

Results of chemical analyses show large variations in composition an also indicate the relative high salinity of the samples, with SEC in the range of 2320–397 µmhos/cm. Sulphate concentrations are quite high for some samples, reaching up to 875 mg/l but most of the samples (75%) have values below 387 mg/l; Cl maximum values are about 384 mg/l, but 25% of the samples have concentrations above 196 mg/l. The distribution of salinity in the horizontal plane shows a clear regional trend and agrees with general groundwater flow direction, lower values (around 300 mg/l TDS) are found at the western part of the area increasing to the east up to 1300 mg/l around the mining operations.

DO in groundwater is generally present (average 4.2 mg/l), mean redox potential values are around 0.3 V, both conditions suggesting aerobic and oxidizing conditions in the aquifer. Most (42%) of the samples are Mixed-Ca waters, however HCO_3–Ca (34%) and SO_4–Ca (17%) compositions are also present; the rest of the samples (7%) do not have a dominant ion.

Geochemical calculations made with PHREEQC (Parkhurst, 1995), indicates saturation of calcite, dolomite, barite and chalcedony for most of the samples, no relation with salinity or groundwater flow was identified. Supersaturation of carbonates and goethite was detected in some samples, specially in recharge water and lagoons in abandoned open pits. Interaction with the atmosphere produces degassing of CO_2 and an increase in dissolved oxygen content in these waters, generating the observed oversaturation in the water. Subsaturation of gypsum, strontianite, celestite, rhodochrosite and siderite is also a common characteristic of most of the samples. In the model, it was specified that calcite, chalcedony, barite and goethite can be precipitated from the mixture when the saturation index is 0.0.

The mixing calculations were executed using fractions of 0.8 and 0.2 of original aquifer water and recharge water respectively, the mixed solution was saved and mixed with another 20% recharge water sequentially (mix numbers in Figure 2). After seven of these mixing procedures the solution has about 78% of recharge water and 22% of original aquifer water. The modeling exercise shows that in both recharge sites, calcite, goethite, barite are precipitated from the solution, chalcedony is precipitated only in the Aqueduct recharge site; however calcite shows the largest amount of precipitation, around 0.6 and 0.9 millimoles per liter for Aqueduct and Santo Domingo sites respectively (Figure 2). The amount of precipitation increases with the mixing, calcite precipitation does not necessarily result from the mixing but from the fact that recharge water in both sites is supersaturated (from 0.7 to 1.0 in recharge waters). Barite is also supersaturated in recharge waters, but barium concentrations are low and consequently its precipitation rate is only a minor component.
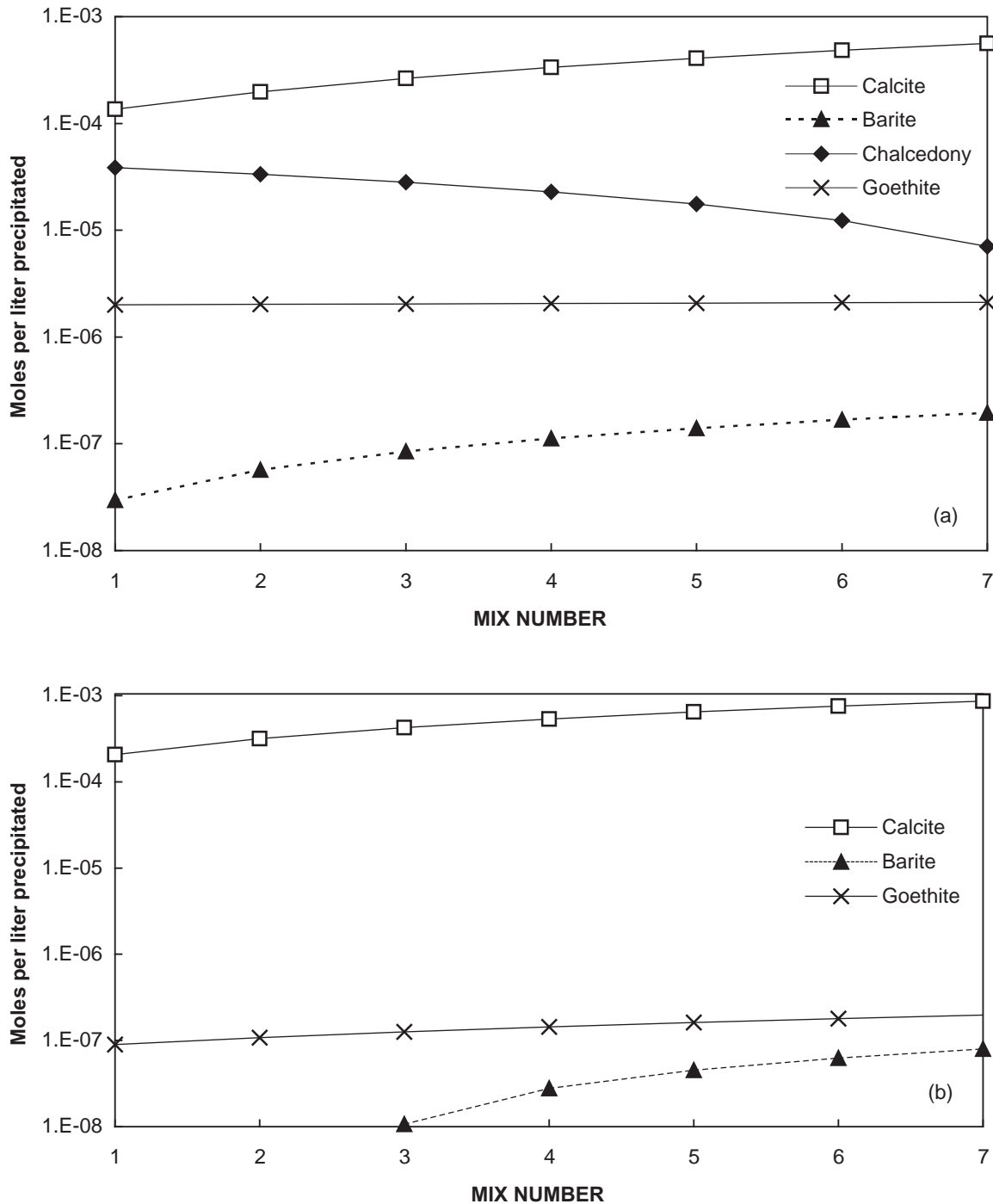
Figure 2. Results of mixing calculations for artificial recharge in: (a) Aqueduct and, (b) Santo Domingo sites

CONCLUSIONS

Considering inorganic chemistry, the geochemical modeling results suggest that calcite precipitation constitutes the major consequence of actual artificial recharge operations. This precipitation as well as clogging derived from suspended matter incorporated along the open channels will be a major a cause of the reduction of the injectivity conditions in the recharge sites.

The understanding of identified chemical reactions controlling the inorganic quality of the intermediate systems in the area during the artificial recharge, allows determining that a reduction in effective porosity around the recharge sites, will be the main side-effect to the aquifer. However, the microbiological quality of injected water was not considered in this study, taking into consideration the karst-conduit network identified in the aquifer, micro-organism could be able to travel long distances down-gradient threatening groundwater resources.

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Artificial recharge in Finland through basin and sprinkling infiltration: soil processes, retention time and water quality

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Abstract

The aim of the 'TEMU' research project (1998–2003) was to optimise the infiltration techniques of artificial recharge (AR) of groundwater in relation to water quantity, water quality and environmental impacts. Special attention was paid to determining the causal relationships between soil processes, aquifer hydraulic characteristics and AR groundwater retention times, and water quality in different phases of percolation and flow. The reduction in total organic carbon (TOC) concentration was mainly restricted to the groundwater zone. There were no marked differences between the infiltration methods – basin or sprinkling – as regards the removal of organic fractions or the reduction in TOC concentration. However, retention time and the flow characteristics significantly influenced the TOC concentration of the AR groundwater. The target TOC concentration (less than 2 mg/l) was attained with varying AR groundwater retention times (7–80 days) and with varying flow path lengths (160–1,300 m). The dimensioning of an AR groundwater plant is controlled by the quality of the infiltrated surface water and the flow characteristics that are dependant on the aquifer particle size distribution, hydraulic conductivity and retention time. Therefore, individual AR groundwater plants may require infiltration areas of different size to produce water of the same quantity and quality.

Keywords

Artificial recharge; basin infiltration; organic carbon; retention time; sprinkling infiltration; TOC.

INTRODUCTION

Today 60% of the water distributed by Finnish waterworks is natural groundwater and artificial recharge (AR) groundwater produced by basin and sprinkling infiltration. The proportion of AR groundwater is 15% and it is estimated that, by the year 2010, it will account for over 20% of the total water use. Basin infiltration has been used since the 1970s in Finland. Since the end of the 1990s, however, several Finnish cities have used a new method, sprinkling infiltration. In this method the surface water is sprinkled directly onto the forest soil from a network of pipes without removing the vegetation and soil as in basin recharge (Helmisaari et al. 1998).

Most of the Finnish AR groundwater plants are situated on forested eskers sedimented during the last ice age. It has earlier been widely assumed that the concentration of TOC in the infiltrated surface water decreases already in the unsaturated surface and percolation zones. This assumption is based on results from e.g. column experiments and basin infiltration studies, which showed that a relatively high proportion (at best 50%) of the organic carbon was retained in the unsaturated zone and in the filter sand located on the bottom of the basins (Frycklund 1995, 1998; Frycklund and Jacks 1997; Jacks 2001). It was considered that half of the retained organic carbon was adsorbed on soil particles, and half decomposed by microbial activity (Jacks 2001). However, in a recent report on column experiments, only $10 \pm 5\%$ of the TOC was removed by the column soil material (Långmark et al. 2004). Apart from a number of studies on bank infiltration (e.g. Kivimäki et al. 1998; Miettinen et al. 1995), hardly any research has been carried out on the changes in water quality at large-scale AR plants in Finland. Therefore, there was a clear need for full-scale studies on AR groundwater.

In 1998, we initiated an extensive research project at five Finnish AR groundwater plants that use sprinkling and/or basin infiltration. The aim of the project was to optimise the infiltration techniques and recharge processes in relation to the quantity and quality of the AR groundwater and to evaluate the environmental effects associated with these techniques.

We concentrated on determining changes in the quality of the infiltrated water in different soil hydraulic zones as the infiltrating surface water is gradually converted into AR groundwater. We determined the hydraulic and hydrogeological characteristics of the aquifer and AR groundwater retention times. This information was used, together with the water quality data, for modelling the AR groundwater flow and for planning and dimensioning the AR groundwater plants.

METHODS

Study areas, lysimeter types and water sampling

We studied the hydrogeochemical variables and hydraulic characteristics of the surface soil zone, the percolation zone and the groundwater zone, at five AR groundwater plants (Table 1). The TOC concentrations in the water in the surface soil and percolation zones under the sprinkling areas and in the percolation zones under the basins, and in the groundwater zones at various distances from the infiltration areas, were determined monthly.

Table 1. Studied artificial recharge groundwater plants in Finland

	Ahvenisto	*Vuonteenharju*	*Pursiala*	*Rusutjärvi*	*Jäniksenlinna*
Studied method of infiltration	sprinkling	sprinkling basin	sprinkling	sprinkling	basin
Operating	basin 1976– sprinkl.1996–	2000–	basin 1974– sprinkling 1999–2001	1997–	1979–
Maximum daily infiltration amount, m^3day^{-1}	20,000	15,000	15,000	20,000	12,000
Mean TOC in raw water, mgl^{-1}	10.3	6.7	7.4	5.8	5.8
Aquifer hydraulic conductivity, ms^{-1}	0.00092	0.00033	0.00029	0.00081	0.0012
Flow distance to uptake wells, m	1,000–1,500	200–480	160	640–780	480–700
Retention time to uptake wells, days	79–103	21–100	16	35–65	36–51

Water samples were taken from the unsaturated zone using three types of lysimeter system. Both zero-tension plate lysimeters and tension cup lysimeters were located immediately below the humus layer, tension cup lysimeters at depths of 20 cm, 40 cm and 1 m measured from the top of the mineral soil (six replicates at each depth on each plot), and specially developed zero-tension tube lysimeters at depths of 1, 2, 10 and 15 meters from the ground surface. Observation wells or tubes were used for water sampling in the groundwater zone. The observation wells and tubes were located at specific distances between the infiltration areas and the water production wells. Between 6 and 25 observation wells or tubes were sampled at monthly intervals at each of the AR groundwater plants during 1998–2001.

Water analyses

Total organic carbon (TOC) in water samples consists of particulate organic carbon (POC) and dissolved organic carbon (DOC). The operational definition of DOC is the organic carbon which passes through a 0.45 µm membrane filter, and it therefore also includes carbon-containing particles with a diameter less than 0.45 µm. In our study, the organic carbon is solely of natural origin. TOC and DOC were analysed on Shimadzu TOC-5000 analysers (standards SFS-EN 1484, SFS-ISO 8245). The molecular size distribution and the chemical characteristics of the organic carbon (Qualls and Haines 1991, Smolander et al. 2001), and other water quality variables (e.g. $KMnO_4$, electrical conductivity, alkalinity, total hardness, CO_2, O_2, Ca, Mg, K, Na, P, Si, SO_4 and Cl) were also determined.

Groundwater flow models

Information about the flow conditions of the groundwater was obtained by means of observation tube flow measurements. These results were used to calibrate the input parameters of the groundwater flow models which, in turn, were used to calculate the AR groundwater retention times and the dilution percentage of raw water by natural groundwater. The input parameters of the flow models, which included aquifer dimensions, hydraulic conductivities, storage coefficients, effective porosities and dispersion factors, were determined on the basis of a number of field studies. The flow model for each study area was calibrated on the basis of several pumping tests and infiltration studies. Retention and flow times of the AR groundwater from the infiltration area to the water uptake wells were calculated using these flow models (e.g. Bear 1979). The AR groundwater retention times were determined on the basis of simulated tracer tests. The retention time is the difference between the simulated tracer injection time and the simulated tracer maximum concentration occurrence time. In addition, a real tracer field test was carried out in the Vuonteenharju study area.

RESULTS AND DISCUSSION

One of the important objectives of the project was to determine changes in the concentration of organic carbon, as well as in its molecular weight and compound group distribution. The studies clearly demonstrated that the reduction in TOC concentrations was completely, or to a very large extent, restricted to the groundwater zone.

During sprinkling infiltration the surface zone, which included the humus layer and underlying organic matter rich mineral soil layer, slightly increased or did not change the concentration of TOC in the percolating water (Fig. 1).

This result is in full agreement with the findings concerning the decrease in the organic matter content of the humus layer caused by the leaching of organic matter down into the underlying mineral soil layers (Derome et al. this volume). However, there were slight differences between the study areas: in Ahvenisto there was a 40% reduction of TOC, which took place in the percolation zone or in the groundwater zone below the infiltration area (Lindroos et al. 2002), while in Vuonteenharju there was no change down to a depth of 10 meters in the mineral soil.

The infiltration water was transported through the 5−30 m thick unsaturated zone within a few hours, which may explain why there was no significant reduction in the
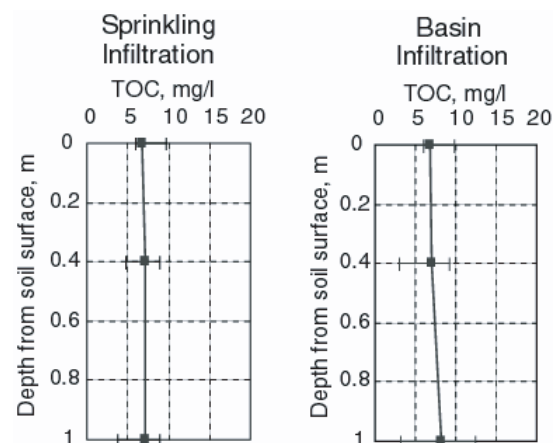
Figure 1. The concentration of total organic carbon (TOC) in infiltrated surface water (0 m) and percolation water (0.4 and 1.0 m) at Vuonteen-harju AR groundwater plant incentral Finland collected using tension cup lysimeters

TOC concentration. The transport of organic carbon in natural soil profiles is often determined by the flow regime and macropore transport (Kalbitz et al. 2000). Thus, the higher hydraulic load in sprinkling areas and basins compared with the natural precipitation load may lead to a reduced contact time between the solid and liquid phases of the soil. This may account for the lack of significant changes in the quantity and molecular weight distribution of TOC in the infiltrated surface water as it passes down through the unsaturated zone.

The values of most of the water quality variables (electrical conductivity, alkalinity, total hardness, Ca, Mg, K, Na, P, Si, SO_4 and Cl) did not change during passage through the percolation zone in either basin or sprinkling infiltration. The upper layers of the soil soon reached a chemical balance with the infiltrated surface water (Lindroos et al. 2001). The dissolved O_2 concentration decreased in both basin and sprinkling infiltration. However, the CO_2 concentration increased in the percolation zone only in the case of sprinkling infiltration. One reason for this could be the presence of large amounts of decomposable organic matter in the humus layer and uppermost mineral soil layers in the sprinkling infiltration areas.

The reduction in the TOC concentration took place in the groundwater zone. The reduction was related to the molecular size distribution of the organic carbon. The largest molecules of organic carbon were removed from the AR groundwater more efficiently than the smaller molecules. The remaining small molecules consisted of chemically and biologically stable groups of compound.

The reduction in the TOC concentration was related to those properties of the aquifer material (particle size distribution and hydraulic conductivity) that also affect the retention time of the AR groundwater. Adsorption on soil particles seemed to be a more important mechanism than microbial decomposition in reducing the TOC concentration. The TOC concentration decreased by 60% of the concentration in infiltrated surface water when the AR groundwater retention time was 10–35 days, and by 70% when it was 20–60 days (Fig. 2). The target TOC concentration (less than 2 mg/l) was attained with varying AR groundwater retention times (7–80 days) and with varying AR groundwater flow path lengths (160–1,300 m). The hydraulic conductivity in the main flow part of the groundwater zone at the AR groundwater plants varied between 0,00029–0,0012 m/s (Table 1). The AR groundwater plants with the shortest retention times (Pursiala and Vuonteenharju) required to decrease the TOC concentration also had the lowest hydraulic conductivities.
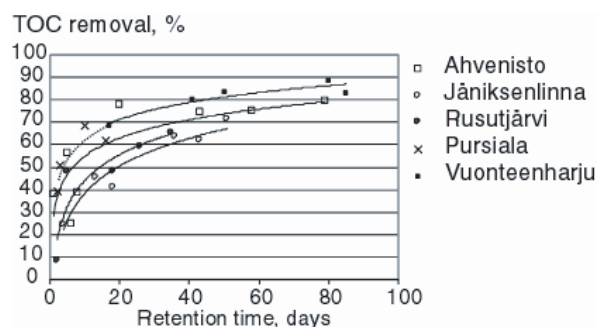
Fig. 2. The change in the TOC concentration in AR groundwater in relation to the retention time at five AR groundwater plants in Finland. The results have been adjusted to take into account the diluting effect of natural groundwater.

We also compared the properties of AR groundwater with those of natural groundwater in the same study areas. Many of the variables (e.g. dissolved O_2 and CO_2 concentrations, electrical conductivity and total hardness) at the individual AR plants varied considerably compared to natural groundwater. However, the TOC concentrations were either similar or only slightly higher in AR groundwater than in natural groundwater (Table 2).

The infiltration method – basin or sprinkling – did not markedly influence the reduction in TOC concentration in the groundwater zone. However, there were differences in several other water quality variables: the dissolved O_2 concentration was higher in the AR groundwater produced by sprinkling infiltration. The electrical conductivity, and the Ca and Mg concentrations were also higher in AR groundwater produced by sprinkling.

Table 2. Mean water quality in infiltrated surface water, natural groundwater and AR groundwater at the AR groundwater plants. Natural groundwater variable values were only available for three of the AR plants before infiltration started (Explanations, see Table 1.)

		Raw water	Natural groundwater	AR groundwater
TOC, mg/l	Ahvenisto	10.3	–	2.0
	Vuonteenharju	6.7	0.8	1.1
	Rusutjärvi	5.7	1.0	1.9
	Jäniksenlinna	5.8	0.5	2.0
	Pursiala	7.4	–	2.1
$KMnO_4$, mg/l	Ahvenisto	43.5	–	3.5
	Vuonteenharju	21.0	< 2	< 2
	Rusutjärvi	21.7	1.4	3.4
	Jäniksenlinna	20.9	1.2	4.4
	Pursiala	24.9	–	2.6
Dissolved, O_2, mg/l	Ahvenisto	5.6	–	1.1
	Vuonteenharju	10.7	8.8	9.3
	Rusutjärvi	9.2	5.5	8.3
	Jäniksenlinna	9.4	7.5	5.5
	Pursiala	9.0	–	1.3
Dissolved CO_2, mg/l	Ahvenisto	3.5	–	12.8
	Vuonteenharju	3.0	11.0	17.0
	Rusutjärvi	2.9	21.6	8.6
	Jäniksenlinna	3.4	34.2	9.4
	Pursiala	4.0	–	22.0

CONCLUSIONS

Our results showed that the planning and dimensioning of an artificially recharged groundwater supply plant should be based on the quality of the infiltrated surface water, and on the groundwater flow characteristics such as aquifer particle size distribution and hydraulic conductivity and the AR groundwater retention time. Therefore, individual AR groundwater plants may require infiltration areas of different size to produce water of the same quantity and quality. Thus, the planning and dimensioning of an AR groundwater plant presupposes comprehensive field studies.

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Subsurface residence times of hyporheic groundwater and mixing with alluvial ground water

Eduard Hoehn

Abstract

This paper summarizes the results of recent investigations in alluvial systems of river-groundwater interaction. Emphasized is the contamination potential of downwelling rivers for drinking-water wells that are located in the near field of the channels. Here at least three ground waters of different residence times mix at variable rates. Conflicts arise between flood control and river restoration efforts, on the one hand, and groundwater use on the other hand. Although the principles of bank filtration flow are well known, a resolution of these conflicts requires more detailed knowledge, given the high heterogeneity of these flow systems.

Keywords

Bank filtration, hyporheos, River-groundwater interaction.

INTRODUCTION

Alpine and perialpine catchments are characterized by a succession of flood plains and knick points (Brunke & Gonser, 1997; Huggenberger et al., 1998). Flood plains are filled with well permeable and coarse-grained glaciofluvial aquifer material, and they carry high-yield ground water reserves. Stream reaches with downwelling water (losing rivers) are found from the upstream end to the center of flood plains, and upwelling sections (gaining rivers) are found near the downstream end of flood plains, when the ground water is released back to the surface into the receiving stream. Ground water that is subject to an intense exchange with the river is called *hyporheic* ground water (Ward and Stanford, 1998). At greater distances from the river, the ground water, which has longer residence times in the subsurface, is called here *alluvial* ground water.

For decades, hydrogeologists and water-supply engineers profited from the interactions between rivers and the ground water in flood plains. Water wells near rivers yield enough ground water, even during droughts. Many countries use systems of induced bank filtration as the first treatment step for drinking water (Sontheimer, 1975; Tufenkji et al., 2002; Ray, 2002). At recharge conditions of induced bank filtration, water is lost from the river only, when a high hydraulic gradient is applied to the well. In the alpine and perialpine flood plains of Switzerland, many rivers lose water naturally. Where the quality of the river water is high, the groundwater resources are used with only minor or without any treatment. The pumped ground water of wells near rivers is characterized by short mixed residence times in the subsurface. Since the rivers are used as receiving courses for wastewater, the groundwater quality is surveyed more carefully.

The Swiss Law on Water Protection relies much on land-use planning and construction measures. Groundwater protection zones (GPZ) are required around public water supply wells. The most important GPZ is the one which is defined by an area within which the ground water requires 10 days or less to reach the well (EC countries: 50 d). In laboratory column experiments, Merkli (1975) found that at the relevant flow of 1 – 10 m/d, about 90 per cent of E. coli were eliminated within 10 days. The GPZ of the 10-days' line is characterized by a construction ban without exceptions, severe agricultural restrictions, and by a ban of all river restoration operations. The concept of the resi-

dence time of 10 days does not allow for a dilution of mobile, persistent toxic chemical contaminants. To this end, restrictions in the zone of contribution (ZOC) around a well are introduced in the planning concept. Within the ZOC of a well, many contaminating industrial activities are banned, which includes high agricultural fertilizer and pesticide applications. Many wells near downwelling rivers do not meet with the requirements of the Swiss Law on Water Protection. Their ground water has mixed residence times of < 10 days, and the percentage of hyporheic ground water is too high for the dilution of contaminants.

GROUNDWATER FLOW FIELD NEAR DOWNWELLING RIVERS

Infiltration of river water into layered alluvial aquifer sediments of glaciofluvial origin is a three-dimensional (3-D) process. Investigations in systems of downwelling rivers with well clusters of screened sections at different depths have shown a stratification of at least three different types of ground water (Fig. 1): Hyporheic ground water stratifies on the top of the saturated aquifer (Hoehn et al., 1983). The vertical component of hydraulic conductivity (H.C.) is orders of magnitude lower than the high horizontal component. The aquifer material has a very high variability in grain size, and less permeable layers can be observed at a scale of 0.1–10 m, along distances of 1 – 100 m (e.g., Huggenberger et al., 1998). Because of the difference in H.C., downwelling of cold river water

(0 – 5 °C.) in winter does not result in a density-driven groundwater flow. The warm summer water (20–25 °C.) results in an increase of H.C. of a factor of about two. Contrary to the high variability of H.C. in aquifers of glaciofluvial outwash, the permeability of river beds and banks does not seem to be as variable: specific infiltration rates of 0.05 - 5 $m^3/m^2 \cdot d$ are reported by Hoehn (2002). Given permeable beds and banks, rivers are downwelling or upwelling, depending on the hydraulic gradient between the water level of the river and the water table.

At downwelling flow conditions, the river is hydraulically connected through a saturated zone up to a difference in water-level elevation of about 2 m, and seepage through an unsaturated zone occurs at greater differences.

Below the freshly infiltrated hyporheic ground water, alluvial ground water predominates, which infiltrated from the river more upstream in the flood plain. The chemical composition of this groundwater of greater depths may be similar to that of the river, but it is older and the temperature varies less in time. The groundwater at the bottom of the aquifer may not be of bank filtration origin, but result from seepage of precipitations and inflow from the slopes to the flood plain valleys. As shown in Fig. 1, the flow directions of the different portions of the groundwater body must not necessarily be the same. In phreatic groundwater systems, the flow direction of the top layer is also dependent on the position of the water table space and the

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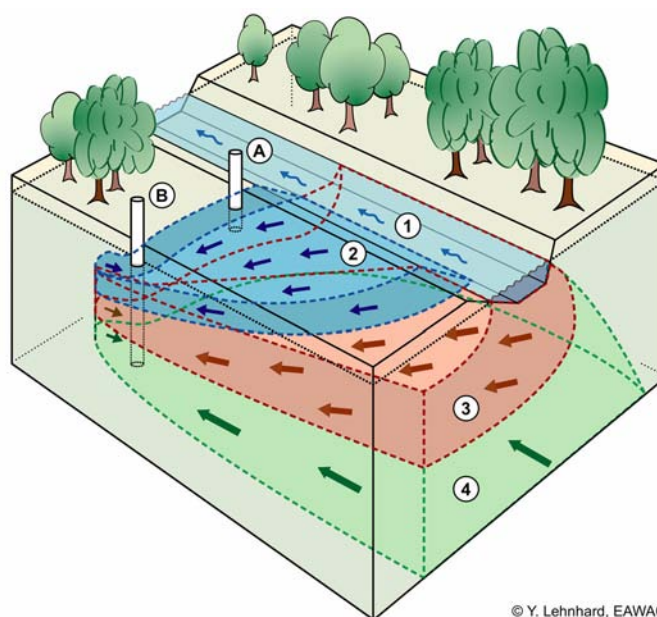
Figure 1. Groundwater stratification near downwelling rivers in alpine and perialpine flood plains. 1, River water; 2, Hyporheic groundwater (τ, < 30 d; τ, subsurface residence time); 3, riverborne alluvial groundwater (τ, 1 – 20 months); 4, Alluvial ground water not of river origin (τ, years). A, Well tapping exclusively hyporheic ground water; B, Well tapping groundwater mixture

ZOC can move time time (Drost et al., 1977; see also Vassolo et al., 1998). Thus a 2-D representation of the ZOC on a map may not be precise enough.

In river reaches with clogged beds and banks, the interaction between river and ground water are restricted or even completely stopped (e.g., Gölz et al., 1991; Schälchli, 1995). The clogging process is characteristic for downwelling rivers more than for upwelling ones because upwelling results mostly in a turbulence of the river-bottom sediments, and the fine material is transported with the flowing water. Clogging has been reported to be of importance i) in pristine systems with high concentrations of suspended sediments and/or organic debris, and ii) in engineered systems of dammed rivers, at conditions of minimal flow due to water diversion, systems of hydropeaking, or at a heavy contamination. River beds and banks are more or less clogged, depending on the substrate, the concentrations in suspended solids, the occurrence and the magnitude of floods, and the structure of the course of the river. The extent of clogging is thus time-dependent, and tracer observations have shown that it is variable over short distances. The dammed river Limmat, which was formerly downwelling, got clogged within two years, which resulted in a decrease of the water table and the anoxic conditions could establish (Harder, 1938).

MEASUREMENT METHODS

The residence time in the subsurface of the riverborne ground water and the age of older ground water can be assessed with isotopic and environmental tracers (e.g., Hofer et al., 1997). Mixing of the various ground waters is assessed with an end-member analysis of the chemical water composition. Among the end members are the river water and the old ground water not of infiltration origin. The Alps consist to a great extent of chemical sediments containing calcite (limestones, marls, dolomites, or evaporitic rocks). Thus alkalinity, chloride or sulfate, yield good tracers. In rivers that are outlets of lakes in which calcite precipitated, alkalinity is much lower than in the ground water. Where the composition of the river water is almost the same as that of the ground water, an analysis of water temperature, or of trace or contaminant compounds is necessary, for an investigation of the mixing process. Data loggers with high-capacity memory (water pressure, temperature, and specific electric conductivity [EC]) make available high resolution time series. This time resolution allows distinguishing seasonal signal amplitudes from events of a smaller time scale. The analysis of the pressure data revealed that aquifers, which were interpreted from borehole logging to be phreatic, turned out to transmit pressure signals from rivers to remote wells within times characteristic for confined ones. It seems that fine-grained layers of a scale of decimeters, which cannot well be distinguished in a bore log, exert a confining pressure on the ground water.

The high-resolution time series of water temperature and EC allow to assess subsurface residence times of ground water near downwelling rivers, by means of cross-correlation analysis and high-pass filtering of two respective data sets (Sheets et al., 2002). The course of groundwater temperature in downwelling systems has long been interpreted in terms of their seasonal sinusoidal amplitudes and the delay of the extreme values. The two relevant temperature end members are the river water and old homothermous ground water (temperature equivalent to mean annual air temperature; Jäckli & Ryf, 1975). Very close to the riverbank, the flow velocity of the hyporheic ground water can be so high that its temperature even fluctuates with the daily amplitude. Temperature is, however, a non-conservative tracer. Its retardation factor with respect to the flowing water depends on the aquifer material and has a value of 2 – 4 (Sauty, 1978). With respect to the flowing water, temperature is thus delayed and damped in its amplitudes. As EC reflects the concentrations of conservative anionic solutes such as alkalinity and/or chloride, it acts as a good groundwater tracer. However, no seasonal or daily sinusoidal variations are commonly found. Most signals are of a pulsed type (e.g., floods, see Fig. 2; snow melt, road deicing with NaCl, or outlets from sewage treatment or waste incineration plants).

The effect of flood events on the ground water near downwelling rivers can be distinguished from the continuous water loss. The residence times resulting from the analysis of seasonal temperature variations and those resulting

from pulsed signal variations do not match. While seasonal temperature amplitudes integrate the transport of both young and older ground water of infiltration origin, the pulsed EC or temperature signals reflect the transport of hyporheic ground water only. When the residence times are interpreted as mixed ages, the pulsed signals from the hyporheic ground water yield smaller values than the integrated sinusoidal signals.
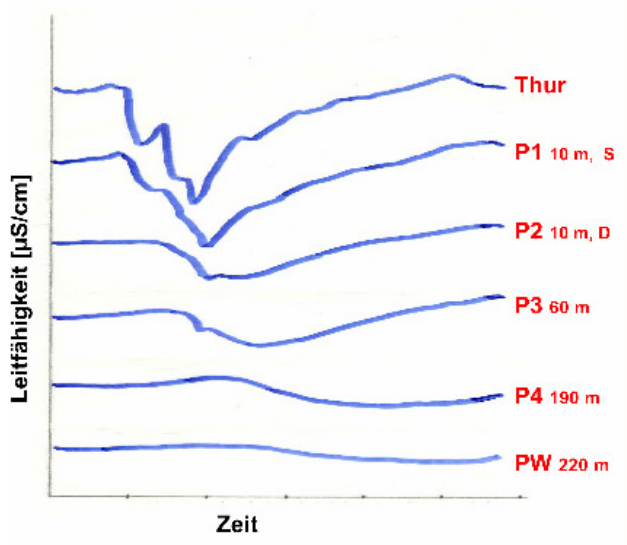
Fig. 2 Transport of dilution pulse of electric conductivity ('Leitfähigkeit' [µS/cm]) during flood event with time ('Zeit', unit: days) from River Thur to 4 wells, at increasing distances from river (flow distance given from river to well). Note delay of minimum, and damping of amplitude along flow path. Well cluster P1, P2: S, shallow well; D, deep well. PW, drinking-water pumping station (courtesy: Dr. Andreas Scholtis, Amt für Umwelt, Kanton Thurgau, Switzerland)

CONCLUSIONS

What do we still study river-groundwater interactions in flood plains in great detail, after all these years of research? Water-supply engineers are more and more faced with trace contaminants in rivers that are toxic even at minute concentrations and that remain mobile and persistent also in the subsurface. With respect to such contaminants, river bank filtration as the first treatment step for drinking water is under debate. In densely populated areas, landscapes with rivers are more and more taken as resorts for urban areas. Furthermore, flowing aquatic systems are taken as diverse ecological habitat for the biota. Thus rivers should be restored and wetlands should be preserved. All this conflicts more and more with the requirements of water works for a safe water supply.

ACKNOWLEDGMENT

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Hydrochemical evaluation of surface and groundwater quality at the coastal Basin (Syria)

A. Kassem

Abstract

Major dissolved constituents (Ca^{++}, Mg^{++}, Na^+, K^+, $Fe^{++,+}$, Cu^{++}, HCO_3^-, SO_4^{--}, Cl^-, PO_4^{3-}, NO_3^-, NH_4^+, F^-) have been determined in 66 water samples (54 of ground water, and 12 of surface water) from the major region of the medial coast basin. Most of these samples have been taken from ground water of the carbonic consolidated aquifer (Limestone's and Dolomites of the Jurassic and Cenomanian/Turonian). The specific characteristics of these hydrochemical groups seem to be related to the lithology of aquifers (dolomites aquifer = Mg/Ca>1).

The elevated Na^+, Cl^-, and SO_4^{--} content of the water is attributed to interaction of marls in the aquifer. The relations between lithology of the aquifer and hydrochemical composition of ground water should be confirmed by systematic sampling. The ground water from the chalks in the area has a higher content of Mg, SO_4^{--}, Cl^- and NO_3^- The elevated contents of Na^+ and Cl^- seem to be related to a higher portion of interbedded marls in the Cenomanian/Turonian aquifer in the southwestern part of the coast mountains, occurring in outcrops in the area near Myssiaf and Dreikich.

The seasonal variations of the HCO_3^- and Mg^{++} Concentrations suggest that the content of HCO_3^- and Mg is generally related to the retention period of the ground water in the aquifer. Mixing of hydrochemically different ground waters occurs as consequence of a hydraulic connection between separated ground water bodies by the weeding out of separating aquicludes, or by change in formations of an aquifer. With the natural recharge the mixing between surface and groundwater may be caused artificially in boreholes by hydraulic connection of separated aquifers.

An obvious influence of surface water can be seen in samples sources and boreholes situated in the range of infiltration of rivers which is indicated by an elevation of the ground water level(NO_3^-, PO_4^{3-}, Na^+, Cl^-,...) in vicinity of river, barge. Alteration by the chemical composition contamination of the ground water is often ocured by biogenesis pollution in wells situated in outcrop areas of the aquifer in the vicinity of villages and rivers. Pollution indicated by an elevated NO_3^-, PO_4^{3-}, SO_4^{--}, Na^+, Cl^-, K^+, COD, Cu^{++}, $Cr^{4,6+}$...content has been noticed particularly in outcrop areas of grand towns and its canalization's (Tartous, Banias,....).

Regional varieties of chemical composition of the ground water are found to be related to differences of lithology of the aquifer and influences of rocks overlying and underlying the aquifer. In dolomites aquifers a relation is assured between the concentration of the ground water, particularly the Mg^{++} and HCO_3^- content, and the retention period of the ground water in the aquifer. Mixing of natural waters of different chemical composition could be recognized in some cases by hydrochemical characteristics. Most of ground water samples represents the carbonate consolidated aquifer of Limestone's and Dolomites rocks of Jurassic-Cenomanian/Turonian period. Polluted shallow and deep ground water resulting from the infiltration of polluted surface water with high chemical composition of NO_3^-, SO_4^{--}, NO_2^-, PO_4^{3-}, F^-, Na^+, Cl^-, $Cr^{4,6+}$, Cu^{++}... etc is resulting with bad quality and unacceptable chemo-physical propriety's. Chloride-sodium chemical type due to the Salinization of ground water (shallow ground water) from the sea water encroachment was observed at North and South Tartous.

We needed to use the management of aquifer recharge (MAR) in this region as natural or artificial recharge (in Dams or wills) in this area and another basins of Syria.

INTRODUCTION

Hydrological and geological studies on the area began in 1924 when a French geological team headed by Deubertret (1933a,b) drew geological maps of Syria and Lebanon at the scale 1/1,000,000. Report of the German mission pay Boeckh E. (1966) and Wolfart R. (1967) identified the essential specification of the general aquifers in western Syria. The most frequent reported sources of pollution of karstic aquifers are urban centers and agriculture. Urban areas cause contamination by sewage systems and when wastes are disposed on surface of the ground or are discharged to watercourses, karstic depressions of swallow holes. The studied area is Mediterranean, rainy of average annual precipitation about 1000 mm/year which varies from 800 mm in coastal region to 1,600 mm on mountain peaks in the east. According to their importance and order from oldest to recent(Figure 1) Selkhozprom Export (1979) and Burdon (1961):

- Aquifers of appear cretaceous (Cenomanien-campanien) – Jurassic which join, in general, cracks and fissures consisting of cracked dolomite limestone rocks of large of more than 40 km. in the coastal range as in the range of Lebanon and Anti-Lebanon which form the most important sources of water recharge for cretaceous springs in Syria and Lebanon (Al-Sien, Abou-Koubiss, Al-Fijeh, Jaetah, Al-Tannour):their water is fresh to little salty;
- appear cretaceous limestone marl dolomite basins which feed sources of high salinity;
- Basalt aquifers, age of Cenomanian-Pliocene (Wadi-Al-Aions, Al-Kafroun) south of Tartous and coastal strip has good water, little salty and limited drainage;
- Aquifers of age Paleogene – Neogene and quaternary aquifer of conglomerate, limestone and sand rocks (Figure 1). Karstic groundwater is the most important drinking water resource in Syria and in many other countries of the world.
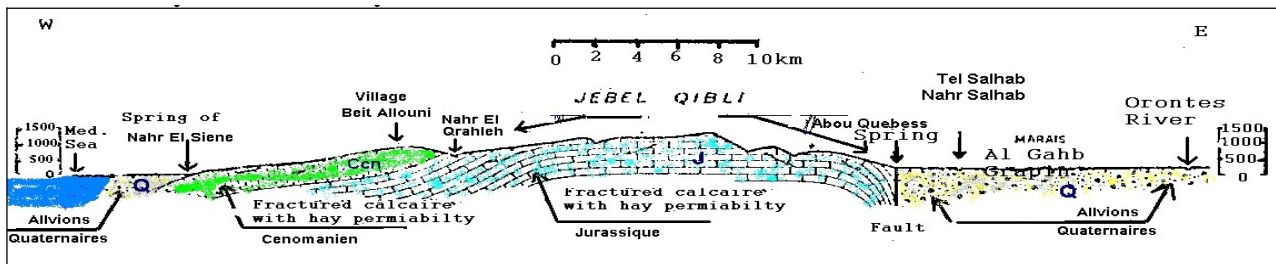
Figure 1. Hydrogeological cross-section along the Syrian coastal range (Deubertret, 1933)

METHODOLOGY

Water samples included most important water sources relevant to these formations /springs, wells/ with surface water samples from major permanent and seasonal rivers running in the area.

The physical and chemical characteristics of water were measured. Measurements took place in field /PH, C.E,T/ and in lab./ NH_4^+, Na^+, K^+, Ca^{++}, Mg^{++}, Fe^{+++}, Cu^{++}, Li^+, Cr^{6+}; F^-, Cl^-, SO_4^{--}, CO_3^{--}, HCO_3^-, NO_3^-, NO_2^-, PO_4^{---}/. For 66 water samples (Figure 2) (54 ground water and 12 surface water) were collected. Analysis was performed at laboratories of Ministry of Housing using the spectrophotometer and another international analysis method.

RESULTS AND DISCUSSION

This study included:

Temperature

The topographic situation and geotectonic condition of the aquifers play a high role in temperature value. The temperature of surface water sample changed frame 8 (at 1,040 m) to 20 in the Jouber polluted river (industrial rejects). But in the groundwater we can notice the effect of the lithology of aquifer and aquiclude with the tectonic situation of aquifers or (Kassem,1990; Bilal and Kassem, 1998) with 12.6, 13.2 in the water of Cretaceous basaltic aquifer to 21.1 and 22 C in the water of some wells at north and south of Tartous town, effected by the intrusion of seawater (Table 1).

PH

The degree of PH is changed between 7 and 8 for many water samples, but it is more than in the polluted, hydrothermal and mixing water.

Electrical Conductivity (µs/cm)

It determines the nature of storing surrounding and nature of surface laid or trespassed; they vary from 1,015 µs/cm in the El Marquieh polluted river to more than 59,200 µs/cm in seawater and great than 4,120 µs/cm in some water wells effected by sea water intrusion (Table 1).

Cations (NH_4^+, Na^+, K^+, Ca^{++}, Mg^{++}, Fe^{+++}, Cu^{++}, Li^+, Cr^{6+})

Concentrations such as cations: NH_4^+, K^+, Fe^{+++}, Cu^{++}, Li^+ and Cr^{6+} (Table 1) increase in the surface and ground water by the effect of swage sludge, industrial and agriculture activities. While the concentration of other cations depend on the natural aquifer rocks as well as the surface water infiltration and sea water intrusion. (Table 1).

Anions (F^-, Cl^-, SO_4^{--}, CO_3^{--}, HCO_3^-, NO_3^-, NO_2^-, PO_4^{---})

Anions concentration such as: F^-, NO_3^-, NO^{2-}, PO_4^{---} (Table 1) increase in some surface and ground water samples by the effect of the housing swage, industrial and agriculture activities (fertilizer, factories air pollution etc.). The most frequent reported sources of pollution of karstic aquifers are urban centers and agriculture. Urban areas cause contamination, particularly when they lack adequate sewage systems and when wastes are disposed on surface of the ground or are discharged to watercourses, karstic depressions of swallow holes.

While the concentration of Cl^-, SO_4^{--} and HCO_3^- (Table 1) depend on the kind of surface soil and aquifer lithology. The groundwater of karstic and no polluted aquifers have a little concentration in Cl^- and SO_4^{--} with a high quantity of HCO_3^- witch is due to the time of the water reserved in the aquifer. We can see the same situation for Chloride, Sodium and potassium concentration. In the surface water the content of these ions change by the housing rejects pollution that exist in chalky marl aquifer and mixing with sea water in some locations.

Spring water is the most likely water source to be developed for this region (El-Sien spring 12 m³/s). The problem is continuity of discharge and potential pollution. To use the permanent Dams water as the domestic water supply, or to management of aquifer recharge (MAR),water treatment is needed. In this area there are more chance for storm water to infiltrate and recharge into the groundwater body at the zones A and B (Fig. 2).

Managements of bank filtration schemes should be incorporated in order to limit potentially polluting activities in the groundwater recharge area and also to balance rivers and dams infiltration losses with ecological needs of the water in this area. 6 and 13 chemical types were identified successfully in surface and groundwater, it shows the change in surface and groundwater quality watch resulting from contamination or mixing with polluted surface water or by the seawater intrusion. The evaluation of the hydrochemistry data, presented by the diagram of Piper, show an increase in the quantity of some ions (Na^+, Mg^{++}, Cl^-, SO_4^{--}, NO_3^-,..), in the groundwater relating from the intrusion of seawater or by the existence of the marl and gray sandstones formations which are rich in these elements, or by the infiltrations of the urban and industrial sewages (Fig. 2).

Table 1. Chemical and physical characteristics of the surface and groundwater at the Coastal Basin

Code	pH	Temp	Turb.	E.C	NH4+	Na+	K+	Ca++	Mg++	Fe+3	Cu++	Li+	F-	Cl-	SO4--	HCO3-	NO3-	NO2-	PO4---	FR. CO2	D.O
		C°	(NTU)	µS/cm	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
1	7	13	3.8	404	0.18	5.98	0.6	52	34.1	0.1	0	0	0.2	21	7	281	20	0.03	0.77	4.4	8.6
2	7	15	6.7	253	0.13	3.22	0.6	32	19.5	0	0	0	0.1	14	8	159	12	0.01	0.39	4.4	8.2
3	8	17	1.3	1636	0.21	128	2.2	288	46.2	0	0	0.3	1.1	28	820	354	9.2	0	0.73	4.4	9.2
4	8	15	0.9	420	0	29.4	0	56	9.7	0	0	0	0.1	28	2	244	2.6	0	0.17	8.8	5
5	7	17	2.5	801	0.15	38.2	1	96	41.3	0.1	0	0	0.3	43	160	305	22	0.01	0.21	2.2	8.7
6	7	16	2.3	463	0.03	18.9	3.6	56	29.2	0.1	0	0	0.5	21	7	305	15	0.02	0.27	0	9
7	8	13	1.6	536	0.19	12.4	0.5	60	38.9	0	0	0	0.1	28	10	329	12	0	0.27		8.7
8	7	16	2.1	609	0.01	4.83	2.5	64	46.2	0	0	0	0.3	28	3	378	13	0.03	0.16	8.8	7
9	7	15	2.8	511	0.03	26.9	1	80	12.2	0.1	0		0.1	21	9	305	21	0	0.29	0	9
10	8	8.2	3.4	120	0.21	15.2	0.3	20	7.3	0	0	0	0	14	10	58.4	11	0.01	0.17		8.6
11	8	14	2.4	401	0.01	2.07	0.4	52	29.2	0.1	0	0	0.2	14	11	256	11	0.01	0.13	0	8.6
12	8	14	3.5	240	0.26	4.83	0.5	20	26.8	0.1	0	0	0.1	14	8	159	9.7	0.01	0.18		8.2
13	7	14	3.1	353	0.01	4.14	0.2	40	24.3	0	0	0	0.1	14	8	207	6.6	0.05	0.11	0	10.2
14	8	16	0.6	807	0.11	39.8	12.5	92	21.9	0.1	0	0	0.3	21	44	366	52	0.01	0.63	0	8.3
15	7	15	1.2	608	0	7.36	1.1	80	43.8	0.1		0	0.2	21	15	403	21	0.05	0.41	0	8.2
16	8	15	3.5	532	0.03	28.5	1.2	80	14.6	0.1	0	0	0.3	28	31	281	21	0	0.28	0	10.2
17	7	15	2.2	554	0.23	30.6	2.6	48	36.5	0.4	0	0	0.2	21	25	317	30	0.05	0.7	2.2	8.6
18	7	16	2.4	323	0.04	14.5	0.4	44	17	0.1	0	0	0.2	28	3	183	20	0	0.58	0	9.2
19	7	13	2.3	411	0.04	16.6	0.9	36	26.8	0.1	0	0	0.3	28	14	183	39	0.01	0.01	4.4	10
20	8	14	2.5	496	0.09	11	0.3	48	31.6	0	0	0	0.2	21	16	244	31	0.01	0.4	0	9.6
21	8	13	2.4	208	0.07	17.9	0.5	28	7.3	0	0	0	0.1	14	2	122	21	0.01	0.68	8.8	9.4
22	8	15	2.4	536	0.05	5.75	0.2	60	36.5	0.2	0	0	0.2	14	2	293	56	0.02	0.03	0	9.5
23	8	15	2.1	366	0.01	4.83	0.3	48	24.3	0.1	0	0	0.4	14	4	220	26	0.01	0.7	0	9.5
24	8	18	0.5	520	0	35.2	1.3	68	12.2	0	0.2	0	0.2	36	21	256	21	0	1.2	8.8	4.8
25	8	18	2.4	820	0.05	40.9	1.3	100	9.73	0.1	0	0	0.6	21	190	293	25	0.01	0.48	4.4	9.2
26	7	19	0.3	550	0	5.06	3.1	80	19.5	0	0	0	0.1	28	18	268	21	0	0.81	8.8	5.2
27	7	17	2	448	0	8.51	0.6	84	12.2	0.1		0.1	0	21	11	281	9.7	0.01	0.26	4.4	7.8
28	7	16	2	601	0	20	10.6	88	19.5	0.1		0	0.1	43	24	329	23	0.01	1.37	0	7.6
29	7	19	2.2	595	0	3.68	0.9	60	58.4	0.2		0	0.2	28	21	403	8.4	0.01	0.42	8.8	7.4
30	8	20	1.5	662	0	14.3	7.6	96	29.2	0.1		0.1	0.1	43	24	354	31	0.01	0.56	2.2	7.2
31	8	22	7	1390	0	61.9	1.7	160	53.5	0.1		0	0.1	213	77	305	153	0.06	0.73	0	7.2
32	7	20	1.5	490	0.22	4.14	0.02	4.2	1.8	0		0.1	0.1	21	10	317	13	0.01	0.42	4.4	7.6
33	7	22	2.1	1480	0.02	83.7	0.6	152	31.6	0.1		0.1	0.3	156	85	366	104	0.02	1.05	4.4	9
34	8	20	2.2	720	0.11	41.4	1	72	9.73	0.1		0	0.1	57	18	244	15	0.01	0.5	8.8	8.8
35	7	22	2.5	520	0	10.6	0.6	100	12.2	0.1		0.1	0.1	50	19	256	29	0.01	0.3	8.8	7
36	7	22	2.5	4120	0.05	519	6.5	264	51.1	0	0	0.3	0.6	951	320	378	29	0.01	0.52	4.4	9.3
37	7	21	2.2	744	0	61.4	2.4	60	26.8	0	0.3	0	0.5	71	27	317	2.4	0.02	0.24	4.4	10
38	7	19	1.2	676	0	9.89	1	108	26.8	0.1		0	0.2	43	20	378	14	0.03	0.41	2.2	8.4
39	8	18	2.2	601	0.03	41.4	1.1	100	4.86	0.1	0.1	0	0.3	36	27	317	22	0	0.76	0	9.4
40	7	19	1.2	730	0.1	8.28	1.2	100	34.1	0.1		0	0.3	50	45	329	22	0.03	0.41	0	8
41	8	20	0.8	651	0.01	2.76	1.1	88	26.8	0.1	0	0	0.3	43	35	268	18	0.01	0.27		8.6
42	7	21	1.6	1930	0	9.2	3.3	184	4.86	0.1		0	0.2	71	80	366	15	0.03	3.2	0	8
43	7	19	1.5	570	0	21.2	0.8	72	21.9	0.6	0.2	0	0.4	43	33	256	15	0.02	0.24	4.4	10.2
44	8	19	2.3	644	0.06	41.6	1.2	80	26.8	0	0	0	0.3	43	21	305	85	0.01	0.61	4.4	8.4
45	8	18	0.5	520	0	29.2	0	76	12.2	0	0.6	0	0	36	7	281	21	0	0.24	8.8	4.9
46	7	17	2.1	519	0.08	28.5	0.6	92	4.86	0		0	0.2	28	14	281	28	0.05	0.33	0	8.8
47	7	16	1	520	0	18.2	0.5	104	4.86	0.1		0	0.1	28	12	317	8.8	0.05	0.33	2.2	8.2
48	9	22	1.5	640	0.03	25.1	3.7	40	46.2	0.1	0		0.1	57	33	183	97	0.01	0.78	0	9.2
49	8	18	1.2	413	0.02	18.6	0.3	72	9.73	0.1	0	0	0.4	21	12	244	18	0.01	0.28		8.6
50	7	22	1.1	836	0	38.2	1.2	100	48.6	0.1		0	0.3	107	50	366	39	0.05	0.52	2.2	8
51	8	18	1.5	465	0.04	15.4	0.3	80	12.2	0.1	0	0.1	0.2	21	17	268	14	0.01	0.25		8.2
52	8	20	3	619	0.04	32.2	1.6	72	31.6	0.1	0	0	0.3	36	48	3.5	36	0.01	0.3		8.5
53	7	18	1.8	144	0.01	18.6	0.3	12	7.3	0	0	0	0.1	14	9	61	7.9	0.02	0.14		10.1
54	7	16	1.5	168	0.03	5.75	0.5	24	14.6	0	0	0	0	14	9	110	14	0.01	0.11		8.3
1*	8	13	3.5	540	0.06	15.2	1	68	31.6	0.1	0	0	0	21	25	317	18	0.01	0.17	0	8.8
2*	8	11	2	502	0.01	58.4	1	60	38.9	0.1		0	0.2	21	44	244	15	0.01	0.2	4.4	7.6
3*	8	8.2	3.4	120	0.21	15.2	0.3	20	7.3	0	0	0	0	14	10	58.4	11	0.01	0.17		8.6
4*	8	13	6.5	221	0.21	8.05	1	24	19.6	0.1	0	0	0	21	14	122	13	0.09	0.19	0	8.6
5*	9	16	1	470	0	29.2	0.5	60	12.6	0	0.1		0.2	36	19	232	5.7	0	0.45	8.8	5
6*	9	16	6	603	0	19.1	0.7	60	21.9	0.1	0.2	0	0.3	18	42	244	9.7	0.02	0.25		10
7*	8	16	2.4	481	0	24.6	0.6	68	19.5	0.1		0	0	21	14	305	12	0.01	0.23	4.4	7.4
8*	9	12	2	501	0	2.07	1.2	80	29.2	0.1		0	0.1	21	19	317	14	0.01	0.39	0	7.2
9*	8	18	2.3	342	0.03	28.1	0.3	44	19.5	0.1	0	0	0.2	21	43	195	14	0.02	0.14	0	8.8
10*	8	18	34	1015	0.3	15.6	5.3	40	26.8	0.3	0	0	0.3	185	51	268	24	0.03	0.42		8.8
11*	8	20	1.5	655	0.05	39.1	1.5	80	17	0	0	0	0.2	50	28	305	24	0.01	0.39		8.4
12*	8	26	1.2	59200	0.1	13153	85	520	2043	0		2	1.8	13809	2600	171	126	0.03	0.31	0	8

Note: 1,2,…54 = Groundwater and 1*,2*,…12* = Surface and sea water.

Orontes river

Turkey

Syria

Iraq

Jordan

Studys area

Coastal Area (Studys area)
International Border
Big Citys
Waters Flow Direction
River
Seasonal Stream
Water Samples:
Sepring
Weel 1,2,... Groundwater
1,2,... Surface and seawater

Qualty of water :

A Good quality with housing pollution rejects
B Fair quality with municipal and agricultural pollution
C Poor qualty with mixing pollution sources(irrigation pay polluted surface water and the intrusion of seawter)

Majorchemical types:	Groundwater Samples N°:	Surface water samples N°:
Bicarbonate - Calium	4,5,6,9,10,11,14,15,16,18,21,23,28,30,35,37,47,49,53	1,3,4,5,6,8,9,11
Bicarbonate - Magnesium	1,2,7,8,12,13,17,19,20,29,49	2,7
Bicarbonate - Calcium or magnesium(rCa=rMg)	2,13,22,54	
Sulfate -Calcium	3	
Chloride - Sodium	36	10 , 12 (Seawater)

Particular chemical types	Groundwater Samples N°:	Surface water samples N°:
Bicarbonate chloride-Mgnesium calcium	1,7,8,12,17,19,20,29,48	
Bicarbonate chloride-or calcium sodium	2 ,13 , 54	
Sulfate bicarbonate-Calcium sodium	3	
Bicarbonate chloride-Calcium sodium	4,10,16,24,32,33,34,37,39,45,46,47,49,50	3 , 5 ,11
Bicarbonate sulfate-Calcium magnesium	5 , 14	6 , 9
Bicarbonate chloride-Calcium magnesium	6,11,15,18,21,26,27,28,30,31,35,38,40,41,43,51,52	1 , 4 , 8
Bicarbonate chloride-Calcium sodium	9 , 53	
Bicarbonate nitrate-Calcium or magnesiumsodium	22	
Bicarbonate nitrate-Calcium magnesium	23,44	
Bicarbonate sulfate-Calcium sodium	25	
Chloride bicarbonate-Sodium calcium	36	
Bicarbonate chloride-Calcium mag.or sodium	42	
Bicarbonate chloride-Calcium or mag. sodium	54	
Bicarbonate sulfate-Magnesium calcium		2 , 7
Chloride bicarbonate-Sodium magnesium		10
Chloride sulfate-Sodium calcium		12(seawater)

Well(in white)
Spring (in red)

Triangular diagramme of Pieper for the groundwater samples in the Study area

A Karst and Baseltic aquifers from Jurassic and Cenomanien Turoien age

B Chalky Marle and Sableuses Aquifers from Cenomanien,Palyogene and Quaternaire age

C Sableuse and conglomeratic aquifers from Quaternaire age

Figure 2. Samples water sites (with chemical types and water quality) of the study's area

CONCLUSION AND RECOMMENDATIONS

There are an assay to use of Management of Aquifer recharge (MAR) by injected the surface or ground water in the wells. But in these area we have a project to store the water of the El-Sien spring in the El-sekabeh Dam at 1 km NE of Banias city (Figs. 1, 2).The water of this dam with anthers dam and wills, will be transfered after treatment for drinking suply to Damascus city.

This study gave us a good idea about the quantity and the quality of surface and ground water that will be utilizing in several water use or in future artificial recharge in some dams or well at basaltic or unconfined karstic rocks at the southern of Tartous city (Figure 2).

In coastal studied area there are about 900 Mm^3/year, from ground water, discharge in sea, and there are about the same quantity of surface water loss in the Mediterranean sea. For that reasons this water need good management projects to use it in natural or artificial recharge in these region or at the arid and semi-arid area of the Orontes basin at East of the studied area (Figure 2).

As a result of this work several recommendations are advised:

- Developing sustainable system for enhancing and protecting groundwater resources.
- Range of methods including bank filtration, aquifer storage and recovery and soil aquifer treatment.
- Using the aquifer to improve the quality and security of drinking water and agricultural water supplies.
- Reducing the toxic load of each contaminant by adsorption and biodegradation as water passé through the soil.
- After considering the environmental damages caused by natural and anthropogenic pollutants, it is necessary to find out whether artificial recharge will be profitable to attenuate these natural processes and artificial pollutants.
- Treatment of the municipal and housing sewages before it's mixing with the surface water.
- Decreasing the utilization of fertilizers or pesticides quantity in the agricultural activity.
- Determination of the pollution causes, chemical types and quality of surface and groundwater for the different aquifers, dams and rivers in these region and the anthers of Syria.

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Exploring surface- and groundwater interactions with the help of environmental tracers and wastewater indicators in Berlin/Germany

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Abstract

The aim of the study was to calculate mixing proportions of treated wastewater in the surface water and production wells during bank filtration as well as the travel times to observation and abstraction wells. For this purpose, a variety of tracers such as the stable isotopes deuterium (D) and ^{18}O and several wastewater indicators like chloride, EDTA (ethylenediaminetetraacetic acid), boron and the rare earth element (REE) gadolinium (Gd) are used and compared to each other. Time series measurements in the surface water could be traced back in bank filtrates and raw water. Gd-DTPA was found to be a useful sewage indicator, even though it is biodegradable at favourable conditions at very slow rates. The travel times of the bank filtrates were obtained by the analysis of the peak shift in time-series of the tracer. Most tracers were found to be applicable but best results were obtained with the stable isotopes.

Keywords

Bank filtration, wastewater, tracer, stable isotopes, gadolinium.

INTRODUCTION

The drinking water production in the Berlin metropolitan is carried out by bank filtration and artificial groundwater recharge. The production well galleries are located near the main rivers and, induced by the pumping of the wells, the surface water infiltrates into the ground. The raw water in the production wells then consist of a mixture of natural groundwater from the landside and bank filtrate of around 70% (Pekdeger and Sommer-von Jarmerstedt, 1998).

The Berlin drinking water production can be characterized as a semi-closed cycle. Drinking water is produced in 9 waterworks (Fig. 1) of the Berlin Water Company (e. g. 215 Mio m³ in 2003). After use, the wastewater is distributed to the 6 Berlin wastewater treatment plants (WWTPs) and after treatment it is released into the surface waters. Despite of the biological treatment stages several substances are not eliminated completely and the wastewater discharge increases the concentrations of boron and EDTA (detergents), chloride and dissolved organic compounds in the receiving surface water. These wastewater indicators can then be used to calculate the proportions of wastewater in the surface water and in the drinking water that is produced by bank filtration as well as the travel time of the surface water to the production wells.

Within a well gallery the proportions of bank filtrate considerably differ because of (a) the heterogenic aquifer conditions in this region and (b) the number and location (depth) of the well screen. Depending on the degradation and sorption capacity of the aquifer material, the wastewater indicators reach the production wells by bank filtration and artificial groundwater recharge, generally in reduced concentrations. The dilution with natural groundwater further decreases the concentrations of these indicators. In some cases the wells are screened in depths were the intrusion of salt water of geogenic origin occurs. Hence, the Cl⁻ and B concentrations in the natural groundwater can exceed the concentrations of the surface water and can therefore locally not be used to determine the proportion of bank filtrate. Temporal and spatial variations of the surface water quality additionally complicate

the calculations of proportions of bank filtrate. All these factors have to be taken into account when estimating the proportions of bank filtrate and recycled wastewater in the raw water or by calculating travel times (Massmann et al., 2004).

The use of one single tracer is not possible in a complicated system as in the watershed of Berlin. The tracers may have more than one source or they are not conservative. The application of conservative tracers in combination with different wastewater indicators is therefore necessary.

The aim of the study therefore is to calculate mixing proportions of treated sewage in the surface water and production wells as well as the travel times to observation and abstraction wells, applying variety of tracer such as stable isotopes of the water molecule and several sewage indicators (Cl^-, EDTA, B, Gd-DTPA). In addition, the applicability of Gd-DTPA as a sewage tracer is examined.

FIELD SITE AND SAMPLING

The study area comprises the Upper and Lower Havel river system in the western part of the Berlin metropolitan (see box in Fig. 1). The surface water samples were taken monthly from at 59 locations over a period of 1½ years. Several transects are installed at representative sites in Berlin to investigate the bank filtrates from the surface water to the production well and the groundwater from the background (Fig. 1). These transects consist of groundwater observation wells screened in various depths between the surface water and the production well and behind the well.
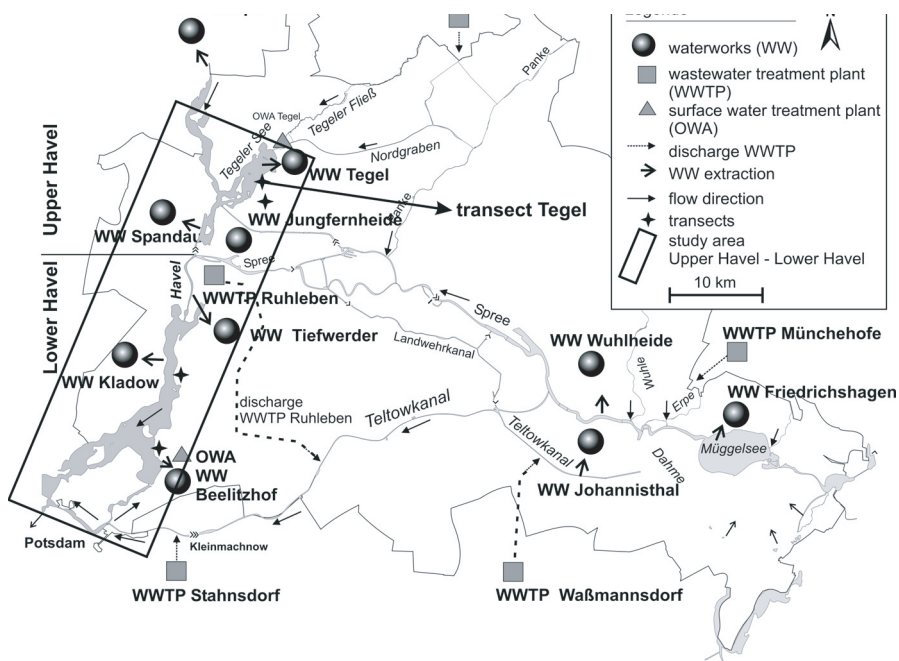
Figure 1.
The Upper and Lower Havel river system in the western part of Berlin and investigation area transect Tegel

METHODS

Surface water samples were taken for chloride, boron, stable isotopes and gadolinium (Gd) analysis. Monthly sampling at the field-transects was conducted over 2 years by the Berlin Water Company (BWB) and samples were analysed for physico-chemical properties, major anions and cations as well as a number of wastewater indicators (e.g. B, EDTA) at BWB.

The stable isotope measurements of deuterium (D) and ^{18}O were done at the Alfred Wegener Institute, Research Unit Potsdam with a H_2 equilibration method for D/H ratios and a CO_2 equilibration method for $^{18}O/^{16}O$ ratios

with a Finnigan MAT Delta-S mass spectrometer. Details on the instrumentation, calibration and the measurement routine are described by Meyer et al., 2000. Samples for rare earth element (REE) analysis including gadolinium had to undergo an enrichment procedure modified after Shabani et al., 1992 and explained in Bau and Dulski, 1996. Analyses were done subsequently by mass spectrometry (ELAN 5000 ICP-MS) at the GeoForschungsZentrum (GFZ) Potsdam. The measurement routine for REE analysis is described in detail in Dulski, 1994.

RESULTS AND DISCUSSION

Surface water

The wastewater influence of the WWTP´s into the Upper and Lower Havel river system can be seen in elevated concentrations of B, Cl^- and Gd_{excess} and in a more negative isotopic composition because of the proportion of groundwater (more negative values) in the drinking water. Figure 2 shows the results from B, Cl^-, $\delta^{18}O$ and anthropogenic Gd (Gd_{excess}) analysis in the Berlin surface water system of the Upper and Lower Havel river (see also Fig. 1) during the sampling campaign in July 2001. At this time the base flow was the lowest in 2001/2002 (11.1 m³/s in July 01 compared to 50 m³/s in December 01 in the Lower Havel). Generally in summer months greater spatial dif-
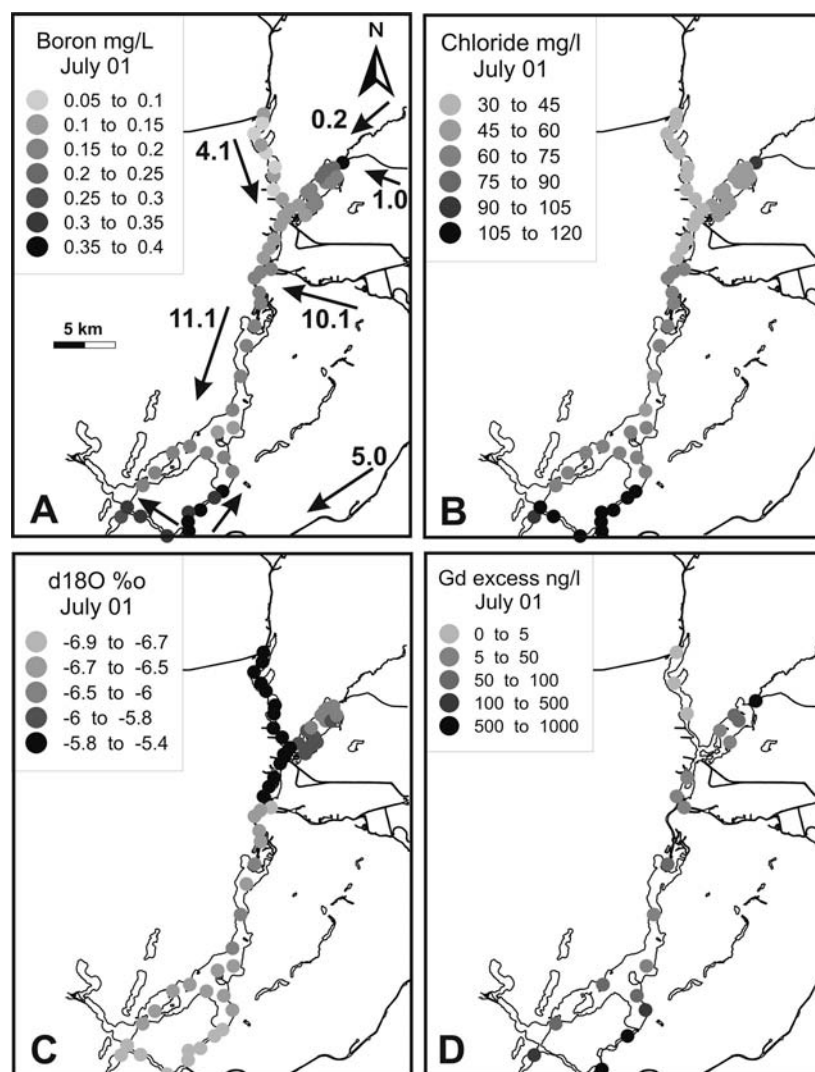
Figure 2. Concentrations of wastewater indicators B, Cl^-, ^{18}O and Gd_{excess}
in the surface water of the Berlin study area. In Figure A the flow directions are indicated by the arrows.
The discharge in July 2001 in m³/s is shown by the numbers beside.

ferences and higher absolute concentrations of the wastewater indicators could be observed than in the winter months.

The Upper Havel River is characterised by lowest concentrations of B, Cl$^-$ and Gd$_{excess}$ and highest δ^{18}O values when it enters the city area from north (Fig. 2A-2D). Here, the wastewater influence is negligible and the inflow of the Upper Havel River represents background values. The highest concentrations and lowest δ-values in the Upper Havel region were detected in the Nordgraben, a small ditch carrying the discharge of the WWTP Schönerlinde before it flows into the Lake Tegel (see Fig. 1). The influence can be traced into the Lake. In the Lower Havel region the Teltow Canal carrying the discharge from the WWTP´s Waßmannsdorf, Ruhleben (only in summer months) and Stahnsdorf (see Fig. 1), shows the highest concentrations and lowest δ-values. The influence of the Teltow Canal can be traced into the Lake Wannsee where the WW Beelitzhof is located (Fig. 1).

The percentage of treated wastewater in the surface water varies strongly depending on the location and on the season with different base flow conditions. Calculations with the data on total discharge and WWTP discharge resulted in a maximum percentage of 77% of treated wastewater for the Nordgraben (45 % average), and up to 90% for the Teltow Canal (average 47%) before the confluence with the Upper Havel. The surface water leaving Berlin in the south was found to contain up to 42 % of treated wastewater (13 % average;Massmann et al., 2004).

Bank filtration (Lake Tegel)

At Lake Tegel, a transect with numerous observation wells and a production well was sampled monthly since March 2001. Details about the hydrogeological conditions are described in Fritz et al., 2000 and Massmann et al., 2000. The position of the transect is given in Figure 1 and a schematic sketch of the transect is given in Figure 3. The observation wells are installed perpendicular to the lake in direction of the production well 13. Observation well 3304, behind well 13 represents the background groundwater, the raw water in well 13 is a mixture of bank filtrate and groundwater from the background. Most tracers were found to be applicable but best results were obtained with the stable isotopes. The stable isotope ratios δD and δ^{18}O as conservative tracers show seasonal variations in the surface water. Thermodynamic fractionation processes cause a depletion or an enrichment of the heavier isotopes. This results are more negative values in winter and more positive values in summer. Once the surface water infiltrates into the ground, the isotopic composition does not change and the summer and winter signals are conserved. Only the mixing with older bank filtrate or natural groundwater and dispersion can smooth the peaks and depths.
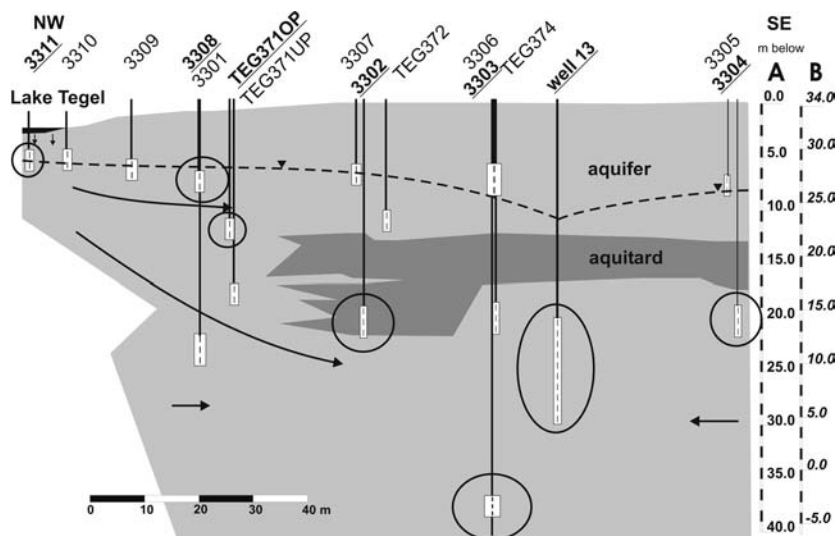
Figure 3.
Study area transect Tegel
with observation wells
and production well 13.
The herein described
observation wells are marked
with circled screens.
Scale A: meters below ground;
scale B: meters above sea level.
Arrows indicate the flow direction

Figure 4 shows an example for time series delineated with $\delta^{18}O$ in the surface water, observation well 3303, well 13 and the background in 3304. The travel time from the surface water to 3303 is 3–5 months, depending of the time of year. In the near production well 13 the temporal signals are smoothed because of the mixing with the background groundwater (3304). The background itself shows more negative values and a more smoothed temporal signal than the surface water. Same results were obtained with δD and are therefore not shown.
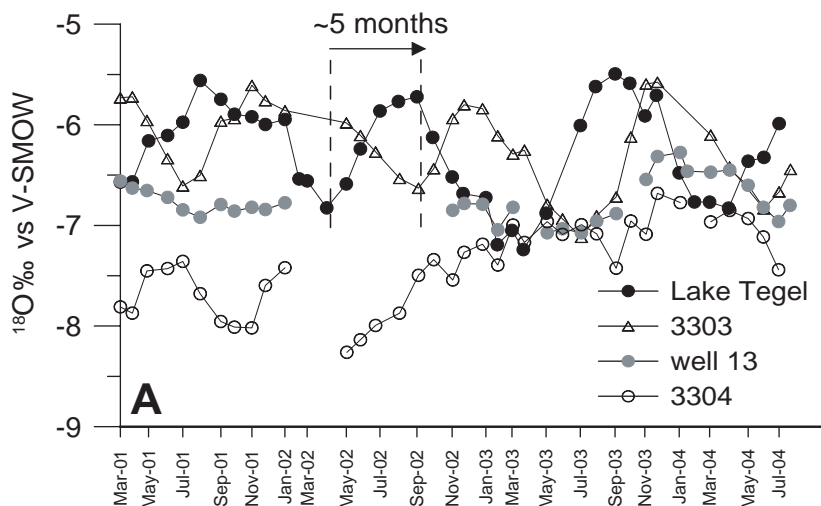
Figure 4.
Time series of ^{18}O in lake water,
deep observation wells
before (3303) and behind (3304)
the production well 13

The anthropogenic input of B, Cl⁻, gadolinium and EDTA in the surface water was traced along the flow path. Figure 5 shows the box plots of some observation wells towards the production well 13 (locations see Fig. 3). Boron concentrations show wide variations in the surface water but the average (~190 µg/L) is comparable with the deep observation wells 3302 and 3303 which consist of 100% bank filtrate.
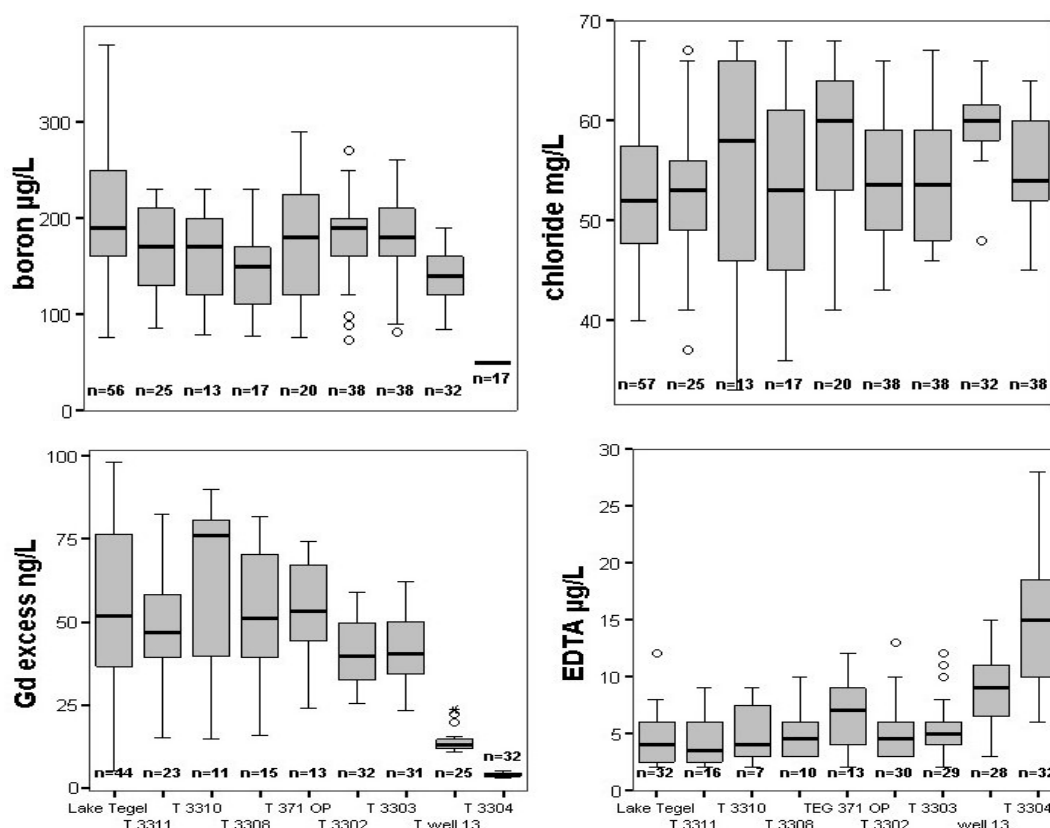
Figure 5. Box plots for concentrations (n= number of analyses) of boron, chloride, Gd_{excess} (except 3304
which represents geogenic background concentrations) and EDTA in Lake Tegel and observation wells
in flow direction towards the production well 13. Locations are given in Figure 3. The distances are not equidistant.

The much lower background value of 3304 (< 50 µg/L) is clearly distinguishable. The differences in concentrations of the two end members make mixing calculations in well 13 favourable. Chloride cannot be used to calculate mixing proportions because the concentrations well 13 always exceeds that of the observation well 3303 and the background well 3304. This indicates the minor occurrence and influence of more saline groundwater from deeper layers, which was pumped during the drinking water production. The rare earth element gadolinium (Gd), which is used as a component in contrast agents (mostly in form of Gd-DTPA), causes anomalously high Gd concentrations in the surface water, expressed as Gd_{excess} (Bau and Dulski, 1996; Knappe et al., 1999). It is one of numerous pharmaceutical residues and passes the WWTPs without significant reduction and is therefore a distinct wastewater indicator. The wide concentration variations in the surface water are smoothed and lowered in the deep observation wells 3302 and 3303. It is known that Gd-DTPA is degradable and not conservative. Column experiments with sandy aquifer material showed that in average 50% of Gd is removed in ~130 days (Holzbecher et al., in press) depending on the conditions in the aquifer material (e. g. microbial activity, organic content). Nevertheless, like boron, the Gd concentrations before and behind the well differ distinctly and can therefore be used for mixing calculations. EDTA concentrations show a reverse picture with much higher concentrations in the background 3304 than in the surface water or in the deep well 3303. Nevertheless, the mixing proportions in well 13 can also be calculated with EDTA. The percentage of bank filtrate in well 13 was estimated with the equation

$$X = [C_W - C_{GW})/(C_{SW} - C_{GW})] * 100$$

where C is the concentration of a suitable tracer in the groundwater (C_{GW}), well (C_W) or surface water (C_{SW}). The surface water in the area of the transect Tegel has an average proportion of 11 to 20% treated wastewater, calculated with Gd_{excess} and chloride concentrations respectively. The percentage of bank filtrate in well 13 was calculated with the average concentrations of B, EDTA and Gd_{excess} and the average $\delta^{18}O$ values and result in 68%, 59%, 53% and 53% respectively. Assuming a proportion of wastewater in the surface water of 20% near the transect, the proportion of wastewater in the raw water of well 13 is around 12% can be calculated.

CONCLUSIONS

The tracer $\delta^{18}O$, δD, EDTA, Gd, and B were found to be applicable for the calculation of travel times and mixing proportions in bank filtration wells in Berlin. The percentage of treated sewage in ground- and surface water can vary strongly in both space and time. Likewise, the travel times to individual abstraction wells is different from site to site, depending on factors such as distance to shore, thickness and characteristics of the clogging layer and hydraulic regime. An advantage of the use of Gd_{excess} is its detection limit in the range of 10 ng/L, whereas the detection limit of most other wastewater indicators (Cl^-, B and EDTA) are much higher.

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Arsenic mobilization and sequestration during successive aquifer storage recovery (ASR) cycle tests in the carbonate Upper Floridan aquifer, South Florida

June E. Mirecki

Abstract

Geochemical reactions between recharge water and aquifer lithologies during cycle testing can affect recovered water quality. In Lee County FL, the Olga ASR system recharges potable water into the Upper Floridan aquifer, specifically a permeable zone of the Suwannee Limestone. Arsenic is mobilized in ASR systems elsewhere in west-central Florida, most likely by pyrite oxidation in the Suwannee Limestone.

During three successive cycle tests at the Olga ASR system, arsenic increased from 3 to 68 µg/L in water recovered from the ASR well. Arsenic remains <8 µg/L in concurrent monitor well samples located 105–120 m away. These two characteristics suggest that dissolved arsenic sequestration occurs along the recharge flowpath, away from the ASR well. During recovery, arsenic apparently is desorbed as water travels to the ASR well. Eh values from sulfide/sulfate and arsenite/arsenate couples range between −0.10 and −0.25 volts, consistent with sulfate reduction. Geochemical modeling of cycle test 3 at the Olga ASR system indicates stable iron sulfides at monitor wells. Arsenic sequestration by iron sulfides is feasible, but probably is reversible. Increasing arsenic during recovery at the ASR well may be due to: 1) reductive dissolution of iron oxyhydroxides; and/or 2) desorption of As(III) from stable iron sulfides.

Keywords

Arsenic transport, geochemical modeling, redox, Upper Floridan aquifer.

INTRODUCTION

Drinking water supplies have been augmented successfully through the use of aquifer storage recovery (ASR) systems to store potable water in several permeable zones of the Upper Floridan aquifer system in South Florida. During ASR, surface or ground water is treated to Federal and state drinking water standards, then is recharged into permeable zones that occur in carbonate or mixed carbonate/clastic lithologies. At a few sites however, interactions among native water, recharge water, and aquifer lithologies can affect water-quality, increase treatment costs, and therefore limit the feasibility of ASR.

Arsenic mobilization has been identified at several ASR systems located in west-central and southwest Florida (Arthur et al., 2001; Arthur et al., 2002; Price et al., 2002; Williams et al., 2002). During recharge, recharge water oxidizes framboidal and microcrystalline pyrite, mobilizing arsenic and other trace metals into the Upper Floridan aquifer system. In recovered water samples at the ASR well, arsenic concentrations sometimes exceed the Federal and state maximum contaminant level for arsenic (10 micrograms per liter, µg/L). However, arsenic concentrations remain relatively constant (<10 µg/L) at monitor well locations. Arsenic mobilization during ASR cycle testing poses a regulatory challenge to ASR feasibility. The Upper Floridan aquifer system is an underground source of drinking water, a designation that prohibits the movement of fluids containing contaminants that exceed primary drinking-water quality criteria (Title 40 Code of Federal Regulations (CFR) part 144.12). In addition, the points of

compliance with drinking-water quality criteria are at the ASR (recharge/recovery) well, within the aquifer, and in recovered water (Richtar, 2004). An understanding of the geochemical controls of arsenic mobilization, sequestration, and transport in the Upper Floridan aquifer system is critical for future development of ASR.

Geochemical modeling methods enable a quantitative assessment of geochemical reaction mechanisms, water-quality evolution, and mass transfer in aquifer environments. Geochemical modeling methods have been applied previously for interpretation of water-quality changes during ASR cycle testing (Castro, 1995; Mirecki et al., 1998; Vanderzalm et al., 2002; Petkewich et al., 2004). Geochemical model development to test hypotheses of arsenic mobilization and sequestration requires complete water-quality data sets for unambiguous interpretation. Minimum analytical requirements that define a complete water-quality data set are major dissolved inorganic anions and cations, pH, carbonate alkalinity, and redox-sensitive species concentrations (total sulfide, arsenate and arsenite, dissolved ferric and ferrous iron). In addition, sample collection throughout the entire cycle test is necessary from both ASR and monitor wells. Unfortunately, few datasets from ASR systems in South Florida fulfill these requirements (Mirecki, 2004). Water-quality data obtained during cycle tests at the Olga ASR system are presented here to further define geochemical conditions that control arsenic transport and fate in the Upper Floridan aquifer system. The objectives of this report are: 1) to define trends in arsenic concentration during cycle tests at a potable water ASR system where water is stored in the Upper Floridan aquifer system of southwest Florida; and 2) to define the geochemical controls for arsenic sequestration in the aquifer geochemical environment.

Hydrogeologic setting

The Olga ASR system consists of one ASR well and two storage zone monitor wells located 105 and 120 m west and southwest of the ASR well. Potable water is stored in a permeable zone within the Suwannee Limestone (one of several permeable zones within the Upper Floridan aquifer system) at depths ranging between 260 and 280 m below land surface. Suwannee Limestone lithologies consist of white to pale-orange to light-brown calcarenic limestone with minor dolostone and sandstone (Weddeburn et al., 1982; Brewster-Wingard et al., 1997; Missimer, 2002).

METHODS

Water-quality data (available from the author) and recharge/recovery volumes from Olga ASR system cycle tests were compiled from consultant's reports and monthly operating reports that were submitted to the Florida Department of Environmental Protection by the water utilities. The water-quality data sets from the Olga ASR system are the most complete, and include analysis of redox-sensitive species (dissolved sulfide, arsenite, arsenate) during cycle test 3 in addition to standard water-quality analytes. All standard water-quality analyses were performed at laboratories certified by the National Environmental Laboratory Accreditation Program.

Arsenic species concentrations were measured in recovered water samples obtained during Cycle 3 recovery at the Olga ASR. In these samples, total arsenic concentration was measured using graphite furnace atomic absorption spectroscopy, with a method detection limit of 1.0 µg/L. Arsenate (As(V)), arsenite (As(III)), and methyl arsenicals were separated using high-performance liquid chromatography (HPLC), and quantified by inductively coupled plasma-mass spectroscopy (ICP-MS) following the methods of Bednar et al. (2002, 2004). HPLC/ICP-MS method detection limits were 0.6 to 1.8 µg/L. When calculating As(III)/As(V) values, concentrations that were below the detection level are reported at the detection level. No methyl arsenical species (monomethyl and dimethyl arsenic) were detected in any sample.

Water-quality data obtained during selected cycle tests were interpreted using the geochemical modeling code PHREEQC (Parkhurst and Appelo, 1999). The approach of this geochemical modeling effort is to define redox controls on arsenic mobility, as a precursor to reactive transport modeling. Arsenic transport and fate is controlled

by the aquifer redox environment, which becomes more reducing over time (during storage and recovery) and space (farther away from the ASR well) as dissolved oxygen is consumed and sulfate reduction produces dissolved hydrogen sulfide in the aquifer.

RESULTS

The goal of this work is to test hypotheses that explain the spatial and temporal distribution of arsenic species measured during ASR cycle tests. A valid hypothesis must explain why increasing arsenic concentrations are observed at the ASR well during recovery, but are not detected at monitor wells throughout the ASR cycle. The recharge water flowpath is defined by chloride trends during each cycle test. Trends in arsenic and major redox couples show how the aquifer redox environment becomes more reducing as each cycle test proceeds.

To assess the spatial and temporal extent of arsenic mobility, the history of cycle testing is presented (Fig. 1). A typical cycle test strategy is to recharge the aquifer with several hundred megaliters (ML) of potable water, and recover a lesser volume, limited by the Federal and Florida chloride maximum contaminant level of 250 mg/L. The result of this strategy is to develop a buffer zone between native aquifer water and recharge water, with the intent to enhance recovery efficiency by minimizing mixing between recharge and native aquifer water.

The geochemical conditions of the buffer zone are not well established with respect to arsenic mobility, because buffer zone water may not be sampled at either monitor or ASR wells during a cycle test. It will be critical to establish whether geochemical conditions at the buffer zone favor arsenic mobility or sequestration, so that ASR systems can comply with underground injection control regulations.
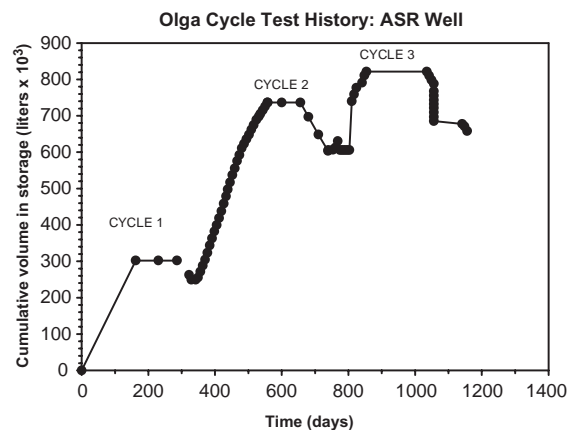
Figure 1. Plot showing cumulative storage volumes for three successive cycle tests at the Olga ASR system

Cycle test characteristics the Olga ASR system

Three successive cycle tests have been completed at the Olga ASR system. Recharge volumes range between approximately 215 and 490 megaliters (ML), with the greatest volume recharged during cycle 2. The typical pumping rate is approximately 2 ML/day. Successive cycle tests at both ASR systems have resulted in the development of a buffer zone between native aquifer water and recharge water, as shown by the positive trend in cumulative storage volume plots (Fig. 1). Cycle 3 recovery efficiency (75.7 percent) was significantly higher compared to previous cycles (Cycle 1, 23.8 percent; Cycle 2, 27.3 percent). It is likely that buffer zone water from previous cycles was recovered during cycle 3.

Chloride breakthrough curves at monitor wells show that recharge water passes their locations during each cycle test (Fig. 2). Contrasting chloride concentrations between native aquifer water ($1,100 \pm 100$ mg/L, n=2) and recharge water (79.1 ± 16.8 mg/L, n=17) allow definition of breakthrough curves. However, chloride concentrations decline at monitor wells at the initiation of each successive cycle test, consistent with buffer zone development.

Redox trends during successive cycle tests at Olga ASR system

Concentrations of major redox couples (O^-/O_2; H_2S/SO_4^{2-}; As(III)/As(V)) indicate that aquifer becomes more

reducing through time (through the cycle test) and space (between the ASR and monitor wells) during cycle test 3. It is likely that some dissolved oxygen is introduced in recovered water samples by cavitation in the well bore (Petkewich et al., 2004). Concentrations of major redox couples measured during cycle test 3 are shown in Table 1.

Redox trends were interpreted with preliminary geochemical models developed using water-quality data obtained during Olga cycle test 3. Eh values were calculated from major redox couples (O^-/O_2; H_2S/SO_4^{2-}; As(III)/As(V)). Recharge Eh values were calculated from the dissolved oxygen couple, and range between +0.72 and +0.76 volts. Recovery Eh values were calculated from the hydrogen sulfide/sulfate and As(III)/As(V) couples. Recovery Eh values at the ASR well range between +0.04 and −0.08 volts (As(III)/As(V) couple), and −0.16 and −0.26 volts (sulfide/sulfate couple). Eh values at monitor wells range between −0.06 and −0.14 v (As(III)/As(V)), and −0.22 and −0.29 volts (sulfide/sulfate). Eh values of approximately −0.2 are consistent with microbial sulfate reduction as a dominant process.

Arsenic trends during successive cycle tests at Olga ASR system

Arsenic transport during three successive cycle tests at the Olga ASR system shows the following characteristics (Fig. 2). In monitor well samples, arsenic concentrations generally remain below 6 µg/L (one sample during cycle 2 was 32 µg/L) throughout three successive cycle tests. In concurrent ASR well samples, arsenic concentrations increase toward the end of recovery, up to 68 µg/L. What is the geochemical process that explains this apparent conundrum: arsenic concentration is low under reducing conditions at monitor well locations throughout the cycle test, but arsenic concentration is greatest during the later stages of recovery, where water farthest from the ASR well (under reducing conditions in the buffer zone) is sampled. The spatial and temporal distribution of arsenic during ASR cycle testing has significant regulatory implications. If arsenic is mobilized during pyrite oxidation, then what is the magnitude and spatial extent of transport?

CONCLUSIONS

Arsenic transport at the Olga ASR system is controlled by the evolution of reducing conditions as the cycle test proceeds. Arsenic that was released to the aquifer by pyrite oxidation during recharge apparently is sequestered along the recharge flowpath between the ASR and monitor wells. Arsenic complexes readily with iron oxyhydroxide minerals under oxic conditions (Grafe et al., 2002; Dixit and Hering, 2003), or sorb to iron sulfide minerals that form under sulfate-reducing conditions away from the ASR well. These processes can sequester arsenic along the recharge flowpath before reaching the monitor wells located 105 to 120 m away. During storage, dissolved oxygen is consumed and total sulfide increases, most likely due to microbial sulfate reduction. Arsenic, mobilized during recharge as arsenate (AsV), reduces to arsenite (AsIII). During recovery, arsenic (primarily as arsenite) is released by reductive dissolution of iron oxyhydroxides, or is desorbed from stable iron sulfide mineral surfaces that occur in zones of sulfate reduction away from the ASR well (Fendorf et al., 1997). Subsequently, arsenic concentrations increase in ASR well samples as recovery proceeds.

Several strategies to minimize arsenic release during ASR cycle testing are under investigation. Most strategies remove dissolved oxygen during disinfection pre-treatment, prior to recharge.

Proposed strategies include degassing technologies (bubbling with nitrogen gas), passage through membranes, or chemical compounds that scavenge oxygen (sodium sulfite). The cost, benefit, and effects on groundwater quality for each method currently are under evaluation. Minimizing arsenic mobility has implications for the feasibility of ASR systems throughout south Florida.
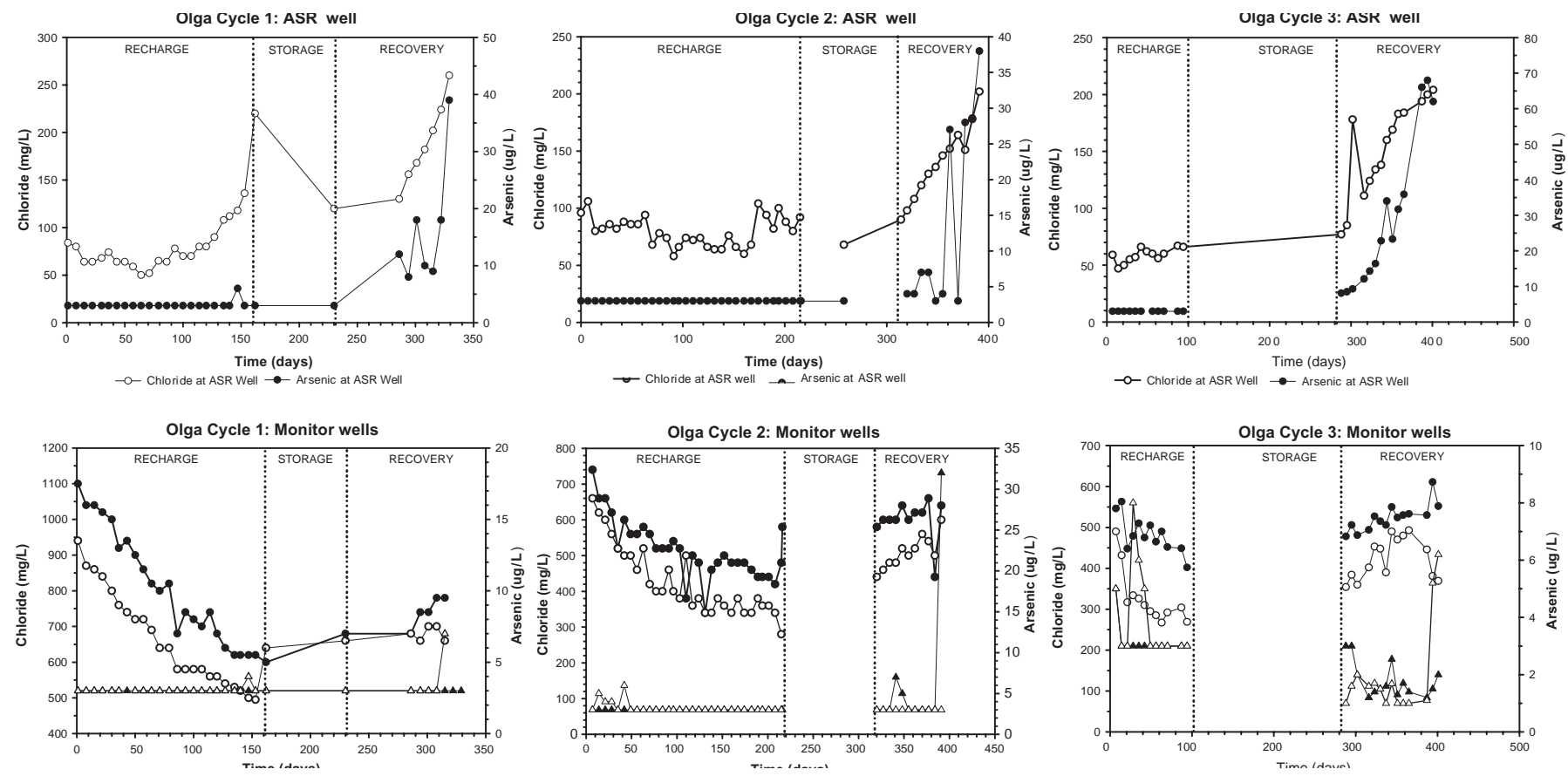
Figure 2.
Plots showing trends in chloride and arsenic in ASR well samples (top) and monitor well samples (bottom)
through three successive cycle tests at the Olga ASR system

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Biological clogging of porous media: tracer studies of non-uniform flow patterns

Dorte Seifert and Peter Engesgaard

Abstract

Bacterial motility and chemotaxis significantly influenced bioclogging in two-dimensional sand box experiments. The bulk porosity and hydraulic conductivity were reduced by 2–5% and 30–40%, respectively. Numerical simulations capture overall features of the bioclogging experiments and demonstrate that preferential flow may be triggered due to motility/chemotaxis in combination with an initial heterogeneous distribution of bacteria in the sand box.

Keywords

Bioclogging, random motility, chemotaxis, preferential flow.

INTRODUCTION

Bioclogging refers to the reduction of the pore space due to growth of microorganisms. Bioclogging results in a reduction of the porosity and hydraulic conductivity, which then affects flow and transport. Bioclogging is of interest in many applications such as artificial recharge, river bank filtration, and engineered bioremediation schemes. The ability of the microorganisms to move themselves towards a substrate is referred to as motility and chemotaxis. These processes have been observed in experiments (Duping and McCarty, 2000), but have generally not been considered in a quantitative manner except for systems without flow (Barton and Ford, 1995; 1997).

We performed 2D sand box experiments to investigate the development of preferential flow due to bioclogging. Preferential flow in this context is defined very general and indicates a change away from a smooth transport behaviour towards a more erratic flow behaviour with, for example, flow bypassing and fingering. The role of motility/chemotaxis in enhancing this transition is explored by using dye tracer studies and numerical simulations.

METHODS

Sand box experiments

Details about the experiments can be found in Seifert and Engesgaard (2005b). The sandbox was packed with quartz sand with a grain size of 0.4–0.8 mm, Figure 1. A 12 cm long strip of sand inoculated with microorganisms was placed approximately in the middle of the sandbox. The clean sand up- and downgradient the strip was sterilized at 121°C for 20 minutes. A substrate was injected into the sandbox through a perforated stainless steel pipe located 3.8 cm downgradient the inlet chamber. The pipe had a diameter of 6 mm with two holes per cm each with a diameter of 1 mm. A thin layer of coarser sand was placed around the pipe to prevent sand to enter the pipe. The background flow from the inlet chamber was constant. The flow rate through the pipe accounted for 10% of the total flow rate. The initial pore water velocity was 8.6 m/d corresponding to an initial retention time in the sandbox of about 75 minutes. The hydraulic head was measured by manometers at three locations: at the inlet, at the perforated pipe, and at the outlet. The outlet was constructed as an overflow giving a fixed hydraulic head.

Acetate and oxygen were used as electron donor and acceptor. The background water consisted of sterilized distilled water containing an oxygen concentration of 0.3 mM O_2. The solution injected through the perforated pipe consisted of sterilized distilled water with an acetate concentration of 1.1 mM CH_3COOH and an oxygen concentration equivalent to that in the background water.
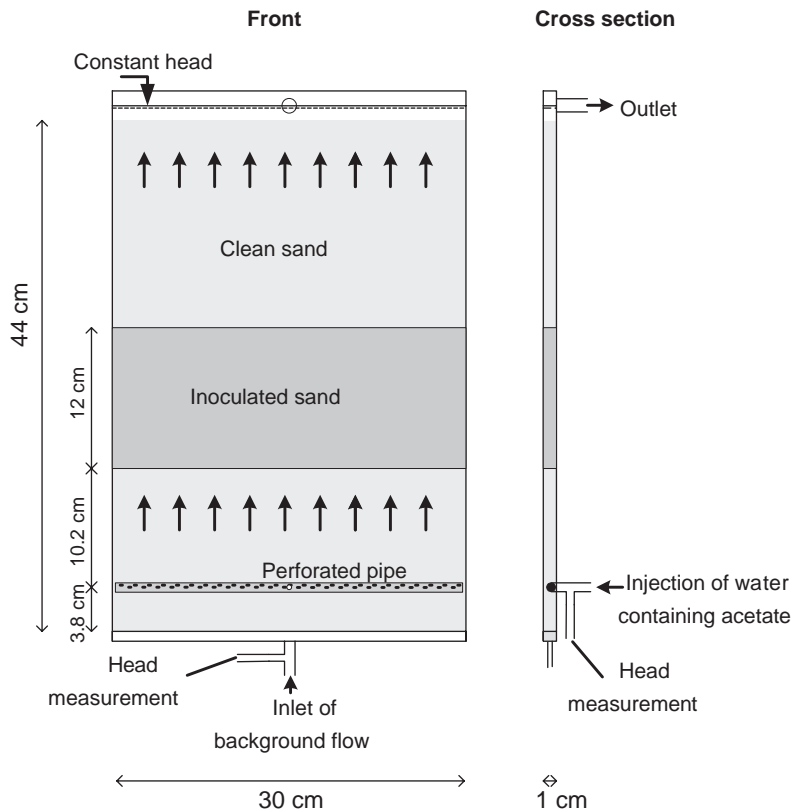
Figure 1.
Sand box
experiment

The experiments ran for 25–30 days with a continuous supply of substrate. The hydraulic head was measured several times a day. A dye tracer (Brilliant Blue) was introduced at the inlet of the sandbox with 1 to 3 days interval. An image of the sandbox was taken by a digital camera every third minute during a tracer test. The digital imaging was used to record the development of the flow pattern.

Numerical model

The model by Kildsgaard and Engesgaard (2002) was extended to account for motility and chemotaxis of bacteria. The governing transport equation for aqueous microorganisms (X_a) is:

$$\frac{\partial X_a}{\partial t} = \frac{\partial}{\partial x_i}\left((D_{ij} + \mu_{eff})\frac{\partial X_a}{\partial x_j}\right) - \frac{\partial}{\partial x_i}\left(f \cdot v_{p,i} + V_{c,eff,i})X_a\right) + \frac{\partial X_a}{\partial t}\bigg|_{mic} \qquad (1)$$

where D_{ij} is the dispersion coefficient, and $v_{p,i}$ is the pore water velocity. The last term on the right-hand-side accounts for microbiological processes except motility and chemotaxis and represents Monod growth kinetics, first-order attachment and detachment, and first-order decay of microorganisms, see Seifert and Engesgaard (2005a) for details.

Random motility and chemotaxis of aqueous bacteria are assumed to occur simultaneously. Random motility can be described as a diffusion term, Eq. (1), where μ_{eff} is the effective value of the random motility coefficient in porous media. The chemotactic velocity can be described as an advective flow term driven by a chemical gradient instead of a hydraulic gradient (Corapcioglu and Haridas, 1984; Barton and Ford, 1997). $V_{c,eff,i}$ is the effective chemotactic

velocity in porous media, Eq. (1), which is a function of the attractant gradient and can be written as (Barton and Ford, 1997):

$$V_{c,eff,i} = \chi_{0,eff} \frac{K_d}{(K_d + C_s)^2} \frac{\partial C_s}{\partial x_i}$$

(2)

where $\chi_{0,eff}$ is the effective chemotactic sensitivity coefficient, K_d is the equilibrium dissociation constant for bacterial receptor-attractant binding, and C_s is the concentration of the attractant, in this case acetate. The chemotactic sensitivity coefficient describes how responsive the specific microorganisms are to the substrate, and the dissociation constant is a measure of the optimal substrate concentration for the bacterial receptors.

The pore water velocity in Eq. (1), $v_{p,i}$, is an average velocity over a continuum of pores. By using the average pore water velocity it is assumed that the microorganisms are distributed uniformly in the pores. The majority of the microorganisms, though, will probably not 'swim' in the middle of a pore against the highest velocity if they have the goal to migrate toward more favorable environments with higher substrate concentrations. Crawling in the immobile pores or near the sand grain surfaces, where the pore water velocity is lowest, the microorganisms would sense the lowest resistance to their motion. Micromodel experiments by Dupin and McCarty (2000) confirm this assumption. Motile microorganisms were observed to migrate slowly upgradient by swimming close to the pore walls, whereas the microorganisms in the center of the pores were flushed downgradient with the flow. The channel size in the micromodel was 20–200 µm, which is in the range of a pore size using sand with grain diameters of 400–800 µm. Microbes with a length of about 1–2 µm occupy 1–10% of the pore throat dimension, and have the possibility to take shelter in the low velocity region close to the pore walls. Thus, it is assumed that the microorganisms are exposed to a reduced pore water velocity corresponding to an average of the velocity at the sand grain surface and in the immobile pores. For simplicity, the pore water velocity affecting the microorganisms will be expressed as $f \cdot v_{p,i}$, with $f \in [0,1]$, Eq. (1). In this study $f = 0.01$, see Seifert and Engesgaard (2005b) for a sensitivity study.

Kildsgaard and Engesgaard (2001) and Seifert and Engesgaard (2005b) provide details about the other solute transport and mass balance equations for the substrate, electron acceptor, and solid biomass and how solid biomass porosity affects the hydraulic conductivity.

RESULTS AND DISCUSSION

Three experiments were carried out (Seifert and Engesgaard, 2005c). Figure 2 shows a few images from selected tracer tests that illustrate the development of flow patterns due to bioclogging. Flow developed from a smooth tracer front to finger-like structures that resemble preferential flow paths. The preferential flow paths became visible after only 3–4 days and developed into permanent structures as bioclogging became more severe. The preferential flow paths can be a result of localized bioclogging where the hydraulic conductivity is significantly reduced in a zone causing very high velocities in the adjacent areas. An initial heterogeneous distribution of the inoculated microorganisms is a likely reason for initiating the development of preferential flow paths.

The amount of tracer visible in the sandbox is observed to increase with time as the porous medium became more affected by bioclogging. After 15 minutes of tracer injection the same amount of tracer took up a larger area at day 24 than at day 0. This can be explained by a porosity reduction in the sandbox due to bioclogging, which made the tracer progress longer than at day 0. The reduction in the bulk hydraulic conductivity (i.e., for the entire sand box) was about 2–5% of the initial value in all experiments and showed almost the same pattern of decrease (data not shown). The hydraulic conductivity in the bioclogged zones, thus, was much lower than observed over the entire length of the sandbox.
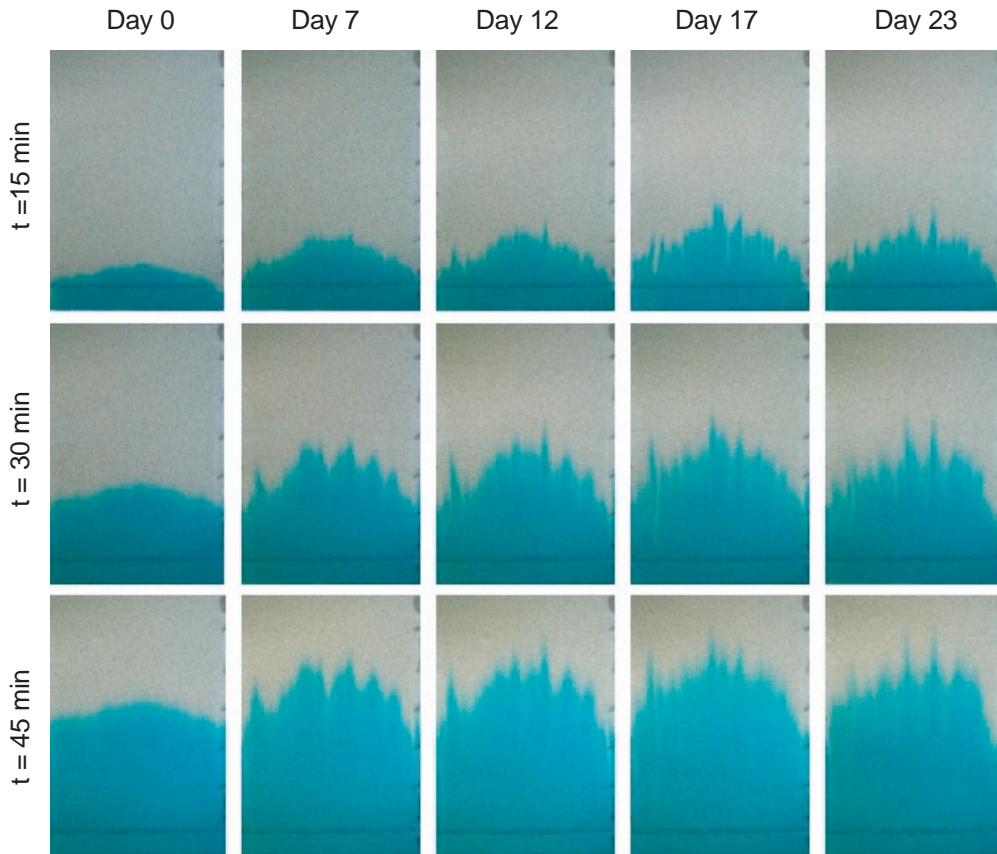
Figure 2. Tracer images (experiment 3)

The visible area with tracer is estimated from the images to quantify porosity reduction. The images were first converted to grayscale, the background removed, the area of the visible tracer estimated, and, finally, compared with the initial area at day 0. The development of the tracer areas for experiments 2 and 3 is similar, Figure 3, with a linear decrease to 60–70% of the initial porosity. This is a significant decrease, as it is a bulk value for the part of the sandbox with tracer, covering both unaffected and bioclogged zones. An alternative explanation to the increase in
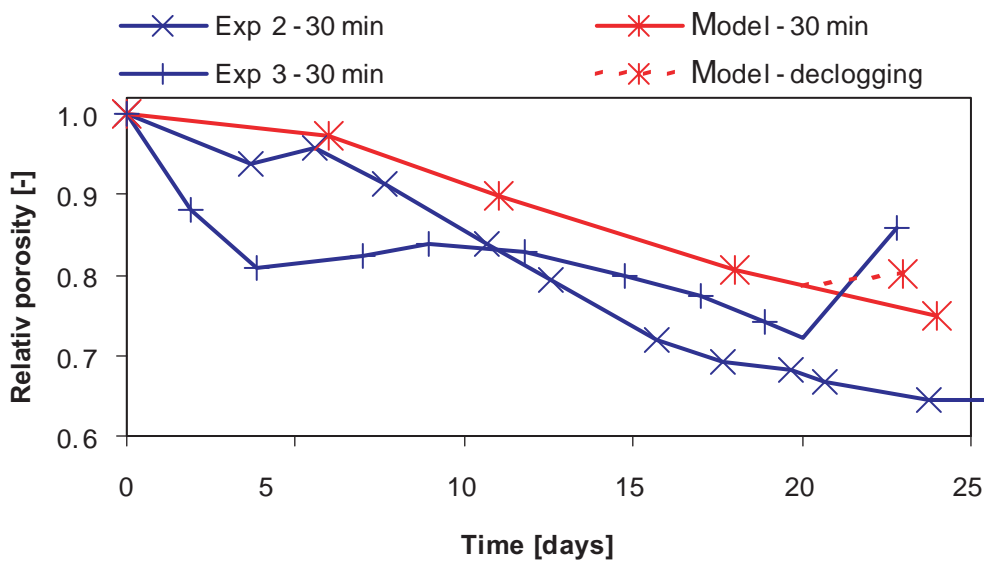
Figure 3. Simulated bulk porosity reduction calculated by the tracer area after 30 minutes of tracer injection with the calculated reduction in porosity from experiment 2 and 3

tracer area with time is that the tracer was affected by three-dimensional flow effects due to the presence of the pipe itself and clogging just downgradient the pipe (Seifert and Engesgaard, 2005b). This may also partly explain why the model is only able to capture some of the reduction in porosity.

The simulated biomass was found to migrate the 0.1 m from the initial inoculated strip of sand to the pipe within 24 days, Figure 4. Notice that we have assumed an initial heterogeneous distribution of the biomass. Assuming a homogeneous distribution would not predict the fingering shown in Figure 2.
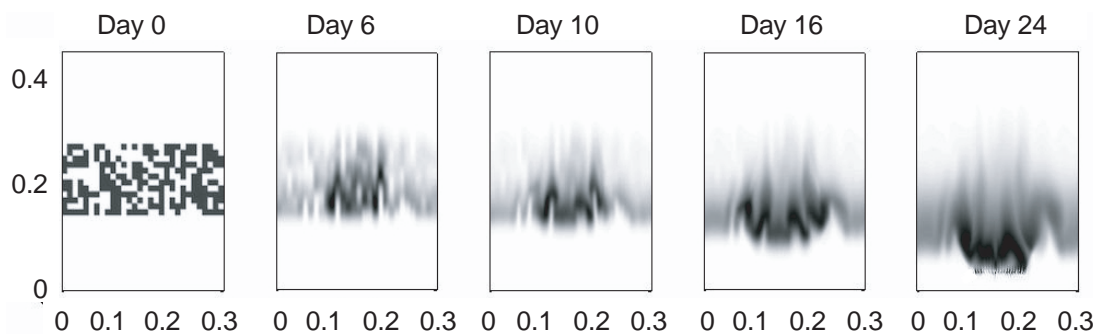
Figure 4 Simulated biomass distributions at different days.
The initial distribution of biomass is assumed heterogeneous.

CONCLUSIONS

Motility and chemotaxis may play a very important role in bioclogging of porous media. Our experimental and numerical results demonstrate that the microorganisms have the ability to transport themselves against the flow with an effective velocity of about 10 cm per 24 days. This, in combination with a heterogeneous distribution of the bacteria at the beginning of the experiment, results in a change in flow toward what resembles preferential flow. The numerical model is able to capture the main features of the observed bioclogging and the reductions in porosity (and hydraulic conductivity, data not shown). A critical assumption in the model is that the microorganisms are able to transport themselves against the flow near the surface of the sand grains or in stagnant zones.

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Behaviour of trace metals and metalloids during Aquifer Storage and Recovery (ASR) of reclaimed water in a carbonate aquifer at Bolivar, South Australia

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Abstract

Aquifer Storage and Recovery (ASR) is used to balance the continual supply of irrigation quality reclaimed water with the seasonal demand for supply. However, the effects of hydrogeochemical processes during subsurface injection and storage on the recovered water quality must also be considered. A full-scale ASR field trial using reclaimed water at Bolivar, South Australia was used to investigate the behaviour of trace species; arsenic, nickel, aluminium and zinc, in a carbonate aquifer. Concentration changes may be due to mobilisation from the aquifer, or conversely, attenuation processes. The injected aluminium (8 µmol/L) was predominantly in the particulate phase while the remaining species of interest were largely soluble. The soluble aluminium (2 µmol/L) and nickel (0.4 µmol/L) concentrations injected were generally greater than in the ambient groundwater (0.5 and 0.01 µmol/L respectively), while the injected arsenic (0.03 µmol/L) and zinc concentrations (1.1 µmol/L) were similar to the ambient condition. Overall, the recovered water quality indicated that particulate aluminium was removed while soluble aluminium concentrations remained the same as injected. Nickel was marginally removed, while zinc attenuation illustrated the removal potential of the carbonate aquifer. Oxidation of pyrite resulted in arsenic mobilisation (average increase of 0.2 µmol/L), but did not limit the suitability of the recovered water for irrigation.

Keywords

Aluminium, arsenic, attenuation, metal mobilisation, nickel, zinc.

INTRODUCTION

Utilisation of reclaimed water in Management of Aquifer Recharge (MAR) takes advantage of an under-utilised resource to augment groundwater resources, while also reducing the discharge of nutrient rich waste water to surface or coastal waters. However the effect of saturated zone geochemical processes on the recovered water quality must be considered. As trace metal and metalloid species can be present within a reclaimed water source, such as waste water or stormwater, subsurface storage has the potential to improve the recovered water quality. The fate of injected species will depend on their presence in particulate or dissolved form and the hydrogeochemistry of the aquifer. In addition, the spatial and temporal variability in redox conditions within the injectant plume resulting from the injection of reclaimed water (Vanderzalm, 2004) influences the speciation and mobility of trace species. Oxygenating a reduced environment can mobilise trace species from the aquifer. This is commonly experienced when pyrite is oxidised, releasing iron, sulphate and a suite of trace metals present as impurities within the pyrite, including arsenic, nickel and zinc (Stuyfzand, 1998; Arthur et al., 2003). Once released, the mobility of these species again depends on the chemical environment within the aquifer, which can also result in subsequent removal prior to recovery. The mobility of metals and metalloids can also be enhanced by formation of soluble complexes with injected organic matter (Banwart et al., 1989) and preferential adsorption of injected ions (Appelo et al., 2002). The fate of the trace species and the implications for the recovered water quality has not been previously

documented for ASR schemes using a non potable injectant where varying redox states are likely to exist. In response, this paper discusses the behaviour of a suite of trace species, aluminium, arsenic, nickel and zinc, when reclaimed water was injected into a carbonate aquifer. Growing awareness in this area has also initiated an assessment of stormwater based species in silicious aquifers (Wendelborn et al., 2005).

METHODS

A full-scale Aquifer Storage and Recovery (ASR) cycle at Bolivar, South Australia between October 1999 and November 2001 (injected 2.5×10^5 m^3 and recovered 1.5×10^5 m^3) investigated the behaviour of trace metal and metalloid species in a carbonate aquifer. The ASR field site is situated adjacent to the Bolivar Water Reclamation Plant, the supply of reclaimed water for injection, and in close proximity to the Virginia horticultural region where the reclaimed water is used as a seasonal irrigation source (Figure 1). The reclaimed water had undergone secondary treatment via trickling filters and waste water stabilisation lagoons, followed by dissolved air flotation/filtration (DAFF) treatment and chlorination. Typical organic carbon, total nitrogen and total phosphorus concentrations in the injectant were 1.5 ± 0.2 mmol/L, 1.3 ± 0.7 mmol/L and 0.02 ± 0.02 respectively.

Figure 1.
Location of the Bolivar ASR site,
indicating the proximity
to the Bolivar wastewater
treatment plant
and the horticultural area,
where reclaimed water
is used for irrigation

The ASR trial targeted the lower Tertiary sediments (T2) of the Port Willunga Formation, an approximately 60 m thick calcarenite aquifer. The mean salinity in the T2 aquifer of approximately 2,100 mg/L at the Bolivar ASR field trial site renders it unsuitable for irrigation. The ASR well was completed and open over most of the T2 aquifer thickness, approximately 100 to 160 m below ground surface (bgs). The mineralogy is dominated by calcite and quartz, with minor contributions from ankerite, hematite, microcline and albite. This work reports hydrogeochemical data from the ASR well, a fully penetrating observation well 4 m from the ASR well and three piezometers 50 m from the ASR well, which intersect the layers of high permeability and injectant breakthrough occurred, 50 m s-1 (104 to 109 m), 50 m s-3 (134 to 139 m) and 50 m n-3 (134 to 139 m). Horizontal hydraulic conductivity varies over almost three orders of magnitude, ranging from 10^{-1} to 10^2 m/day when measured on cores samples over a 10 cm scale (Wright et al., 2002). The average hydraulic conductivity derived from a pumping test over the whole aquifer thickness is 3 m/day (Martin et al., 1998).

All water samples were taken after stabilisation of pH, temperature, DO, EC and Eh, in accordance with appropriate well purging (approximately 3 well volumes). These physical parameters were measured *in situ* using a TPS-FL90® probe in a flow-through cell, directly from the pump discharge to prevent atmospheric exposure. Eh values were reported relative to the standard hydrogen electrode. Sampling, preservation and analysis methodology was based on the Standard Methods for the Examination of Water and Wastewater (APHA, 1998), with samples maintained below 4°C and transported to the Australian Water Quality Centre (AWQC), Bolivar, South Australia for analysis within 24 hours. Metal samples were acidified immediately after sampling with nitric acid to below pH 2. Dissolved metal samples were filtered through a 0.45 µm filter immediately after sampling and prior to acidisation. Arsenic was analysed by continuous hydride atomic emission spectroscopy with a detection limit of 1 µg/L. Emission spectroscopy was used for analysis of aluminium, chromium and zinc to a detection limit of 5 µg/L, for nickel to a detection limit of 1 µg/L and for cadmium and lead to a detection limit of 0.5 µg/L.

RESULTS AND DISCUSSION

Within the ambient groundwater and the reclaimed water injectant, aluminium was predominantly in the particulate phase, while arsenic, nickel and zinc were principally soluble (Table 1). Well redevelopment is undertaken in ASR schemes to manage physical clogging and removed approximately 60% of the particles injected at Bolivar (Dillon et al., 2003). Much of this particulate material was organic carbon, which was physically retained near the point of injection and is capable of scavenging metals from solution by sorption. Metal cations are known to be adsorbed by iron oxides (such as goethite and hematite), organic matter and aluminium hydroxides. In a competitive environment, zinc and nickel were shown to have less potential for removal than other heavy metals such as chromium, lead and copper. Also the order of preference for adsorption was generally $Zn > Ni$, except for soils with higher cation exchange capacity (Gomes et al., 2001). Hence it can be expected that injected metals may also be concentrated near the injection well and the proportion removed by redevelopment will exceed the proportion of injected particles. While the removal of trace species was not quantified during redevelopment events, this was shown to be the case for iron (Dillon et al., 2003).

Table 1. Mean water quality comparison of the ambient groundwater in the T2 aquifer,
the reclaimed water injectant and the water recovered from the Bolivar ASR field trial

Parameter (μmol/L unless otherwise stated)	Ambient [22]	Injectant [24] †	Recovered water [21] †
Temp (°C)	26	20	22
pH	7.3	7.1	7.1
EC (mS/cm)	3.5	2.3	2.3
Eh (mV SHE)	50	250	30
Al-total	20 [8] 8	1.6 [10]	
Al-soluble	0.5 [8]	2	1.4 [10]
As-total	0.07	0.03	0.19
As-soluble	0.05	0.03	0.19
Ni-total *	0.010	0.4	0.32 [10]
Zn-total	1.5	1.1	0.10 [10]
Zn-soluble	1.0	0.9	0.10 [10]

Notes:
[n]   number of samples
*   all in soluble form
†   flow-weighted average

The total quantity (in moles) of each metal in the recovered water can be compared to the quantity expected to be recovered in the absence of reactions, i.e. conservative behaviour in the aquifer (Table 2). Chloride was used as a conservative species to calculate the proportions of injectant and ambient groundwater in each sample of recovered water and thereby determine the overall mass balance for the trace species. Removal of particulate aluminium introduced with the injectant can be explained by physical filtration soon after injection followed by permanent removal from the aquifer during redevelopment of the ASR well. As a result, 95% of the particulate aluminium was removed during the ASR cycle and soluble aluminium became the dominant fraction within the recovered water. Soluble aluminium was introduced into the carbonate aquifer at higher concentrations than the ambient condition. At the observed pH and Eh conditions throughout the ASR cycle, the dominant speciation was within the soluble hydroxy complex $Al(OH)_4^-$, which did not undergo any net removal through precipitation.

Table 2. Ratio of quantity of trace species recovered to quantity expected if behaviour in the aquifer was conservative (measured/conservative=1 in absence of reactions). Chloride was used as a tracer of conservative behaviour and to calculate the proportions of injectant and ambient groundwater within samples of the recovered water.

Parameter	Conservative total recovered (mol)	Actual total recovered (mol)	Measured/ Conservative
Al-total	1,100	260	0.2
Al-soluble	220	220	1
As-total	4.9	30	6
Ni-total [*]	59	49	0.8
Zn-total	160	15	0.1

Nickel was also injected at higher concentrations than the ambient condition, while arsenic and zinc concentrations were comparable in the injectant and the native groundwater. Through the ASR cycle soluble aluminium remained unchanged, arsenic was gained, the removal of nickel was marginal, and zinc removal was quite efficient.

Cadmium, chromium and lead concentrations were monitored less frequently through the ASR cycle than the other species, but remained below their analytical detection limits and therefore their behaviour does not warrant further consideration here.

To understand the mechanisms responsible for attenuation or mobilisation it is necessary to outline the hydrogeo-chemistry within the injectant plume. The varying redox status within the injectant plume, both temporally and spatially throughout the ASR cycle (Vanderzalm, 2004), influenced the solubility of the trace species. Immediately upon commencement of injection, an oxic zone was evident around the injection well. This was short-lived, with conditions rapidly progressing to denitrifying, without any evidence to suggest more reducing conditions such as manganese or iron mobilisation or sulphate reduction. In general terms, the redox state within the injectant plume was comparable to the condition within the aquifer prior to injection. The buffering capacity of the carbonate aquifer maintained a relatively constant pH within the injectant plume near 7.1. It was only during storage that more reducing and acidic conditions (minimum pH of 6.5) were evident directly in the vicinity of the ASR well, where the microbial activity was highest. Here both sulphate reduction and methanogenesis were reached. Despite this, groundwater in the bulk of the plume, including at the 4 m radius, remained in the denitrifying state.

Arsenic mobilisation was apparent through the ASR cycle, increasing the arsenic concentration in the recovered water and consequently it was necessary to investigate the dominant mechanisms behind this process. Arsenic was released from the aquifer by two mechanisms and initiated under differing redox conditions. Approximately 0.16 µmol/L (equating to 40 mol in total) of arsenic was released by pyrite oxidation in the near well zone (evident at the 4 m well), under aerobic and denitrifying conditions. With migration of the injectant plume to the 50 m radius, there was no evidence left of the arsenic release previously seen at 4 m (concentrations < 0.04 mmol/L). In this case, additional aquifer passage allowed arsenic to be scavenged from solution, probably by sorption on iron oxide surfaces, an important removal process for arsenate (Stuyfzand, 1998). Thus in a dual well scheme (Aquifer Storage Transfer and Recovery), arsenic release would not be evident at the recovery well unless the sorptive capacity of the aquifer was exceeded or inhibited by competition from other oxyanions, such as phosphate and carbonate (Appelo et al., 2002). However in this ASR cycle, reversal of flow during the recovery phase resulted in desorption from the aquifer and the arsenic concentration recovered was approximately six times that expected from conservative behaviour. Despite release of approximately of 0.16 µmol/L (12 µg/L) arsenic, the recovered water remained compliant with the Australian long term trigger value in irrigation quality water of 100 µg/L (ANZECC and ARMCANZ, 2000).

The second mechanism for arsenic release had the greatest, but short-lived, effect on the magnitude of the soluble arsenic concentration and was observed under the reducing conditions that developed during storage. While these storage phase changes affected only the water directly around the ASR well and were alleviated with the first flush of recovery, it was of interest to understand how the changes altered the mobility of arsenic, along with nickel and zinc. Arsenic and nickel concentrations were higher in the ASR well than in the rest of the injectant plume as the reductive dissolution of iron oxides released previously sorbed species. The effect was greatest for arsenic with a peak concentration of 2.5 μmol/L, while the nickel peak was lower at 0.8 μmol/L. If we consider the magnitude of the desorbed concentrations to be indicative of the aquifer's capacity to retain individual species, a preference for retention of arsenic was evident. Following on from this, there appeared to be little tendency for nickel adsorption within the carbonate aquifer. There was a trend toward lower nickel concentrations in the 104 to 109 m depth interval than in the 134 to 139 m interval at the 50 m radius. As this shallow layer illustrated a tendency to remove injected ammonium and potassium by cation exchange (Vanderzalm, 2004), the marginal removal of nickel was attributed to cation exchange (Gomes et al., 2001). Cation exchange was limited to the initial pore flushes, which were achieved within a matter of days in the near well zone. Therefore the potential for retention near the ASR well for subsequent removal through the well redevelopment was low.

During the injection phase, the zinc concentration had declined by 30% 4 m from the ASR well, indicating some retention in the near well zone and thus possible removal with particulate matter during redevelopment. However, the greatest zinc removal within the ASR cycle occurred between the 4 m and 50 m radius, as with arsenic. In contrast to arsenic, the removal mechanism was maintained during the recovery phase resulting in efficient removal of zinc through the ASR cycle. Speciation indicated that zinc was present as carbonate species under the neutral conditions maintained within the injectant plume and thus solubility was likely to be controlled by the precipitation of zinc carbonate species. Zinc concentrations in the near well zone showed a net decline during storage, likely to be due to removal by precipitation as zinc sulphide. Thus under reducing conditions sulphide precipitation had a greater influence on zinc concentrations than potential release from iron oxides.

CONCLUSIONS

Reclaimed water Aquifer Storage and Recovery highlighted the variable behaviour that the trace metals and metalloids can display through transitional redox zones. Removal of particulate aluminium was achieved through physical filtration during the injection phase, whereas soluble aluminium concentrations remained unaltered. The potential for attenuation of injected nickel via adsorption onto iron oxides or cation exchange was slight, while zinc attenuation by precipitation of zinc carbonate illustrated the removal capacity of the carbonate aquifer. Arsenic was mobilised upon injection with the subsequent redox conditions controlling the aqueous concentration. Despite an approximate increase in arsenic of 0.16 μmol/L (12 μg/L), the recovered water remained compliant with the Australian long term trigger value in irrigation quality water of 100 μg/L (ANZECC and ARMCANZ, 2000) and thus the beneficial use of the water was not hindered.

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Research on metals in stormwater for aquifer storage and recovery in alluvial aquifers in Melbourne, Australia

Anke Wendelborn, Gavin Mudd, Ana Deletic and Peter Dillon

Abstract

Aquifer storage and recovery (ASR) can provide efficient temporary storage of stormwater for recycling and reduce evaporation losses. The injection of stormwater introduces oxygen, nutrients, suspended solids, organic carbon and other pollutants like heavy metals into an aquifer. This may change the hydrogeochemical equilibrium in the subsurface and influence the behaviour of heavy metals. Stormwater samples from Melbourne had Cr, Cu, Pb and Zn concentrations that exceeded Australian guidelines for freshwater aquatic ecosystems. Of special concern are Zn and Cu as they are found predominantly in the soluble fraction and are attached to the finer suspended solids that are difficult to remove. The overall objective of this recently commenced study is to evaluate the behaviour and fate of heavy metals during ASR. This will require characterisation of metals in stormwater, identifying their association with particulates, correlating their concentrations with surrogate variables, exploring spatial and temporal changes in near-well geochemical composition at a demonstration ASR site, using batch and column tests under laboratory conditions to evaluate factors affecting behaviour of metals and developing models to predict their long-term fate and identify the need for any corrective actions to ensure sustainability of stormwater ASR operations.

Keywords
Stormwater quality; heavy metals; aquifer storage and recovery; geochemical interactions.

INTRODUCTION

Melbourne is a city with growing population, increasing water demand and limited freshwater supply from surrounding catchments. It overlies a range of productive aquifers, but beneficial use of groundwater is often limited by salinity levels (Leonard, 1992). On the other hand the city produces a large volume of stormwater runoff during the winter months from the impervious areas and discards this water into rivers and Port Phillip Bay, impairing its water quality. This situation can be improved and one measure is the recycling of stormwater (DSE, 2004). As most of the rain falls in winter, but is needed mostly in summer the key issue for harnessing it is storage. Aquifer storage and recovery (ASR) seems a potential option to limit surface storage space, evaporation losses and to protect water quality (Pyne, 1995). Stormwater includes a range of pollutants that may affect the groundwater quality and aquifer geochemistry, so a long-term risk assessment regarding groundwater pollution is warranted. Stormwater constituents include oxygen, organic carbon, total suspended solids (TSS) and electron donating substances like nitrate. These may change the geochemical conditions in the aquifer significantly (Stuyfzand et al., 2002) and influence the mobility and speciation of other pollutants like heavy metals.

Heavy metals are often present in urban runoff in concentrations exceeding Australian guidelines for freshwater aquatic ecosystems and are priority pollutants due to their phyto- and cytotoxicity (Makepiece et al., 1995). They undergo various biogeochemical processes during ASR that are mostly reversible. The main attenuation process, sorption, is controlled by the aquifer material, while metal speciation, metal concentration and binding forces are influenced by pH, Eh, ionic strength, co-sorbents etc. (Appelo and Postma, 1999; Bradl, 2004). Nevertheless ASR schemes are usually based on technical feasibility, meaning they are implemented in transmissive aquifers with

limited clay content and therefore limited sorptive capacity. In addition previous ASR studies showed that geogenic heavy metals have been released due to the induced changes of the redox state, with arsenic often being of special concern due to limited adsorption (Stuyfzand, 1998).

In one stormwater ASR study (Herczeg et al., 2004) conducted in South Australia, mass balances were determined for particulate matter. Although the mass of particulates recovered was similar to the mass injected, the recovered mass was largely composed of aquifer material (Dillon and Pavelic, 1996). A major proportion of the metals injected are retained within the aquifer. Metals of stormwater origin may potentially be remobilised if exposed to further redox or pH changes, as with geogenic metals. Most redox reactions produce protons (Appelo and Postma, 1999) and depending on the buffering capacity of the aquifer this might reduce pH, and increase the solubility of heavy metals. In a reclaimed water ASR study Vanderzalm (2004) found that dissolved organic carbon (DOC) decreased in recovered water. This suggests that organic complexation of metals within the aquifer may be possible and could further increase solubility of heavy metals. Geochemical reactions that occur when recharged water interacts with the ambient groundwater and aquifer matrix are expected to be site-specific. Such information for Melbourne aquifer systems is unknown and aquifer geochemical properties are poorly defined. This paper presents the first part of our study focusing on the hydrogeology of Melbourne and specific aspects of stormwater quality in Melbourne relating to metals, while outlining the future research programme.

METHOD

A literature review was conducted on hydrogeology of Melbourne, mostly from reports of the Geological Survey of Victoria. Relevant data were also gathered from the Victorian Geoscientific Database (Department of Primary Industries), Groundwater Database (SKM), the Victorian Water Resources Data Warehouse and from personal communication with senior hydrogeologists.

Table 1. Stormwater monitoring sites in Melbourne

Site	Primary land use	Area (m²)	Impervious fraction
Monash Roof (MR)	coated aluminium roof	50	1.00
Gilby Rd (GR)	commercial	28,200	0.80
Shepherds Rd (SR)	medium density residential	37,980	0.45
Richmond (RI)	high density residential	89,120	0.74

As not many comprehensive data sets for Australian urban stormwater runoff have been collected so far, the Institute for Sustainable Water Resources in conjunction with the Co-operative Research Centre (CRC) for Catchment Hydrology has been monitoring stormwater quality and quantity around Melbourne for different catchment sizes and land uses. So far only flow hydrographs, TSS, total nitrogen and total phosphorus for rainfall events have been measured. The automated samplers are triggered via increased flow and subsequent samples are taken at set times after the first trigger with high sampling frequency in the rising limb of the hydrograph and decreased sampling in the hydrograph tail. According to the hydrograph volumes composite samples for event mean concentration (EMC) measurements are made. Of these, four sites (Table 1) have been chosen for more detailed water quality measurement to enable characterization of Melbourne's stormwater for the ASR assessment. Since December 2004 incoming samples are now analysed for pH, electrical conductivity (EC) and alkalinity (Gran-plot). Composite samples are analysed for EMCs of major ions (Na, K, Mg, Ca with flame spectroscopy and Cl, SO_4 with flow injection analysis) and total/dissolved organic carbon (TOC/DOC with TOC analyser). EMCs of heavy metal

(Al, As, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn) are determined for the soluble fraction (< 0.45 µm), acid-extracted size fraction 0.45–63 µm and acid-extracted size fraction > 63 µm measured with ICP-OES. To date 11 samples have been analysed and sampling is ongoing.

RESULTS AND DISCUSSION

Review of hydrogeology of Melbourne

The site selection for an ASR scheme depends on hydrogeological characteristics, availability of land to harvest the stormwater, and a localised demand. Three broad classes of aquifer system occur; confined sedimentary aquifers, fractured rock systems and unconfined alluvium. The implementation of ASR would be technically most feasible in confined sedimentary aquifers with large storage potential and very limited impact on surface features. Fractured rocks often prove to have limited recovery efficiency due to their heterogeneous characteristics. Unconfined aquifers have constraints on recharge volumes, but are cheaper to construct. The area of most interest is the central section of the sandy unconsolidated (semi-)confined Fyansford Formation-Brighton Group aquifer system with brackish groundwater in the southeast of Melbourne. The Tertiary Fyansford Formation is made up of marine coarse to fine grained sand, gravel and discontinuous sandy limestone layers about 30–50 m below the surface (Leonard, 1992). The total formation is up to 60 m thick that thins towards the north. The transmissivity of 20–30 m^2/d makes it suitable for small scale ASR schemes (AGT, 2002). Salinity levels vary greatly (~100–6,000 mg/L). Areas with salinity above 1,500 mg/L are targeted for ASR as this reduces potential impacts on beneficial use of the groundwater. The relatively low hydraulic gradient (~0.001) in the central section of the aquifer system is favourable for ASR schemes as this increases the recovery efficiency in brackish aquifers.

Stormwater quality

Table 2 shows a wide range of values for each parameter between events and between sites. This is due to the fact that many parameters e.g. catchment characteristics, antecedent dry-weather period, rainfall volume and intensity

Table 2. Selected stormwater data sets of sampled stormwater events

	Total rainfall (mm)	Peak rainfall intensity (mm/h)*	Peak flow rate (L/s)	Duration (h)	ADWP (days)	pH	EC (µS/cm)	TOC/ DOC (mg/L)	TSS (mg/L)	Cu (total/ < 0.45µm) (µg/L)	Zn (total/ < 0.45µm) (µg/L)
d.l.							1	1	0.5	0.79	3.00
	2	2	2.3	0.2	2	5.6	37	3/3	11	9.2/3.8	101/84.5
MR	3.4	1.8	1.5	1	11	5.7	29	5/4	91	27.1/4.7	343/267
	4.8	0.6	0.5	3	1	5.3	17	2/2	8	3.6/3.2	127/124
	2.6	1.4	340	2	11	6.5	95	17/9	138	84.5/19.4	1,456/849
GR	3.6	0.6	160	5	1	6.4	58	9/6	17	17.5/12.4	738/695
	4.8	1.2	450	2	5	6.9	54	6/3	54	10.7/4.7	335/221
SR	6.2	3.4	490	3	10	6.5	96	17/9	432	74.3/6.6	897/142
	144.4	4.4	470	20	4	6.7	74	6/4	220	11.2/4.5	175/136
	9.4	8.4	2,850	2	1	6.6	52	9/4	318	89.6/5.0	1,826/83.8
RI	3	1.4	185	3	12	6.4	128	12/10	196	57.3/9.7	1,395/676
	4.2	0.8	190	4	1	6.5	114	20/10	167	63.9/13.9	1,664/744

ADWP: antecedent dry weather period; d.l. detection limit; *over 10 min interval.

have a bearing on them (Wong et al., 2003). The urban stormwater pollutant concentration ranges are comparable to those found in the literature (Table 3). The samples exceed the Australian guidelines for freshwater aquatic ecosystems and for drinking water

Table 3: Comparison of guideline values and stormwater concentrations

Parameter		Freshwater aquatic system[1]	Australian Drinking water guideline[2]	Melbourne stormwater (this study)	Mean storm-water [3]	Mean storm-water [4]
TSS	mg/L	20	10	8–432	164	155
TOC		15		2–20	23.6	32
Al (soluble)		55	200	153–634	900	
As (total)		24 [AsIII]/ 13[AsV]	7	< 1.4–5.3	2	
Cd (total)		0.2	2	< 1–1.4	2	8
Cr (total)				0.3–25	70	33
Cr (VI)		1	0.05			
Cu (total)	μg/L	1.4	2,000	3.6–90	80	62
Fe (total)			300	390–13,040	4,000	4,500
Pb (total)		3.4	10	1.8–243	250	140
Mn (total)		1,900	500	10.5–198		
Ni (total)		11	20	< 2 - 23.5		32
Zn (total)		8	3,000	101–1,826	910	320

Notes:
1 ANZECC and ARMCANZ, 2000: level of protection 95%;
2 NHMRC, 2004;
3 Dillon and Pavelic, 1996 (mostly Australian data);
4 Duncan, 1999 (worldwide data)

(ANZECC and ARMCANZ, 2000) for most parameters, but for the ASR scheme the environmental values (beneficial uses) of the ambient groundwater and recovered water will define the concentrations of metals and other constituents that are acceptable. These stormwater samples have lower concentrations of arsenic than have been found in Port Phillip Bay (Fabris et al., 1999). The latter data suggests that As cannot be ruled out as an issue. If pyritic material was to be present in anaerobic aquifers, sulfide oxidation caused by injection of oxygenated stormwater could lead to As mobilization (Vanderzalm et al., 2005). It can be seen that all concentrations are significantly reduced in events with a very short ADWP (Table 2), as the pollutant load is a function of build-up and wash-off. It can also be concluded that the roof catchment has generally lower pollutant concentrations compared to the other catchments. TSS concentrations are of great importance as many other toxic pollutants are associated with them, especially with the fine fraction (Liebens, 2001; Muthukaruppan et al., 2002), and is often used as an indicator of runoff pollution in general. Inverse relationships between the dissolved fraction of Zn (R^2 = 0.77) and Cu (R^2 = 0.54) with TSS could be established in agreement with findings from Sansalone & Buchberger (1997).

pH is a very important parameter influencing metal solution and soil-surface chemistry as well as complexation (Bradl, 2004). Metal solubility is also determined by the most abundant anion in the water, which can inhibit adsorption (Fic and Isenbeck-Schröter, 1989). Even though significant relationships could not be established due to the limited number of samples, nevertheless an inverse trend with salinity and pH with the dissolved fraction is apparent for most samples. The significantly lower pH and lower salinity of the roof catchment leads visibly to an

increased fraction of dissolved heavy metals (Fig. 1). It was found that zinc has the highest soluble fractions with up to 98 % dissolved. Due to its high pH for the adsorption edge this was found in other studies as well (Zobrist et al., 2000). The high solubility of copper is often due to its formation of organic complexes (Van Dijk, 1971). Lead was mostly associated with the particulate matter (data not shown) in accordance with other studies (Harrison and Wilson, 1985).

With regard to stormwater ASR this means that it is likely that most of the heavy metals in dissolved form and attached to the small size fraction will be introduced to the aquifer. Even deposited solids in detention ponds cannot be regarded as permanent pollution sinks, as the majority of contaminants that are particulate-associated can be transferred to the water phase especially if the physico-chemical conditions change (Pitt, 1996). The attenuation of heavy metals is a combination of different processes which are conservative and mostly reversible especially if there are significant changes in redox and pH conditions during the ASR cycle. They may accumulate adsorbed in the aquifer, be transported away as organic complexes or adsorbed to colloidal matter or be pumped back out in particulate or soluble form.
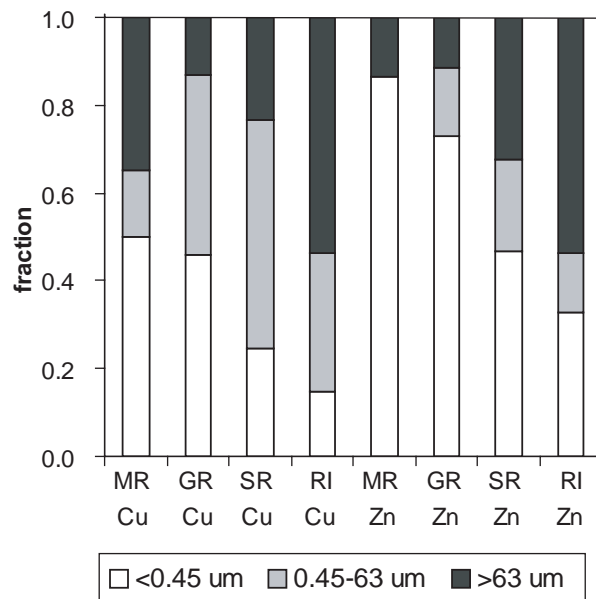
Figure 1. Copper and Zinc mean fractionation at the different sites

CONCLUSIONS AND FUTURE RESEARCH

From the limited set of stormwater samples analysed so far it can already be seen that stormwater quality is highly variable in time and space. This great variability poses the main challenge regarding characterisation. It was shown that heavy metals like Zn, Pb, Cu, Cr, and Ni are rather abundant in Melbourne's urban stormwater in particulate and dissolved form. Although behaviour of heavy metals at stormwater ASR sites has not been an issue under geochemical conditions experienced in Adelaide's carbonate aquifers, a precautionary approach suggests this should also be evaluated under the different geochemical conditions prevailing in Melbourne, to assess any potential for long-term variations in the quality of groundwater and recovered water, and allow management strategies to be devised if any problems are predicted.

Aquifers are important assets in a holistic urban water cycle and impacts that would impair their role must be prevented. Therefore an understanding and assessment of possible risks is warranted as a precautionary measure. Consequently the proposed activities within this research project are to:

- identify concentrations and associations with particulates of heavy metals in Melbourne stormwater;
- identify mineralogy and geochemical condition of an aquifer in Melbourne prior to ASR at a demonstration project site;
- identify mechanisms of attenuation and remobilization of these metals during ASR, making use of column and batch studies under a specified range of laboratory conditions with collected aquifer material and groundwater and using synthetic stormwater;
- quantify mass fluxes of metals into and out of the well and measure water quality in monitoring wells to determine fate of metals, and the geochemical changes that may affect metals within the ASR system;
- demonstrate the relevance of the laboratory experiments to field observations and develop a conceptual model that characterises the observed short-term behaviour of metals;

- apply an existing coupled flow and transport and geochemical model (e.g. PHT3D, Prommer et al., 2003) to simulate the observed parameters and subsequently to predict the long-term geochemical behaviour of metals at stormwater ASR sites for various scenarios, with a view to establishing any factors that may affect sustainability of operations.

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TOPIC 3

Modelling aspects and groundwater hydraulics

Unsaturated zone processes

Flow, transport and reaction models

Obtaining reliable aquifer and well performance hydraulic parameter values in a double porosity aquifer: examples from artificial recharge trials in South London

Malcolm Anderson, Michael Jones, Scott Lewis and Keith Baxter

Abstract

Double porosity aquifers present both advantages and pitfalls to artificial recharge design. Perhaps the biggest pitfall comes from the inappropriate but common use of conventional homogeneous radial drawdown models to determine hydraulic parameters from pumping test data. These models account for porous flow only; and in South London can lead to miscalculation of aquifer storage coefficients that are magnitudes greater than true values, and overestimates of the total recharge capacity in error by a similar margin. The double porosity hydraulics of the Chalk are complex but in the context of artificial recharge operations proved to be beneficial. Fracture flow was found to limit well clogging whilst sustaining high well yields (10 to 20 Ml/d per well) that significantly reduce the costs of artificial recharge per Ml/d supplied. Testing also showed that recharge injection rates could be reliably calculated from the hydraulic results from abstraction testing. Abstraction tests are easier and cheaper to perform whilst the locations of testing are not limited by the availability of appropriate water supplies. Case study examples from recent artificial recharge investigations in South London are presented that illustrate double porosity hydraulics and the methods used to effectively analyse drawdown and draw-up responses.

Keywords

Artificial recharge; chalk; dual porosity; hydraulic analysis; London Basin.

INTRODUCTION

The potential for artificial recharge in the southern area of the London Basin is currently being evaluated by Thames Water as part of the South London Artificial Recharge Scheme (SLARS) investigation, (Jones et al., in press). The Chalk, the primary aquifer in the SLARS area, transmits groundwater flow both through connected pore spaces and through a well developed fracture network, and is a classic double porosity aquifer. This paper provides examples from the SLARS investigations at Streatham, Bell Green and Ladywell Fields, Catford that show how aquifer hydraulic parameter values were obtained for the Chalk, and comments on the consequences to artificial recharge design of ignoring double porosity hydraulics. The structure, fracture pattern and fracture density of the Chalk in the SLARS study area reflects the differential settlement of the Jurassic formations below the Chalk, (Andrews et al.. 1995). Interpretation of the substantial number of borehole lithology logs available from the SLARS area, (Andrews et al., 1995), show that faulting, and therefore the fractures and joints, occur primarily in two orthogonal vertical planes. Enhanced jointing is likely in the more competent beds and may be substantially developed by borehole acidisation, producing preferential flow in the third horizontal plane. Thus flow to the borehole approximates to the idealised 3D orthogonal fracture system of Warren and Root (1963). Effectively the Chalk consists of two hydraulic components: a) an orthogonal fracture network, and b) porous matrix blocks. Each porous matrix block is bounded by individual planar fractures of the fracture network, whilst within-block fracturing is poorly developed. Although it can be argued that this conceptual model is idealised, in the SLARS area experience shows the double porosity model is an effective approximation. Firstly, pumping test data provide consistent results once double porosity issues are taken into account. Secondly, Thames Water has found that systematic well development by

injection of acids is successful in the SLARS area, as it is elsewhere in the Chalk. The effectiveness of acidisation is significant as it works by enlarging the local fracture system so that the well becomes better connected to the macro scale fracture network, (Banks et al., 1993).

METHODS

Although more theoretically robust techniques have been developed by Bourdet and Gringarten (1980) and Moench (1984), the straight-line methods of Warren and Root (1963) and Kazemi et al. (1969) are easier to use, yield definitive curve matches, and lend themselves to explanation of double porosity responses. All of the methods used in this paper are described in detail in Kruseman and de Ridder (1990) and are not reproduced here. However due to the complexity of double porosity hydraulic responses it is necessary to use more than one method in a sequence as follows:

1) *Distance-drawdown analysis*

Pumped boreholes in double porosity aquifers have an effective radius (r_e) that is significantly greater than either the screened or drilled radius. Values of r_e can be determined from the straight-line inflection point on a distance-drawdown graph (Jacob 1947). Since the response of a double porosity aquifer at late pumping times approximates to the response of a homogeneous porous aquifer (Kazemi et al., 1969), the distance-drawdown method of Jacob (1947) can also be used to determine the aquifer transmissivity, by selecting a late pumping (or recharge) time and fitting a straight line to data from observation boreholes located outside r_e.

2) *Time-drawdown analysis*

Fracture networks in dual porosity aquifers have significantly higher transmissivity than the matrix blocks. In contrast most of the available storage is in the pore spaces of the matrix blocks. This results in double porosity aquifers having three distinctive straight-line segments to the time-drawdown response. In the first segment drawdown in the fracture network inside r_e is much faster than storage can be released from the matrix blocks. The second segment is effectively a transitional response between the first and third segment responses. The third segment occurs when the matrix drawdown at each location first becomes fully coupled to the fracture network drawdown and approximates to the porous model drawdown function of Theis (1935). The method of Kazemi et al., (1969) has been used to analyse the three time-drawdown responses.

3) *Recovery response analysis*

Recovery responses in double porosity aquifers also have three segments. Analysis is problematic as no general analytical method is available. Although, the third segment late time response approximates to the porous model recovery response of Theis (1935), recovery is often not measured to completion so that the third segment can not be reliably matched. Furthermore available monitoring wells will often be located inside r_e, data from which produces erroneous results. However recovery data provide a useful validation of results from other methods.

4) *Leaky response analysis*

Since the third segment hydraulic response of a double porosity aquifer approximates to the classical Theis function, any leaky or boundary responses that occur after the third segment begins can be analysed with relevant methods, e.g. Hantush and Jacob, (1955).

DISTANCE-DRAW-UP ANALYSIS

Figure 1 shows data typical of the SLARS area responses taken from a recharge test at Streatham. Distance-draw-up is flat in the vicinity of the well and steep in the outer part of the draw-up cone. The radius at which the transition

occurs is 700 to 1,000 m and may be considered to be the effective radius during recharge of the borehole. The aquifer transmissivity calculated from radial distances greater than re gave values of 500 to 900 m^2/day. Aquifer storage coefficient values should not be calculated using this method as the geometry of the solution in double porosity aquifers is invalid.

RECOVERY ANALYSIS

Figure 2 shows the recovery responses measured in the Streatham abstraction borehole (ABH), observation borehole (OBH) and the former water supply well (WELL). All three recovery responses are identical following a short initial adjustment period. As the ABH, OBH and WELL radii are respectively 0.36, 5.6 and 48.6 m, but the recovery responses are identical, the aquifer can not be a homogeneous porous system. However as all the measurement points are located well inside r_e, this response can be explained using the double porosity model. Essentially, the high transmissivity of the fracture network redistributes groundwater within the effective radius of the fracture network in response to the end of pumping, much faster than the storage inside the effective radius can be replenished by flows from outside. As a result, the redistribution of groundwater levels inside r_e is achieved rapidly and thereafter all the water levels recover together. Consequently superposition of recovery data from a pumping borehole and a single observation borehole can be used to confirm the double porosity status of the aquifer, or conversely, whether the observation borehole is located inside or outside the effective radius.
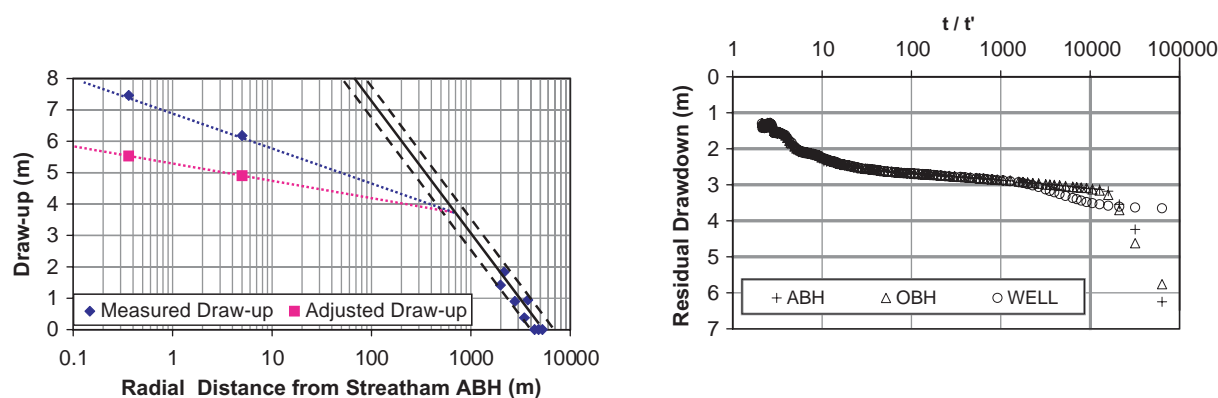
Figures 1 and 2. Distance-draw-up and recovery responses from the Streatham tests

TIME-DRAWDOWN ANALYSIS

Storage coefficient values in double porosity aquifers can only be obtained from time-drawdown analysis. Of the time-drawdown methods, Kazemi et al. (1969) is the easiest method to apply. Care is required in the identification of the correct drawdown segment. In the SLARS area, the first segment takes 1 to 5 minutes to complete and occurs in the pumping borehole and observation boreholes located inside r_e only. Outside r_e only two segments occur in the time-drawdown response so that the second transitional time segment occurs first. The third segment occurs at elapsed times of 1,000 minutes and typically can be reliably identified providing the pumping test is long enough. The results from this segment must then be checked for consistency by comparison with the results from monitoring boreholes located
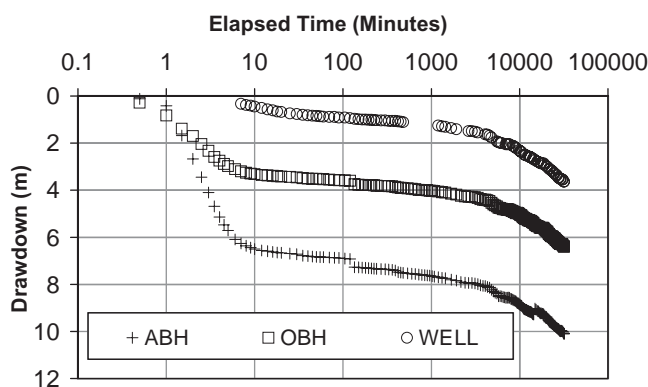
Figure 3. Time-drawdown at streatham

both within and outside r_e. However it should be noted that segments 2 and 3 occur at progressively greater elapsed times as the distance between the monitoring borehole and the pumping well increases. Inside the effective radius, Warren and Root (1963) note that r_e must be used instead of the value r, whilst r_e may be determined from distance drawdown analysis (Figure 3).

DOUBLE POROSITY ANALYSIS OF LEAKY HYDRAULICS

Providing data is taken from the third hydraulic segment from an observation borehole located outside r_e, other conventional leaky or boundary hydraulic analytical methods, e.g. the leaky method of Hantush and Jacob (1955), can also be used to determine aquifer hydraulic parameters. This is feasible because the response of a double poros-

ity aquifer at late pumping times approximates to the response of a homogeneous porous aquifer (Kazemi et al., 1969). Selection of an observation borehole just outside r_e, will maximise the period of data that can be used. The leaky response of the pumping test at Ladywell Fields was therefore best analysed using data from the Catford Town Hall borehole (Figure 4). This figure shows it is difficult to obtain a definitive log-log leaky response fit. However this problem can be overcome if the start time of third segment is determined from a semi-log graph, and only the third segment data values are then fitted on the log-log response. Respectively transmissivity, storage and leakance values of 1,590 m^2/day, 1 x 10^{-3} and 5,500 m were obtained, all of which were consistent with the double porosity analysis.
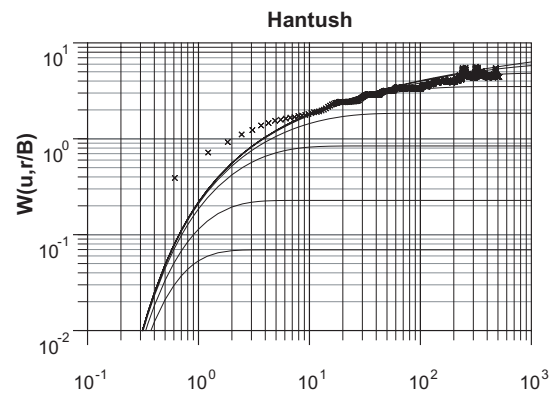
Figure 4. Analysis of the leaky time-drawdown response in the Catford Town Hall OBH

CONTOUR MAP ANALYSIS

Contour maps of drawdown in the study area are another useful method of analysing double porosity aquifers. Fracture systems will often be better developed in the main fracture direction creating marked azimuthal variation in transmissivity. This was a feature of all of the pumping tests carried out in the SLARS area (Streatham, Ladywell Fields and Bell Green). Figure 5 shows the drawdown map for the Streatham test. Here the drawdown is not circular but elliptical, with the long axis showing the orientation of maximum transmissivity. Contour maps from the Bell Green and Ladywell Fields tests produced ellipses with the
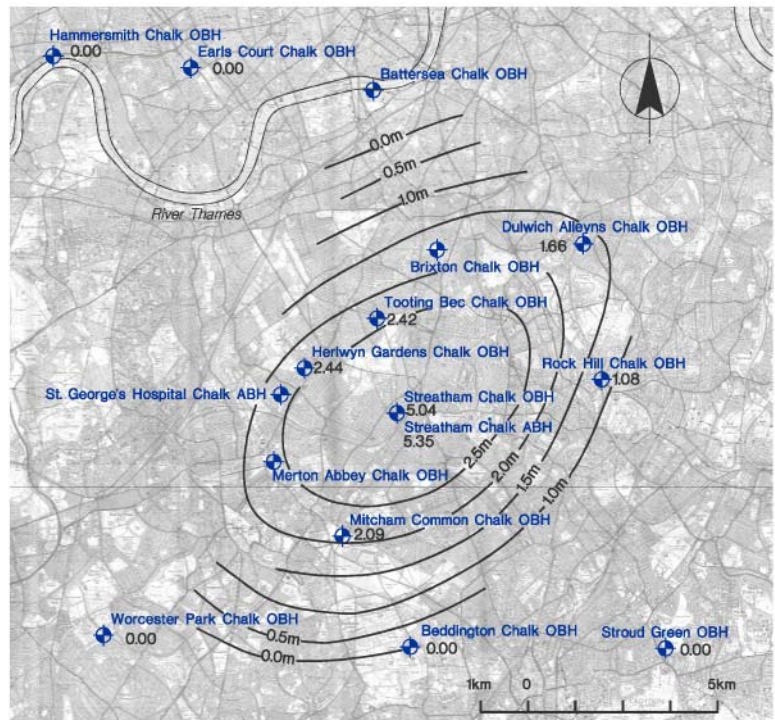
Figure 5. Drawdown contours from the Streatham tests

same orientation as the Streatham test and which also coincide with the strike of the major (Greenwich and Wimbledon) faults and structural features of the SLARS study area. Once the azimuth of maximum transmissivity was identified, much greater consistency in the time-drawdown and distance-drawdown results was identified. Wells located on the azimuth of maximum transmissivity were found to form one cluster of higher transmissivity results whilst those on the perpendicular formed a cluster of lower values.

DISCUSSION

In the Chalk of the SLARS investigation area, double porosity aquifer hydraulics can be reliably identified from: a) the three segments of the drawdown response; b) the very large values of r_e; and c) the identical recovery responses in observation boreholes located inside r_e. In order to quantify the risk of using conventional analytical methods, hydraulic parameter values were determined using double porosity methods from pumping test data available from various technical reports and the results were compared with the hydraulic parameter values contained in the reports determined from conventional porous model analysis. The transmissivity and storage values using conventional analytical methods were found to overestimate the storage coefficient value by up to a factor of 360, and transmissivity by factors of up to 10. Typically the analytical errors occurred where the wrong segment of the drawdown curve was selected, or ignored altogether, such that all drawdown segments were fitted, as appropriate in conventional analysis. Therefore if the SLARS study had applied conventional analysis, it is probable that the storage capacity of any future artificial recharge scheme would have been substantially over-estimated.

Furthermore the much greater hydraulic detail that can be determined from double porosity analysis can be used to great effect in the interpretation of the significance of clogging processes. In our accompanying paper, (Anderson et al., in press), we have provided detailed data of the progression of well clogging at the Streatham test that occurred during recharge testing when injected water was mixed with native groundwater. The results of the recharge testing showed that the wells clogged instantaneously, yet well performance deteriorated by less than 20%, and progressively improved during individual injection tests and during subsequent injection events. The only effective explanation was instantaneous clogging of the matrix blocks contained in a fracture network that was effectively insensitive to clogging. Essentially, this inference could only be made because the hydraulic analysis presented in this paper proved the existence of a fracture network containing matrix blocks approximating to the orthogonal fracture system of Warren and Root, 1963. This meant that clogging was unlikely to be a long-term problem and a recharge solution was feasible, as the relatively expensive treatment of recharge water prior to injection was not required to prevent the clogging. However as the geochemical analysis showed that the $CaCO_3$ mineral precipitation occurred as a result of the relatively high pH (7.5 to 8.0) of the water contained in Thames Water's water mains, it was reasonable to assume that $CaCO_3$ clogging could be a widespread issue in the SLARS area. Consequently there was a significant concern that clogging could limit the feasibility of artificial recharge at many locations. However the identification of double porosity hydraulics at the Streatham site through analysis, and linkage of this phenomenon to clogging insensitivity, suggested that clogging was unlikely to be an issue wherever double porosity hydraulics were identified. Fortunately testing also showed that recharge hydraulics were very similar to the abstraction hydraulics and therefore recharge rates could be calculated from abstraction test data. Consequently it was possible to re-analyse pumping test data in Thames Waters' technical reports using the double porosity analyses presented in this paper and to show that the orthogonal fracture network hydraulics was a widespread feature of the Chalk in the SLARS study area.

CONCLUSIONS

Despite the fact that various methods for analysing pumping tests in double porosity aquifers have been available for 40 years or more, they are not yet widely used. This reticence may be explained by a lack of familiarity from

'the lack of a unified approach' and 'an enormous overlap of equations', (Kruseman and de Ridder, 1990), but also from a lack of confidence in the methods. It is perhaps not surprising that practitioners are often sceptical of the applicability of the various double porosity methods and have a preference for continuing to apply conventional porous analytical techniques. However this approach has serious consequences, particularly in the investigation and design of artificial recharge systems. Although the double porosity methods presented here do not have a firm theoretical justification (Kruseman and de Ridder, 1990), the risk from calculating erroneous storage values using conventional homogeneous radial drawdown methods is much greater. Furthermore, more sophisticated double porosity methods; Bourdet and Gringarten, 1980; and Moench, 1984) do have a sound theoretical basis and can be used if the approximations inherent in the methods of Kazemi et al. (1969) and Warren and Root (1965) are of concern.

Clearly, the application of double porosity analytical methods to aquifers that can unmistakably be shown to have double porosity aquifer hydraulics, such as the Chalk in South London, is important. Providing data are obtained from sufficient monitoring boreholes located both within and outside the effective radius, relatively simple but generally ignored double porosity aquifer analytical methods can be used to reliably determine the aquifer transmissivity and storage. In the context of artificial recharge, the use of porous model analytical methods will typically result in mis-calculation of aquifer storage coefficients that are magnitudes greater than true values; and overestimates of the total recharge capacity in error by a similar margin. Clearly failure to use double porosity analytical methods engenders unacceptable risks to the eventual recharge scheme not meeting its designed capacity, particularly during sustained groundwater demand periods such as droughts, when the scheme water supply is likely to be most needed. Lastly, testing showed abstraction tests had similar double porosity hydraulics to recharge tests in the SLARS area. Consequently it should be feasible to determine recharge potential from abstraction tests which are easier and cheaper to perform, and which are not limited to locations where appropriate water supplies are available.

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Development of a microbial pathogen attenuation model to aid management of aquifer recharge systems

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Abstract

One of the principal improvements in water quality that can occur during aquifer storage is pathogen inactivation. Our aim is to develop a predictive tool for estimating required retention times for the natural attenuation of microbial pathogens, mainly human enteric viruses and *Cryptosporidium*. Recent studies reveal that pathogen decay is predominantly controlled by the activity of indigenous groundwater microorganisms, while temperature, redox state and nutrient levels play a secondary role. Our focus is to understand several critical environmental controls on the survival of microbial pathogens in the presence of groundwater microorganisms. Pathogen removal by other physical and chemical processes are not being considered at this stage as we confine our attention to lab-based survival experiments before making field comparisons. Our main hypothesis is that pathogen die-off rates (and minimum aquifer retention times) can be estimated reliably using empirical equations derived from survival experiments. The experimental design controls for temperature, redox state and nutrient level, and considers a suite of pathogens (i.e. coxsackievirus, adenovirus, rotavirus, *Cryptosporidium,* hepatitis A virus, norovirus) and the indicator microorganism MS2. Cellular activity levels of the groundwater microorganisms are being measured concurrently. The formulation of a predictive tool is helping to advance our understanding of the fundamental controls on microbial pathogen attenuation in managed aquifer recharge systems.

Keywords

Cryptosporidium, groundwater microorganisms, human enteric viruses, pathogen attenuation model.

INTRODUCTION

There has been a major interest in predicting the fate of microbial contaminants in the subsurface as communities move toward using artificial recharge or managed aquifer recharge (MAR) with different water types to boost urban water supplies. River water, stormwater and treated effluent are commonly considered for MAR projects and among these the water quality can vary considerably. Among the major health risks to humans are enteric infections from contact with faecally contaminated water, which can persist in treated reclaimed water (Toze, 2004). Although conventional water treatment systems can eliminate most pathogens prior to aquifer recharge, there remain potential health risks from water-borne microbial pathogens that resist disinfection. Our focus is on the enteric viruses and protozoan parasites as they pose an immediate human health risk. Few studies have examined the survival of enteric protozoa (e.g. *Cryptosporidium parvum*) in groundwater, as the emphasis has been mainly on filtration mechanisms to remove oocytes (Harter et al., 2000; Kim and Corapcioglu, 2004; Tufenkji et al., 2004). In contrast, there is much more information about the fate and transport of viruses (e.g. Schijven and Hassanizadeh, 2000; Bhattacharjee et al., 2002) and physical factors influencing pathogen removal based on colloid transport studies that neglect microbial interactions (Bradford et al., 2002; 2004). The aim of this study is to quantify reductions in several enteroviruses and *Cryptosporidium* in non-sterile groundwater due to inactivation, rather than by attachment to aquifer surfaces. Inactivation or 'die-off' refers to disruption of the protein coat leading to a loss of nucleic acid in the case of viruses and nonviable oocytes in the case of *Cryptosporidium*. While it is understood that a range of physical and chemical conditions affect the persistence of microbial pathogens (Toze, 2004), it has also been concluded that antiviral activity of indigenous bacteria in groundwater (Nasser et al. 2002; Toze and Hanna, 2002;

Gordon and Toze, 2003), and mixed human and animal wastes (Deng and Cliver, 1995; Nasser and Oman, 1999) have a major role in the inactivation of microbial pathogens. In the case of *Cryptosporidium*, it has also been shown that autochthonous microorganisms (and exo-enzymes from bacteria or fungi) in river water can influence the survival of the oocytes (Medema et al., 1997), but there are relatively few investigations using groundwater (John, 2003). The formulation of a computer model for determining minimum required retention times for microbial pathogen die-off during aquifer storage is one piece of a complex puzzle. As managed aquifer recharge projects become increasingly utilised, it is critical that water resource managers have a quantitative index of the risk of human exposure to viable pathogenic microorganisms. This is one of the first studies to systematically investigate some of the fundamental controls on the inactivation of microbial pathogens due to the activity of indigenous groundwater bacteria with the aim of developing a reliable framework for predicting minimum aquifer retention times.

METHODS

There are a multitude of environmental stresses in the subsurface that could influence the survival of microbial pathogens (Yates and Yates, 1988). In a recent survey of published decay rates for microbial pathogens, Toze (2004) identified temperature, dissolved oxygen, water chemistry, source of water, as well as the type of pathogenic microorganism, as the most commonly cited factors influencing pathogen survival. The viruses are more resistant to pH and electrical conductivity (salt species and concentration) than bacteria, such that it has been postulated that water chemistry directly impacts the activity of indigenous bacteria, which in turn has a secondary affect on the persistence of viruses (Toze, 2004). As different studies of microbial survival have examined different pathogens under a variety of environmental conditions, it is difficult to use published decay rates to formulate a predictive model. The approach we are using is to systematically vary several environmental stresses identified by Gordon and Toze (2003) as potential factors influencing the survival of enteric viruses: temperature, redox state and nutrient levels are currently being considered (Sidhu et al., in press). Although treated effluent can have a range of nutrient types and concentrations, we are first considering a complex organic carbon source and its effects on several pathogenic microbes through the activity of non-pathogenic microorganisms native to groundwater.

Collection of data from pathogen survival experiments

Survival experiments are being undertaken to provide data to be used in the generation of the computer model as well as using data from previous experiments. Current experiments are focusing on the influence of microbial activity on the decay of a range of microbial pathogens. As it has been established that the activity of indigenous groundwater microorganisms is the major influence on pathogen decay (Gordon and Toze, 2003), our focus is on evaluating the influence of other biological, chemical and physical conditions on groundwater microbial activity and the subsequent impact on the decay of enteric pathogens in groundwater. The biological processes affecting a range of enteric pathogens that are commonly found in reclaimed water are being considered. The type of pathogen has been shown to be an important factor as differences in decay rates have been observed between different pathogens, even those that are genetically similar (Toze and Hanna, 2002; Gordon and Toze, 2003). As shown in Figure 1, the survival of poliovirus and coxsackievirus under a similar set of environment stresses are quite different, despite the microorganisms having genetic similarities. A list of some of the major pathogens to be investigated is given in Table 1.

The major chemical and physical parameters of interest are the type of degradable organic carbon present, the redox condition of the groundwater and a range of temperatures (5 - 20 °C). It is hypothesized that these parameters impact on the metabolic activity and population structure of the indigenous groundwater microorganisms, thus influencing the rate of decay of the different pathogens. The experiments on pathogen survival under the different groundwater conditions are being undertaken using non-sterile groundwater obtained from a bore in the superficial aquifer at the CSIRO laboratories in Perth, Western Australia. Groups of the selected pathogens are added to bio-
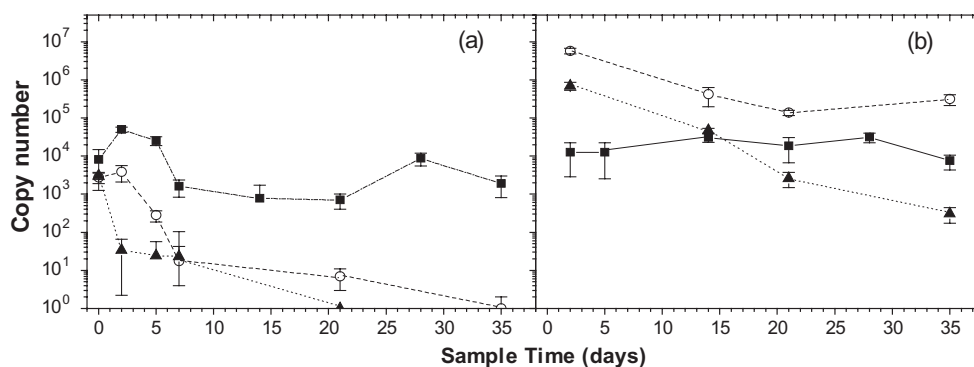
Figure 1. Decay of (a) poliovirus and (b) coxsackievirus in the presence of oxygen at 28°C (s), oxygen at 15°C (O), and anoxic conditions at 28°C (n)

Table 1. Microbial pathogens to be tested for persistence in groundwater during MAR

Pathogen type	Selected pathogens
Viruses	rotavirus, norovirus, hepatitis A virus, adenovirus
Bacteria	*Salmonella* spp., *Shigella dysenteriae*, and *Campylobacter* spp.
Protozoa	*Cryptosporidium* spp. *Giardia lamblia*

reactors containing groundwater that has been manipulated to achieve the conditions to be studied (see Sidhu et al., in press for an example of the experiments). Samples of the pathogens are collected at specified time intervals and the number of viable pathogens is determined (Toze and Hanna, 2002; Gordon and Toze, 2003). At the same time, the metabolic activity of the groundwater microorganisms present in the collected sample is determined by a combination of measurements of ATPase levels and Fluorescein Diacetate degradation. The rate of decay of each of the pathogens studied can then be determined over the entire incubation period and correlated with the metabolic activity levels.

Modeling microbial population decay rates

A common approach to describe the inactivation rate of microbial pathogens is to use a first-order reaction; however it will also be necessary to evaluate whether nonlinear survival curves are more suitable by plotting the decrease in numbers of microorganisms over time and determining the slope of the regression curve (Yates and Yates, 1988; Hurst, 1992). In terms of quantifying the amount of variation in inactivation rates due to temperature, redox state and nutrient status, linear regression analysis to obtain correlation coefficients is one approach; however, other methods will be considered: Hurst et al. (1991) compared the accuracy of different regression equation formats that are used to empirically model viral population decay rates and found that the greatest modelling accuracy was obtained using multiplicative error regression, meaning that the independent variables controlling virus survival are multiplicative rather than additive as in linear regression models. This approach has also been used to estimate decay rates for *Cryptosporidium* due to different environmental stresses (Walker et al., 2001).

RESULTS AND DISCUSSION

The conceptual basis for the computer model is to provide a management tool that takes into full account the results from pathogen survival experiments (Figure 2). A preliminary modelling framework has been developed in a

Microsoft Excel workbook that will access experimentally-determined decay rates stored in a separate worksheet for each of the pathogenic microorganisms for each set of unique conditions (temperature, redox state, and nutrient level). The demonstration version of the workbook includes the linkages and essential calculations for determining required pathogen decay times. The software structure is intended to be flexible, transparent and simple to use to allow water resource managers to readily access experimental data to aid decision-making about minimum aquifer residence times for microbial pathogen attenuation. As more experimental data are acquired throughout the project, regression analysis and investigation of nonlinear survival curves will be conducted. The model will include an uncertainty range on the calculated residence times. In this preliminary version of the model, decay rates can be readily added to the appropriate look-up tables that are linked to the Excel workbook.

Required Pathogen Decay Time

1. Model Input Parameters and Conditions
Select one from each category with an "X" or enter data where applicable

Pathogen	Coxsackievirus	Adenovirus	Rotavirus	Cryptosporidium	Hepatitis A	Norovirus
	x					
or Indicator	MS2					
Initial population	very low	low	medium	high	very high	or specify (Viral No./litre)
						5.71E+06
Final population	very low; 0	low; ≤ 10	or specify (Viral No./litre)			
			4.15E+05			
Redox state	Aerobic	Methanogenesis	Sulfate reducing	Nitrate reducing	or specify (mV)	
	x					
Organic carbon concentration (mg/l)	0					
Groundwater temperature (deg. C)	12					

2. Pathogen Decay

Pathogen/Indicator	Coxsackievirus	
Decay rate	0.038	log10(Viral No./litre)/day
Required pathogen decay time	29.7	days
Error bars (+/-)	to be completed	days

Figure 2. The basic front-end to the computer model developed as an Excel workbook.
The user enters data in field 1 and the spreadsheet provides the calculated pathogen decay time in field 2.
This front-end is linked to a spreadsheet containing data from the pathogen survival experiments.

CONCLUSIONS

The formulation of a predictive tool for determining minimum required retention times for microbial pathogen decay during aquifer storage is one piece of a complex puzzle. There are other processes that can further attenuate microbial pathogens (e.g. adsorption, straining) within the porous media during groundwater transport, but we confine our attention at this stage to understanding how several critical environmental controls impact the survival of pathogenic microorganisms. The preliminary model set forth in this paper is one of the first attempts to system-atically investigate some of the fundamental controls on the inactivation of microbial pathogens due to the activity of indigenous groundwater bacteria to facilitate the management of aquifer recharge systems. The approach and synthesis of data for predicting the fate of microorganisms could be adapted to manage in-situ bioremediation or estimate setback distances to prevent faecal contamination.

ACKNOWLEDGEMENTS

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Excess air: a new tracer for artificially recharged surface water

Jordan F. Clark and G. Bryant Hudson

Abstract

As part of a deliberate gas tracer experiment, noble gas concentrations were examined at an artificial recharge site in Ventura County, CA. Surface water samples collected within a spreading basin were in equilibrium with the atmosphere. Noble gases were supersaturated in groundwater samples collected directly below the spreading pond due to the dissolution of trapped air within the unsaturated zone. This added fraction is commonly referred to as excess air. The amount of excess air ranged between 9 and 17 cm^3/L and averaged 13.7 cm^3/L. Under natural recharge conditions, excess air concentrations in groundwater are typically less than 5 cm^3/L, although excess air concentrations as large as observed here have been reported elsewhere away from artificial recharge sites. The enrichment in noble gas concentrations indicates that excess air is a valuable tracer of surface water artificially recharged water from spreading ponds.

Keywords

Excess air; geochemical tracers; noble gases; recharge temperature.

INTRODUCTION

Tracing the movement of groundwater artificially recharged at engineered sites is critical for evaluating *in situ* water quality changes and the potential for future contamination of near by wells. A number of methods have been developed over the years for determining flow paths and travel times from recharge locations including geochemical finger printing (Lee et al., 1992; Ma and Spadling, 1996; Williams, 1997; Clark et al., 2004) and deliberate tracer experiments (Gamlin et al., 2001; Clark et al., 2004, 2005; Avisar and Clark, 2005). The former technique exploits differences in the geochemistry (e.g., stable isotope composition, chlorinity, etc.) of the recharge and native groundwater while the latter uses controlled releases of trace substances (e.g., sulfur hexafluoride, isotopes of noble gases). Deliberate tracer experiments are limited by dilution with the native groundwater and by the travel time within the subsurface. Travel time to wells can be longer than the project length and, thus, these experiments can be impractical. Geochemical fingerprinting does not have these limitations because it is not dependent on travel time. However, when there is little difference between the chemistry and isotope composition of the native and recharge waters, geochemical finger printing is unable to trace the recharge water. Here, we examine a new tracer, excess air, which has characteristics of both types of tracers discussed above and can be use to trace artificial recharge water from spreading basins. Excess air forms during recharge and is characterized by gas concentrations greater than equilibrium values (Heaton and Vogel, 1981). Thus, it is added to the percolating surface water like a deliberate tracer whenever recharge is occurring. Because its signature in artificially and naturally recharged groundwater is different, its fingerprint can be used to distinguish the origin of groundwater near engineered recharge sites.

Excess air forms when air becomes trapped below the water table and dissolves due to the increased hydrostatic pressure as a result of rising water tables. The rise is proportional to the amount of recharge and is typically much larger during periods of artificially recharge at spreading ponds than during natural recharge events. Non-reactive dissolved gases record the amount of excess formed. Most commonly, it is determined using measurements of

dissolved noble gases (Ne, Ar, Kr, and Xe). During the inversion process noble gas recharge temperatures are also calculated (Aeschbach-Hertig et al., 1999, 2000; Stute and Schlosser, 2000).

Field location

The El Rio Spreading Grounds are located in the regional recharge area (the Forebay) of the Oxnard Plain (Ventura County, CA, USA) within the Santa Clara-Calleguas Hydrologic unit. The upper aquifer system is composed of discontinuous layers of gravel, sand, and silt and has generally been split into an upper and lower zone of production (Hanson et al., 2003). The upper 50 m below the spreading grounds is free of impermeable layers and, thus, the ground has a high infiltration capacity. Artificial recharge at the El Rio Spreading Grounds has taken place since their construction in 1955 and during the 1990s about 40 x 10^6 m^3 of water was recharged annually (United Water, 2001).

Ten shallow ponds (~3 m maximum depth) and a series of distribution channels make up the spreading grounds (Fig. 1). Very little surface water recharges from either Pond 1, which is used as a desilting basin, or Pond 9, which is used in the potable supply system. The other eight ponds are scraped periodically to ensure rapid infiltration. The source of recharge water is primarily diverted seasonal run-off and released stored water from a local reservoir (Lake Piru).

The spreading basins are surrounded by nine production wells and a set of seven nested monitoring wells (Fig. 1). Due to intensive pumping at the production wells, a regional groundwater depression is found surrounding the spreading grounds, although local mounding beneath the ponds can

Figure 1. Map of the El Rio Spreading Grounds. The dual gas tracer experiment was conducted from Pond 2 while recharge was occurring from Pond 2 and 3

occur during periods of artificial recharge. On average, groundwater production is higher during the summer and artificial recharge is higher during the winter and spring. The seasonality in the recharge and production combined with the very steep local hydraulic gradients create a complex pattern of flow beneath the El Rio Spreading Grounds (e.g., Avisar and Clark, 2005).

Methods

Noble gas samples were collected primarily for ^{3}He analysis as part for a dual gas tracer experiment designed to examine gas transport below spreading ponds. Results from that experiment can be found in Clark et al. (2005). In September 2002, about 24 x 10^6 m^3 of water was released from Lake Piru down Piru Creek and into the Santa Clara River. About 20% of this water was diverted into the El Rio Spreading Grounds, flooding Ponds 2 and 3 for 35 days

beginning on 8-Sept, day −19 of the experiment (Fig. 2). The spreading area had received very little water for the three months prior to this release. Between 27-Sept-02 and 4-Oct-02, a gas mixture containing SF_6 (~900 liters) and ^3He (~1 liter) was injected into Pond 2 (surface area = 3.8 hectare) by bubbling through a diffusion stone. During this time, the average percolation rates of surface water into the ground from Pond 2 and 3 were 1.8 $m^3 s^{-1}$ and 1.7 $m^3 s^{-1}$, respectively. The September percolation rate was about an order of magnitude higher than the long-term mean because the ponds had recently been cleaned of clogging material. At the time of the injection, the water table measured at nearby monitoring and production wells (El Rio #7) was ~12 m below the bottom of the pond (Fig. 2).
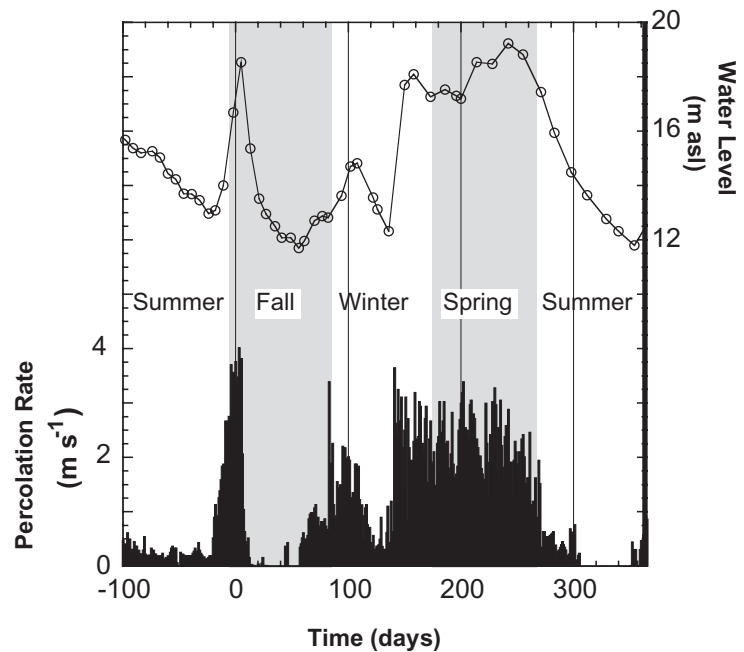
Figure 2. The temporal rate of artificial recharge (percolation) and the local water table elevation at the El Rio Spreading Grounds. The bottom of Pond 2 is about 30 m above sea level. Time zero is Sept. 27, 2002.

Surface and groundwater samples for noble gas analyses were collected in 20 cm long, ~1 cm diameter copper tubes sealed with steel pinch-off clamps. Pond water samples were collected ~0.2 m below the surface from two locations using a submersible pump. Groundwater samples were collected from two production wells, El Rio #5 and #6 (Fig. 1). Both of these wells are located adjacent to Pond 2 and are screened between about 45 m and 90 m below ground surface. During the first two months after the tracer injection, groundwater samples were collected every 3 to 7 days. For the next six months, they were collected every 2 to 4 weeks. Thereafter, samples were collected every 6 to 8 weeks.

In the laboratory, the copper tubes were attached to a high vacuum/clean-up system that led to a VG5400 noble-gas mass spectrometer. The noble gases were extracted from the water samples by boiling under vacuum and separated from other gases with a series of cold traps and titanium getters. The $^3He/^4He$ isotope ratio and the concentrations of He, Ne, Ar, Kr, and Xe were determined. The mass spectrometer was calibrated with equilibrated water samples and known quantities of air. The uncertainties of $^3He/^4He$ isotope ratio, the He, Ne, and Ar concentration, and the Kr and Xe concentration measurements were ±0.5%, ±1%, and ±2%, respectively. The amount of excess air and noble gas recharge temperatures were calculated with inversion methods described by Aeschbach-Hertig et al. (1999, 2000). The uncertainties of the calculated noble gas temperature and excess air amount were, respectively, ± 0.5°C and ± 0.3 cc STP/L or better.

RESULTS AND DISCUSSION

Dissolved Ar, Ke and Xe concentrations in Pond 2 were in equilibrium with the atmosphere assuming a water temperature of ~17.5°C (the pond temperature was not measured). He and Ne were slightly supersaturated ([He] = 101 to 105% saturation and [Ne] = 103 to 107%). Relative to the pond water, all five gases were enriched in the groundwater samples due to the dissolution of trapped air. Average excesses, defined here as $([X]_{measured}/[X]_{equilibrium}-1)$, were for He, 153% ± 27%, Ne, 130% ± 26%, Ar, 42% ± 10%, Kr, 22% ± 5%, and Xe, 10% ± 4%. As measured by the Ne

excess, the amount of excess air formed below the spreading grounds is larger than generally observed under conditions of natural recharge where typically Ne excess are less than 50% (Wilson and McNeill, 1997; Stute and Schlosser, 2000). Ne excesses as large or larger than observed below the El Rio Spreading Grounds have been found in a few aquifers away from spreading ponds, most notably in the Stampriet aquifer, Namibia (Stute and Talma, 1998).

Oxygen and other components of air will dissolve along with the noble gases during excess air formation and, therefore, the recharge process will also increase their concentrations above equilibrium values. The noble gas most similar to oxygen is Ar. Groundwater Ar concentrations were 142% ± 10% of saturation. Thus, dissolved oxygen concentrations should have increased by a similar amount. Because dissolved oxygen is a critical reactant in many biogeochemical reactions that removes contaminants, excess air formation is an important process that leads to improve water quality near sites where surface percolation is the primary method of artificial recharge.

Excess air concentrations in the groundwater ranged between 9 and 17 cm^3/L and averaged 13.7 cm^3/L. During the September recharge event, ~3 x 10^6 m^3 of surface water was recharged from Pond 2, thus more than 4 x 10^7 liters of trapped air within the vadose zone dissolved during excess air formation. Assuming a porosity of 33% and a depth to water table of 12 m (Fig. 2), air from more than 30% of the total pore volume dissolved. Hence, the amount of trapped air contained below the spreading pond was significantly reduced during the recharge event. This reduction most likely leads to changes in the amount of excess air formed. More excess air should form early in the wetting cycle of the spreading ponds. Given enough time, the amount of excess air should approach zero and excess air formation should end.

Time series measurements at El Rio #5 and #6 showed that the amount of excess air varied systematically, although the two wells were not in phase (Fig. 3a). These variations reflect the decreasing amount of excess air during the wetting cycle, the differing transit time between recharge locations and well, and differences in mixing between water recharged at different times due to the relatively long screen intervals (~45 m).

Noble gas recharge temperatures ranged between ~20° and ~13°C during the 7-month long study period (Fig. 3b). The warmest samples were observed at the beginning of the study (Oct-02) and the coolest at the end (April-02). The temperatures were consistent with the season of recharge assuming a very short travel time. Results from the deliberate tracer study indicate that travel from Pond 2 to these wells were on the order of weeks (Clark et al., 2005). Early in the record, noble gas temperatures at El Rio #6 were significantly warmer (~3°C) than El Rio #5. These calculated temperatures were also warmer than Pond 2 samples. Noble gas temperatures generally reflect the
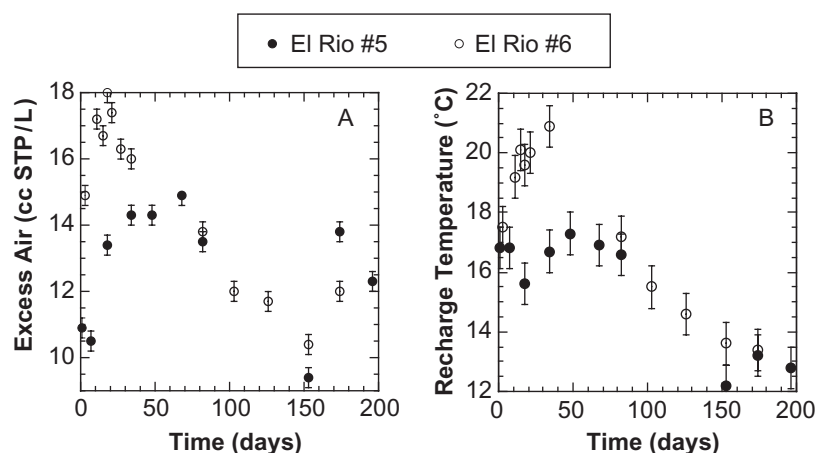
Figure 3. (A) Excess air and (B) noble gas temperatures from groundwater samples collected beneath the El Rio Spreading grounds. Time zero is Sept. 27, 2002.

soil temperature near the water table (Stute and Schlosser, 2000). Given the large quantities of water recharged it is likely that the temperature of the soil was not very different from that of the pond. The more likely explanation is El Rio #6 was drawing in a greater fraction of water recharged early during the September recharge event when the temperature was warmer.

SUMMARY

Excess air concentrations below the El Rio Spreading grounds are usually high. Dissolution of trapped air within the unsaturated zone is the source of the excess air. Because it is added during the recharge processes, it can be used to trace artificially recharged surface water through the groundwater system. The amount of excess air observed was more than a factor of two greater than the typical amount found in groundwater under natural recharge conditions.

ACKNOWLEDGEMENTS

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> Colloid transport and deposition in natural sand

C. Fischer, N. Weisbrod and A. Yakirevich

Abstract

Tracer experiments were conducted to compare colloid transport in natural and cleaned dune sand at saturated conditions. Negatively charged fluorescent microspheres of three sizes (0.02, 0.1 and 1 µm) together with a soluble tracer (LiBr) were added to artificial rainwater solution and applied to the surface of vertical small and large columns. During the experiments effluent samples were taken at the column outlets. Additionally, the columns were dissected and samples were collected at different depths at the end of the experiments. Breakthrough colloid concentration curves and the final spatial distribution of colloids retained by the porous media were found to be dependent on the colloid size and cleanness degree of the sand. Overall breakthrough of colloids was found to be less in natural sand compared to washed sand. Differences were especially significant for the smallest colloids. In the large column experiment colloids were retained in the upper meter, while the solute tracers showed similar behavior like in the small columns.

Keywords

Colloid-facilitated transport; colloid transport; heterogeneous porous media; saturated flow.

INTRODUCTION

Colloidal particles exist in most natural sediments in high quantities (Kretzschmar et al., 1999; McCarthy and Zachara, 1989; Ryan and Elimelech, 1996). Due to their large specific surface area, mobilization of colloidal particles may lead to enhanced transport of hydrophobic contaminants (with high affinity to the solid matrix). These contaminants, such as radionuclides, toxic metals, and many of the pesticides, are likely to be immobile unless there is a migration of colloids (Ryan and Elimelech, 1996). Considering the importance of aquifers as a potable water resource, contamination of groundwater by hydrophobic pollutants and biocolloids is of the utmost public concern. The fact that most of the potential contamination sources are located at the soil surface or within the vadose zone requires transport through the unsaturated zone before the groundwater is reached. Despite this, flood-water percolation like in arid regions, can lead to temporary saturated conditions, which increases the likelihood of colloid facilitated transport (Chen and Flury, 2005; Jin et al., 2000; Wan and Wilson, 1994).On the other hand, most natural surfaces have an uneven, heterogeneous charge distribution (Song et al., 1994), which could lead to retention of colloids. We compared colloid transport in natural and washed saturated sand. Washing of the sand reduces chemically charged surfaces like oxide coatings (as for acid washing) and removes the smallest dust fraction from the sand. In this paper we present the results of small and large column experiments, which evaluate the effect of cleaning treatment of natural dune sand on colloidal transport at saturated conditions. Colloid transport experiments were carried out in columns, which were packed with differently treated dune sand: (a) no washing (natural); (b) washing with Distilled Water (DW); and (c) acid washed sand (acid). Breakthrough Curves (BTC's) were obtained for colloid and solute tracer.

MATERIALS AND METHODS

For the experiments we used natural dune sand, originating from the Israeli Coastal Plain 20 km south of Ashkelon. The grain size distribution of the sand is shown in Fig. 1, accounting for the three different washing procedures. The porosity of the sand was 0.32 ± 0.01. The moisture retention characteristics were determined by the pressure plate method (pF=0-2) and at high pressure in the desiccator at pF=4.445 (Shein et al., 2001). The water content was determined gravimetrically. The saturated hydraulic properties were determined in small saturated column experiments (Jury et al., 1991). The unsaturated hydraulic properties (van Genuchten) were evaluated by the code APW02-1 (A. Yakirevich, in preparation). The sand was coarsely sieved (2 x 2 mm) in order to remove roots and stones, but to conserve the natural soil texture. The sand denoted as 'natural' was not further treated. The sand named DW was washed and rinsed with DW water for about ten times until no turbidity was observed. The 'acid washed' sand was treated with nitric acid (50%) and rinsed with Double Distilled Water (DDW) as described by Wan and Wilson (1994). The experimental set up for the small and the large column experiments is shown in Figs. 2 and 3, respectively.
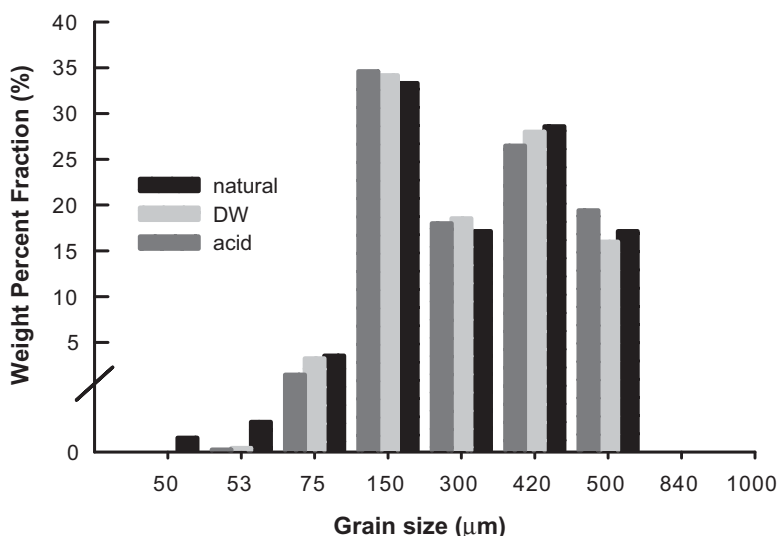
Figure 1. Grain size distribution of the three differently treated sands by weight percent fraction
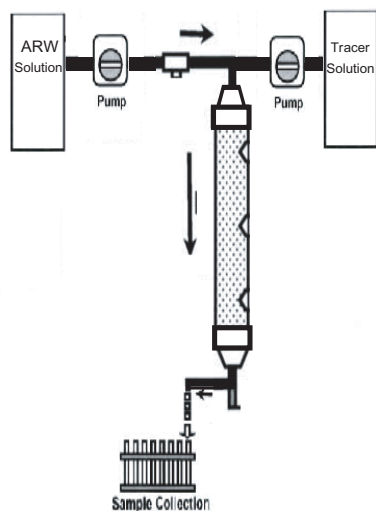
Figure 2. Small column set up
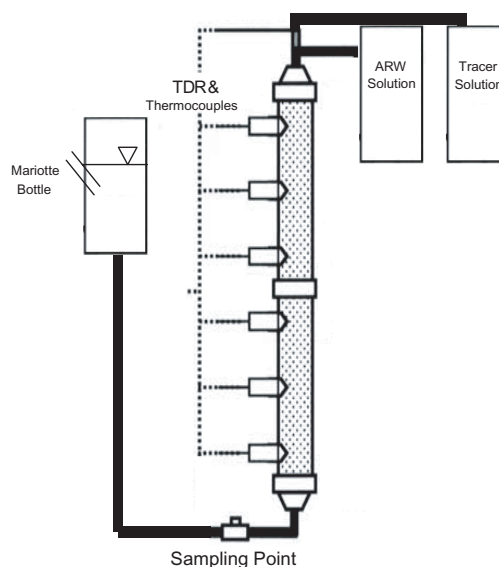
Figure 3. Large column set up

The small columns were made of Plexiglas with overall dimensions of 20 cm length and 5.7 cm inner diameter. They were dry packed homogenously, separately filled with the differently treated sands, and then saturated from the bottom, first with CO_2 and afterwards with Artificial Rain Water (ARW, (Livshitz, 1999). The tracer solution consisted of fluorescent polystyrene carboxylate microspheres (FluoSpheres®, Molecular Probes Inc., Eugene, Oregon) of three different sizes (1, 0.2, and 0.02 µm) and LiBr (Lithium Bromide) dissolved in ARW (I=0.0023M, pH=7.3). After the injection of the tracer as a pulse (1/4 of the column pore volume) the column was flushed with ARW for at least 4 Pore Volume (PV). The solutions were pumped from top to bottom with a peristaltic pump, which was calibrated to a flow rate resulting in pore velocity of 19.5 cm/h, prior to the experiments. Similar pore velocities were observed from floodwater infiltration experiments carried out in the Arava Valley, Israel (Dahan, O., in preparation). Column outflow was collected in a fraction collector and analyzed for solute and colloidal tracer, pH and Electrical Conductivity (EC) at the end of each experiment. Colloid particle concentration was determined using a fluorescent spectrometer (Fluoremeter®) and soluble tracer concentration by spectrophotometer and atomic adsorption spectrometer (Li$^+$ and Br$^-$, respectively). The columns were sliced in the end of each experiment and the slices were sonicated in NaCl 0.005N solution for 20 minutes. Soil sample solution was extracted and particle concentration was analyzed with Fluoremeter®.

The large column consisted of rigid PVC and had a length of 250 cm with an inner diameter of 23.7 cm. Ten TDR probes were installed in the column (from top to bottom: 10, 20, 30, 40, 60, 80, 100, 130, 160, 210 cm). The TDR probes were used to ensure that saturated conditions were reached and maintained before and during the whole run of the experiment. In addition to that, thermocouples monitored the wetting front of the tracer pulse and heat fluxes during the experiment (from top to bottom: 5, 10, 15, 70, 120, 170, 220 cm). The column was dry packed homogeneously and saturated as the small columns. Water flow was induced by water head differences between the pond level at the top of the sand and the water level in a hanging column that was connected to the bottom of the large column. The two levels were kept constant - at the top of the column by an electrical tap and at the bottom by a peristaltic pump. The gradient was calibrated such, that pore velocity would result in 19.5 cm/h, as for the small column experiments. In the first large column experiment we used natural sand and the same tracer pulse (1/4 of the column pore volume) as in the small columns. The tracer solution was applied at the top of the column and afterwards flushed with ARW for at least four PV. In the second large column experiment we used DW sand and twice the volume of tracer solution (1/2 column PV). Outflow was collected manually at 5-minute intervals and analyzed as the small column outflow. Soil samples were collected in 5 cm resolution by a special soil sampler (Soil Sample Ring Kit, Eijkelkamp). The sample extract was analyzed as described for the small column experiments.

RESULTS AND DISCUSSION

BTC's from the small columns are presented in Fig. 4 for 1, 0.2, and 0.02 µm for each sand treatment. In acid washed sand colloid breakthrough is always earlier than in DW washed and natural sand. There is no significant difference between 1 µm and 0.2 µm in breakthrough for all treatments. The smallest colloid though, 0.02 µm, shows later breakthrough for DW and natural sand than the larger colloids. In addition, colloid recovery decreases most distinctively for the smallest microspheres (0.02 µm). The recovery of the middle-sized colloids (0.2 µm) is least influenced by the sand washing procedure.

The solute tracer BTC for the small columns is shown in Fig. 5, for lithium and bromide. Overall lithium breakthrough is retarded compared to colloidal breakthrough. In DW and natural sand tailing is observed. This is because the lithium cation attaches to the mostly negative charged sand grain surfaces of the more impure sands. Acid washing results in decrease of electrical charge, which leads to early breakthrough and higher recovery. Bromide with its negative charge is least affected by the washing. Still there is earlier breakthrough for the acid washed sand.

BTC 1μm

BTC 0.2μm

BTC 0.02μm
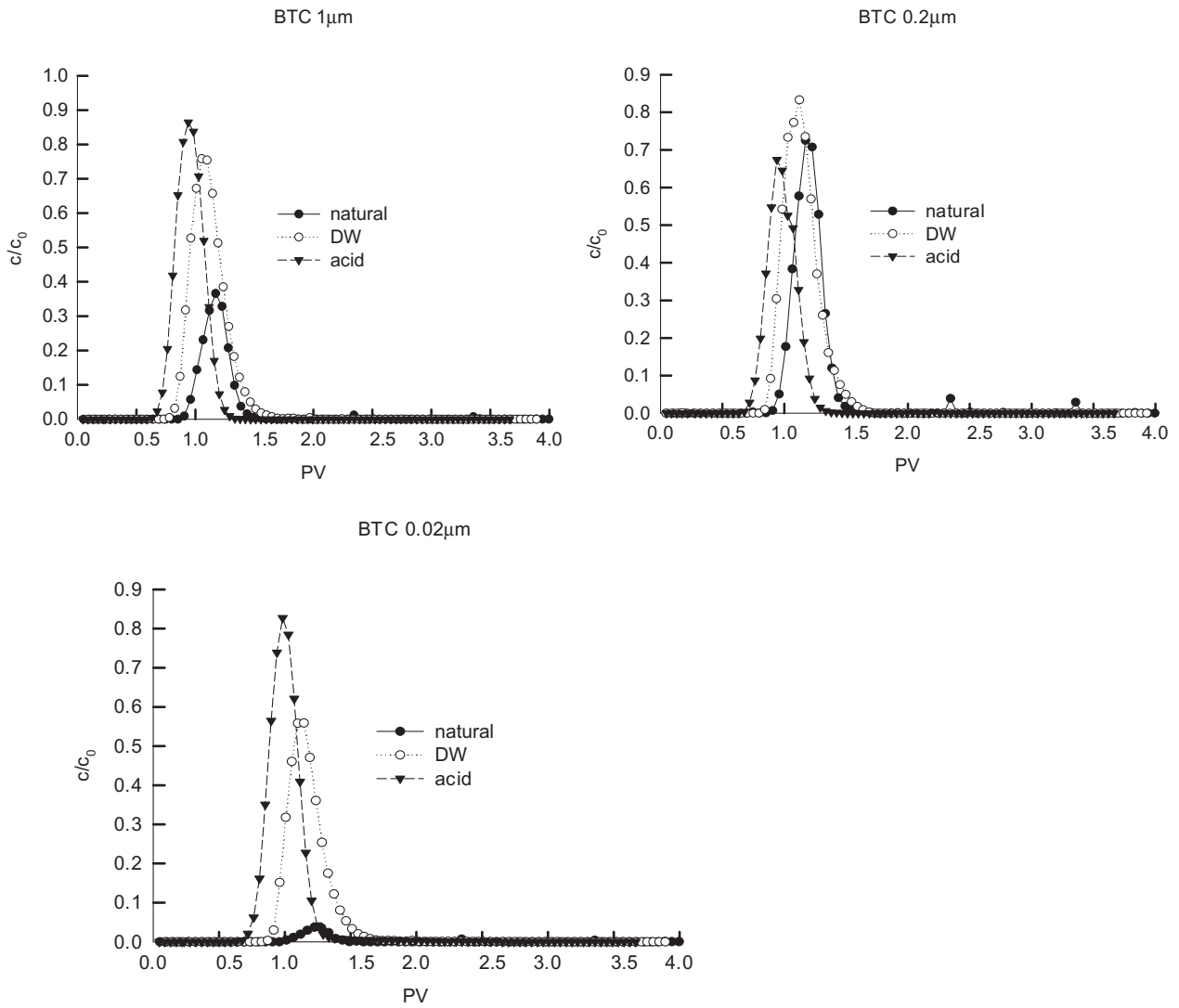
Figure 4. BTC's from experiments conducted at 19.5 cm/h average water velocity

Bromide

BTC Lithium
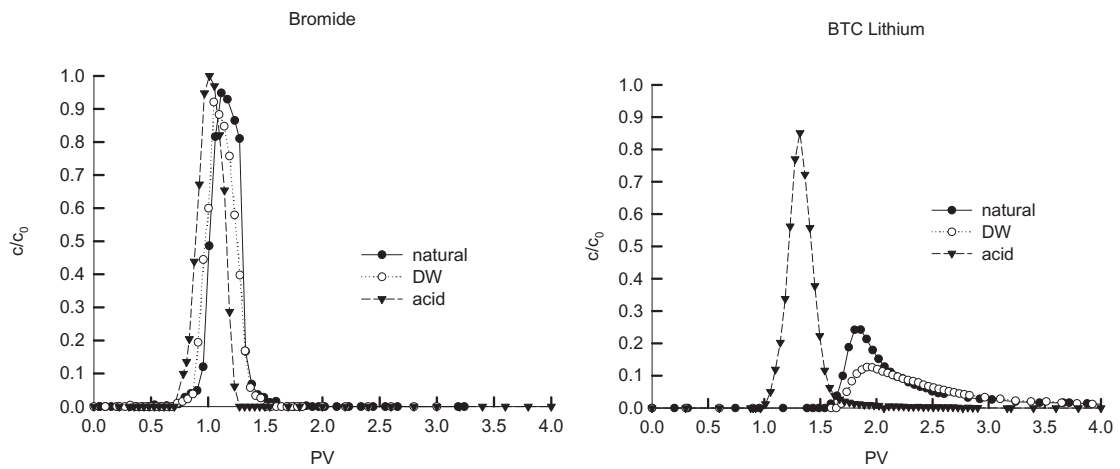
Figure 5. LiBr BTC from experiments conducted at 19.5 cm/h average water velocity

Breakthrough of bromide was slightly earlier than that of colloids in natural and DW sand and similar to colloid breakthrough in acid washed sand.

In the up scaled experiment with the same natural sand, colloids were retained completely, while the solute tracer showed similar behavior to that observed in the small column with natural sand, see Fig. 6.

Concentration distribution of colloids retained with depths (see Fig. 7) shows that colloids are retained in the upper meter.

This finding suggests that at field scale colloid facilitated transport may occur, but for the layers close to the soil surface. Afterwards small dust particles and chemical heterogeneity lead to retention of the colloids.
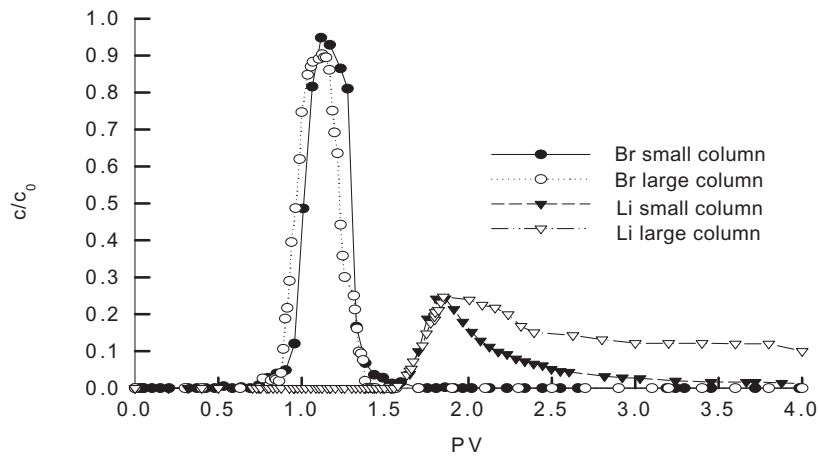
Figure 6. LiBr in natural sand, when comparing small and large column experiments
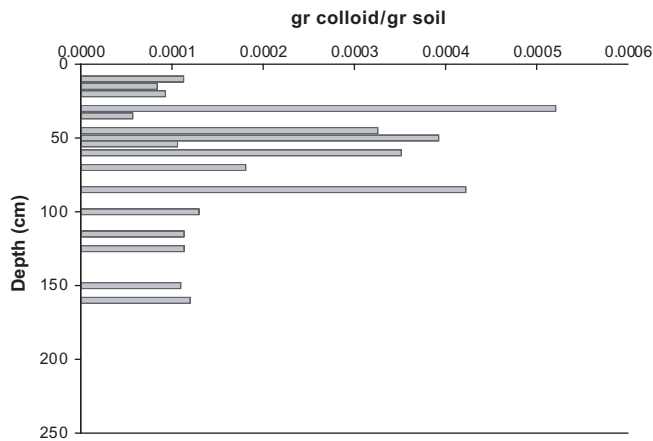
Figure 7. Colloid concentration distribution with depths in gram colloids/gram soil

SUMMARY AND CONCLUSIONS

BTC's of the small column experiments show significant differences in arrival time and mass recovery of colloids for cleaned and natural sand. While breakthrough in acid washed sand is earlier than the conservative tracer, break-

through in natural sand is retarded and mass recovery is reduced. Up scaling to large columns reveals that with travel time colloids are retained by either the matrix properties or clogging of flow channels by fine sand particles.

In arid environment where Aeolian dust is abundant, large amounts of such fine particles are likely to accumulate on the land surface and in the upper soil layer. Intensive rain events and floods may carry these fine particles, and contaminants sorbed to them, deeper into the vadose zone. However, while salts are likely to infiltrate fast toward the water table, particles are more likely to accumulate within the upper meter. Particle accumulation could partially clog the matrix and inhibit vertical contaminant migration. Overtime, this phenomenon could control infiltration rate and even play a major role in the active recharge process.

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> # Hydrogeochemical changes of seepage water during artificial recharge of groundwater in Berlin, Germany

J. Greskowiak, G. Massmann, G. Nützmann, H. Prommer

Abstract

The spatial and temporal evolution of the seepage water chemistry below an artificial recharge pond was investigated to identify the impact of dynamic changes in water saturation and seasonal temperature variations. Geochemical analysis of the pond water, suction cup water and groundwater showed that during summer, nitrate and manganese reducing conditions dominate as long as saturated conditions prevail. Iron and sulphate reduction occur only locally. When the sediment below the pond becomes unsaturated, atmospheric oxygen penetrates from the pond margins leading to re-oxidation of previously formed sulphide minerals and enhanced mineralisation of sedimentary particulate organic carbon. The latter promotes the dissolution of calcite. During winter, both the saturated and the unsaturated stage were characterised by aerobic conditions. Thereby, nitrification of sedimentary bound nitrogen could now be observed because nitrate is not immediately consumed, as is the case during summer. This suggests that nitrification below the pond might be less affected by seasonal temperature changes than nitrate reduction.

Keywords

Artificial recharge, geochemistry, redox zoning, saturated-unsaturated zone, seasonal temperature changes.

INTRODUCTION

Increasing water demands and pollution of water resources are some of the most serious challenges of the modern world. Facing these problems, techniques of artificial groundwater recharge such as river bank filtration (e.g., Ray et al., 2004), aquifer storage and recovery (ASR, e.g., Pyne, 1995), deep well injection (e.g., Stuyfzand et al., 2002) and infiltration ponds (e.g., Bouwer, 2002) are becoming increasingly popular. Like other recharging techniques, infiltration ponds are commonly used either to enhance the quality of surface water or to purify partially treated sewage effluent (Bouwer 1991; Asano, 1992). While the fate of specific organic substances during surface water infiltration into unsaturated porous media has been investigated intensively (e.g., Fujita et al., 1998; Lindroos et al., 2002; Långmark et al., 2004), only a few studies exist which simultaneously give attention to the prevailing inorganic chemistry, including redox conditions (e.g., Brun and Broholm, 2001). However, as the local redox state is known to affect the fate of various organic pollutants such as pharmaceutically active compounds (PhAC's, e.g., Holm et al., 1995, Massmann et al., 2005), pesticides (e.g., Tuxen et al., 2000), or halogenated organic compounds (e.g., Bouwer and McCarty, 1983), it is important to understand the development of redox zones and their constraints in artificial recharge systems.

The aim of this study was to characterise and understand the dynamics of the hydrogeochemical evolution of the seepage water below an artificial recharge pond, which result from transient hydraulic conditions and from seasonal temperature variations of the surface water.

FIELD SITE

The study site is one of three recharge ponds surrounded by 44 production wells located near Lake Tegel, Berlin, Germany. Its infiltration surface extends over an area of approximately 8,700 m^2 and has an elevation of 3 m below the adjacent ground surface. Surface water of Lake Tegel is discharged into the pond after it has passed a microstrainer. With time, clogging processes at the pond's floor lead to a continuous decrease of infiltration rates i.e. from 3 m/d to as low as 0.3 m/d. As soon as the infiltration rate decreases to below 0.3 m/d the site operator Berliner Wasserbetriebe (BWB) abrades the clogging layer to restore the original hydraulic conductivity of the bottom sediments. This operational cycle is repeated every 3–4 months.

The sediments in the adjacent area of the pond are of Quaternary age and consist of fluvial and glacio-fluvial, medium sized sand deposits. Fragments of an up to 5 m thick till layer are locally found in depths of approximately 15 m below the ground surface (Pekdeger et al., 2002). The hydraulic conductivities of the aquifer sediment are about 10–100 m/d. A more detailed description of the sediment hydraulic properties is given in Pekdeger et al. (2002).

The organic carbon content of the sediment below the pond is highly variable and ranges from 0.2 g/kg to 20 g/kg. Reducible forms of iron and manganese minerals are found in concentrations of 0.2–1.2 g/kg and 0.01–0.1 g/kg, respectively. Total sulphur concentrations are in the range of 0.1–2.1 g/kg and are highest in organic-rich layers. A more detailed description of the geochemical composition of the sediment below the pond can be found in Greskowiak et al. (in press).

METHODS

A detailed monitoring program was carried out over a period of more than one year. As part of this program (i) the pond water, (ii) groundwater at a depth of 7 m below the pond and (iii) water extracted from four ceramic suction cups located at depths of 50 cm, 100 cm, 150 cm and 200 cm below the pond were analysed for major ions and dissolved organic carbon (DOC) every week. Anions and cations were measured by ion chromatography (IC, DX 100) and atomic adsorption spectrometry (AAS, Perkin Elmer 5000), respectively. DOC was measured photometrically with a Technicon autoanalyser. Dissolved oxygen (DO) was measured by optical oxygen sensors (Hecht and Kölling, 2001) placed next to the suction cups. Water contents and pressure heads at 50 cm and 150 cm depths below the pond were recorded continuously by TDR (Time Domain Reflectometry) probes and pressure transducers respectively. The temperature of the pond water and the groundwater as well as the piezometric head at 8 m below the pond were continuously measured daily with data loggers. The location of the measurement devices are schematically shown in Figure 1.
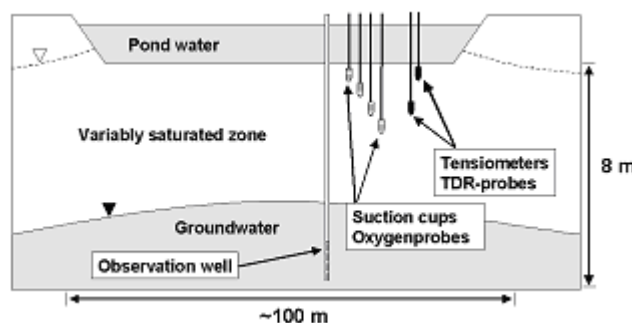
Figure 1. Schematic cross-sectional view of study area and locations of measurement devices

RESULTS AND DISCUSSION

Hydraulic characteristics

The hydraulic regime immediately below the pond was characterised by cyclic changes between saturated and unsaturated conditions, as indicated by (i) the varying piezometric head resembling the approximate groundwater
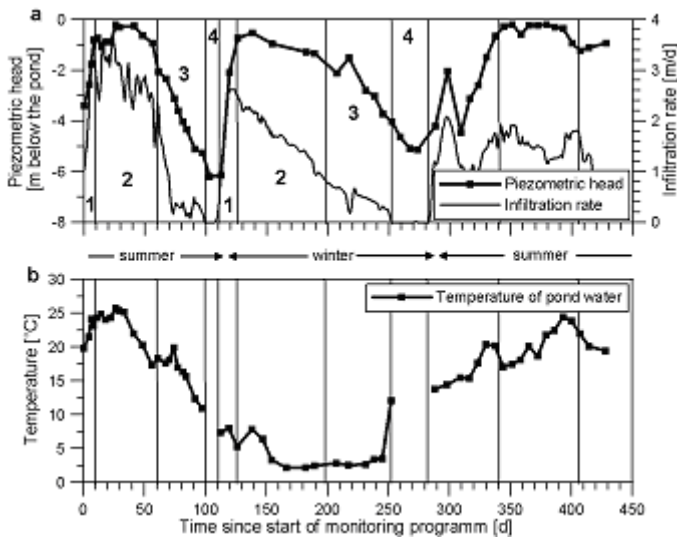
Figure 2. a. Infiltration rate and piezometric head;
b. temperature of the pond water;
the numbers 1–4 refer to Stages 1–4.

table (Figure 2) and (ii) the varying water contents and pressure heads below the pond (data not shown). These changes, which occurred during each operational cycle, resulted from the repeated formation of a clogging layer at the pond bottom (Greskowiak et al., in press). Each operational cycle was hydraulically classified into four different stages (Figure 2). Stage 1 was characterised by a rising groundwater table resulting from the refilling of the recharge pond. During Stage 2 the groundwater table rose to the bottom of the pond and saturated conditions were established. As soon as the hydraulic resistance of the clogging layer became too high (around day 60 and day 200 after start of the monitoring program), air penetrated from the pond margins beneath the clogging layer, which resulted in a sudden drop of water contents (not shown) and piezometric head (Figure 2). Thus, unsaturated conditions established below the pond (Stage 3). During Stage 3 the groundwater level continuously declined to approximately 4–5 m below the pond (Figure 2). Note that infiltration still took place during Stage 3, but continuously decreases to approximately 0.3 m/d (Figure 2). During Stage 4 the pond was dry and the site operator abraded the clogging layer. During this Stage, the groundwater level was at its minimum.

Geochemistry

Surface water composition

Throughout the entire observation period, the pond water typically contained 7–16 mg/l of DO. Nitrate concentrations were highly variable and ranged from 0–12 mg/l. The lowest concentrations were attributed to nitrogen uptake by algae blooms during summer. With a few exceptions, dissolved iron (data not shown) and manganese were typically below detection limit. Sulphate concentrations were relatively high (~140 mg/l, data not shown). With calcium concentrations of 80–90 mg/l and total dissolved inorganic carbon (DIC) concentrations of 30–37 mg/l, the pond water was oversaturated with respect to calcite ($SI_{Calcite}$= 0.4–1.2). Dissolved organic carbon was rather low with concentrations of about 5–11 mg/l. The pH was relatively stable and generally in the range of 8–8.5.

Summer cycle

As soon as saturated conditions established below the pond in summer (stage 2), DO became entirely depleted at all observed depths below the pond (Figure 3). Nitrate and manganese reducing conditions were dominant beneath the pond as long as saturated conditions prevailed. This was indicated by the total depletion of nitrate and the subsequent increase of dissolved manganese at several observation points (Figures 3 and 4). Iron and sulphate reduction and subsequent formation of iron sulphides occurred locally as a result of the sediment's chemical heterogeneity and non-uniform flow (Greskowiak et al., in press). At the beginning of Stage 3, when the sediment below the pond became unsaturated, atmospheric oxygen entered the region leading to an increase of DO (Figure 3) and subsequent cessation of nitrate and manganese reduction (Figure 4). The iron sulphides that had formed during saturated conditions immediately became re-oxidised and led to a short peak of elevated sulphate concentrations at various depths below the pond (data not shown). During Stage 3, nitrate concentrations at some observation points

below the pond were considerably lower than the concentrations in the pond water and groundwater (Figure 3). However, throughout the entire Stage 3, the presence of atmospheric oxygen greatly enhanced the mineralisation of labile particulate organic carbon (POC) below the pond originating from seasonal algae blooms, which promoted the dissolution of calcite (Greskowiak et al., in press). As as result, DIC concentrations of the groundwater steadily increased relative to the surface water concentrations until the end of Stage 3 (Figure 4).

Winter cycle

During winter, DO was not entirely consumed in the presence of fully saturated conditions (Figure 3), presumably because very low surface water temperatures (Figure 2) caused a reduction in microbial activity. Therefore the consumption of electron acceptors such as oxygen and nitrate proceeded at much lower rates (see also, e.g., Prommer and Stuyfzand, 2005). Unlike in summer, aerobic conditions prevailed throughout the entire Stage 2 as a result of the lower water temperatures during winter (Figure 2). Neither consumption of nitrate nor production of dissolved manganese occurred. Instead, during Stage 2 and Stage 3, nitrate concentration increased by 1–5 mg/l within the seepage water on its path from the pond bottom to the groundwater monitoring well (Figure 3). Since ammonium concentrations within the pond water are generally small and range from 0 mg/l to 0.2 mg/l (unpublished data of BWB), additional sediment bound nitrogen might be oxidised under the prevailing aerobic conditions. The nitrogen source could either be exchange-

Figure 3. Concentrations of a. dissolved oxygen (DO),
b. nitrate (NO3-) in the pond water, groundwater
and at depths of 50 cm;
the numbers 1–4 refer to Stages 1–4.
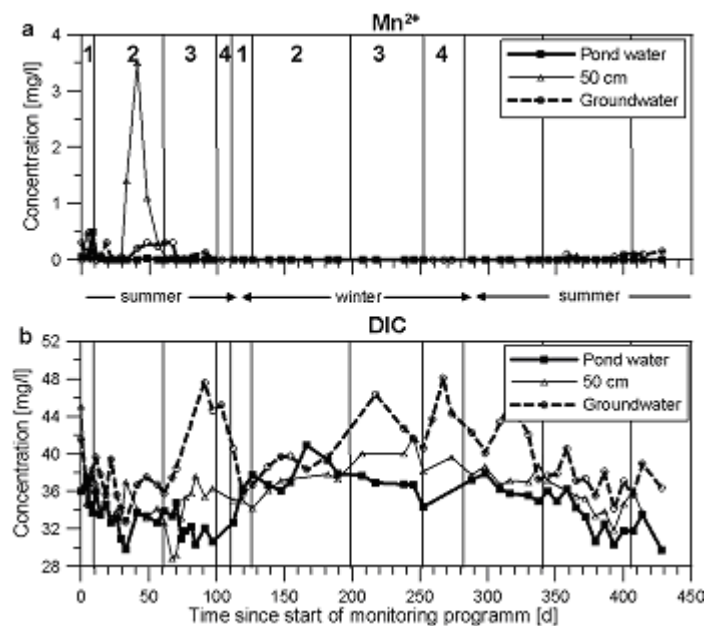
Figure 4. Concentrations of a. dissolved manganese (Mn^{2+}),
b. total dissolved inorganic carbon (DIC) in the pond water,
groundwater and at depths of 50 cm;
the numbers 1–4 refer to Stages 1–4.

able ammonium that was oxidised, as previously observed for other recharge basins (e.g., Bouwer et al., 1980), or it might also be organic nitrogen that was oxidised during breakdown of POC (e.g., von Gunten et al., 1991). The production of nitrate below the pond was only observable during winter when nitrate was not consumed simultaneously. During Stage 3, an enhanced production of inorganic carbon is observed, though not as intense as during the summer period (Figure 4).

CONCLUSIONS

This study investigated the geochemical evolution below an artificial recharge pond with respect to its transient hydraulic behaviour and seasonal temperature changes. The results show that during the summer period the spatial and temporal development of different redox environments is impacted considerably by the hydraulic conditions prevailing below the pond. During the entire winter period, the redox environment below the pond remains aerobic despite variable hydraulic conditions. As a side effect of decreased biodegradation rates, nitrification of sedimentary bound nitrogen is only observed during winter conditions. This could indicate that nitrification below the pond is less affected by temperature changes than nitrate reduction. Since local redox conditions are the key control for the attenuation of some trace organic compounds, the study provided the base for further investigations on the fate of such compounds during artificial recharge, as demonstrated for the case of PhAC's by the companion paper by Massmann et al. (2005).

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Quantifying biogeochemical changes during ASR of reclaimed water at Bolivar, South Australia

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Abstract

A modelling study was carried out to provide a process-based quantitative interpretation of the biogeochemical changes that were observed during an ASR experiment in which reclaimed water was injected into a limestone aquifer at a field-site near Bolivar, South Australia. A site-specific conceptual model for the interacting hydrodynamic and biogeochemical processes that occur during reclaimed water ASR was developed and incorporated into an existing reactive multi-component transport model. The major reactive processes considered in the model were microbially mediated redox reactions, driven by the mineralisation of organic carbon, mineral precipitation/dissolution and ion exchange. The study showed that the geochemical changes observed in the vicinity of the ASR well could only be adequately described by a model that explicitly considers microbial growth and decay processes, while an alternative, simpler model formulation based on the assumption of steady state biomass concentration failed to reproduce the observed hydrochemical changes. However, both, the simpler and the more complex model approach were able to reproduce the geochemical changes further away from the injection/extraction well. These changes were interpretated as a result of the combined effect of ion exchange, calcite dissolution and mineralisation of dissolved organic carbon.

INTRODUCTION

Aquifer Storage and Recovery (ASR, Pyne, 1995) is an increasingly popular technique to augment groundwater resources and secure and enhance water supplies. During ASR, physical, chemical and biogeochemical processes modify the water quality within the target aquifer. For instance, the injection of both oxic potable water and oxic nutrient-rich reclaimed water into an anaerobic aquifer can lead to a number of microbially mediated redox reactions (Stuyfzand, 1998; Vanderzalm et al., 2002), which in turn may trigger further geochemical reactions that have a considerable effect on the water quality and the composition of the aquifer matrix. (Eckert and Appelo, 2002). In order to design and operate efficient, sustainable and safe ASR schemes, a qualitative and quantitative understanding of those processes is important. Therefore, several field-scale investigations have been carried out over the past years, some of them specifically investigating the geochemical response to the injection of high quality water (e.g., Stuyfzand, 1998; Mirecki et al., 1998) but also of reclaimed wastewater (e.g., Valocci et al., 1981; Vanderzalm et al., 2002). However, numerical models which investigate the interacting hydrodynamic and biogeochemical processes during ASR (e.g., Valocchi et al., 1981, Saaltink et al., 2003; Prommer and Stuyfzand, 2005) and assist in the analysis and interpretation of observed field data are rather scarce to date.

As part of a larger-scale study (Dillon et al., 2005), Greskowiak et al. (submitted) developed a site-specific modelling framework that provides a process-based interpretation of the biogeochemical changes that occurred during a field experiment in which pre-treated, nutrient-rich reclaimed water was injected into a limestone aquifer at Bolivar, South Australia.

The present paper addresses the ubiquitous question of selecting an appropriate level of model complexity under such circumstances. To illustrate this issue we compare and discuss two alternative conceptual biogeochemical models and their numerical implementation in relation to their respective capability of describing the observed field data.

BACKGROUND

The Bolivar ASR site is located in the Northern Adelaide Plains, South Australia and used to investigate the viability of storage and recovery of reclaimed water intended to compensate the greater demand of irrigation water during summer (Vanderzalm et al., 2002). During the experiment, the reclaimed water was injected into a brackish limestone aquifer, which is separated from the overlying fresh water aquifer by a 7.5 m thick confining clay layer. Discontinuous injection of 250 ML took place between October 1999 and April 2000, followed by a storage period of 110 days. Subsequently, about 150 ML were recovered within the following 130 days (Figure 1). The injection took place over the entire depths of

Figure 1. Cumulative volume of injected water

the target aquifer, i.e., from 103 m to 160 m below the ground surface. Several multilevel wells have been installed at various radial distances from the ASR well, monitoring the geochemical evolution of the groundwater during injection, storage and recovery.

The hydrogeochemistry of the target aquifer is characterised by anoxic conditions and the aquifer matrix is composed of about 74% calcite, 18% quartz and small amounts of ankerite and hematite (Vanderzalm et al., 2002). The average total cation exchange capacity (CEC) is 20 meq/kg.

Pumping tests, flowmeter, temperature and anisotropy measurements revealed that the target aquifer could be classified into four stratified zones of similar thickness but of distinctly different permeability with an average horizontal hydraulic conductivity of about 3 m/d (Pavelic et al., 2001). Thereby, lateral flow occurs preferentially in two layers, referred to as Layer 1 and Layer 3.

Throughout the trial period, the quality of the injectant varied significantly with time. For example, oxygen and nitrate concentrations ranged between <0.02 and 0.32 mmol/L and < 0.0004 and 0.34 mmol/L, respectively. Correspondingly, ammonium concentrations varied between <0.02 and 2 mmol/L. However, dissolved organic carbon concentrations (DOC) were relatively constant with an average concentration of 1.40 mmol/L. The average total organic carbon concentration (TOC) was slightly higher (~1.51 mmol/L) and it is assumed that organic matter in particulate form (POC) compensated the concentration difference between DOC and TOC. The injection water is generally under-saturated with respect to calcite ($SI_{Calcite}$ = −1.44 to 0.13). As the chloride concentration of the injectant is approximately 50% of the ambient groundwater concentration, chloride was thought to be a suitable tracer for the identification of physical transport. More details on the monitored hydrochemistry are given in Vanderzalm et al. (2002).

GROUNDWATER FLOW AND REACTIVE TRANSPORT MODEL

In a first step a three-dimensional flow and conservative transport model of the ASR trial was set up and calibrated. In a second step the results were used to determine the groundwater flow within the most permeable layer (Layer 3) and to construct a computationally more efficient quasi-radial flow and transport model for this particular layer. The quasi-radial flow model formed the basis for the subsequent simulations with the reactive multi-component transport model PHT3D (Prommer et al., 2003). The reaction network considered in those simulations included all major ions, oxygen, one type of ion exchanger site, two forms of mobile organic carbon (i.e., DOC, POC), immobile organic carbon, four minerals (i.e., calcite, hematite, siderite and amorphous iron sulphide) and two microbial groups. The two microbial groups were defined as facultative aerobic/denitrifying bacteria and facultative iron/sulphate-reducing bacteria, respectively. The reaction stoichiometry of the redox reactions, which incorporate microbial growth and decay were adapted from Prommer et al. 2002 and linked to a standard Monod-type microbial growth model (e.g., Barry et al., 2002). The POC contained in the injectant was expected to rapidly become immobile in the close vicinity of the ASR well due to filtration (Skjemstad et al., 2002) and attachment was simulated using first order kinetics. From there it was assumed to solubilise and to form a continuous source of DOC. The solubilisation was simulated by a kinetic approach adapted from Kinzelbach et al., 1991. More details are given in Greskowiak et al. (submitted). For the present study two alternative conceptual biogeochemical models of differing complexity were investigated:

In alternative (A), the simplifying assumption of a steady state microbial concentration was incorporated and the mineralisation of DOC releases ammonia (Jacobs et al., 1988).

In alternative (B), microbial growth and decay were explicitly modelled and the release of ammonium was assumed not to be associated with the solubilisation of the filtered POC. Instead, ammonium was cycled through the occurrence of biomass growth and decay. That is, ammonium was included as a nitrogen source during biomass formation while biomass decay was assumed to release ammonium back into the aqueous phase.

For both alternatives, adjustable model parameters, i.e., the rate constants of the kinetic reactions were fitted such that the residual between simulated and observed concentrations was minimised. For this process the model-independent nonlinear parameter estimation program PEST (Doherty, 2002) was coupled to PHT3D.

RESULTS AND DISCUSSION

The simulation results of both calibrated alternatives (A) and (B) were compared with the hydrogeochemical data collected at the ASR well and the 50 m well. The simulation results indicated that both alternatives were capable of reproducing the key features of the hydrochemical changes which occurred at the 50 m well during injection, storage and recovery. In the model, the degradation of DOC is accompanied by the consumption of oxygen (not shown), nitrate (Fig. 2) and sulphate (Fig. 2), as measured in the field. Furthermore, the observed retarded ammonium breakthrough at the 50m well can be decribed accurately by the simulated ion exchange reactions (Fig. 2), while the increased concentrations of calcium (relative to nonreactive transport, see Fig. 2) result from the dissolution of calcite.

However, alternative (A) was unable to account for some of the highly dynamic changes of the local geochemistry observed in the close vicinity of the ASR well during the storage period. While the observed rapid increase of DOC was well matched by the simulations (Fig. 2), the increase of ammonia (Fig. 2), alkalinity (Fig. 2), calcium (Fig. 2) and dissolved iron (not shown) as well as the drastic decrease of sulphate (Fig. 2) and pH (Fig. 2) was not reproduced by alternative (A).

On the other hand, all of the geochemical changes observed at the ASR well could be adequately simulated by alternative (B), providing evidence that the observed hydrochemistry is strongly affected by the dynamics of microbial growth and decay. The model results suggest that in particular during the storage period, microbial decay, i.e., the mineralisation of highly degradable biomass, plays a key role and consumes considerable amounts of oxidation capacity, as discussed in Prommer et al. (2002). Correspondingly, inorganic carbon and protons are produced during decay, as was observed in the field. In the model this effect was caused by the decay of facultative aerobic/denitrifying bacteria (Fig. 2), which reduced sulphate (Fig. 2) and hematite (not shown) during the storage period.
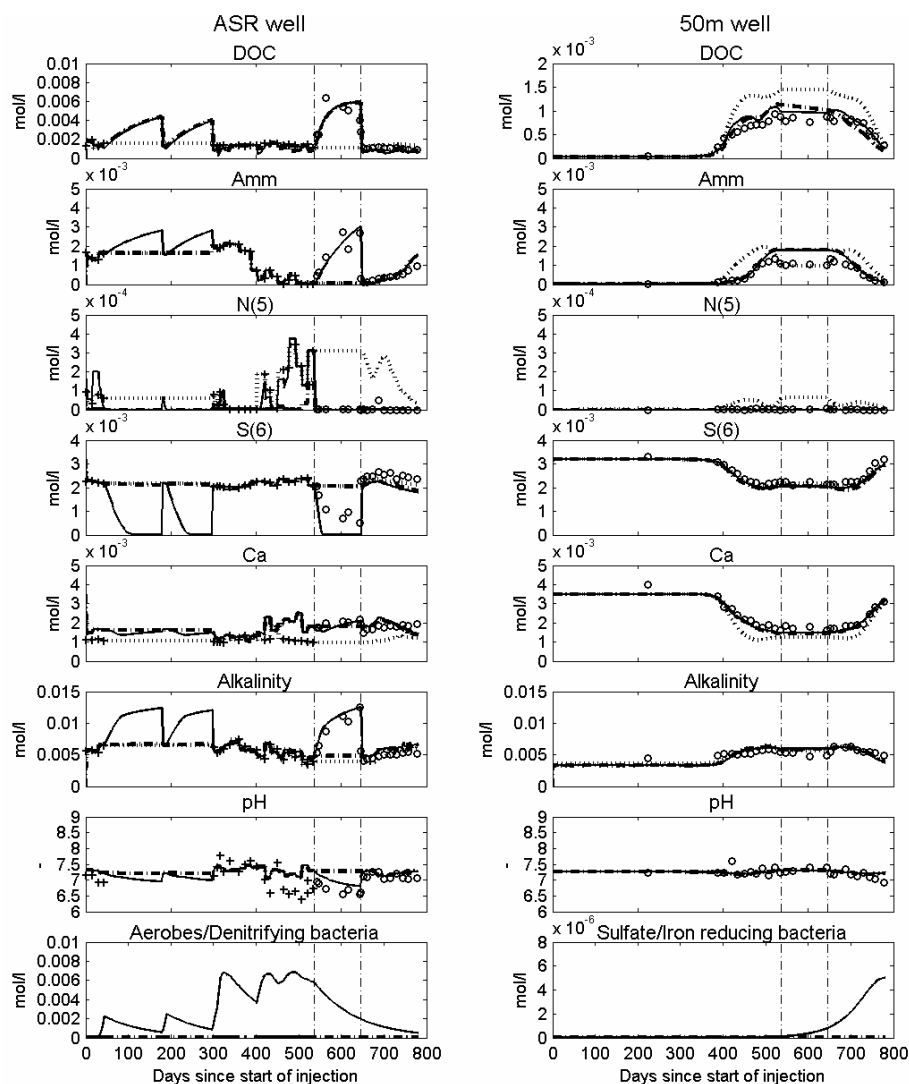
Figure 2. Concentrations of DOC, ammonia/ammonium, nitrate, sulphate, calcium, alkalinity, pH and bacteria at the ASR and the 50 m well. Concentrations of the injectant (crosses), observations (circles), alternative A (dash-dot lines), alternative B (solid lines) and non-reactive simulations (dotted lines) are shown.

CONCLUSIONS

The present study investigated two alternative conceptual biogeochemical models with respect to their capability of providing a quantitative and consistent description of the biogeochemical processes at a reclaimed water ASR research site in South Australia. The results indicate that an adequate simulation of the geochemical changes in the vicinity of the ASR well could be achieved by a more complex conceptual biogeochemical model, while geochemical

changes at larger distances were well-described by both models. This suggests that dynamic changes in bacterial mass may be important in interpreting near-well hydrochemical data from reclaimed water ASR schemes and should at least be considered during formulation of conceptual and numerical models. However, the identified processes and their respective parameters are highly nonlinear and may be subject to non-uniqueness. Thus, the possibility cannot be excluded that other conceptual models might describe the data equally well (Reichert and Omlin, 1997). Generally, the benefit of mechanistic multi-component reactive transport models is seen primarily in their capacity to constrain or reject hypotheses on interactions of physical, chemical and biological processes and to a lesser extent in their predictive capabilities.

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Case studies on water infiltration processes in the unsaturated zone with a multi-dimensional multiphase flow model

Song Pham-Van and Reinhard Hinkelmann

Abstract

In this contribution, we briefly introduce the model concept of two-phase flow in porous media including the governing equations as well as the numerical simulator MUFTE-UG. Case studies on water infiltration into the unsaturated zone and its further movement to the groundwater table are investigated. The results show the strong influences of a lens with a lower permeability and they emphasize the necessity of the two-dimensional approach.

INTRODUCTION

Water infiltration in the unsaturated zone and its further movement to the groundwater table is of major importance for aquifer recharge concerning water quantity and water quality. If complex aquifer systems which consist of different layers, low permeable lenses and preferential flow paths are considered, the flow and transport processes in the unsaturated zone are very complex and of course multi-dimensional, and they can considerably be influenced by small-scale heterogeneities.

The flow processes in the unsaturated zone are mostly modeled with a one-dimensional Richardson model concept in medium and large-scale domains. With this approach, lateral flow processes and their influences on the aquifer recharge cannot be described in an adequate way.

Multi-dimensional two-phase flow models consisting of the phases water and soil air simulate the flow processes in the unsaturated zone and the groundwater with the same model concept and include the horizontal spreading of infiltrating water. Up to now, such model concepts have hardly been used for large-scale hydrological questions, but for small-scale problems, for example in the context soil remediation or local contaminant infiltration.

TWO-PHASE FLOW MODEL CONCEPT FOR POROUS MEDIA

The model concept of two-phase flow in porous media assumes that the two fluids, here water (w) and air (a), are not miscible in each other and that mass transfer processes, for example evaporation or condensation, are negligible. The processes on the microscale are averaged to the REV scale (see Fig. 1).

The continuity equation for each phase $\alpha = w$, a reads as:

$$\frac{\partial(\phi\rho_\alpha S_\alpha)}{\partial t} + div\left(\rho_\alpha \underline{v}_\alpha\right) - \rho_\alpha q_\alpha = 0 \tag{1}$$

The validity of the extended *Darcy* law is assumed:

$$\underline{v}_\alpha = \frac{k_{r\alpha}}{\mathrm{\mathì}_\alpha}\mathrm{K}\left(grad\, p_\alpha - \tilde{n}_\alpha \underline{g}\,\right) \tag{2}$$

In these equations, ϕ stands for the porosity, S for the saturation, ρ for the density, t for the time, q for a sink/source term, $\underline{v}$ for the *Darcy*-velocity vector, k_r for the relative permeability, μ for the dynamic viscosity, K for the permeability tensor, p for the pressure and g for the gravity vector.

Two further conditions are required to close the system of equations. The saturations, which describe the ratio of the fluid volume to the volume of the void space, add up to one:

$$S_w + S_a = 1 \qquad (3)$$

At the interface between the two fluids, a jump in the pressure occurs which is given by the capillary pressure p_c:

$$p_a - p_w = p_c \qquad (4)$$

The capillary pressure and the relative permeability are (highly) non-linear functions of the saturations and must be determined experimentally; they are called constitutive relationships. The most general parameterizations are given by Brooks, Corey and Van Genuchten; an overview is found in Helmig (1997).

The resulting equations are strongly coupled, highly non-linear and of mixed parabolic/hyperbolic type. A so-called p_w-S_a formulation is used i.e. that the water pressure and the air saturation are chosen as primary variables. The equations are discretized by an implicit *Euler* scheme in time and a so-called box method in space. The box method is a node-centered Finite-Volume Method using a fully upwinding technique for the advection terms. Further information is found in Helmig (1997) and Hinkelmann (2005).
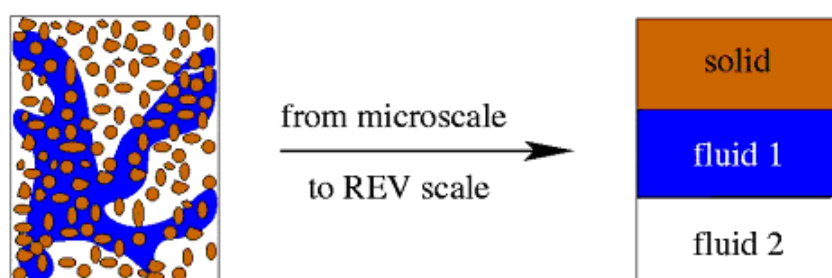
Figure 1. REV concept

THE NUMERICAL SIMULATOR MUFTE-UG

The two-phase flow model concept is implemented in the modeling system MUFTE-UG that is a combination of MUFTE and UG (see Fig. 2). MUFTE stands for MUltiphase Flow, Transport and Energy model. UG is the abbreviation for Unstructured Grids. The numerical simulator is developed in the University of Stuttgart, TU Berlin (part MUFTE) and the University of Heidelberg (part UG). This software toolbox mainly contains the physical model concepts and discretization methods for isothermal and non-isothermal multiphase / multicomponent flow and transport processes in porous and fractured-porous media, while UG provides the data structures and fast solvers based on parallel, adaptive Multigrid Methods. Further information is found in Helmig (1997), Breiting et al. (2002) and Hinkelmann (2005).
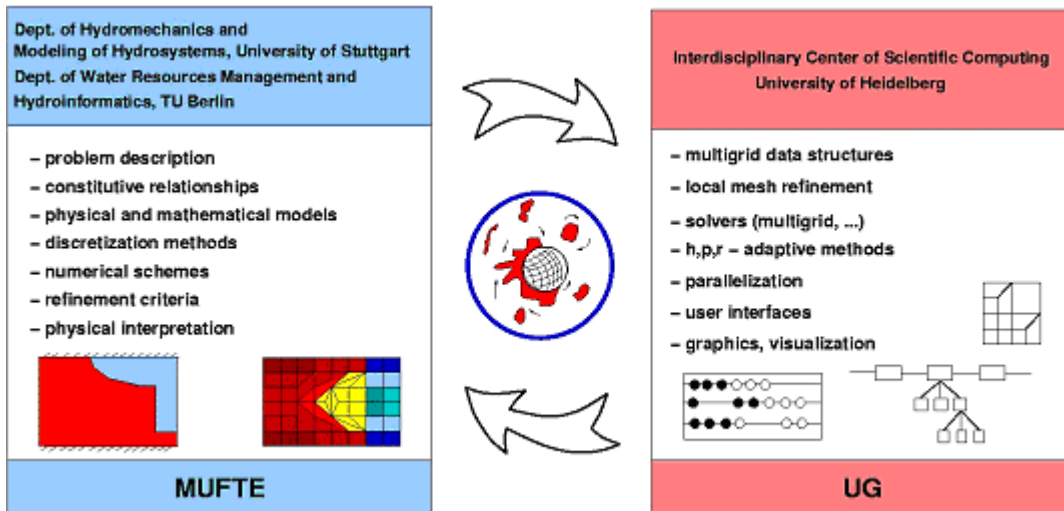
Figure 2. Numerical simulator MUFTE-UG

APPLICATIONS

A simplified typical aquifer for the area of Berlin is chosen for the test cases (see Fig. 3). The actual domain has a depth of 50 m and a length of 100 m. The groundwater table is 35 m below the surface of the earth. We will investigate the water infiltration processes from rainfall in to a homogeneous system and in to the same system with a low permeability lens.
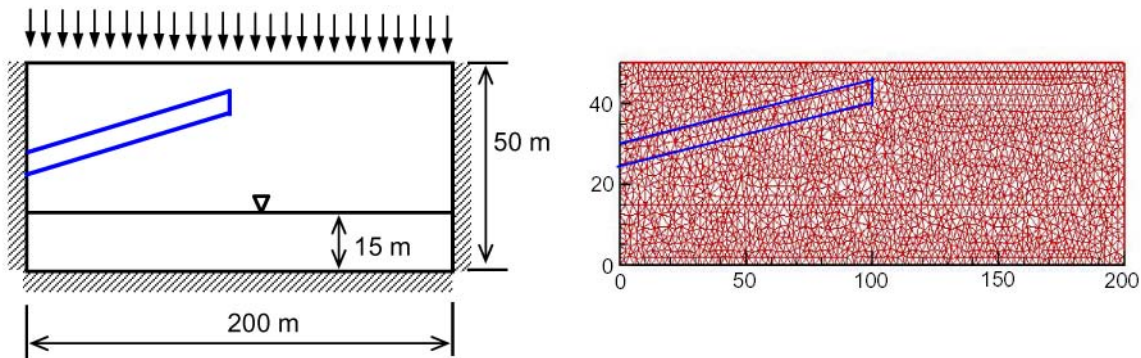
Figure 3. Sketch of the aquifer system (left) and computational mesh (right)

The fluid properties of water are $\rho_w = 10^3 \, kg/m^3$, $\mu_w = 10^{-3} \, Pa$ and of air are $\rho_w = 1.2 \, kg/m^3$, $\mu_w = 1.02 * 10^{-5} \, Pa$. The physical parameters are set to $\phi = 0.40$, the domain has a permeability $K = 10^{-10} \, m^2$, this correspondents to a hydraulic conductivity $K_f = 10^{-3} m/s$, and the lens has a lower permeability $K = 10^{-8} \, m^2$, this correspondents to a hydraulic conductivity $K_f = 10^{-1} \, m/s$. The constitutive relationships of Brooks Corey are chosen with the parameters $\lambda = 2.0$, $p_d = 1000 \, Pa$. Additionally, residual water and air (or gas) saturations must be given $S_{wr} = 0.1$, $S_{ar} = 0.01$.

As initial conditions, the gas saturation is set to $S_{nw} = 0,9$ in the unsaturated zone and the system is fully saturated with water in the domain filled with groundwater. On the upper side, the boundary is open, and the water level i.e. the water pressure is prescribed with a height of 20 cm. On the other sides, the boundaries are closed.

Figures 4, 5, 6 show the different saturation fields for the two test cases. In the left side of the result figures, first, the infiltration front is observed (see Fig. 4), then the infiltration front reaches the capillary fringe (see Fig. 5) and

finally, the infiltration front is melting with the groundwater table (see Fig. 6). In the right side of the result figures, the interference of the infiltration processes caused by the lens can be seen. The infiltration processes are decelerated and the flow is directed to another region. Such a heterogeneities has an important impact on the behavior of the subsurface flow. In this case, 40% of the water which infiltrated into the system is trapped in the domain above the lens.

Figure 7 shows the Darcy velocity fields of the domain with lens (right) and without lens (left) at time t = 484 min. In the left, we can see one-dimensional infiltration processes, while these processes are two-dimensional in the right.
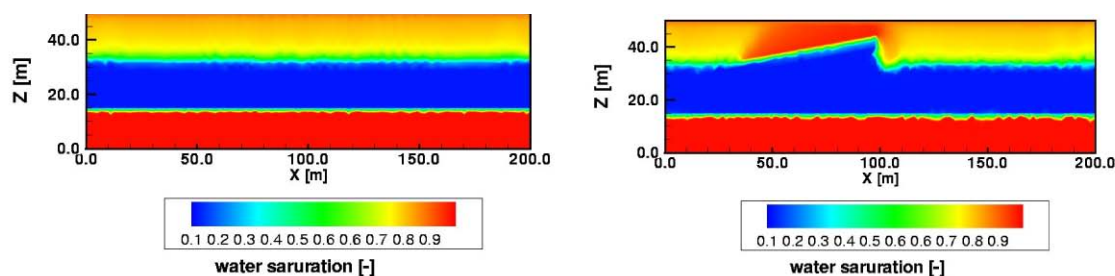
Figure 4. Water saturation fields at t = 183 min, homogeneous case (left), case with lens (right)
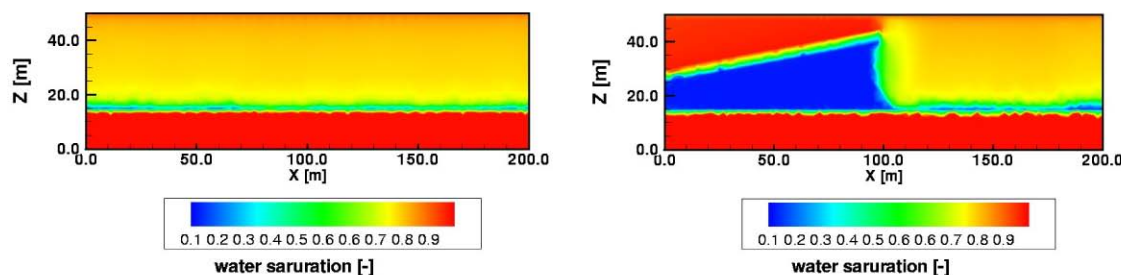
Figure 5. Water saturation fields at t = 429 min, homogeneous case (left), case with lens (right)
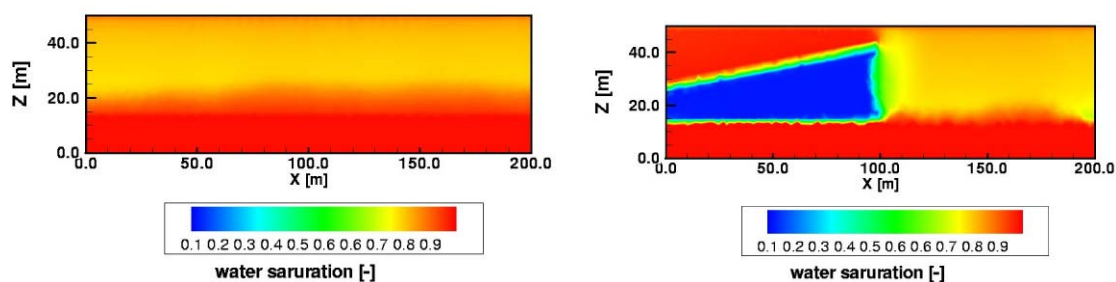
Figure 6. Water saturation fields at t = 484 min, homogeneous case (left), case with lens (right)
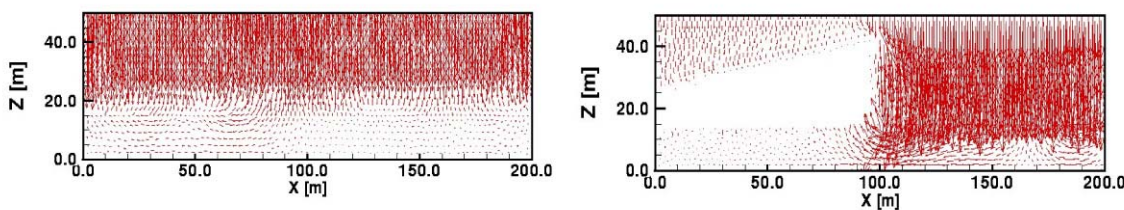
Figure 7. Darcy velocity at t = 484 min, homogeneous case (left), case with lens (right)

CONCLUSIONS

The two-phase flow model MUFTE-UG is capable to simulate two-dimensional water infiltration processes into heterogeneous aquifers. Case studies on water infiltration into the unsaturated zone and its further movement to the groundwater table are investigated. The results show the strong influences of a lens with a lower permeability and they emphasize the necessity of the two-dimension approach.

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On the construction
of flowpath vector fields

E. Holzbecher

Abstract

A software coupling with interfaces, written in MATLAB, is presented, which can be applied to produce flowpaths as vector fields from a MODFLOW model. The main task of flowpath calculation is performed by MODPATH, a MATLAB module transforms the results in a format which can be used by other programs. The details are outlined in the paper. Another task, which can also performed by a MATLAB module, is to transfer flowpath starting positions from Processing MODFLOW's PMpath to MODPATH.

Keywords

Groundwater modeling, post-processing, visualisation, particle tracking.

INTRODUCTION

MODFLOW (1988) probably is the most used code for groundwater models worldwide. Several software shells have been wrapped around MODFLOW, making it even more popular: Processing MODFLOW (2001) in Europe, VisualMODFLOW (2005) in Canada and GMS (2005) in the United States, just to name the most prominent products. Using these packages, the most common pre- and postprocessing steps are simpler to perform and the presentation of model results becomes visually more impressing. Nevertheless the connection with other programs is still sometimes awkward or impossible. A prominent example is that flowpaths or streamlines, calculated by ProcessingMODFLOW as post-processing, can not be transferred to GIS-programs,[1] which are often used for data representation within application projects. In the paper a solution for this problem is given, connecting the simulation code MODFLOW (1988), the post-processor MODPATH (1994) and the visualisation tool SURFER (2002). The proposed procedure can be applied in two and three space dimensions.

METHODS

Different types of software are included in the presented approach. Figure 1 provides a schematic view on the various connections. Independent codes included in the approach are Processing MODFLOW (2001), MODPATH (1994), SURFER (2002) and MATLAB (2002). Processing MODFLOW is a graphical user interface, from which the simulator code MODFLOW (1988) as an integrated part is called. MODPATH is a postprocessor for flowpath computation (not part of Processing MODFLOW), with the advantage that flowpath and traveltimes are stored in files for further processing. Such processing is performed by a module, implemented in MATLAB, here for the graphical representation of flowpaths as vectors in the SURFER program.

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1. Other codes, for example VisualMODFLOW (2005), allow an easy connection with GIS programs.

SURFER is one of the most popular programs used for GIS and data representation in the geosciences. The package is able to represent bitmaps and vector graphics. The advantage of the vector format is that the graphical resolution of an object is not effected by changing the scale of the entire plot. The described connection with SURFER at the end of the line of codes is an example. A similar method would work with other GIS-tools at the end.

The study site is one of three recharge ponds surrounded by about 44 production wells located near Lake Tegel, Berlin, Germany. Its infiltration surface extents over an area of approximately 8,700 m^2 and is located 3 m below the adjacent ground surface. Surface water of Lake Tegel is discharged into the pond after it has passed a microstrainer. With time, clogging processes at the pond's floor lead to a continuous decrease of infiltration rate, i.e., from 3 m/d to as low as 0.3 m/d. As soon as the infiltration rate becomes lower than 0.3 m/d, the site operator Berliner Wasserbetriebe (BWB) abrades the clogging layer to restore the original hydraulic conductivity of the bottom sediments. This operational cycle is typically repeated every 3–4 months.

The sediments in the adjacent area of the pond are of Quaternary age and consist of fluvial and glacio-fluvial, medium sized sand deposits. Fragments of an up to 5m thick till layer are locally found in depths of approximately 15 m below the ground surface (Pekdeger et al., 2002). The hydraulic conductivities of the aquifer sediment are about 10–100 m/d. A more detailed description of the sediment hydraulic properties is given in Pekdeger et al. (2002).

The organic carbon content of the sediment below the pond is highly variable and ranges from 0.2 g/kg to 20 g/kg. Reducible forms of iron and manganese minerals are found in concentrations of 0.2–1.2 g/kg and 0.01–0.1 g/kg, respectively. Total sulphur concentrations are in the range of 0.1–2.1 g/kg and are highest in organic-rich layers. A more detailed description of the geochemical composition of the sediment below the pond can be found in Greskowiak et al. (submitted).

The procedure is explained in more details in the following. A data flux diagram is depicted in Figure 1. MODFLOW performs the numerical calculations and stores it in several output files. Relevant for particle tracking are the files: *head.dat, budget.dat, wel.dat* (in case of wells) *rch.dat* (in case of groundwater recharge).

Before the start of MODFLOW some further options have to be set to enable MODPATH to perform. MODPATH is a particle tracking tool that requires some additional special output files from MODFLOW. These are *mpath30* and *main30.dat*. Based on all mentioned output files MODPATH calculates flowpaths. The semi-analytical method is described by Pollock (1988) and was independently implemented by Holzbecher in the FAST code, as described in Holzbecher (1996a, 1996b, 2002). MODPATH allows tracking of streamlines from steady-state and pathlines from unsteady simulations.

Particle tracking is based on start- or endpositions (depending on tracking direction; forward tracking requires start positions, backward tracking end positions). MODPATH allows the user to implement those positions in a dialog before the computing of the flowpaths. Nevertheless, if there are lots of flowpaths to be drawn it is more appropriate to read the positions from file, which is implemented in MODPATH as an option.

Major output of MODPATH is a *pathline* file, which contains the pathlines as multi-lines, connecting calculated positions on the flowpaths by straight lines. Moreover it is possible to store locations for markers on the flowpath to mark traveltimes; the file also has the name *traveltime*.

The output files of MODPATH can not be used in a GIS software directly, but have to be transformed into an appropriate format. For this purpose a MATLAB module is implemented. The flowpath multi-line is read and stored in a format, which can be read by SURFER as an vector element. In Figure 1 new (extra) modules are marked. The
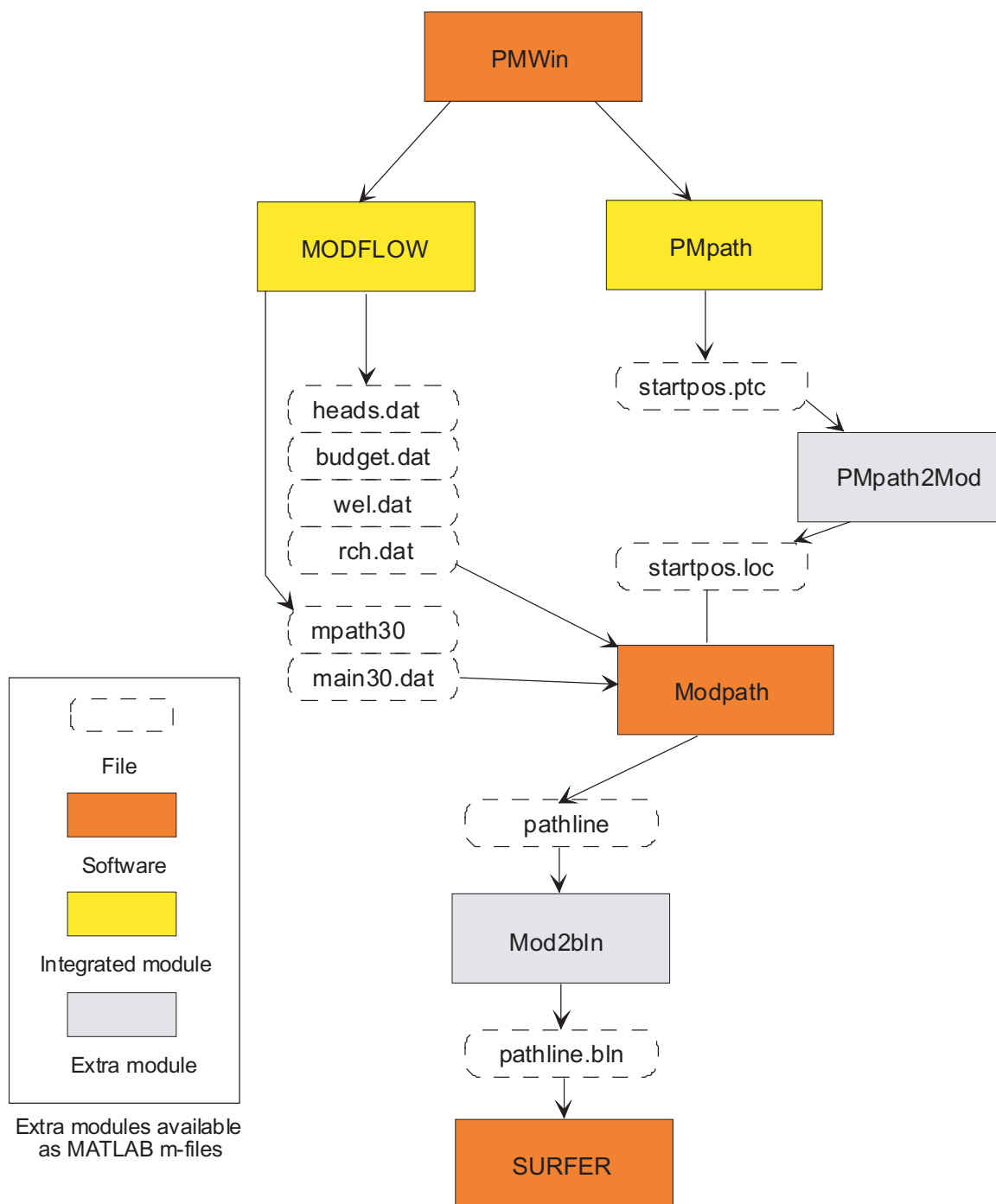
Figure 1. Data flux diagram for the software coupling to create flow path vector plots

module that makes MODPATH pathlines readable for the SURFER software, is called *Mod2bln*. The module itself is written using MATLAB's m-language.

Two additional modules have to be mentioned. The module *Pmpath2mod* is included in Figure 1, connecting the PMPath (2001) postprocessor, which is included in Processing MODFLOW, with MODPATH. It performs the special task to transform start/end positions for flowpaths from one format into another. Another module, *Mod2dat*, transforms the *traveltime* output file of MODPATH into a format which can be read by the visualisation software. Both modules are also implemented using MATLAB.

RESULTS

The software connection has been used during the interdisciplinary NASRI research project (Natural and Artificial Systems for Recharge and Infiltration) dealing with river bank filtration processes and during the KORA research project (Kontrollierter natürlicher Rückhalt und Abbau) dealing with natural attenuation processes in the subsurface.

Figure 2 provides an example in which all programs and modules, which are shown in Figure 1 have been involved. The figure shows flowpaths in a system of two aquifers in the city of Berlin, The aquifers are partially connected, partially separated by highly impermeable boulder clay, a characteristic situation in alluvial terrain, formed by
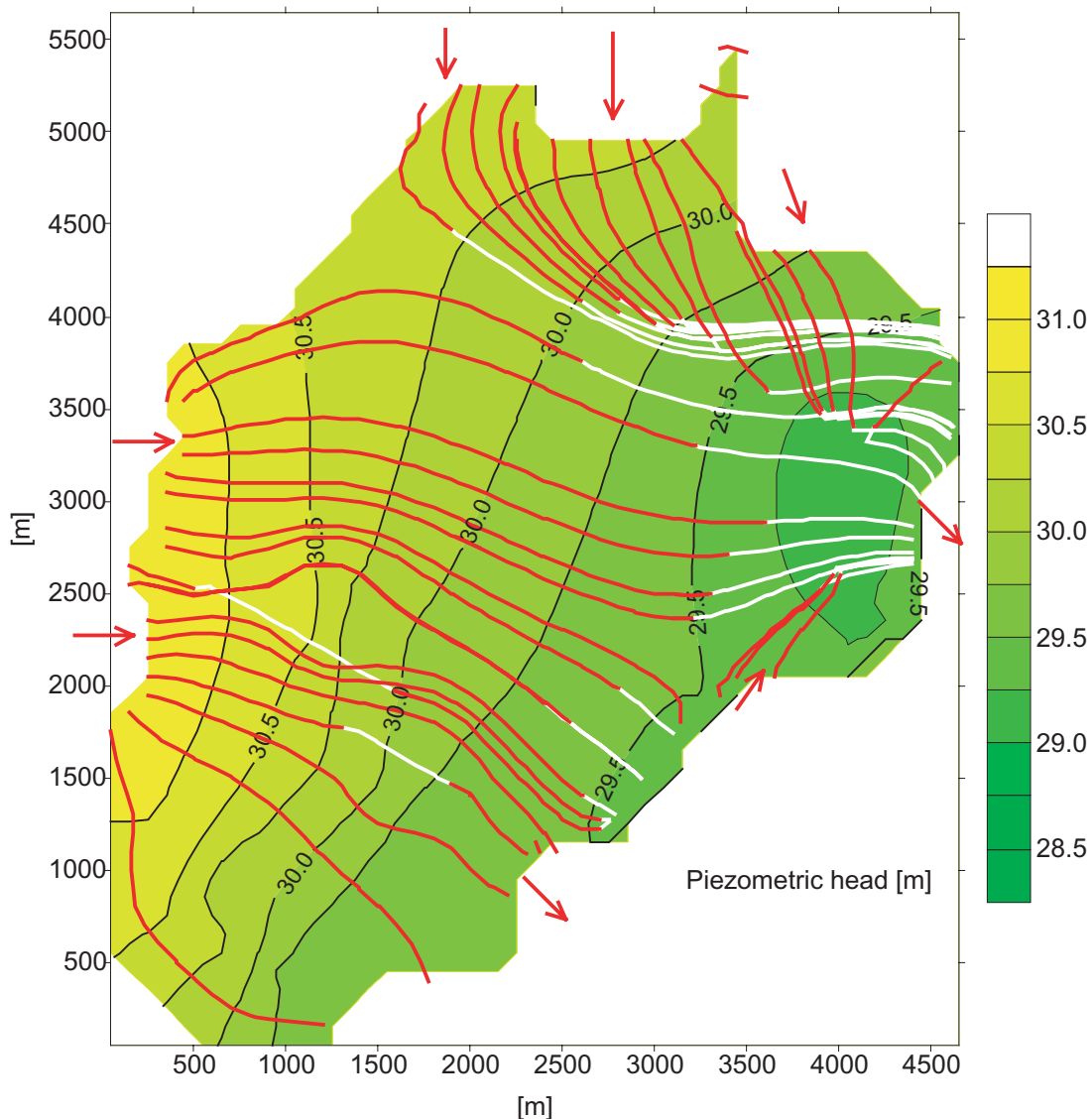
Figure 2. Example plot for an aquifer with bank filtration in Berlin-Kladow; black contour lines and fill pattern show head distribution, corresponding values can be taken from the bar on the right side; dark pathlines are within the upper aquifer; light flowpaths within the lower aquifer; directions are indicated by arrows; the south-east boundary of the depicted model region is the shoreline of Lake Wannsee (Berlin); while groundwater enters the lake in the southern part of the shoreline, there is bank filtration in the middle part; in the northern part the flowpaths show that groundwater 'dives' into the lower aquifer to pass below Lake Wannsee to the water works well galleries on the Eastern shore of Lake Wannsee (not depicted in the figure).

glacial activities during several ice-ages. The upper aquifer is connected to Lake Wannsee in the East (see Figure 2), where two bank filtration galleries of the Berlin Water Works are located.

Flowpaths in Figure 2 clearly show the complex pattern within the system, which is very much influenced by the extension of the nearly impermeable boulder clay layer within the aquifers. Starting points for the flowpaths in Figure 2 were selected in the upper aquifer along the inflow boundaries. Flow pattern in the upper aquifer follows the head contour field, which is visualised in the background and the colour bar on the side. When the flowpath reaches the edge of the boulder clay, it dives down into the second aquifer, where is may follow a completely different path. Such behaviour is most obvious in the upper left part of the model region, where some flowpaths make a turn of more than 90°. A similar behaviour strong influence of the separating layer was observed in other modeling projects in Berlin, for example in Lake Tegel and on the Eastern shore of Lake Wannsee.

CONCLUSIONS

Flowpaths as vector fields, used for an optimal visualization of groundwater flow in the subsurface, can be constructed for use in GIS programs. Model output can be presented together with maps or measured data fields. The advantage of the vector representation is that the graphic is scalable without loss of resolution.

Another advantage of the described procedure is that flowpaths are obtained as one-dimensional line-elements, allowing further post-processing. Thus it is possible to treat even reactive transport models along the flowpath, which requires much less computer resources than a 2D or 3D model. However the simplified method is restricted to situations in which transversal dispersive transport can be neglected. Nevertheless the described method opens the way to examine complex biogeochemical systems in combination with transport more easily.

It is intended to combine the three different modules, which have been implemented, in one graphical interface. In one window it will be possible to select, which of the modules is to be performed, and to set all relevant options.

ACKNOWLEDGEMENTS

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Simulation of bacteriophage populations during sub-surface passage

E. Holzbecher, H. Dizer, J. López-Pila, R.G. Szewzyk and G. Nützmann

Abstract

In several slow-sand-filter experiments the behaviour of phages during the subsurface passage was measured and modelled. Here the focus is on the effect of the velocity. The observed data show a strong effect of decreasing filter efficiency with increasing velocity. Using a modelling approach, which is based on the transport differential equation, the theoretical influence of velocity on filter efficiency is examined. Finally an extrapolation of the results to large scale filtration units or bank filtration processes is attempted.

Keywords

Phages, transport, modelling, dimensionless formulation, filters.

INTRODUCTION

In urban areas surface water is often contaminated with pathogens, mainly due to the discharges of waste water treatment plants or to run-off water from agricultural fields. The passage of surface water through soil by bank filtration or slow sand filtration is an effective way to eliminate pathogens and produce safe drinking water. Both approaches have been applied world-wide for more than a century in the treatment of surface water for drinking water supply (Jekel 2002, Grischek et al. 2003).

When the surface water enters the sub-surface, either due to natural conditions or induced by an imposed outer regime, the survival conditions for microorganisms, and the physical circumstances acting upon them, change. The retention of suspended and soluble contaminants of surface water by sand filtration bases on the physical (mech-anical) filtration, chemical adsorption or detachment and biological degradation on the filter matrix (Ginn et al., 2002). The size, molecular weight and polarity of contaminants, the porosity and polarity of sandy soil grains, as well as the quantity and biodiversity of micro organisms in filter bed determine the affectivity of retention processes (Azadpour-Keely, 2003). The flow rate regulates the contact time for all physical, chemical and biological inter-actions between the surface structures of solid filter matrix and soluble or suspended particles of percolated water (Fundenberg et al., 1981; Schijven and Hassanizadeh, 2000).

Due to its large size, the efficiency of sand filtration for bacteria and protozoa has been reported to be higher than that for viruses (Ginn et al., 2002). Viruses are able to migrate considerable distances through soils depending on not only their small size but also to their adsorption characteristics and degree of inactivation in capillary water (Keswick and Gerba, 1980). Processes like adsorption and inactivation that determine removal of viruses during subsurface passage apply therefore to all pathogens, albeit to a different extent (Schijven and Hassanizadeh, 2000).

As the velocity seems to play a crucial role, its influence is in the focus of this paper. The effect of velocity is exam-ined theoretically, by numerical and analytical simulation and compared to the outcome of experiments in slow sand filters, performed within the interdisciplinary NASRI research project (Natural and Artificial Systems for Recharge and Infiltration) concerning all aspects of river bank filtration.

SIMULATION

The following analysis shows that the complex interaction of the different processes may be lumped into a small set of dimensionless parameters. Modelling is based on solutions of the one-dimensional transport equation, which is a partial differential equation for the phage population c, taking into account the processes of advection, dispersion, equilibrium sorption and decay:

$$R \frac{\partial c}{\partial t} = \frac{\partial}{\partial x} \alpha_L v \frac{\partial c}{\partial x} - v \frac{\partial c}{\partial x} - \lambda R c \tag{1}$$

In the differential equation (1) four parameters appear: the retardation R, the longitudinal dispersion length α_L, the interstitial (real) flow velocity v and deactivation rate λ, which is valid in the fluid and the solid phase. Equation (1) can be derived from the formulation, given by Schijven and Hassanizadeh (2000) under the assumption that attachment and detachment processes are fast. The retardation factor always exceeds 1. For the following it is assumed that parameters in equation (1) are constant. Additional parameters are usually introduced by the boundary and initial conditions. In order to classify the solution space, it is convenient to reduce the number of unknowns. The first step is to non-dimensionalize the equation. The unit length is the length L of the model region. Using the notation $\xi = x / L$ and the inner derivative $\partial / \partial x = (\partial \xi / \partial x) \partial / \partial \xi = (1 / L) \partial / \partial \xi$ one obtains:

$$R \frac{\partial c}{\partial t} = \frac{\alpha_L v}{L^2} \frac{\partial^2 c}{\partial \xi^2} - \frac{v}{L} \frac{\partial c}{\partial \xi} - \lambda R c \tag{2}$$

Analogously the time transformation can be used to non-dimensionalize time also. One obtains:

$$\frac{v}{L} \frac{\partial c}{\partial \tau} = \frac{\alpha_L v}{L^2} \frac{\partial^2 c}{\partial \xi^2} - \frac{v}{L} \frac{\partial c}{\partial \xi} - \lambda R c \tag{3}$$

which by multiplication with L/v leads to a differential equation with non-dimensional coefficients:

$$\frac{\partial c}{\partial \tau} = \frac{1}{Pe} \frac{\partial^2 c}{\partial \xi^2} - \frac{\partial c}{\partial \xi} - Da_2 c \tag{4}$$

where the dimensionless Péclet-number Pe and (second) Damköhler number Da_2 are defined as:

$$Pe = \frac{L}{\alpha_L} \qquad Da_2 = \frac{\lambda R L}{v} \tag{5}$$

The concentration variable c is usually non-dimensionalized by introducing the normalized concentration $\theta = (c - c_0) / (c_1 - c_0)$ with appropriate values for c_0 and c_1. For situations with moving fronts it is convenient to use the background concentration for c_0 and the inflow concentration for c_1. The linear transformation does not change the validity of the transport equation:

$$\frac{\partial \theta}{\partial \tau} = \frac{1}{Pe} \frac{\partial^2 \theta}{\partial \xi^2} - \frac{\partial \theta}{\partial \xi} - Da_2 \theta \tag{6}$$

This is an equation with dimensionless parameters and dimensionless variables only. Instead of seven parameters R, α_L, v, λ, L, c_0 and c_1 in the original formulation only two parameters appear in equation (6), which simplifies the discussion of parameter influences. Note that the velocity v appears in the Damköhler number only. According to formula (5) the degradation, as represented in Da^1 as combination of relevant parameters, is proportional to the inverse of v. If no other parameter changes a decrease of deactivation proportional to $1/v$ can be expected.

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1. In the following the abbreviation of the Damköhler number is written without subscript, as it is always the second Damköhler number, which is referred to. A derivation of an equivalent dimensionless formulation using the first Damköhler number is given by Holzbecher (in preparation).
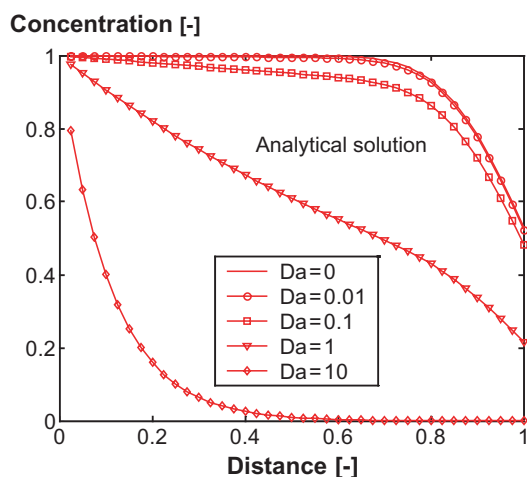
Concentration [-]

*Figure 1. Characteristic solutions
for the concentration profile for Péclet number
Pe = 100 and different Damköhler numbers
at dimensionles time = 1*

Figure 1 presents the characteristic concentration profiles along the flowpath for Pe = 100 and various Damköhler numbers. Clearly for $Da < 0.1$ only slight differences to the zero-degradation situation ($Da = 0$) can be observed. For $Da = 1$ the front is visible only as a bend near to the outlet. For $Da = 10$ no front can be recognized anymore. The shown profile is identical to the steady state of the system.

Figure 1 depicts analytical solutions of the transport equation (6), calculated using MATLAB (2002). Analytical solutions with initial and boundary conditions for an advancing front in an semi-infinite strip are implemented in the formulation, given by Wexler (1992). The MATLAB module can also be used for parameter estimation. An automatic procedure for inverse modelling is implemented using the MATLAB optimisation toolbox. More details of the module are provided by Holzbecher (submitted). The optimisation is performed using the original parameters and variables from equation (1) and utilizes numerical solutions instead of analytical solutions.

Modelling of phages turns out to be a challenge for the described model, as population gradients for phages are much steeper than for chemical species. High temporal and spatial fluctuations in the data make it impossible to use an automatic parameter estimation procedure – even after smoothing original data (using splines). The parameters obviously change during the underground passage and thus have to be evaluated from one breakthrough curve to the next anew.

It is important to note that an expected efficiency of bank filtration systems is obtained using the steady state of the analytical solution for equation (6), given by the formula

$$\theta(\xi) = \frac{\mu_2 \exp(\mu_2)\exp(\mu_1\xi) - \mu_1 \exp(\mu_1)\exp(\mu_2\xi)}{\mu_2 \exp(\mu_2) - \mu_1 \exp(\mu_1)}$$

$$\text{with} \qquad \mu_{1,2} = \frac{1}{2}Pe \pm \sqrt{\frac{1}{4}Pe^2 + Da_2}$$

(7)

(Holzbecher, in preparation). The formula is derived using standard methods for ordinary differential equations with constant parameters. Based on equation (7) combinations of Péclet- and Damköhler-numbers can be computed, for which an aimed rate of total deactivation, say in terms of log-units, can be achieved.

EXPERIMENTS

The removal of pathogen viruses under field conditions can not be studied easily because the contamination levels with viruses are often not high enough to be followed up along the filters. On the other hand, seeding pathogenic viruses in the filtration fields poses too high a risk for humans. Therefore, bacteriophages, viruses which attack bacteria and are not pathogenic for humans, were taken as surrogates for human pathogenic viruses, since both viral arts have similar morphological and structural properties (Haveelaar, 1993). We therefore seeded somatic and f specific coliphages (two kind of phages with different entry mechanisms into the bacterial bodies) and carried several filtration experiments.

Coliphages infect only *E. coli* species as host bacteria. The infection of host bacteria occur not in the aquatic environment, only in the intestinal tract of human and animals. Therefore they can be used as indicator for faecal contamination of surface water. Both test coliphages are wild strains isolated from Teltow Canal, mean wastewater discharger in Berlin and high resistance in the aquatic environment. Coliphage 138 is a F+phage and to be cultivated and detected only in the host bacterium culture of *E. coli* K13. As the somatic coliphage 241 infect only *E. coli* WG5 as host organism and can be detected in a fresh culture of E. coli WG5.

Surface: ca. 40 m^2
Surface water body: ca 40 m^3
Flow rate: 120 cm/d, 3000 L/h

Culture suspension of Coliphage138 and 241

samples of surface water

$10^3 - 10^4$ pfu /ml

samples from 10 cm depth

1 cm/h

samples of effluent
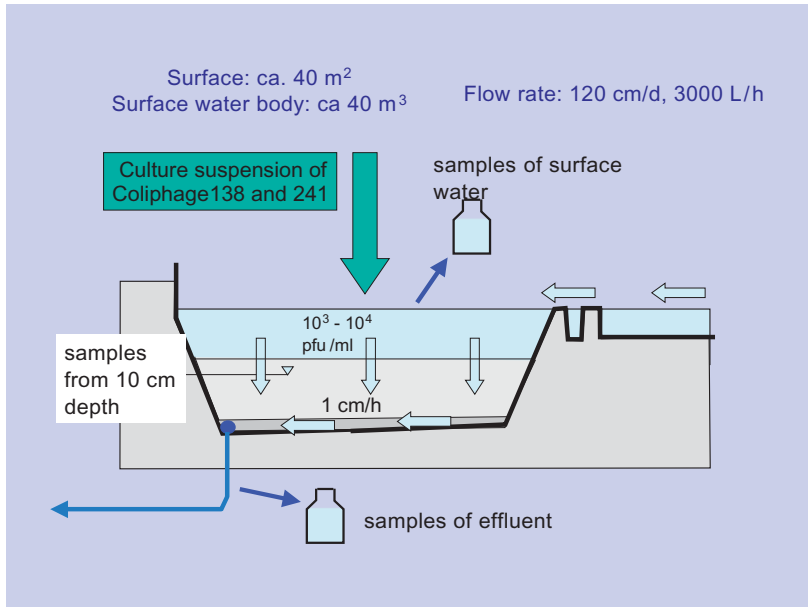
Figure 2. The slow-sand-filter set-up,
in which the phage experiments were performed

The experiments were performed in slow sand filters located at the experimental field station Marienfelde of the Federal Environmental Agency (UBA) of Germany. The chosen sand filtration pond in Marienfelde was a semi-technical plant with a quadratic filtration surface of 60,4 m^2 filled with sand. A water column of 45 cm height was constantly hold over the sand surface. A sketch of the set-up is given in Figure 2. The porosity of the sand was 0.335. The filtration pond was fed with groundwater extracted in a close-by located water work.

The experiments were each started by suspending a high amount of phages into the water standing over the sand ('sluggish pulse') and letting afterwards the water percolate at the respective flow rate. Three filtration rates were chosen at 60, 120, and 240 cm/day respectively, corresponding to true filtration velocities of ca. 180, 380, and 720 cm/day. Before each experiment, the flow rate was adjusted to the desired velocity and run during 4 weeks previously to the experiment.

RESULTS AND DISCUSSION

The slow sand filtration process at a flow rate of 60 cm/d led to a breakthrough of 0,02% and 0,4% for f-specific and somatic coliphages 138 and 241, respectively. An increase of flow rate to 120 cm/d increased the virus breakthrough of both coliphages to 0,2% and 2,4%. At a rate of 240 cm/d the breakthrough of the test organisms increased drastically; to 33% for the phage 138 and to 57% for the 241.

In accordance with the kinetics expected after a sluggish pulse, the initial filtrate fractions contained the highest amount of viruses (Fig. 3). The cumulative breakthrough of the viruses was calculated from the concentration in the single fractions and from their volumes. After an infiltration time of 12 h at all flow rates tested, the breakthrough curve reached a stationary value that did not change considerably during the further percolation time of 150 h. The retention or breakthrough rate of test organisms after 12 h percolation time was therefore taken as a reference value and used for extrapolation of virus removal during further percolation time through the filtration path.

The MATLAB model was applied for the simulation of the slow-sand-filter experiments with phages 138 and 241 for all three experiments. The result for phages 138 in the slow velocity experiment is shown in Figure 3 as an example. The major breakthrough of the phages is represented quite well in the simulations. But the irregular shape of the inlet concentration remains visible in the simulated results, which leads to an underestimation of concentrations in the first part of past-front results, and an overestimation afterwards. The first automatic fit was improved several times by manual selection of better starting values for the optimisation procedure. For the shown case the optimized values are $R = 1.3$ and $\lambda = 0.89$ 1/h.
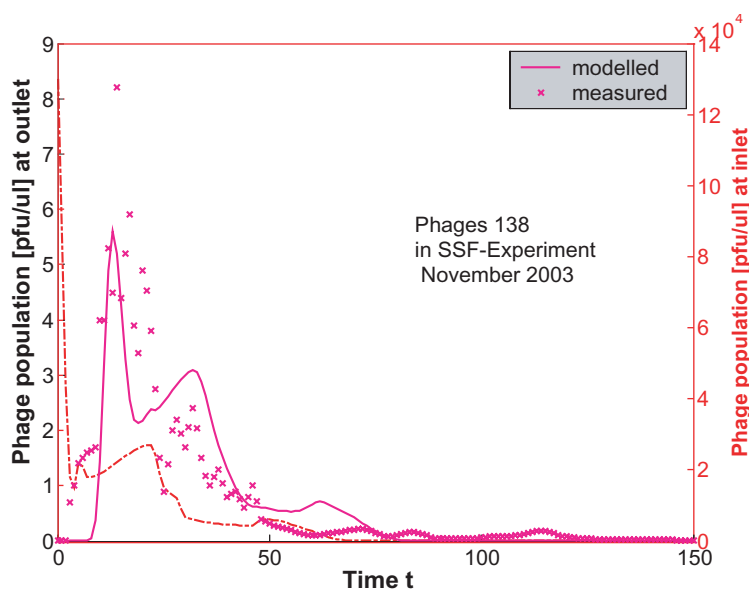
Figure 3. Comparison of measured and modelled results for the phages 138 in the slow velocity experiment; time [h]; broken line shows the inlet concentration

The outcome of all simulations is gathered in Table 1. Velocities v and dispersivities α_L were taken from tracer measurements, retardations R and deactivation rates λ were obtained from inverse modelling runs with the MATLAB module. Resulting Péclet- and Damköhler numbers are listed in the last columns.

Table 1. Results from parameter estimation runs using the MATLAB module

Phages	v [cm/d]	α_L [m]	R	λ [1/d]	Pe	Da_2
138	60	0.07	1.3	0.89	100	16.5
	120	0.13	1	1.36	43.5	10.5
	240	0.2	1.25	0.68	22.2	2
241	60	0.07	1.08	0.63	100	9.72
	120	0.13	1.3	0.62	43.5	6.2
	240	0.2	1.36	0.125	22.2	0.85

According to Table 1 the deactivation represented in the Damköhler number as combination of all interacting processes, decreases with increasing velocity in the experiments. Though the velocity changes by a factor of 2 from one experiment to the other, deactivation changes are different due to the change of the other processes. Altogether there is a decrease of deactivation by a factor of 8.25 for phages 138, and of 11.4 for phages 241, compared to a velocity change by a factor of 4.

Thus for both phages the observed degradation is overproportional, either for the transition from the low to rapid velocities, or from medium to rapid velocities. For the transition from low to medium velocities the observed change in deactivations is underproportional. The observed decrease of deactivation rate, and thus of filter efficiency, from the medium velocity to the rapid velocity case amounts to 5.25 for phages 138 and to 7.3 for phages 241; both values significantly exceed the theoretically expected factor of 2.

CONCLUSIONS

With the presented model approach, based on dimensionless numbers, it is possible to classify filters by their efficiency much easier than in the primitive variable formulation, as the outcome depends on two dimensionless parameter combinations only: the Péclet- and the Damköhler-number. In the view of this theory also deviations from the conventional extended transport equation approach can be discussed much easier.

Results for the idealized simple constant parameter situation can be used as a first guess for more complex bank filtration systems; for example that flow velocity and effective deactivation are inverse proportional, as expressed by the Damköhler number. As in bank filtration systems the flow velocity is controlled by the pumping rate, the proportionality provides a hint on the change of deactivation or degradation by an altered pumping regime.

The theoretically expected decline of total deactivation with velocity could be confirmed by experiments. However the degree of the observed decline was slower between the low and medium velocity cases, and much higher between the medium and rapid velocity cases. This shows that extrapolations from one velocity range to another may be erroneous. Here an extrapolation from the slow velocity range to the rapid velocity range would lead to an overestimation of filter efficiency.

We ignore the reason(s) for the insufficient predictability obtained from the above model approach and suspect that the long time passed by between the different experiments might be responsible for it. During the four weeks elapsed during each experiment, a considerable amount of biomass might have build up in the top layer as well as in the interstitial space of the filter. This biomass might have been responsible for the better-than-model inactivation of the viruses found during filtration.

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A coupled transport and reaction model for long column experiments simulating bank filtration

Christoph Horner, Ekkehard Holzbecher and Gunnar Nuetzmann

Abstract

Within the scope of the interdisciplinary NASRI research project (Natural and Artificial Systems for Recharge and Infiltration) dealing with river bank filtration processes at Berlin water works, a semi-technical column experiment is ongoing since January 2003 to simulate river bank filtration. Here a 30 m long ensemble of 6 soil columns is operated by surface water sampled from Lake Tegel (Berlin, Germany). In April 2004, oxic conditions in the column ensemble were forced to change in anoxic by gassing the flushing solution supply with nitrogen gas. The objective of reactive transport modelling was to (i) identify the main biogeochemical processes and the governing redox conditions within the soil column system during flushing as a conceptual model for river bank filtration and to (ii) verify the observed hydrochemistry of the pore water.

Keywords

Biodegradation, column experiment verification, double porosity, reactive transport modelling, river bank filtration.

INTRODUCTION

Von Gunten and Zobrist, (1996) and Schäfer et al. (1998a, 1998b) present examples of soil column flushing simulation studies involving biogeochemical degradation reactions. To provide unique results these models require a large database including an adequate reaction framework and exact knowledge about soil matrix composition, otherwise reaction models can be partially ambiguous (Appelo et al., 1998). Upscaling of microcolumn studies leads consequently to field-site scale models studying riverbank filtration or contamination of groundwater by reducing leachate plumes originating from landfills or oil spills (Hunter et al. 1998; Prommer et al., 1999a, 1999b). Large-scale semi-technical soil columns are of intermediate scale between small laboratory columns and field sites. In contrast to small laboratory column experiment simulations and field-site scale models there are only few references for large soil column experiments simulating river bank filtration (Porro et al. 1993; Drewes and Jekel, 1996; NASRI, 2003).

SEMI-TECHNICAL SOIL COLUMN EXPERIMENT

The experimental set-up of the large soil column consists of 6 soil columns 5 m in length connected in line. Sampling is possible on 21 sampling ports along and between the columns. Initially the column soil matrix proved to contain interstitial residual air over its whole length, and it took about 10 months until the interstitial air was nearly completely removed from the column sections apart from the inlet. The experiment started in April 2003 and is still ongoing. A decrease of the DOC concentration resulting from oxygen consumption due to biodegradation

was noticeable only near the inlet, while in the column sections (apart from the inlet) DOC concentrations remained nearly constant and independent on oxygen concentrations. Due to the very slow progression of biodegradation (oxic redox conditions were stated still after 1 year), anoxic redox conditions were forced in April 2004 by gassing the influent permanently with nitrogen gas. Pore water samples are taken monthly and analysed on O_2, DOC and inorganic parameters.

MODELLING CONCEPT AND REALISATION

Two approaches to model biodegradation are common: the singe step and the two step process model. The single step process models formulate the biodegradation reaction like a geochemical reaction using a definite electron acceptor (for example Schäfer et al., 1998a). The two step process models conceptualise the overall biogeochemical degradation as a fermentation process consuming available terminal electron acceptors depending on the actual redox state using a partial equilibrium approach for the redox half reactions. Using this approach a close coupling of biodegradation kinetics to inorganic hydrochemistry is straightforward (Prommer et al., 1999a, 1999b; Prommer, 2002; Brun and Engesgaard, 2002). Here, the two step approach is followed. An adequate strategy to solve the reactive transport equation results by coupling both subsets sequentially, e. g. by coupling MATLAB (The MathWorks, 2003), as a powerful intrinsic solver for partial differential equation systems such as the advection-dispersion equation, with PHREEQC as chemical speciation solver.

Difficulties result to provide reliable initial conditions over the whole column system. Besides biodegradation (which was formulated by a simple first order approach) the removal of oxygen in the inner column was formulated using a double porosity approach analogous to formulations by Parkhurst and Appelo (1999). The biodegradation reaction front movement near the inlet seems considerably retarded as compared to the apparent flow velocity determined by tracer experiments performed in the column (NASRI 2003). Therefore, sorption of DOC had to be introduced as an additional process. Three differently reactive fractions of DOC as indicated by chromatographic analysis (NASRI 2003) – two dissolved and one particulate DOC species – were considered. To verify the observed pore solution hydrochemistry, all analysed inorganic components were considered by the PHREEQC module of the reactive transport model. Calcite, pyrolusite (MnO_2), and ferrihydrite ($Fe(OH)_{3(amorphous)}$) were also included as pH and redox buffering minerals into the reactive transport model.

The soil column was modelled as a 1D finite-difference domain. The pore velocity was specified at 1 m/d as confirmed by tracer tests (NASRI, 2003). A simulation time of 589 d was set up for the model simulation (i.e. the time interval ranging from April 2003 until September 2004, for which measurements were available).

RESULTS AND DISCUSSION

Simulation results are exemplarily shown as O_2 breakthrough curves in Figs 1–3. Fig. 1 focuses on both biodegradation and remobilisation of interstitial O_2 as competing processes. Figs. 2 and 3 demonstrate the effects of both time variable O_2 inflow concentrations and biodegradation after consumption of interstitial O_2. In relation to observation, the simulation overestimates the biodegradation at later flushing stages, especially when oxic conditions change to anoxic conditions (see Fig. 3). From Fig. 4, time variable DOC inflow results so that the DOC balance related to O_2 seems obscured. The simulated breakthrough of DOC representing the sum of both simulated DOC shows the numerical effect of changing the DOC inlet but verifies the observed DOC for the inner part of the column. Gassing with N_2 did not provide a further decrease of DOC. Therefore, the inert fraction of DOC is interpreted to prevail over the column extent whereas biodegradation using the reactive DOC fraction seems to take place at the column inlet.

The observed pore water hydrochemistry was also verified by the simulation confirming the key role of calcite as main pH buffer (here not shown), but due to the lack of data basis about the calcite matrix composition and only minor changes in HCO_3^- concentrations, balancing the mineralization of organic carbon was not possible.
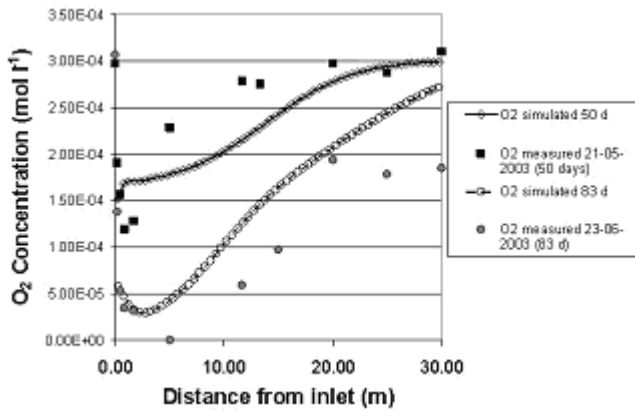
Figure 1. Measured and simulated oxygen breakthrough during soil column flushing at the beginning of the flushing experiment
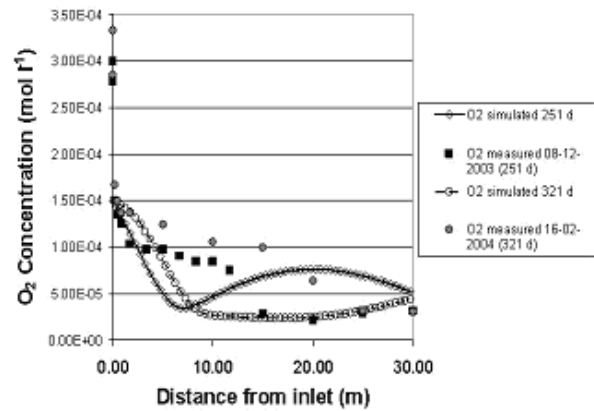
Figure 2. Measured and simulated oxygen breakthrough during soil column flushing after 251 and 321 days
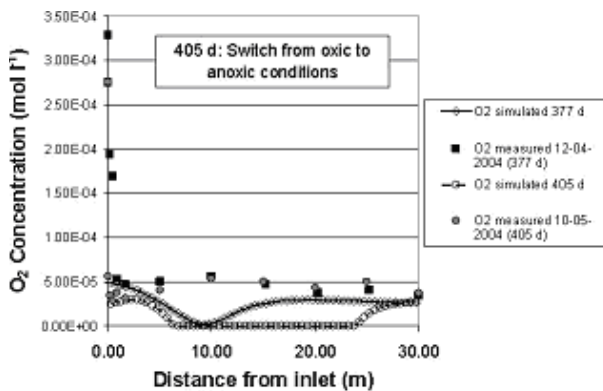
Figure 3. Measured and simulated oxygen breakthrough during soil column flushing after 377 and 405 days
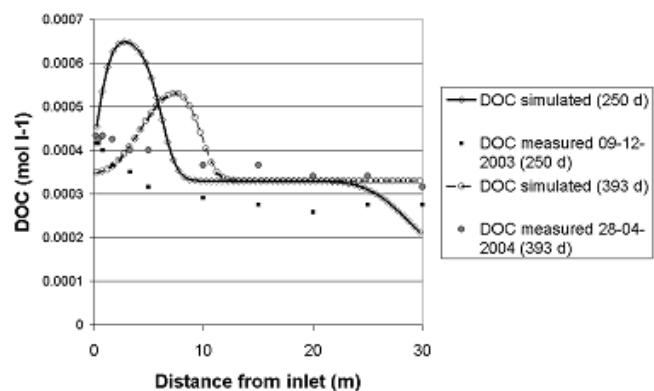
Figure 4. Measured and simulated DOC breakthrough during soil column flushing after 250 and 393 days

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Simulation modeling
of salient artificial recharge techniques
for sustainable groundwater management

Madan K. Jha and Stefan Peiffer

Abstract

Artificial groundwater recharge and rainwater harvesting are two basic techniques for the sustainable management of shrinking freshwater resources. Numerical modeling is a vital tool for the quantitative analysis of groundwater systems, which is not subject to many of the restrictive assumptions required for analytical solutions. The present paper focuses on the development of a groundwater model for the Takaoka groundwater basin, Japan which is afflicted with overdraft problem, and its application to the selection of suitable artificial recharge techniques for improving groundwater condition of the basin. A transient, two-dimensional FEM-based groundwater-flow model was developed for this basin, which was successfully calibrated and validated. The validated groundwater-flow model was then used to examine the efficacy of four feasible artificial recharge techniques. Based on the simulation results, two recharge techniques viz., 'subsurface barrier' and 'river modification' are selected as promising remedial measures for overdraft. The effect of the proposed subsurface barrier was found to be limited both in terms of areal extent and the extent of groundwater rise, but a weir of height 4.5 m from the dry period water level in a perennial river bordering the basin was found adequate to meet the current groundwater demand. The implementation of these two artificial recharge techniques is strongly recommended for the sustainable management of scarce groundwater resources of the Takaoka basin.

INTRODUCTION

Excessive exploitation of groundwater has resulted in groundwater lowering, with concomitant problems of land subsidence and saltwater intrusion in both developed and developing nations. Artificial groundwater recharge and rainwater harvesting have emerged as two basic techniques for sustainable management of shrinking freshwater resources. Artificial recharge of groundwater can reduce the vulnerability of natural groundwater recharge to changes in the amounts of precipitation, especially in areas with low precipitation (ASCE, 2001). In many areas, high land costs and environmental problems encountered with large surface reservoirs have enhanced the attractiveness of artificial recharge as a means of regulating and sustaining water supplies.

Numerical modeling is an important tool for the quantitative analysis of complex groundwater systems. Numerical models have been extensively used for groundwater analysis since mid-1960s (Mercer and Faust, 1980a; Das Gupta and Onta, 1994). The variety and complexity of mathematical models used in groundwater applications have increased dramatically during past four decades. The numerical methods commonly used in groundwater applications are actually variations of two general methods: finite-difference method (FDM) and finite-element method (FEM). Occasionally, specialized techniques such as the method of characteristics (MOC) are also used. Faust and

Mercer (1980) and Van der Heijde et al. (1989) discuss in detail the pros and cons of various numerical methods. For any given class of problems, the choice of the best method generally depends on the processes being modeled, the accuracy desired, and the effort that can be expanded on obtaining a solution. Thus, in order to prepare an appropriate model, a kind of model that suits the desired precision level and the amount of data available should be selected (Shibasaki, 1995). The effective application of groundwater flow models to field problems involves several interrelated phases such as model selection, computer program use, sensitivity analysis, system conceptualization, data collection design, calibration, validation, prediction, and post-audit. It is worth mentioning that a model is never exact and the complete data (as comprehensive as desired for solving a given problem) are never available. Therefore, the quantitative results of a numerical analysis only serve as qualitative guides, and considerable scientific judgment of a subjective or intuitive nature is necessary for any degree of success (Mercer and Faust, 1980b).

The Takaoka groundwater basin of Kochi Prefecture, Japan is afflicted with overdraft problem, which is posing a serious threat to the sustainability of scarce groundwater resources. The present paper deals with the development of a groundwater model for the Takaoka groundwater basin, Japan and its application to the selection of suitable artificial recharge techniques for improving groundwater condition and ensuring its sustainable utilization in the basin.

STUDY AREA

The study area, 'Takaoka groundwater basin' is located in Tosa City, Kochi Prefecture, Japan, and it has a mountainous landscape (Fig. 1). The basin encompasses an area of 510 ha and is comprised of paddy fields (242 ha), greenhouses (77.3 ha), and built-up land (190.7 ha). The Niyodo River flows to the east of the basin and the Hake River to the south, which are major river systems of the study area. There is a 2-m high inflatable rubber dam called 'Fabri dam' across the Hake River. A small and shallow intermittent river, the Kamo River, flows through the basin as shown in Fig. 1.

A total of 20 major production wells are under operation, of which 17 wells supply water to the paper industry and the remaining three are used for domestic water supply; totaling a groundwater withdrawal of about 53.4 x 10^3 m^3/day (Jha et al., 1996). In the Takaoka basin, groundwater occurs extensively under unconfined conditions within the alluvium lithologic unit comprising sand and gravel. This lithologic unit has a mean thickness of 14.5 m and constitutes the major aquifer of the study area. Most wells in the basin tap this aquifer. The average hydraulic conductivity and specific yield of the Takaoka aquifer are 9.35 x 10^2 m/d and 15%, respec-

Figure 1. Map of the study area

tively (Jha et al., 1996). Groundwater flows predominantly from north to south into the Hake River. Interested readers are referred to Jha et al. (1996) for further details about the study area.
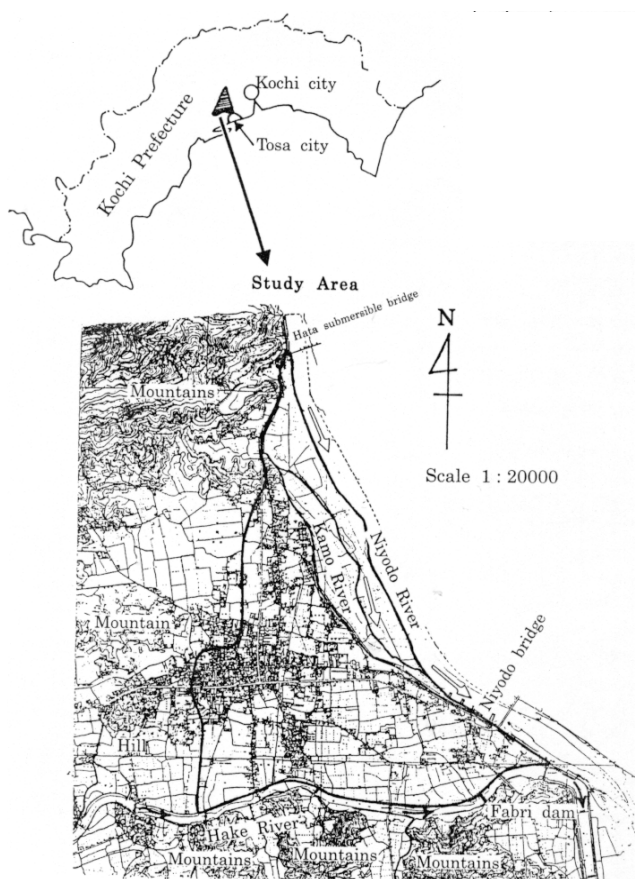
METHODOLOGY

Model development and solution technique

A conceptual model of the Takaoka aquifer was developed based on the field investigations and the analysis of the hydrogeologic data. Based on the conceptual model, a groundwater numerical model for transient, two dimensional groundwater flow in the isotropic, heterogeneous unconfined aquifer system was formulated using the following governing equation:

$$\frac{\partial}{\partial x}\left(K(x, y)\, b(x, y)\, \frac{\partial h}{\partial x} \right) + \frac{\partial}{\partial y}\left(K(x, y)\, b(x, y)\, \frac{\partial h}{\partial y} \right) = S_y\, \frac{\partial h}{\partial t} + Q \qquad (1)$$

Where, K(x,y) = aquifer hydraulic conductivity as a function of space, b(x,y) = aquifer thickness as a function of space, S_y = specific yield of the unconfined aquifer, and Q = source or sink.

Finite-element method (FEM) being more flexible than the FDM (Yu and Singh, 1994), a finite-element formulation for modeling two-dimensional groundwater flow was used in this study. The Takaoka basin was discretized into 1,138 two-dimensional triangular elements with 627 nodes having finer discretization around pumping wells (Fig. 2). The nodal spacing was 100 m, except for the boundary nodes and pump nodes. With the aquifer depth and elevation data specified for each node, the model provides a 3-D spatial representation of the basin, while modeling groundwater flow in two dimensions. Eqn. (1) was solved by the Galerkin method using a linear interpolation function. A fully implicit discretization in the time variable was used. The groundwater model thus developed allowed the computation of transient hydraulic heads in the aquifer, starting from an initial hydraulic head distribution under constant boundary and recharge/discharge (source/sink) conditions. The aquifer hydraulic con-ductivity and specific yield were assumed constant over each element, but the former was considered as a spatially-varying parameter. The details on the formulation of FEM-based groundwater flow models can be found in Kinzelbach (1986).
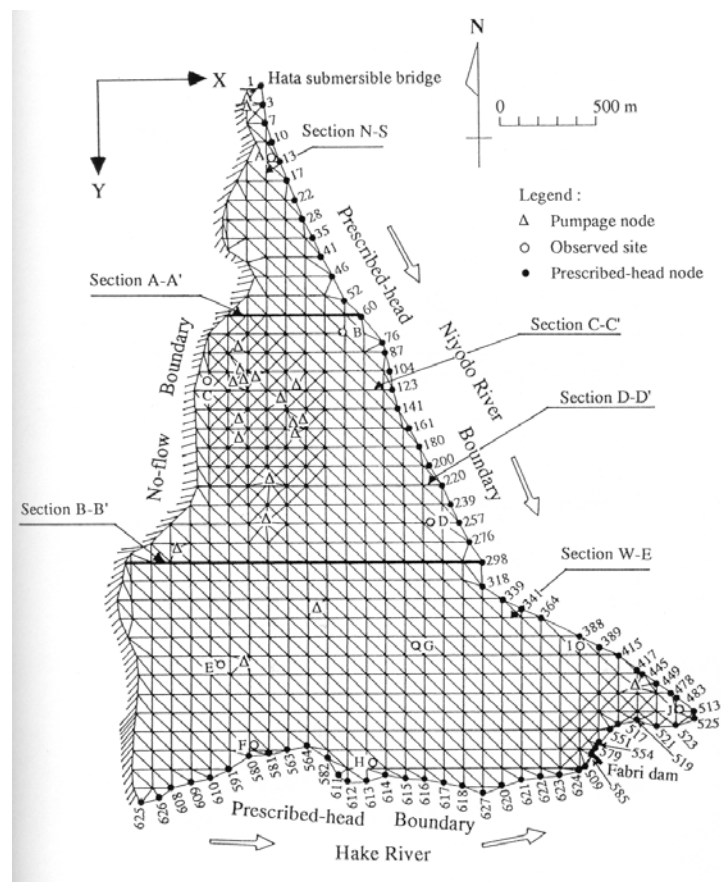
Figure 2. Finite-element mesh of the Takaoka basin with boundary conditions and well fields

Calibration and validation of the model

Considering the severity of the overdraft problem during dry periods (October to February) in the study area, the modeling effort aimed at analyzing the groundwater scenarios of dry periods only. Model parameters, boundary

conditions, initial condition, and source-sink terms were obtained or estimated from existing records or field studies. Prescribed-head boundary conditions were assigned along the east and south boundaries of the basin based on the historic dry period water level information. Because of the dominating silts along the west boundary, it was modeled as a zero-flux boundary. The aquifer hydraulic conductivity and the specific yield were determined from the time-drawdown data obtained by field pumping tests conducted at selected sites over the basin. The saturated aquifer thickness was estimated with the help of available stratigraphy and water table depth over the basin. In addition, well pumpage was obtained from the data of groundwater withdrawals by the industry and water supply plants.

First of all, the formulated two-dimensional transient groundwater flow model was run to simulate steady-state groundwater flow in the basin under 100% pumping and constant Niyodo River stage conditions. Water tables yielded by the model and those measured in the field were compared, and the two input parameters (K and S_y) of the model were adjusted until the simulated and measured values of water table matched satisfactorily at observed sites. The model was calibrated by trial-and-error method using the dry-period water table data from 1988 to 1992. An independent set of the water table data from 1993 to 1994 was used to validate the calibrated model.

Simulating impacts of salient artificial recharge techniques

In order to explore the basin's response to artificial recharge, the impacts of four feasible artificial recharge techniques were simulated using the developed FEM model: (i) subsurface barrier, (ii) modification of the Niyodo River, (iii) recharge from existing irrigation and drainage channels, and (iv) shifting of the Niyodo River course. A subsurface barrier is usually a subsurface dam at the downstream end, which obstructs and detains groundwater flow. Several types of subsurface barrier have been used worldwide to detain groundwater flow in different aquifer systems (Oaksford, 1985). The subsurface barrier in the study area is proposed to be a metallic sheet or concrete dam underground along the downstream of the Fabri dam. The second artificial recharge technique involves the construction of a weir across the Niyodo River at a suitable location. Increasing the rate of recharge through streams has successfully been implemented, especially in Los Angeles (Helweg, 1985). The proposed weir could be a gravel mound (semi-permanent-type structure) or a permanent-type structure.

The possibility of using existing irrigation and drainage channels for artificial recharge purposes was also explored as a third artificial recharge technique. Since the observed data were not available, based on the local experience, a constant recharge rate of 15 mm/day from the drainage channels was assumed, and the aquifer response to irrigation-canal recharge was simulated for three prescribed recharge rates viz., 5, 10 and 15 mm/day under existing conditions. Furthermore, bearing in mind considerable groundwater recharge from the Niyodo River, a new and feasible river course was delineated. This new Niyodo River course was considered as a fourth possible artificial recharge technique for the study area.

RESULTS AND DISCUSSION

Calibration and validation results

Figs. 3(a,b) show a comparison of simulated and measured groundwater levels at ten sites over the basin for calibration and validation periods; the location of the sites is shown in Fig. 2. It is obvious from these figures that the agreement between the simulated water tables and measured water tables is reasonably good for both the calibration and validation periods. Thus, the developed groundwater model was calibrated and validated successfully.
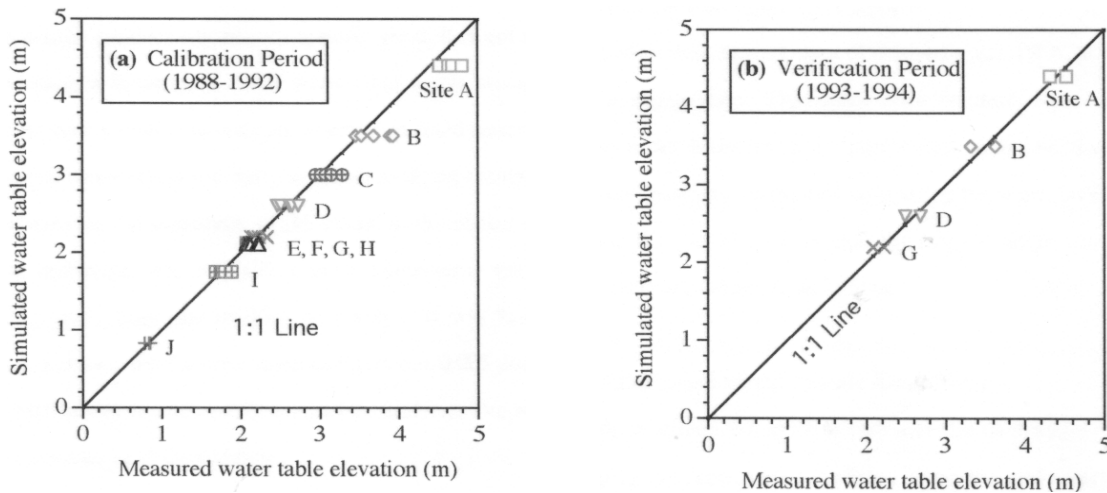
Figure 3. Simulated versus observed groundwater levels for calibration and validation periods

Subsurface barrier vis-à-vis rroundwater

Considering 100% pumping and a constant Niyodo River stage, the effect of a subsurface barrier on the basin's groundwater is illustrated in Fig. 4. A comparison of this figure with a simulated contour map without the barrier revealed that the downstream water table is significantly improved due to the barrier, with a rise of 70 cm in downstream groundwater levels. The barrier also decreases the hydraulic gradient downstream of the Fabri dam, suggesting slow movement of groundwater towards the Niyodo River (i.e., less subsurface outflow). However, the influence of the barrier was found confined to approximately 1.5 km from the Hake River. Overall, the proposed barrier is effective for improving overdraft by maintaining a relatively high water table in the basin.

Impact of Niyodo River modification

The Niyodo River is hydraulically connected with the Takaoka basin, and its upstream reach length of 2.9 km from the Hata submersible bridge significantly contributes to groundwater. A weir of proper height can be constructed across the Niyodo River at the southern end of the recharging river reach, i.e., at the location nearby node 298 (Fig. 2). Such a structure will increase the pressure head and will retain water longer. The increased residence time means more water can be recharged and thus higher groundwater levels in the basin can be maintained. To determine an appropriate height for the weir, its various heights were simulated for the present rate of pumping using the developed groundwater model, and their contributions to the groundwater supply were compared. It was found that a weir height of 4.5 m from the dry period water level in the Niyodo River is adequate to meet the present groundwater demand (i.e., 53,400 m³/day). The predicted groundwater rise due to the proposed weir is shown in Fig. 5. It is worth to mention that the actual groundwater build-up will be more than that illustrated in Fig. 5, because the backwater effect created by the proposed weir was assumed to be horizontal during numerical simulation.

Aquifer response to the recharge from irrigation and drainage channels

The simulation results revealed that even for a recharge rate of 15 mm/day from the irrigation and drainage channels, the water table does not rise appreciably. Thus, the recharge from existing irrigation and drainage channels is not effective for the aquifer rehabilitation unless infiltration rates of these channels are artificially enhanced and maintained on a long-term basis.
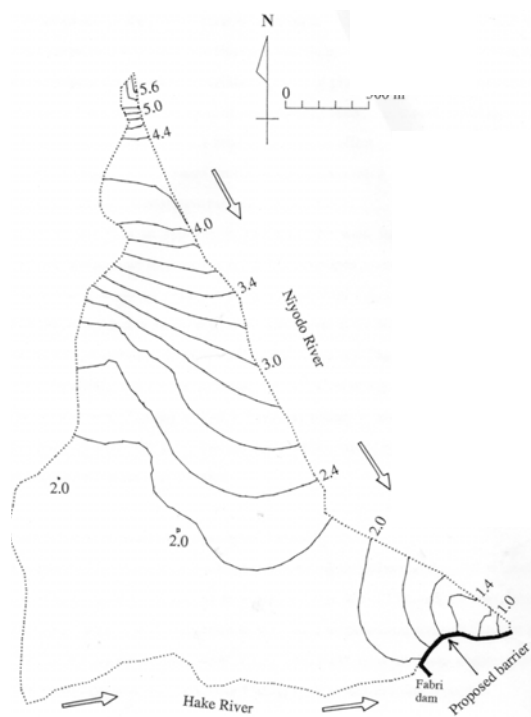
*Figure 4. Groundwater response
to the subsurface barrier
(Contour interval = 0.2 m)*
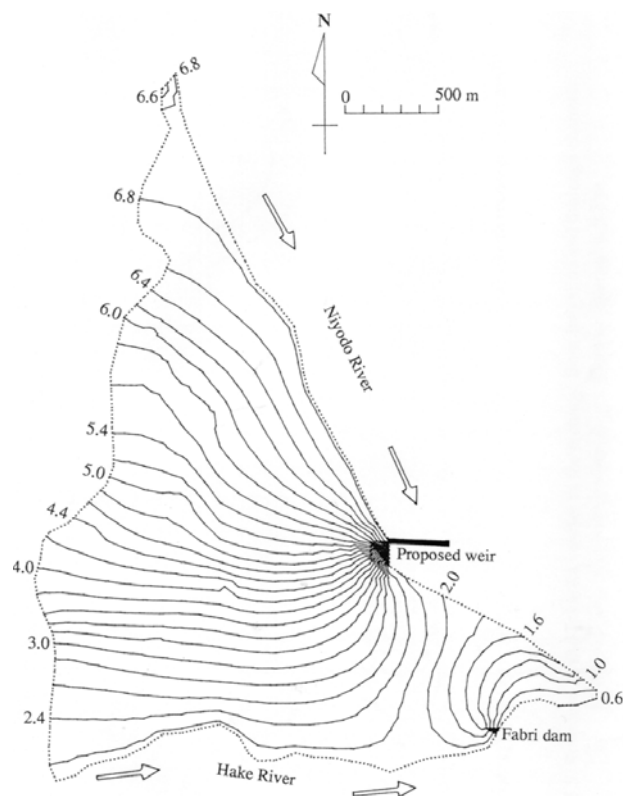
*Figure 5. Improved groundwater scenario
due to the proposed weir
(Contour interval = 0.2 m)*

Impact of the shifted Niyodo River course

Simulation results of the impact of shifted Niyodo River course indicated that the basin's groundwater level rises at most by 20 cm along the north-south section. However, the maximum differences between the water levels for the two river courses are about 22 cm along Section C-C' and 15 cm along Section D-D' (Fig. 2), except for the eastern boundary nodes along Section C-C' having a difference of 27 cm. In this case, an interesting observation was that the amplitude of variation between the groundwater levels under existing river-course and new (shifted) river-course conditions gradually increases from west to east along Section C-C', whereas it starts decreasing after 360 m from the western boundary along Section D-D'. The concentration of pumping wells in the western portion above Section D-D' is attributed to the unique water-table profile along this section. Moreover, the simulated water table contour map under shifted Niyodo River-course conditions revealed that the water table does not rise significantly due to the proposed Niyodo River course. Thus, contrary to our expectation, the shifted Niyodo River course doesn't offer a favorable alternative for improving groundwater condition in the study area.

CONCLUSIONS

In this paper, the application of simulation modeling to assess the efficacy of salient artificial recharge techniques is discussed for sustainable groundwater management in the Takaoka basin of Kochi Prefecture, Japan. A transient, two-dimensional FEM-based groundwater-flow model was developed for this basin, which was calibrated and validated successfully. The validated model was then used to examine the efficacy of four feasible artificial recharge techniques.

The analysis of the simulation results indicated only two promising artificial recharge techniques for rehabilitating the Takaoka basin, and thereby stabilizing the depleting aquifer. These artificial recharge techniques are: (i) a 600-m long subsurface barrier along the downstream of Fabri dam, and (ii) a weir across the Niyodo River at 2.9 km south of the Hata submersible bridge. It was found that the impact of the proposed subsurface barrier is confined to about 1.5 km from the Hake River, while the proposed weir in the Niyodo River is enough to meet the present groundwater demand. However, for long-term sustainability of the scarce groundwater resources of the Takaoka basin, a combined application of these two techniques is strongly recommended. As suggested by Bouwer (1988), small artificial recharge schemes must be implemented in near future in order to acquire field experience with the proposed techniques, which in turn would help develop proper design and management criteria for successful operation of full-scale prospective groundwater recharge projects. Such a recommendation is also valid for the Takaoka groundwater basin in order to effectively implement the recommended measures.

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Influencing factors on the removal efficiency of DOC and ammonium in 2-D pilot-scale soil aquifer treatment (SAT) model

J.-W. Kim, J. Won, S.D. Seo and H. Choi

Abstract

Soil aquifer treatment (SAT) is a promising technology for wastewater reclamation and reuse using natural remediation process in the aquifer. This technology takes advantage of physicochemical and biological processes in the subsurface. The performance of SAT is governed by (i) aquifer characteristics, (ii) source water characteristics, and (iii) operation condition. Two-dimensional pilot-scale SAT model was designed and operated to see the major mechanism of SAT in both unsaturated and saturated aquifer. The dimension of the model is 4-m in length, 2-m in height, and 10-cm in width. As groundwater is induced to the half parts of the model, the unsaturated and saturated aquifers are differentiated. As loaded membrane bioreactor (MBR) effluents into the model basin with the cycle of 4-days loading and 3-days drying, we monitored hydraulic characteristics and the fate and transport of the contaminants in terms of dissolved organic carbon (DOC) and ammonium. Infiltration rate was decreasing continuously, but high infiltration rate causes insufficient retention time for the reasonable remediation. Therefore, it is important to find the optimal infiltration rate and to control the infiltration rate.

Keywords

2-D model; ammonium; DOC; infiltration rate; pilot-scale model; soil aquifer treatment (SAT).

INTRODUCTION

Soil aquifer treatment (SAT) is a promising technology for wastewater reclamation and reuse using natural remediation process in the aquifer. Because of its economical efficiency and operational simplicity, SAT has been studied and applied, especially, in arid area of the world (Kanarek and Michail, 1996; Fox et al., 2001; Nema et al., 2001; Cha et al., 2004). In a typical SAT system, the secondary or tertiary treated wastewater percolates through the unconfined or confined aquifer using various physical or biochemical soil processes. A major concern in SAT systems is the fate and transport of organic compounds during the infiltration of treated wastewater to the groundwater. A secondary concern is the nitrogen species that include ammonium and nitrate (NCSWS, 2001). The performance of SAT is governed by (i) aquifer characteristics such as hydraulic conductivity, intrinsic soil microbiology, etc., (ii) source water characteristics such as contaminants concentrations, loading rate, etc., and (iii) operation condition such as features of operation cycle (NCSWS, 2001; Kim et al., 2004). In the previous work, lab-scale SAT column study was accomplished to evaluate the feasibility of SAT application in terms of the source water and the site characteristics (Cha et al., 2004). Since, however, the study was conducted in 1-m 1-D columns, the long-term effects of SAT which are related on saturated aquifer cannot be considered. Although the most removal in SAT is occurred in unsaturated aquifer, specifically within 1-m (NCSWS, 2001), the impact of the treated water on the existing groundwater and another removal trends of remained contaminants in saturated aquifer should be explored subsequently. Therefore, in this study, the 1-D column study is extended to vertical 2-D pilot-scale model in which unsaturated and saturated aquifers are combined.

METHODS

2-D pilot-scale SAT model domain

The image and the schematic are presented in Fig. 1. The dimension of the model is 4-m in length, 2-m in height, and 10-cm in width. The 2-m height is differentiated by unsaturated and saturated aquifer. The basin is located at the top of the model and is a little biased to the side where groundwater is injected in order to consider the groundwater flow direction in saturated aquifer. The sampling ports are regularly distributed on the back side of the model. However, sampling was not accomplished from every sampling ports, but from 34 pre-selected sampling ports which can be representative of neighbors. Buffer spaces, in which coarse grains are filled and groundwater is overflowed to maintain predetermined constant head, are located in the both sides of the model.
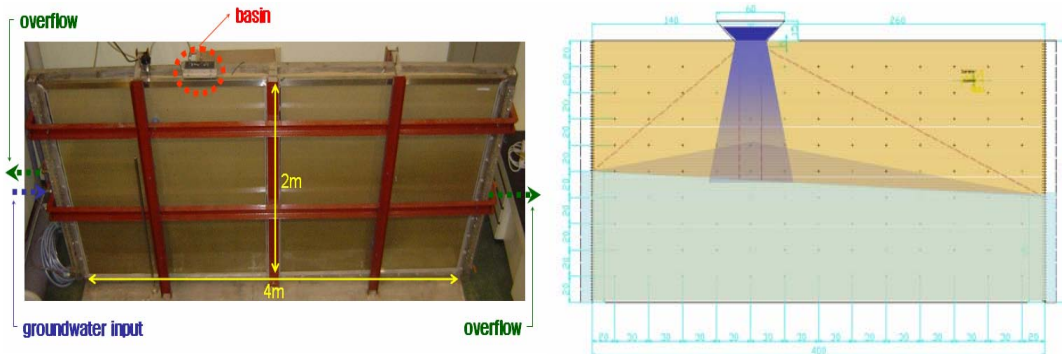
Figure 1. Image and schematic of 2-D pilot-scale SAT model

Soil and groundwater characteristics

Soil filled in the model was from Youngsan River, Korea, which is a candidate of SAT field application since a waste water treatment plant (WWTP) is located near the river. The main portion of the soil is sand whose size is less than 2-mm. Preliminary analysis for the soil characteristics was accomplished, and the results are listed in Table 1.

Groundwater was pumped up from 33-m deep of alluvial layer which is located near our laboratory in GIST. It is directly connected to the groundwater input port of the model. As maintained 20-cm of head difference between both sides, about 0.33-m^3/day of flow rate was obtained in saturated aquifer when only groundwater flows. Preliminary results of groundwater quality are listed in Table 1.

Table 1. Characteristics of soil and groundwater used in the SAT model

Soil		Groundwater	
Source	*Youngsan River*	*Source*	*33-m alluvial layer*
Size	< 2 mm	DOC	0.80 mg/l
SOM	0.84 % (g/g)	NH_4^+-N	0.79 mg/l
Particle density	2.619 g/ml	NO_3^--N	2.21 mg/l
Bulk density	1.522 g/ml	DO	4.9 mg/l
Porosity	0.419	pH	6.55

Operation conditions and source water

In typical SAT system, the operation consists of wetting period and drying period as a cycle in order to prevent clogging layer under the basin due to continuous loading (Nema et al., 2001). From our previous column test, 4-day

wetting / 3-day drying (W4D3) cycle is optimal to get the highest efficiency. In this study, the model was operated by W4D3 cycle for the first 4-weeks, and continuous loading (W7D0) followed it to compare the results. Instead of secondary effluent from a WWTP, membrane bioreactor (MBR) effluent was used as source water (Jang et al., 2004). The effluent of MBR was linked to the SAT basin in online using a pump.

RESULTS AND DISCUSSION

Hydraulic characteristics

During the operation, variations of water contents and infiltration rates were monitored (Fig. 2). Water contents were measured at 10-cm and 30-cm deep from the basin, and infiltration rates were measured at the basin as an input and the right-side overflow as an output. In the water contents variation, the different operation cycles (W4D3 and W7D0) are clearly differentiated. The range of water contents variation at 30-cm deep is relatively narrow than at 10-cm deep. It means that the aquifer under the basin is unsaturated and clogged locally. In the infiltration rates variation, the infiltration rates are continuously decreasing during both cycles. This continuous decrease implies the possibility of clogging layer formation. When the decreasing rates of infiltration rates of each cycles were compared, cycle W4D3 (-0.0361 m^3/d/d) has relatively higher than cycle W7D0 (-0.0228 m^3/d/d) unlike our expectation. As cycle W7D0 followed cycle W4D3, the difference of initial infiltration rates can induce the unexpected result. In addition, the difference between input and output can be explained by mounding effect. It advocates the result that the flowrate in the left-side overflow was increased as basin loading was continued (data is not shown). In typical SAT, the water table under the basin is mounded because of basin loading, and this leads groundwater flow path is changed.
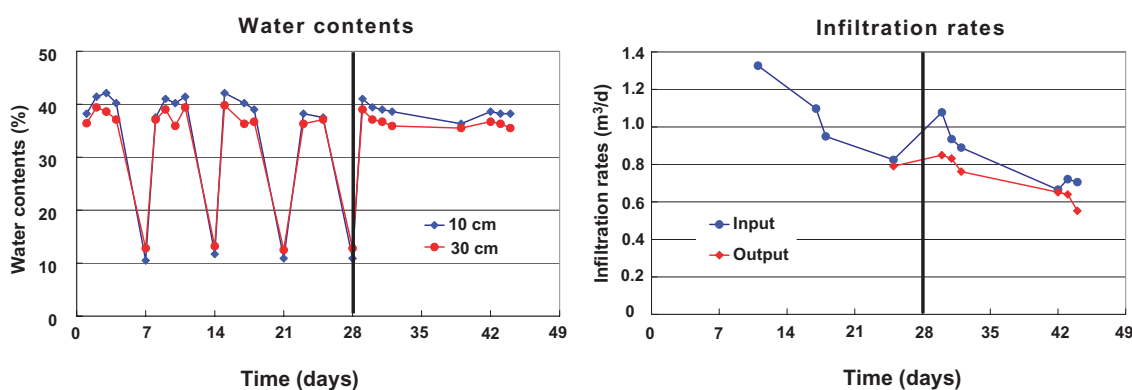
Figure 2. Water content variation at 10-cm and 30-cm deep from the basin (left),
and infiltration rate variation at the basin (input) and the overflow (output) (right)

Contaminants

DOC and ammonium were contaminants considered in this study, and DO was also monitored as an electron acceptor. As shown the results in Fig. 3, we compared concentrations distributions at day 25 and day 39 which represent cycle W4D3 and cycle W7D0, respectively. The removal of DOC and ammonium was not completed although DO was remained. This means that the retention time for their removal reactions was not sufficient. This can be also explained by the result that removal efficiency at day 25 in which infiltration rate was relatively higher is lower than that at day 39. The insufficient retention time leads that biological reaction under the basin is uncompleted and that the contaminants concentrations are decreased mainly by dilution in saturated aquifer rather than by adsorption or biological reaction.
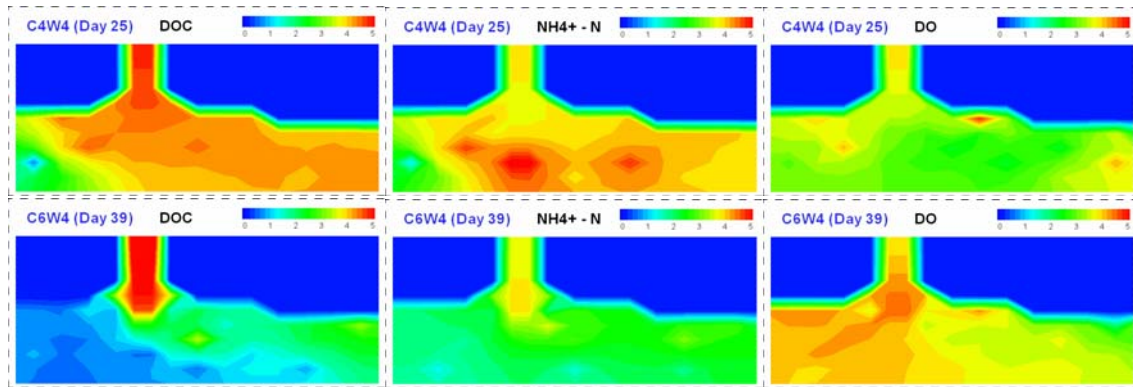
Figure 3. Distributions of DOC, ammonium and DO concentrations at day 25 (cycle W4D3; upper) and day 39 (cycle W7D0; lower)

CONCLUSIONS

2-D pilot-scale SAT model was designed and operated in order to extent the research scope into the combination of unsaturated and saturated aquifer. As an operation schedule, 4-day wetting/3-day drying cycle was employed and this was compared with following continuous loading cycle. The infiltration rate was decreasing continuously during the operation time, and the retention time was not sufficient for the reasonable remediation. This indicates that the infiltration rate, especially under the basin, is an influential factor in terms of removal efficiency. In the other way, however, if the infiltration rate is too low the quantity of treated water will be small although the water quality is better. Therefore, it is important to find the optimal infiltration rate and to control the infiltration rate. These will be left as future work.

ACKNOWLEDGEMENTS

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Estimating of the solute transport parameters retardation factor and decay coefficient of pharmaceutical residues using the program visual CXTFIT

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Abstract

In the course of the interdisciplinary research project NASRI (natural and artificial systems for recharge and infiltration) many investigations are currently being carried out to assess the risk of break through of persistent organic substances into raw water used for drinking water supply. One part of these studies is the determination of the transport behavior of pharmaceutical residues in test sand filters, so called enclosures, equipped with sampling points at various depths. Breakthrough curves were determined for carbamazepine, primidone (both antiepileptic drugs), clofibric acid (a metabolite of blood lipid lowering agents), diclofenac, ibuprofen (both analgesic drugs) and for chloride, used as a conservative tracer. Retardation coefficients and degradation rates were obtained by using the software Visual CXTFIT. Degradation rates between 0.7 h^{-1} and 1 h^{-1} were observed for ibuprofen whereas clofibric acid, primidone, carbamazepine and diclofenac showed no or very little degradation ($\lambda < .06\ h^{-1}$).

Keywords

Pharmaceutical residues, retardation factor, decay coefficient, Visual CXTFIT.

INTRODUCTION

The occurrence of pharmaceuticals in surface water used for drinking water production after groundwater recharge is an increasing problem for public water supply (Daugthon and Ternes, 1999; Heberer, 2002). Sources for the occurrence of pharmaceutical residues are discharges from sewage treatment plants runoff (Ternes, 1998), landfills (Eckel et al. 1993), superfund sites (Reddersen et al. 2002) or applications in farms and fishery. Such residues may then also leach into groundwater by groundwater recharge (Heberer and Stan 1997). Some residues of pharmaceuticals and their metabolites may cause problems mainly because of their persistence and low tendency to adsorbe.

The knowledge of the transport behavior of the pharmaceutically active compounds (PhACs) is important to minimize the concentrations of drug residues by optimizing the pumping regime of raw water production and /or taking measures to reduce source water concentrations.

The experiments were carried out by adding dissolved substances to the supernatant water of an enclosure constructed similar to a large scale column (diameter = 1.13 m; depth = 1.0 m) in a pulsed application. The breakthrough was measured at the sampling points in different depths (0.4 and 0.8 m) over about three days (for further details see Grützmacher et al., in press).

The chemical analyses were carried out by using solid-phase extraction, chemical derivatization with MTBSTFA,

and detection of the analytes by capillary gas chromatography-mass spectrometry (GC-MS) with selected ion monitoring (SIM) (Reddersen and Heberer, 2003). The influence of the biologically active upper layer of the filter, the so called Schmutzdecke or clogging layer, was investigated by removing it before the second experiment. For interpretation the pore velocity and dispersion coefficient of the conservative tracer chloride were determined by inverse modelling with Visual CXTFIT. Subsequently retardation factors and decay coefficients were obtained for the pharaceuticals by fitting the breakthrough curves to the measured values on the basis of the previously determined hydraulic parameters.

TRANSPORT BEHAVIOR OF THE TRACER

The results of the inverse modelling of the tracer breakthrough curves are shown in Table 1. The calculated values for the effective pore volume and dispersion length are given in Table 2.

Table 1. Results for the modeling of the breakthrough curve of the tracer using the Visual CXTFIT software

Sampling point	P1 (0.4 m)	P1 (0.4 m)	P2 (0.8 m)	P2 (0.8 m)
Clogging	visible	not visible	visible	not visible
Filter velocity v_f [m/h]	0.045	0.051	0.045	0.051
Pore velocity v_a [m/h]	0.50	0.38	0.35	0.32
Dispersion coefficient D [m²/h]	0.040	0.0080	0.080	0.020

Table 2. Results for the modeling of the breakthrough curve of the tracer – calculated values from Table 1

Sampling point	0.4 m	0.4 m	0.8 m	0.8 m
Clogging	visible	not visible	visible	not visible
Dispersion length α_L [cm]	8.00	2.11	22.9	6.3
eff. pore volume n_f [%]	9.00	13.4	12.9	15.9

The average effective pore volume calculated from all values obtained in both experiments is 12.8 %. This is much less than the total porosity of 35 % which was determined by differential weighing in the laboratory. This may be caused by adhesively bound capillary water, small gas bubbles in capillary range or dead-end pores. The higher values of porosity and the lower values of dispersion length in the experiment without clogging are probably caused by mechanical influences due to the removal of the upper filter layer.

TRANSPORT BEHAVIOR
OF THE PHARMACEUTICALLY ACTIVE COMPOUNDS

Tables 3 and 4 show the values for retardation factors and degradation rates calculated with Visual CXTFIT. All values are based on the data for dispersion coefficient D and pore velocity va of the tracer shown in Table 1.

Table 3. Values of the retardation factor R calculated using Visual CXTFIT

	Clofibric acid	Ibuprofen	Primidone	Carbamazepine	Diclofenac
0.4 m without clogging	1.4	1.6	1.0	1.5	1.4
0.4 m with clogging	1.2	2.0	1.0	1.3	1.6
0.8 m without clogging	1.5	1.5	1.5	1.7	1.6
0.8 m with clogging	1.5	2.9	1.3	1.5	1.5

Table 4. Values of the degradation rate λ [h-1] calculated by Visual CXTFIT

	Clofibric acid	Ibuprofen	Primidone	Carbamazepine	Diclofenac
0.4 m without clogging	0.00	1.0	0.00	0.00	0.06
0.4 m with clogging	0.00	1.0	0.00	0.00	0.00
0.8 m without clogging	0.03	0.75	0.00	0.02	0.06
0.8 m with clogging	0.00	0.70	0.00	0.00	0.02

Only little retardation was observed for all of the investigated substances (R = 1.0 to 2.9). Ibuprofen showed considerable degradation or irreversible adsorption (λ = 0.7 to 1.0 h^{-1}). The other substances showed only very little degradation (λ < 0.06 h^{-1}).

Partially, the values of retardation and degradation are higher in the experiment without clogging which might be explained by analytical variations. As a reference for the analytical uncertainty, Table 5 shows the results of inter-laboratory tests of the years 2000 to 2002 within the BLAC-study (BLAC 2003).

Table 5. Average relative standard deviation of the inter-laboratory tests

Clofibric acid	Ibuprofen	Primidone	Carbamazepine	Diclofenac
16 %	19 %	not specified	20 %	32 %

It has to be considered that the results are based on a confidence interval of 66%. For common tasks in practice the values have to be multiplied by 1.96 to get a confidence interval of 95%. This also shows the high analytical requirements. It has to be noted that analytical methods used for the determination of pharmaceutical residues in aqueous samples were elaborated to acquire field data on the occurrence of such compounds in various different aquatic compartments. They have, however, not been designed to monitor little changes of concentrations as they often have to be monitored in processing studies. The analytical method used in this study delivers qualitatively highly reliable results. The results are also quantitatively acceptable even if a certain variation of the results caused by random errors is not avoidable.

With the exception of ibuprofen, no significant influence of the clogging layer was seen. This is reasonable because for the other PhACs only a marginal degradation was observed even in sewage treatment plants that show a much higher biological activity.

Table 6 shows the comparison of the concentrations of PhACs between the influents and the effluents of sewage treatment plants (BLAC 2003). A comparison with the results obtained in the enclosure experiments described above shows a similar behavior for ibuprofen during sand passage and in sewage treatment plants. In both cases it is readily degradable.

Table 6. Concentrations in sewage treatment plants (BLAC 2003)

	Inflow (Median) [µg/l]	Outflow(Median) [µg/l]	Degradation [%]
Carbamazepine	1.60	1.41	12
Clofibric acid	0.27	0.22	18
Diclofenac	1.63	1.70	0
Ibuprofen	3.00	0.09	97

APPLICABILITY OF THE RESULTS TO BANK FILTRATION

For the interpretation of the results obtained for groundwater samples it has to be taken into account that the breakthrough curves obtained for the individual substances have phase shifts compared to other substances or conventional tracers like chloride or the gadolinium-EDTA complex. As in nature the concentrations can vary distinctively at the sources the knowledge of the retardation coefficient is important.

Assuming that dispersivity and field velocity are constant over the distance the time difference between tracer and substance can be estimated with the following equation:

$$\Delta t_{Peak} \approx t_{RT}(R-1)$$

Δt_{Peak}	time difference between tracer and substance
t_{RT}	retardation time of the tracer
R	retardation coefficient

With this equation it is only possible to make a rough estimation assuming that the conditions are homogeneous and in a steady state. In addition, it has to be guaranteed that dispersivity and effective pore volume are the same as they are at the bank filtration sites.

The following table 7 was calculated assuming the above mentioned conditions.

Table 7. Lag time between tracer and substance after transport times of one week and one month

	Clofibric acid	Ibuprofen	Primidone	Carbamazepine	Diclofenac
Average retardation coefficient R from Table 3	1.3	degradation	1.2	1.4	1.4
Time lag after transport time of one week	50 h		34 h	67 h	67 h
Time lag after transport time of one month	9 d		6 d	12 d	12 d

The time lags become obvious at longer distances. The retardation for carbamazepine is two times longer than that of primidone. Only at monitoring wells close to the source (e.g. a river bank) with transport times of only few days the time shift is negligible.

In general, it can be concluded that the determination of retardation factors and decay coefficients in an enclosure is a good method for the prediction of the transport behavior of dissolved substances in groundwater aquifers and to estimate the phase shifts between different substances or conventional tracers.

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Robustness of microbial treatment during groundwater recharge

Peter Fox and Roshan Makam

Abstract

The removal of dissolved organic carbon (DOC) has been proposed as a surrogate for the removal of unregulated contaminants of concern. Since the concentration of most trace pollutants is too low to support growth, trace pollutants maybe co-metabolized as DOC is degraded. Studies of both SAT and bank filtration systems demonstrate that the majority of trace pollutants are removed after sub-surface travel times of 2–6 months. From a regulatory perspective, travel time criteria are based on the potential for virus survival. Travel times of 50 days and 70 days are criteria used for bank filtration systems in Germany and the Netherlands, respectively while six-month travel times are in the California proposed guidelines for indirect potable reuse. In a biologically active sub-surface system, the travel time criteria for virus survival might also be suitable for the removal of trace organics. Since most microbial reactions occur on the surfaces of the aquifer matrix, this study examines relationships between surface area and travel time. A relationship between the relative importance of surface area with respect to sub-surface travel time mathematically demonstrates that surface area contact does not vary significantly for a range of common aquifer materials subject to identical hydraulic gradients.

Keywords

Robustness, organics, surface area, travel time.

INTRODUCTION

Groundwater recharge systems using impaired waters rely on transport through the soil matrix for purification of the water. Reclaimed waters may contain numerous potentially harmful organic compounds. These compounds undergo many potential reactions and transformations during transport through the subsurface. Dissolved organic carbon (DOC) is often used as a surrogate to monitor the removal of organic compounds. DOC is considered to be removed primarily by biodegradation. Trace organic contaminants may be cometabolized by bacteria during wastewater recharge which requires time since their concentrations are too low to support growth. Cometabolism refers to the transformation of a secondary substrate by metabolic reactions that do not directly support the growth of microorganisms. Trace organic contaminants are usually present at concentrations well below those required to sustain a microbial growth. Current research has focused on water quality parameters and the biodegradation of DOC has demonstrated that the majority of persistent DOC resembles natural organic matter (Drewes and Fox, 2001, Quanrud et al., 2003). Numerous studies have also demonstrated the relationship between primary substrate utilization and biomass growth (Rittmann, 1984; Rittman and Manem, 1992; Rittman and McCarty, 1981). Yet information regarding the mechanistic rate-limiting step for the removal of DOC during sub-surface transport and its impact on the removal of trace contaminants has not been clearly elucidated. A portion of this study examined the mechanistic rate-limiting step for the removal of DOC during sub-surface transport and its potential impact on the transformation of trace organic compounds.

The robustness of microbial transformation during sub-surface transport were examined by analyzing the relationship between surface area and travel time. While no definitive survey results are available, the lack of failures of bank filtration systems and other groundwater recharge systems in terms of improving water quality is readily apparent. Travel times of 2 to 6 months have been effective at removing trace organics during sub-surface transport in both bank filtration systems and groundwater recharge systems using spreading basins. The majority of microbial activity occurs on surfaces and most biological reactors that use attached microbial growth are designed based upon surface area. Based upon the potential survival time of viruses, travel time criteria are already in use for groundwater recharge systems. These travel time criteria are in the same range of times as the travel times that are apparently effective for the removal of trace organics. Therefore, it might be possible that existing travel time criteria for virus survival are also appropriate for the removal of trace organics during sub-surface transport. A travel time criterion for trace organic removal could potentially eliminate the need for inappropriate surrogates for organic carbon transformations such as DOC. Persistent DOC concentrations can vary significantly between different systems since background natural organic matter concentrations and other factors are highly variable. A scientific basis for using travel time criteria for organic removal may be developed to eliminate concentration based criteria that are sometimes inappropriate. During this study, a range of common aquifer materials, hydraulic conductivities and associated surface areas were examined to develop a relationship that normalizes travel time to surface area for a wide range of aquifer materials.

METHODS

An experimental investigation of the removal of high molecular weight organic compounds consisted of 4 reactors operated with two particle sizes and two different flowrates. The reactors were one meter in length with 8 cm inner diameter and were constructed of Plexiglass. Two different sizes of clean silica sand were obtained from Border Products, Arizona and were sieved to obtain geometric mean particle sizea of 0.6 mm and 0.353 mm. The sand was packed in the above columns to an average dry packing density of 1.4 g/cm^3. The four reactors were acclimated with tertiary effluent from Mesa Northwest Reclamation Plant, Arizona for a period of 50 days. A synthetic feed consisting of dextran (average M.W. = 1,000) at a concentration of 6.8 mg/L was used. Dextran was used since it is representative of high molecular weight compound that is completely biodegradable and it can be easily measured with a carbohydrate analysis. Routine monitoring of turbidity, UV254, dissolved organic carbon and carbohydrate was done.

A modified biodegradable organic carbon (BDOC) reactor was used to measure the kinetics of dextran utilization using sand originally acclimated to tertiary effluent. The BDOC reactors used were 500 ml Erlenmeyer conical flasks containing 100 g of biologically active sand in each of the reactors. The reactors were incubated with 300 ml of dextran to mimic the synthetic feed used in the column studies. The DOC and carbohydrate was monitored every day since day zero for dextran utilization for a period of 5 days.

An analysis of hydraulics during sub-surface transport was done by combining basic groundwater hydrology with an estimation of surface area contact during transport. For the estimation of surface area, cubic packing of soil particles was assumed. The specific surface area (A, m^2/m^3) of the particles as shown in equation 1 is the surface area of a sphere per volume occupied by the soil particles,

$$A = \frac{4\pi r^2 n^3}{x^3} = \frac{4\pi r^2 n^3}{(2nr)^3} = \frac{\pi}{2r} \tag{1}$$

where r is radius of the particles, n is the number of particles and x is the distance in the direction of flow. To account for particle size distributions, the surface areas of different sieve fractions were used to develop a range of

potential surface areas for different aquifer materials. Darcy's Law (equation 2) was used to estimate the travel times based on a range of hydraulic conductivities associated with different aquifer materials.

$$q = K \frac{dh}{dx}$$
(2)

where q is the darcy velocity, K is hydraulic conductivity and dh/dx is the hydraulic gradient. Knowing the porosity η, one can then obtain the linear velocity V using equation 3.

$$V = \frac{q}{\eta}$$
(3)

The distance x traveled is simply $x = V * t$ where t is the travel time. For typical ranges of hydraulic conductivities with different aquifer materials, the surface area contact as a function of travel time for a constant hydraulic gradient was calculated.

RESULTS

The results for BDOC tests using Dextran are presented in Figure 1. While data was collected on days 4 and 5, the concentrations were below detection limits and were not considered for the kinetic analysis. The most striking result from this test is the linear relationship with time suggesting a zero-order reaction. Since dextran is a high molecular weight compound, hydrolysis by extracellular enzymes may be necessary before biodegradation can occur. If hydrolysis is the rate-limiting step, then zero-order kinetics are often appropriate to model substrate utilization. No significant production of soluble microbial products was observed as the DOC concentration and carbohydrate concentrations correlated for the duration of the experiment. The majority of persistent DOC in

Dextran - BDOC Reactors - Zero Order Reaction

Concentration = 5.169 mgC/L
k = 1.6 mgC/L-day

$y = -1.6432x + 5.3648$ $R^2 = 0.9891$
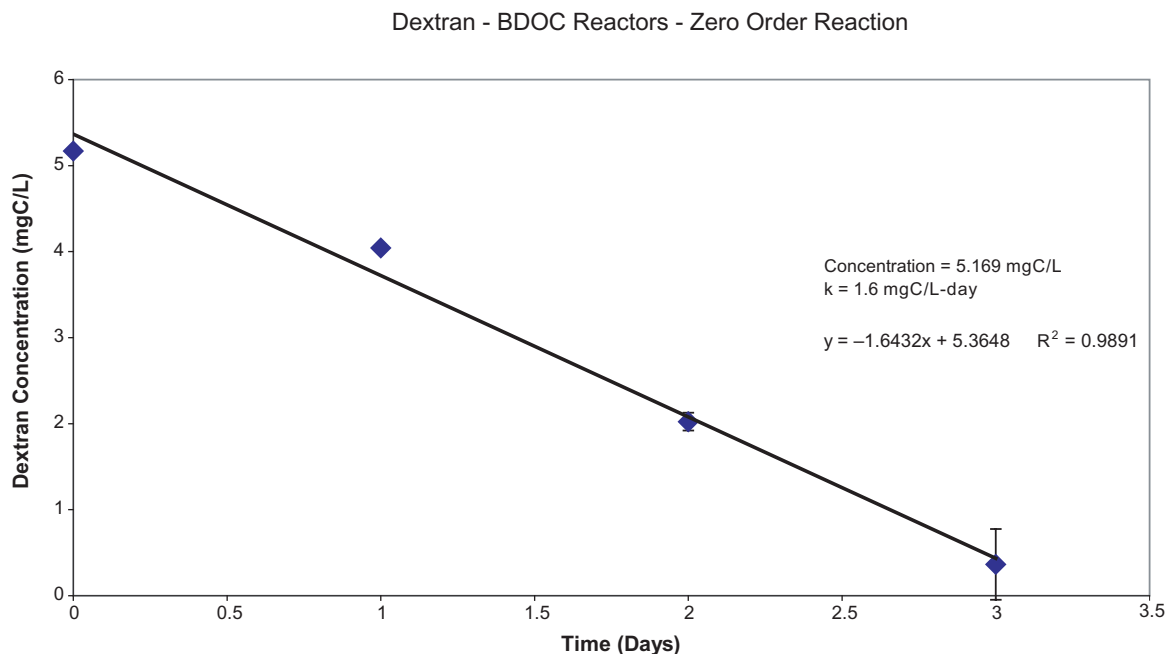
Figure 1. Dextran Concentration (Expressed as C) as a function of time in a BDOC reactor

reclaimed waters consists of high molecular weight compounds that would also require extracellular hydrolysis for biodegradation. However, both BDOC tests and soil column studies have demonstrated first-order kinetic relationships for the biodegradation of DOC in reclaimed waters. It is still possible that hydrolysis is the rate-limiting step and zero-order kinetics actually occurs although first-order kinetics are observed. This is possible during BDOC tests since sorption and desorption occurs with the complex matrix of organic compounds present in reclaimed waters. During column studies, both sorption and microbial growth may make the kinetics of DOC biodegradation appear to be first-order. However, the column studies done using dextran also followed zero-order kinetics and zero-order kinetics are observed in the BDOC reactors. This is logical since zero-order substrate utilization kinetics will eliminate spatial distribution of biomass and the biomass activity in the columns was not a function of distance.

Other analyses were also done to examine the potential rate-limiting steps for microbial removal of DOC. An analysis of the fluxes present in the column studies demonstrated that biofilm thicknesses were less than one micron thick if only one percent of the surface area was available for microbial attachment. This demonstrates that diffusion through a biofilm should not be occurring. Mass transfer resistance between the bulk liquid and the soil surface was demonstrated to be negligible based on typical groundwater velocities. Therefore, the most important rate-limiting factors for substrate utilization appear to be extracellular hydrolysis and microbial growth which will both depend on surface area contact.

Figure 2 is a plot of normalized surface area which represents the surface area coverage divided by travel time as a function of the average particle diameter. One assumption was the range of particle size distributions for different aquifer materials obtained from the United States Department of Agriculture data base may be generalized to common aquifer materials. The second key assumption was that all systems are subject to a constant hydraulic gradient. The normalized surface area increases sharply until a 1 mm average particle diameter and then falls down gradually

Normalized Surface Area vs. Average Particle Diameter
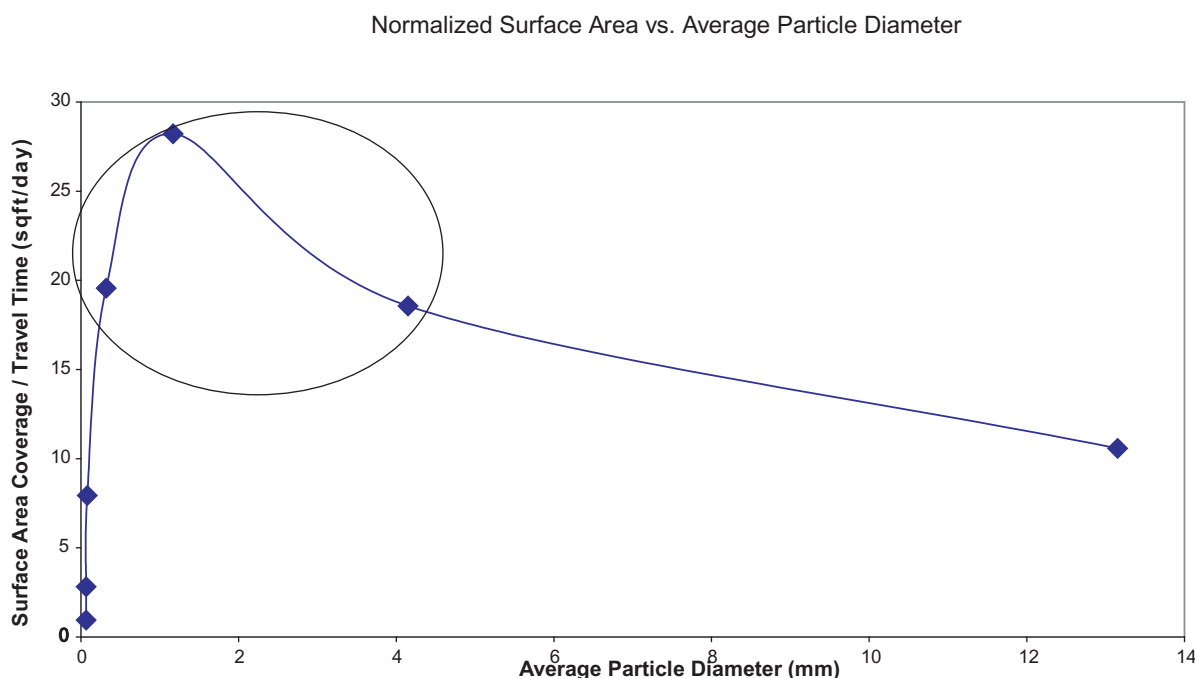
Figure 2. Normalized Surface Area as a function of common aquifer materials.
Surface area contact is normalized to travel time for a uniform hydraulic gradient.

with respect to the average particle diameter. The circle in Figure 2 indicates the normalized surface area in the range for typical aquifer materials found in bank filtration and aquifer recharge systems. There is less than a factor of 2 difference in normalized surface area for this range of aquifer materials. The results indicate that if a travel time criteria were used for systems with these aquifer materials, they would all have similar surface area contact if the hydraulic gradients were similar. While an analysis of different hydraulic gradients has not been done, typically larger hydraulic gradients develop with smaller particle sizes that have lower hydraulic conductivities and vice-versa. Based on the results presented in Figure 2, variations in hydraulic conductivity should not have a major impact on surface area contact. This could be a key reason why SAT and Bank Filtration systems tend to function well in improving water quality independent of the different operating conditions and mineralogy.

CONCLUSIONS

The use of a synthetic substrate to simulate dissolved organic carbon transformations during sub-surface transport provided insight into the rate-limiting mechanisms. Zero-order substrate utilization kinetics commonly associated with the hydrolysis of high molecular weight compounds were observed during batch tests. The rate limiting step for DOC biodegradation is dependent on surface area contact during sub-surface transport.

The robustness of groundwater recharge systems with respect to the removal of organic compounds has been observed around the world. An elementary analysis of hydraulics to develop a relationship that normalizes surface area contact to travel time was completed. The results demonstrated that major variations in surface area contact for a specific travel time with a uniform hydraulic gradient do not occur with aquifer materials representative of groundwater recharge systems. Therefore, travel time criteria presently based on virus survival might also be appropriate for the removal of organic compounds during subsurface transport.

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Visual CXTFIT – a user-friendly simulation tool for modelling one-dimensional transport, sorption and degradation processes during bank filtration

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Abstract

Successful predictions of the fate and transport of solutes during bank filtration and artificial groundwater recharge depends on the availability of accurate transport parameters. We expand the CXTFIT code (Toride et al., 1995) in order to improve the handling by pre- and post processing modules under Microsoft EXCEL. Inverse modelling results of column experiments with tracers, pharmaceutical residuals and algae toxins demonstrate the applicability of the advanced simulation tool.

Keywords

Advection-dispersion equation, analytical solution, inverse modelling, graphical user interface, sorption, degradation.

INTRODUCTION

Within the interdisciplinary NASRI research project (Natural and Artificial Systems for Recharge and Infiltration) dealing with river bank filtration processes, a set of column experiments were carried out to understand mechanisms of transport, sorption and biodegradation of different compounds. To identify these parameters the CXTFIT code was selected and embedded in a pre- and postprocessing routine programmed with Visual Basic for Applications. With the help of a graphical interface and using EXCEL worksheets and graphs, experimental data sets can easily be transferred and results – observed and fitted breakthrough curves – are depicted simultaneously. Thus, particularly experimentally working groups are enabled to handle this user-friendly simulation tool both for analysing and planning of experiments.

The transport code CXTFIT (Toride et al. 1995) utilises analytical solutions for transport parameter estimation. Solutions of the one-dimensional transport equation for the fluid phase concentration c:

$$R \frac{\partial c}{\partial t} = D \frac{\partial^2 c}{\partial x^2} - v \frac{\partial c}{\partial x} - \mu c + \gamma \qquad \text{1)}$$

with retardation factor R, dispersivity D, velocity v, decay constant μ and production γ are implemented for different boundary and initial conditions. Both boundary and initial conditions are optional: the Dirac delta input, multiple pulse input (with constant concentration as special case), and exponential input. Alternatively the transport equation can be taken in dimensionless or in stochastic form and the concentrations in flux-averaged form. The non-equilibrium sorption situation can also be treated as a generalisation of equation (1).

The implemented analytical solutions are extensions of the classical solution of the advection-diffusion equation of Ogata and Banks (1961), some of which were also presented by van Genuchten (1981).

Analytical solutions have the advantage that the accuracy of the results does not depend on certain numerical conditions to be fulfilled. In contrast the accuracy of numerical solutions is coupled to stability conditions as the grid-Péclet condition, the Courant- or the Neumann condition (Holzbecher 1996). The user of CXTFIT thus does not have to keep an eye on such criteria for temporal or spatial step size.

USER INTERFACE

Visual CXTFIT is implemented in Visual Basic for use in Microsoft EXCEL. The implementation work was performed at the IGB Berlin. The program is started either by double click directly on the EXCEL file *VCxtfit.xls* or from EXCEL directly using an Add-In. At the right of EXCELs' menu bar at the top of the screen the item 'Visual Cxtfit' appears, as shown in Fig. 1.
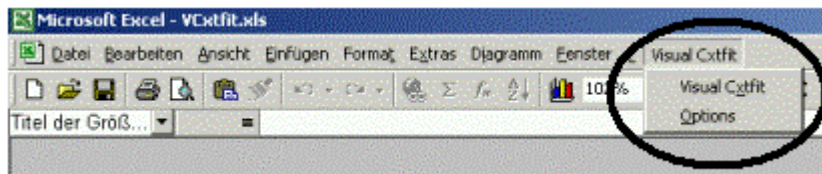
Figure 1. Appearance of CXTFIT in EXCEL

Fig. 1 depicts the new Excel menu bar after opening the worksheet. The item 'Visual Cxtfit' runs the program, the item 'Options' opens a window, where path and file settings can be entered. The Visual CXTFIT main window opens after selection of the 'Visual Cxtfit' menu, and it is shown in Fig. 2.
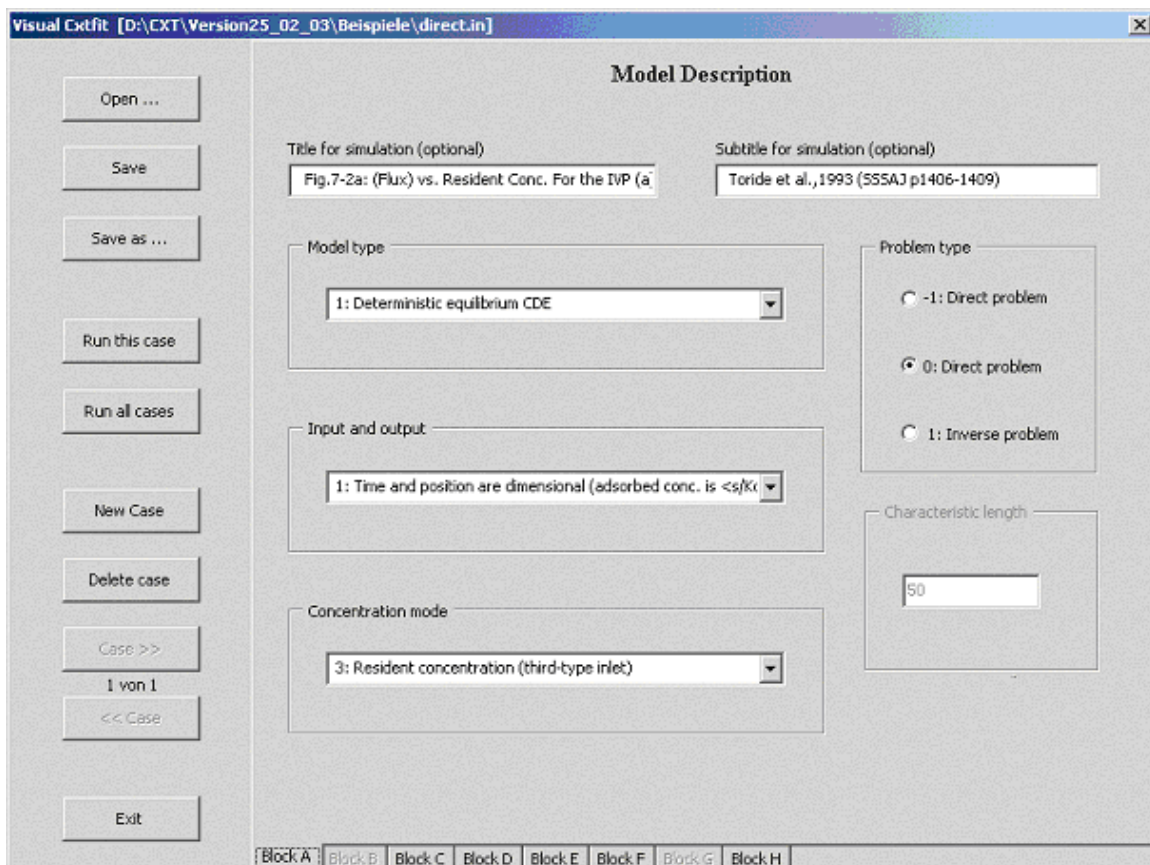
Figure 2. User Interface of Visual CXTFIT

In the main window all operational commands are gathered, like 'open', 'save', 'save as', 'new', 'exit'. The 'run' button starts the CXTFIT code in the background and presents the results in EXCEL and ASCII-formats. In addition major options can be set in the main window: the type of the differential equation and of the concentration, if independent variables are dimensionless or not, or if the direct problem is solved only.

Within the CXTFIT code these options are gathered in Block A, which appears in the GUI view in Fig. 2 as a footer name at the bottom. Concerning further input data the block-structure of the CXTFIT code is represented in the user interface. In Block B various inverse modelling parameters can be specified (in Fig. 2 this block is deactivated, because the direct problem option is selected. In Block C transport parameters are set, which the program fits during an inverse modelling run, if the parameter is to be determined, or remain unchanged otherwise. Parameters selected for estimation should be given minimum and maximum constraints. In Block D the boundary conditions are set, in Block E the initial conditions. Block F deals with the production term, Block G with observed data for the inverse problem. Finally in Block H temporal and/or spatial increments for the output of the estimated analytical solution are specified by the user.

The CXTFIT program starts after a click on the 'run'-button (see Fig. 2). There is an option to treat several cases in one run. After the run a report is given on an ASCII file, as well as tabular and graphical output in EXCEL. Example outputs are shown in the next sub-chapter.

APPLICATIONS

Different kind of field, semi-technical and column experiments in the NASRI research project were carried out with tracers (e.g. Bromide, Gadolinium), pharmaceutical residuals (e.g. Primidone, Carbamazepine) and blue algae toxins (Microcystin). First of all the model was used to estimate best fits for the transport parameters velocity v/φ and dispersion length $\alpha_L = D/v$ (with porosity φ) regarding different types of input functions. Especially for field experiments these parameters are used as first approximations to be improved with the help of numerical modeling. For column experiments, based on these analyses inverse modeling of retardation and decay leads to a better understanding of sorption/desorption and biodegradation effects, sometimes coupled with mobile-immobile transport behavior. Results of Visual CXTFIT modeling often are used further for reactive transport simulation. Three examples are given in the following sub-chapters.

Dual tracer experiment in a semi-technical field site

The aim of this study is to identify the spatial distribution of the hydraulic conductivity and dispersivity of an artificial semi-technical experimental site, where bank filtration can be modelled in a combined surface water –-groundwater system (Nützmann et al. 2004). The shallow aquifer is 3–4 m thick and consists of two layers of coarse and fine gravel sediments. A dual tracer experiment with Bromide (NaBr) and Gadolinium (Gd-DTPA) was carried out and the breakthrough curves (BTC's) observed at 15 groundwater observation wells with different filter depth in two transects over a travel distance of 32 m enabled a three-step analysis comprising (1) estimation of non-steady tracer input function, (2) inverse modelling of one-dimensional groundwater transport with Visual CXTFIT and, (3) modelling of fully 3-D flow and transport in the aquifer. Here, only results of the second step are reported.

Because of very similar boundary conditions at the shore line, the shapes of both tracer breakthrough curves differ very slightly. In Fig. 3 the temporal distribution of observed and simulated Gd-DTPA concentrations for three single wells with increased distance from the injection location can be seen. Fig. 3A and 3B show a close agreement between measured and modelled BTC's but with slightly different values of flow velocities and dispersion. From Fig. 3C an additional mechanism like mixing of water bodies with different concentrations as a result of 3-D flow
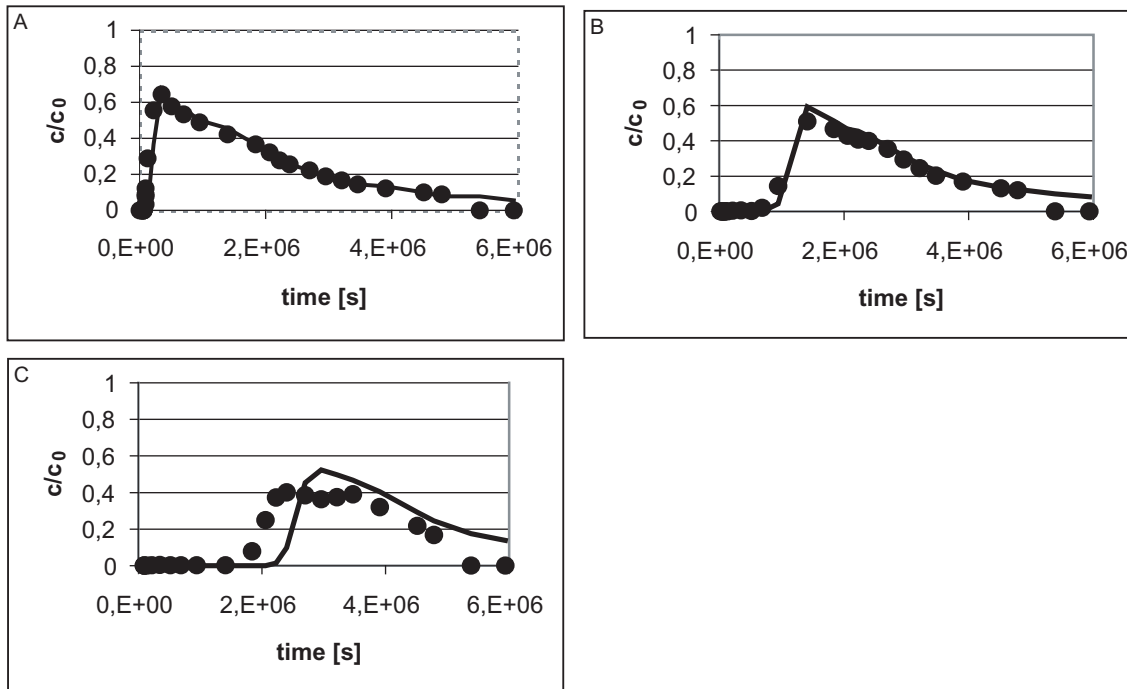
Figure 3. Measured (dots) and simulated (line) Gd-breakthrough curves
(A: 2m, B: 10m, C: 22m distance from the tracer injection)

structures is detectable, what leads to a non sufficient approximation. With the help of the one-dimensional simulator Visual CXTFIT a likely range of variation for the flow velocities of the sandy and gravel layer could be identified (from 0.3 to 7.1 m per day). Again, the variation of the dispersion lengths is in a higher range than the variation of the velocities. The identified parameters demonstrate a first approximation of the spatial distribution of transport coefficients, which could be improved only by using a fully three-dimensional modelling approach.

Fate of PhAC's during enclosure experiments

Several experiments were carried out with so-called enclosures (cylindrical pipes with about 1 m height and an area of 1 m^2) to study the fate of selected pharmaceuticals in a sandy soil without and with a thin clogging layer. Five pharmaceuticals, namely clofibric acid, ibuprofen, primidone, carbamazepine and diclofenac were chosen for this study. All substances have already been found in the aquatic environment and have been selected as model substances due to their physico-chemical properties and their expected or reported different behaviour during groundwater recharge. Apart from their initial concentrations the observed change of concentration was quite similar for all four compounds, but the identified degradation coefficients are unequal (Table 1). Whereas all

Table 1. Estimated values of the decay coefficient μ [1/h] for different PhAC compounds during enclosure filtration experiments (from Mechlinski et al., 2005)

Depth below surface [cm]	Clofibric acid	Ibuprofen	Primidone	Carbamazepine	Diclofenac
40	0.0	1.6	0.0	0.0	0.09
80	0.05	2.0	0.0	0.04	0.1
40 *	0.0	2.0	0.0	0.0	0.0
80 *	0.0	2.0	0.0	0.0	0.03

* Experiments with a clogging layer.

pharmaceuticals are retarded (1.2 < R < 2.9), only ibuprofen was efficiently degraded during slow sand filtration. This may indicate, that ibuprofen is subject to a microbial degradation and the removal is not only a result of an adsorption to the sediment (Heberer *et al.* 2004).

Transport and degradation of cyanobacterial toxins

Laboratory and technical scale experiments have also shown, that breakthrough of particle-bound as well as extra-cellular cyanotoxins is possible (Bricelj and Sedmak 2001, Chorus et al. 2004). Similar to the PhAC experiments discussed above enclosure experiments are carried out to identify transport parameters like dispersion, adsorption and decay of microcystins (MCYST). Pulse-type injection of tracer and MCYST into the water column above the sediment layer leads to a decrease of concentration of both substances due to mixing with fresh water. First, the normalized input concentration was estimated as a function of time, see Fig. 4.
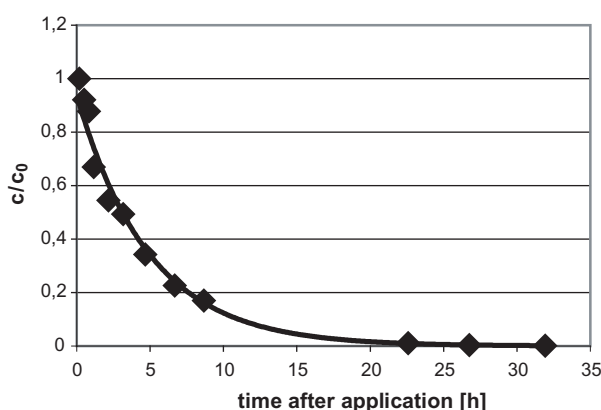
Figure 4. Measured and approximated MCYST input function

Then, the BTC's of tracer and MCYST at 4 different sampling points (20, 40, 60, 80cm depth) and the outlet (120 cm depth) of the enclosures are estimated with VisualCXTFIT using the fitted input function. The results are shown in Fig. 5 for two sampling points.
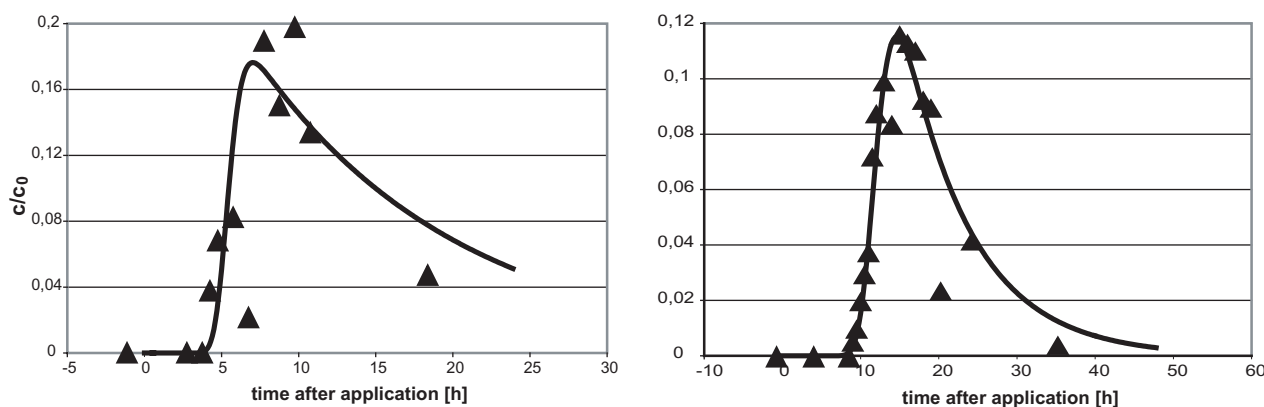
Figure 5. Measured and modeled MCYST breakthrough curves (left: 60 cm below the inlet, right: outlet)

The identified values of retardation varies from 1.2 < R < 1.8, whereas their distribution is not correlated with the depth. In contrast to that, the decay coefficient range from $\mu = 1.0$ [1/h] at 20 cm to $\mu = 0.17$ [1/h] at the outlet showing a decrease of microbial activity with increasing depth and a partial removal of MCYST during slow sand filtration under saturated conditions. Recent infiltration experiments during the NASRI project confirm, that MCYST breakthrough can occur at different depths depending on the biological activity at the infiltration surface, pore water flow velocity, geochemical composition of the sediments and prevailing redox conditions during infiltration.

CONCLUSIONS

Using the Visual CXTFIT add-in measured breakthrough curves for tracers, pharmaceuticals and algae toxins were successfully reproduced. Especially for column and enclosure experiments mechanistic studies are carried out to find basic transport parameters. Even in the case of more complex flow situations during field experiments the identified parameters are used as initial values to be improved with numerical modelling. The simulation results make evident that this easy-to-use program can be a fundamental tool not only for modellers but for experimental working groups also.

ACKNOWLEDGEMENTS

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On the use of reactive multicomponent transport modelling for assessing water quality changes during managed aquifer recharge

H. Prommer and P. Stuyfzand

Abstract

Managed aquifer recharge is an increasingly used technique to supplement water supplies. Following recharge the water quality of the injectant is typically altered by a multitude of geochemical processes during subsurface passage and storage. Relevant geochemical processes that affect the major ion chemistry include microbially mediated redox reactions, mineral dissolution/precipitation, sorption and ion-exchange. Reactive multi-component transport modelling provides an ideal platform for the simultaneous quantification of these processes and to develop an understanding of how they interact. The present paper demonstrates this for the analysis of a comprehensive data set that was collected during the deep-well injection of aerobic surface water into an anoxic, pyritic aquifer in the Netherlands.

Keywords

Deep well injection, geochemical transport modelling, pyrite oxidation, temperature, PHT3D.

INTRODUCTION

Managed aquifer recharge is increasingly used to enhance the sustainable development of water supplies. Common recharge techniques include aquifer storage and recovery (ASR), infiltration ponds, river bank filtration and deep-well injection. Following recharge the water quality of the injectant is typically altered by a multitude of geochemical processes during subsurface passage and storage. Relevant geochemical processes that affect the major ion chemistry include microbially mediated redox reactions, mineral dissolution/precipitation, sorption and ion-exchange. The hydrochemical conditions and changes that occur under these circumstances, in particular the temporal and spatial changes of pH and redox conditions, are in many cases the controlling factor for the fate of micropollutants such as herbicides and pharmaceuticals. Similarly, changes in mineralogical composition such as dissolution and precipitation of iron- or aluminiumoxides may affect the mobility of trace metals as well as the attachment and subsequent decay of pathogenic viruses (Schijven et al., 2000, Zhuang et al., 2003). Laboratory and field-scale experimental studies are aimed at investigating such processes under controlled conditions and to eventually develop a better qualitative and quantitative understanding of their complex interactions, both site-specific and at a fundamental level. However, a comprehensive and integrated in-depth analysis of the recorded experimental data is often omitted, despite the sometimes significant cost for the experimental work and despite the potential long-term benefits for designing and operating efficient, sustainable and safe recharging schemes. The present paper gives a summary on how reactive multicomponent transport modelling was used at a recharge site as a technique for a more rigorous analysis and the quantification of physical and biogeochemical processes that can explain the observed data. A full description of the modelling study can be found in Prommer and Stuyfzand (2005). A second modelling example is presented in the companion paper by Greskowiak et al. (2005).

DEEP-WELL INJECTION EXPERIMENT

The technical feasibility for deep well recharge of canal water was investigated by two Dutch water supply companies (Brabant Water and WML) in order to halt and reverse water table drawdown and to minimise the potential negative impact on wetlands within the capture zone of water supply wells. A pilot plant was built near Someren in Southern Netherlands and a trial injection of initially 720 m^3 day^{-1} (between day 0 and 726) and later of 960 m^3 day^{-1} (between day 727 and day 854) was carried out. It was accompanied by a detailed site characterisation and a comprehensive water quality monitoring program. Pre-treated surface water from a canal was injected by one well at ~300 m depth (273 m–312 m below the land surface) and withdrawn by an extraction well located 98 m away from the injection well. The target aquifer is anoxic (pyritiferous), very low in calcite (probably absent) and has a near-neutral pH. Its geochemical reactivity is largely defined by locally varying concentrations of pyrite and sediment-bound organic matter (BOM). Both, the hydrochemical composition and the temperature of the injection water changed significantly during the course of the trial period. Most notably the temperature varied seasonally between a minimum of 2 °C and a maximum of 23 °C The field observations show that oxygen depleted rapidly in the vicinity of the injection well while nitrate was removed at a slower rate. However, no elevated nitrate concentrations were found in the extraction well. More details can be found in Stuyfzand (1999).

FLOW, CHLORIDE AND HEAT TRANSPORT SIMULATIONS

The deep well injection and extraction scheme created a three-dimensional flow field that strongly determined the hydraulic gradients and mass transport in the vicinity of the wells. The symmetrical flow-field that resulted from the injection was approximated by a half-model of 263 m length (in flow direction) and 124 m width, i.e., perpendicular to the main flow direction. In vertical direction the model was discretized into 12 layers of different conductivities between −273 m and −340 m depth, as proposed in the conceptual model by Stuyfzand (1999). Initially nonreactive transport of chloride was simulated and the results compared with the measured concentrations. A good agreement between simulated and observed data was achieved after small adjustments of the initially attributed hydraulic conductivities were made. This step was followed by a fine-calibration that used the simulation of heat transport and its comparison with measured temperature data as an additional constraint for the flow and nonreactive transport model. Heat transport was modelled in a simplified manner by defining the water temperature as a retarded, mobile species.

REACTIVE MULTICOMPONENT TRANSPORT

Based on the field observations a reaction network was defined and a corresponding reaction module was formlated for the multicomponent transport model PHT3D (Prommer et al., 2003), a model that couples the transport simulator MT3DMS (Zheng and Wang, 1999) with the geochemical model PHREEQC-2 (Parkhurst and Appelo, 1999). The reaction network included complexation reactions of all major ions, dissolution/mass transfer of sediment bound organic matter to dissolved organic carbon (DOC), kinetically controlled DOC mineralisation, ion-exchange reactions and the kinetically controlled, temperature-dependant oxidation of pyrite by oxygen and/or nitrate. The duration of the pilot test was subdivided into 39 stress-periods of variable length and the measured injection water compositions were used to define the source concentrations at the injection well. The ambient water composition was assumed to be homogeneous, however, pyrite and BOM contents as well as the cation exchange capacity varied with depth, according to the zonation proposed by Stuyfzand (1999). The model calibration was used to determine both the magnitude and the relative importance of pyrite oxidation in comparison with the organic carbon mineralisation. Matching simulated and measured sulfate concentrations served as the major constraint for the estimation of the parameters that determined the magnitude of the pyrite oxidation rates, while

matching simulated and observed oxygen and nitrate concentrations served as the main constraint for the combined effect of pyrite oxidation and DOC degradation.

MODELLING RESULTS

The calibrated model reflects in great detail the spatial and temporal changes of the major ion and redox chemistry that occurred in response to the injection of the aerobic surface water. For example, the model is capable of accurately reproducing the extent and the seasonal variability of the oxygen and nitrate plumes that are created during injection. Fig. 1 illustrates the extent of those plumes 730 days after the start of the experiment. The snap-shot shown corresponds to a period where low temperatures of the injected water leads to comparatively large oxygen and nitrate plumes. The degree of variability is indicated in Fig. 2, which shows the breakthrough curves of
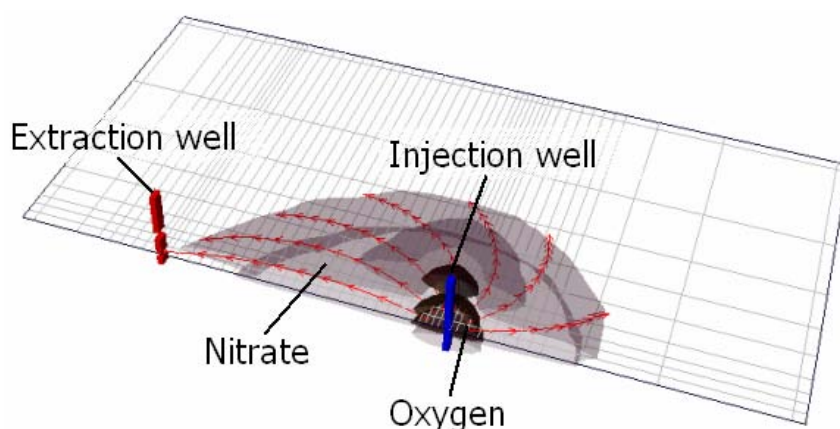
Figure 1. Illustration of the modelled redox-zonation - Iso-concentration surfaces
of oxygen and nitrate at 730 days after the start of the deep well injection experiment.
Also shown are pathlines between injection and extraction well.
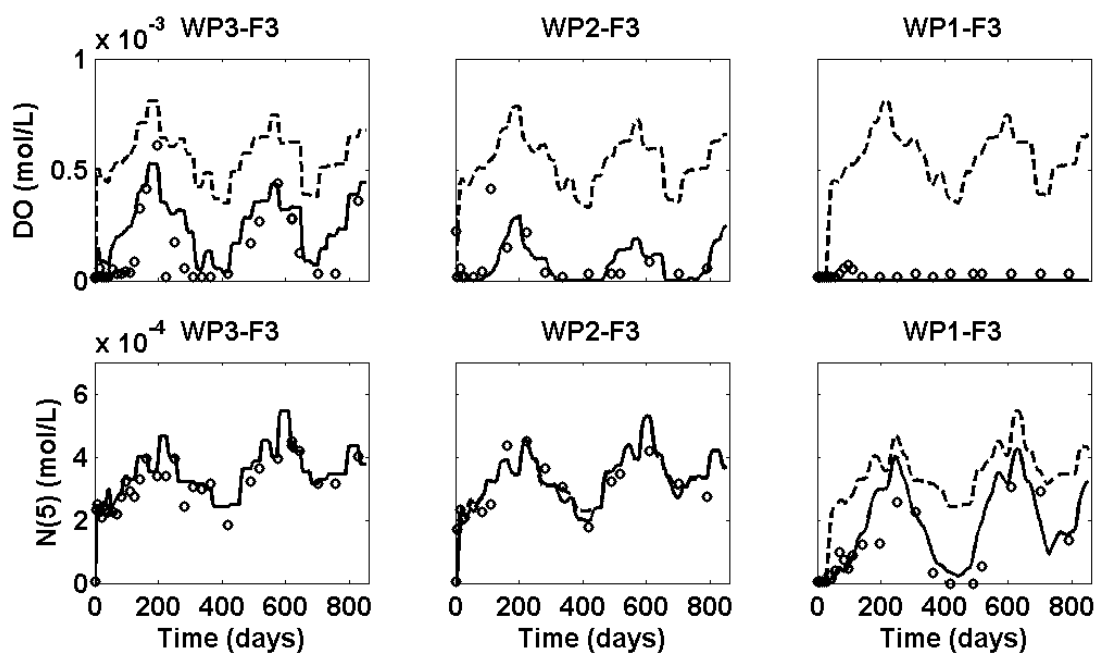
Figure 2. Measured (circles) and simulated (solid lines) breakthrough curves of oxygen and nitrate
at three different distances from the injection well (WP3 = 8 m, WP.2 = 12 m, WP.1 = 38 m).
Dashed lines show the simulated breakthrough curves for advective dispersive only (no reactions).

oxygen and nitrate at three different distances from the injection wells, at 8 m, 12 m and 38 m, respectively. The plots show how, influenced by a seasonal variability, initially oxygen and with increasing distance also nitrate concentrations are decreasing. The results of model runs where only advective dispersive transport was considered, that is, all reactions were switched off, are plotted in comparison. The results illustrate that nitrate is transported conservatively as long as oxygen is present.

The model calibration identified pyrite oxidation as the dominant process with respect to oxygen and nitrate removal, consuming 6.34 of the 9.11×10^6 mol electron equivalents (Eeq) injected into the aquifer. This is indicated in Fig. 3, where the simulated rates of the spatially integrated oxygen and nitrate consumption are plotted in comparison with the removal rates of pyrite and BOM. Through a more detailed modelling analysis (Prommer and Stuyfzand, 2005) it was shown that pyrite oxidation rates exhibit a strong temperature dependency, which can explain a large fraction of the observed seasonal variability of the breakthrough curves at the monitoring wells. For that study the temperature-dependency was approximated from the rates of measured oxygen consumption in the proximity of the injection well and incorporated into the appropriate rate expressions through an Arrhenius-type term in the equations for pyrite oxidation and BOM mineralisation.
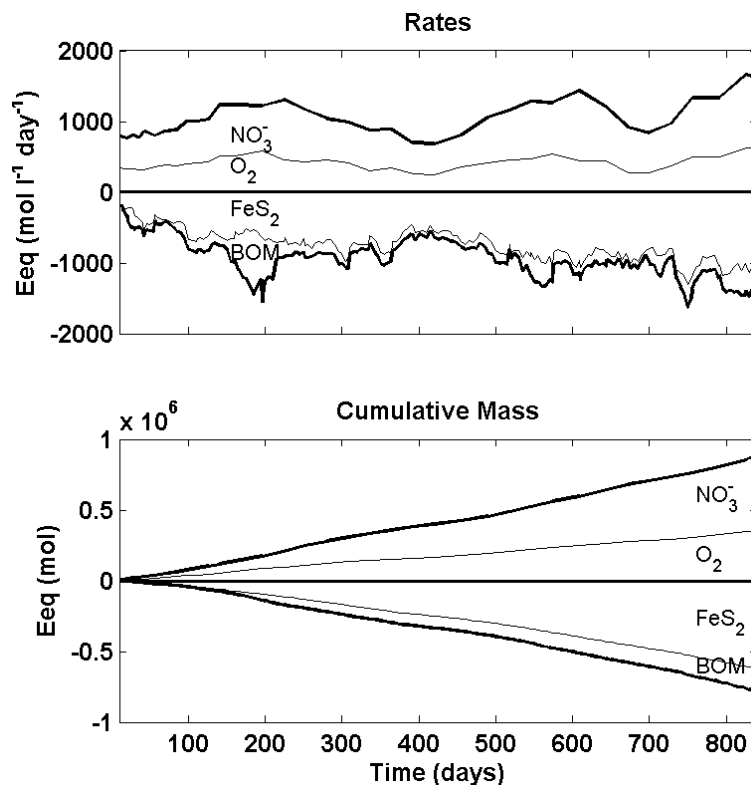
Figure 3. Modelled rates and cumulative mass of electron equivalents (Eeq) added to the aquifer via deepwell injection (as O2 and NO3-) in comparison with the modelled, spatially integrated consumption of pyrite and BOM, respectively.

In contrast to the observed variability of concentrations at the monitoring wells the extraction water quality showed only a minor impact of seasonality and of temperature variations. This can be seen in Fig. 4, where the results of the reactive transport simulations are plotted in comparison with measured concentrations for some of the key species and water quality parameters. The extraction well collects water with a wide range of differing travel times, which explains why the locally observed variability observed in the monitoring wells are dissipated in the extraction water. Through the comparison of the modelling results for reactive transport with those of conservative (i.e., advection-dispersion only) transport simulations, the impact of physical and chemical processes on the extraction water

quality can be distinguished. For example, it can be seen that 30–40% of the increase in sulfate concentrations can be attributed to pyrite oxidation, whereas the remainder is determined from the dynamically changing mixing ratio between ambient water and the more sulfate-rich injectant. Similarly it can be seen that the dissolved inorganic carbon concentrations in the extraction water remain essentially unaffected by reactive processes.
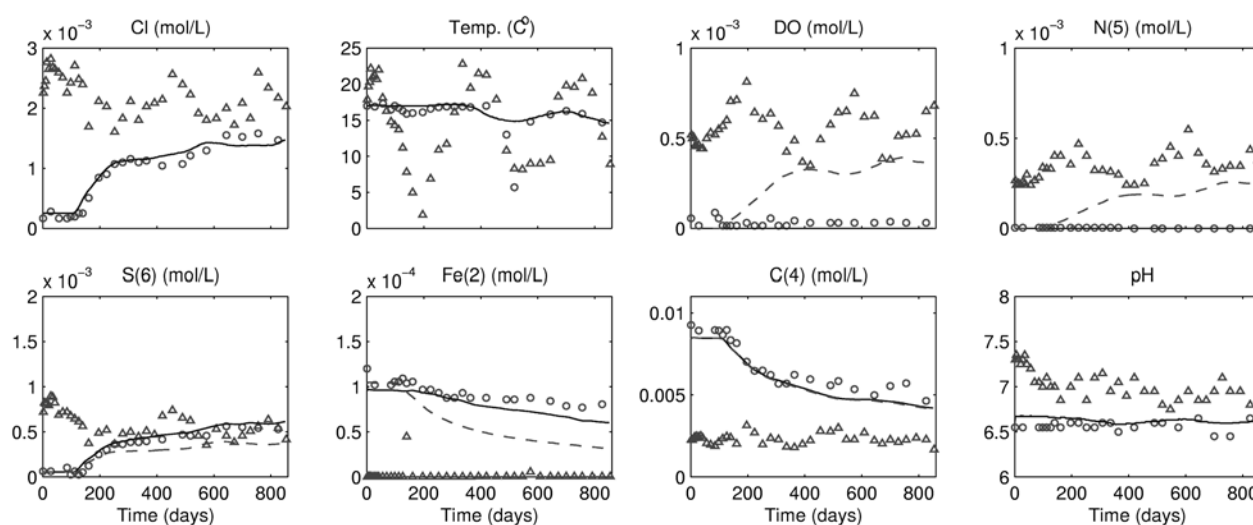
Figure 4. Selected simulated (solid lines) and measured (circles) aqueous component concentrations at the extraction well PP.1 in comparison with measured concentrations at the injection well IP.2 (triangles). The dashed lines indicate the simulated extraction well concentrations for purely advective-dispersive transport (no reaction).

CONCLUSIONS

Reactive multicomponent transport modelling combines the more widely established modelling techniques of groundwater flow modelling and that of batch-type geochemical modelling. It was shown here to be an adequate tool for the in-depth analysis of the comprehensive data set collected during the deep-well injection experiment. The model was for the particular case here, like for the case described in the companion paper by Greskowiak et al. (2005), used in a mode that included the development and testing of hypotheses how physical, chemical and biological processes interact. In the deep-well injection case the resulting major outcome was the model-based identification and quantification of the impact that the variable temperature of the injection water had on reaction rates. This allowed to accurately describe the spatial-temporal evolution of the major ion chemistry, pH and redox changes that had developed during the trial injection. The capacity to reliably quantify those hydrochemical parameters for artificially recharged aquifers provides the key for the assessment of the fate of potential micropollutants and, for example, to pursue a process-based description of the mobility of pathogenic viruses. The mobility of the latter, which might be controlled by the electrostatic repulsion between viruses and grains, is strongly affected by hydrochemical parameters such as pH, ionic strength and the concentration of divalent cations. Those parameters are of course all intrinsically determined by reactive multicomponent transport models, making it an ideal platform for more detailed studies of those issues.

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Simulating effect of successive cycles in aquifer storage and recovery well in India

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Abstract

Aquifer storage recovery ASR well is a dual purpose well, used for recharge and recovery sequentially. Field experiment conducted cycles in a highly brackish semi-confined aquifer. Calibaration and validation of finite element flow model, HYDRUS-2D was done by drawup/drawdown in pressure heads during recharge/recovery in successive ASR cycles in observation well 10 m away from well. For simulation variable flux boundary conditions were applied at axisymteric elliptical cavity surface. Top, bottom, and side boundaries were closed for flow. Good fit data, high modeling efficiency, high correlation coefficient, and fairly low RMSE in the computed and measured heads during the 2nd, 3rd and 4th ASR cycles validated the model. 20 m pressure isoline advanced during recharge and receded during recovery in x and z direction. So the study suggested that HYDRUS 2D may be adopted for groundwater recharge studies in ASR wells.

INTRODUCTION

ASR systems include ASR well and its influencing zone, piezometer and observation well and are designed to meet seasonal variations in supply, demand and quality (Pyne, 2003). This technique is being increasingly utilized for reducing salinity of brackish aquifers for irrigation (Gerges et al., 2002 a, b; Malik, 2002); to prevent surface ponding in standing crops (Malik, 2000) and for maintaining the desired water levels in fresh water aquifers (Pyne, 1995; Gale et al., 2002; Pavelic et al., 2002) at relatively small cost, that would be economically viable at individual farmer level. Improvement in the quality of water in the aquifer surrounding the ASR prompts the farmer to extract more groundwater for irrigation (Malik et al., 2002) thereby implementing the recommended conjunctive use strategy.

Modelling water pressures around the ASR well would not only quantify this temporal and spatial rise or fall in water levels but also be helpful in assessing the environmental impacts (Gale et al., 2002). Simulation models can integrate information on different geological and hydrological situations and operational factors to help devise appropriate operational schedule for the success of ASR technology. Temporal and spatial water pressure responses near recharge/recovery ASR strainer wells have been modelled using analytical (Anonymous, 1993; Simpson et al., 2003) in saturated flow conditions in homogeneous areas and numerical 2D or 3D approaches (Chiang and Kinzelbach 1998; Williams 2000; Kohfahl et al., 2002; Jorgensen and Helleberg, 2002; Bogdanov et al., 2003) under steady and transient saturated flow conditions. To our knowledge, a few numerical modelling studies have been reported for cavity type ASR wells. A scientifically documented and evaluated (Diodato, 2000), HYDRUS-2D software package of (Simunek et al., 1996) having extensive interface capabilities for simulating saturated and unsaturated water, solute and heat flow under bare and cropped condition. A study was therefore, initiated with the objectives to calibrate and validate the Hydrus-2D model for water level responses in successive ASR cycles in a cavity type ASR well.

MATERIALS AND METHODS

Field experiments

The field experiment was conducted at Soil Research farm of CCS Harayana Agricultural University Hisar, Haryana, India (28°59' to 29°49' N latitude and 75°11' to 76°18' longitude at an elevation of 215.2 m above mean sea level to study the effect of successive cycles of recharge-recovery on water level responses and spatial influence. The ASR well and piezometer were installed under the Indo-German collaborative project on artificial recharge of groundwater through integrated sand filter- injection well technique at CCS HAU Hisar India. First cycle was used for calibration and next three cycles were used for validation.

Parameter estimation

Soil water retention functions were derived from pressure head and water content data measured on pressure plate apparatus using Van Genuchten-Mualem equations (Van Genuchten, 1980) as:

$$\theta(h) = \frac{\theta s - \theta r}{[1 + |\alpha h|^n]^m} \quad h<0 \quad (1) \qquad \theta(h) = \theta s \quad h \geq 0 \quad (2) \qquad K(h) = Ks\, Se^k \left[1 - \left(1 - Se^{k/m}\right)^m\right] \quad (3)$$

Where θ is the volumetric water content, h is the pressure head, α, n, m (=1=1/n), and k (=0.5) are empirical parameters, Se = $(\theta - \theta r)/(\theta s - \theta r)$ is the degree of saturation. The calibration of numerical experiments to generate piezometric pressure heads h time pairs for optimizing effective saturated hydraulic conductivity Ks by comparing with the field observed data using θr, θs, α, n, and m as fixed parameters (Table 1).

Table 1. Hydraulic parameters used in numerical experiments

Parameters	$\theta r\ (m^3 m^{-3})$	$\theta s\ (m^3 m^{-3})$	$\alpha\ (m^{-1})$	$n(-)$	$Ks\ (md^{-1})$	k	$Ak*$
Loam	0.03	0.47	0.0500	2.6	0.8040	0.5	1
Loamy sand	0.05	0.43	0.1000	2.9	0.7290	0.5	1
Clay	0.07	0.54	0.0150	2.5	0.0064	0.5	1
Loamy sand	0.03	0.40	0.0700	3.1	0.1884	0.5	1
Clay	0.06	0.53	0.0013	2.1	0.0002	0.5	1
Loamy sand	0.05	0.10	0.0010	2.9	0.2100	0.5	1

* Ak is anisotropy in hydraulic conductivity.

Numerical experiments

Four successive cycles (see Table 2 for recovery- recharge time schedule) were conducted using HYDRUS-2D to obtain axisymetric three dimensional view of recharge and recovery processes. HYDRUS-2D solves the modified form of Richard equation (4) for variably saturated flow numerically.

Relatively coarse numerical mesh involving nodes depicting the geometry of the flow region, 70 m depth with radius of 500 m and an exocentric elliptical cavity with 1m horizontal radius and 1 m vertical radius as shown in Fig. 1. The observation node was marked at the Piezometer (1) as shown in Fig. 1. Recovery and recharge fluxes as in Table 2 were applied as variable flux boundary condition at axisymteric elliptical cavity surface. The top, bottom, and side boundaries were closed for flow. Recharge and recovery rates were constant i.e. 24 m^3/hr and 55 m^3/hr. Fig. 2 showed the recharge (–) and recovery (+) fluxes as Hydrus 2D works in –ve Z direction.

Table 2. Recovery and recharge schedule for each calibration and validation days

Cycle No	Volume (m^3)					Duration (days)			
	Recharge	Rate (m^3/h)	Recovery	Rate (m^3/h)	%	Recharge	Storage	Recovery	Total
1	2000	24	2000	55	100	4	0	2	6
2	2000	24	2000	55	100	4	0	2	6
3	2000	24	2000	55	100	4	0	2	6
4	2000	24	2000	55	100	4	0	2	6

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial r}\left(K(h)\frac{\partial h}{\partial r}\right) + \frac{K(h)}{r}\left(\frac{\partial h}{\partial r}\right) + \frac{\partial}{\partial Z}\left(K(h)\left(\frac{\partial h}{\partial z}+1\right)\right) \qquad (4)$$
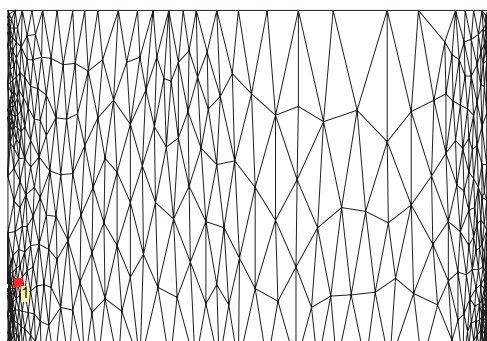
*Figure 1. Observation node number 1
is piezometric points at 10 m
concentric radial distances*
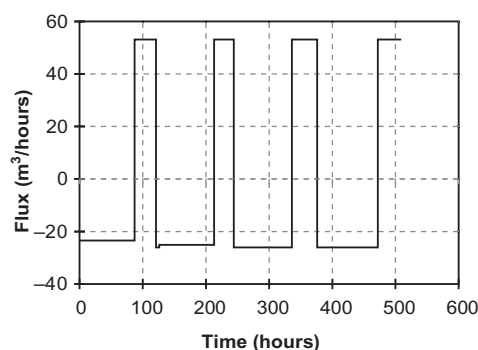
*Figure 2. Recharge and recovery fluxes
as a function of time during*

RESUTS AND DISCUSSION

Calibration

The calibration process was performed for first cycle using inverse modelling on the on the hydraulic conductivity of the aquifer. The correlation coefficient (R^2) 0.81 wasobtained between experimental and observed pressure heads by inverse modelling.

The estimated parameters by inverse modelling are given in Table 1. Goodness-of-fit and statistical measures as modeling efficiency (ME) (eqn. 6) and RMSE (root mean square error) (eqn. 8) as function of time were used to assess simulation performance.

maximum error $(me) = \max \left(\left| Oi - Si \right|\right)_{i=1}^{n}$ (5)   modeling efficiency $(ME) = \left[1 - \frac{\left\{\sum_{i=1}^{n}\left(Oi - Si\right)^2\right\}}{\left\{\sum_{i=1}^{n}\left(Oi - \bar{O}\right)^2\right\}} \right]$ (6)

where Oi and Si represent the observed and simulated time series of heads in the observation well at 10 m from ASR well, n represents the number of observed and simulated values used in the comparison, and $\bar{O}$ the observed average. one is considered to be the best modelling efficiency. Negative values of modelling efficiency are considered

as unacceptable (Ghulam et al. 2004). The root mean square error is useful to quantify the differences between observed data and simulated data with the optimized parameters:

$$\overline{O} = \frac{\sum\limits_{i=1}^{n} Oi}{n} \qquad (7)$$

$$RMSE = \sqrt{\left(\frac{1}{N}\sum_{i=1}^{N}\left[M_i - S_i(b)\right]^2\right)} \qquad (87)$$

where Mi and Si (b) are measured and simulated values for an output variable.

During recharge there was decrease in RMSE with the time (Table 3). RMSE decreased from 1.24 m after 1 hr of recharge to 0.83 m after 87 hrs of recharge. Similar trends were shown by ME also. ME increased from 0.97 to 0.98 from 1 hr of recharge to 87 hrs of recharge. Initially high RMSE and low ME revealed that during the start of recharge there was much deviation between the experimental head and simulated head. Similar trends were observed during the recovery also i.e. from 1 hr to 34 hrs recovery, RMSE decresed from 3.57 m to 3.05 m and ME increased from 0.85 to 0.87 (Table 4). Overall matching between observed and simulated data points was quite satisfactory during calibration except during start of recharge and recovery in each cycle (Fig. 3). It might have been due to the facts: 1) that hydrus 2D model does not take into account the storage coefficient in the aquifer and/or 2) the entry points resistances of piezometers might have delayed their response to drawup and drawdown pressure heads under actual field conditions.

Table 3. Number of observations N, RMSE and ME in drawup during calibration and validation in pressure heads

| | Calibration | | | Validation | | | | |
| | | | | | RMSE (m) | | | |
Time (h)	N	RMSE* (m)	ME**	N	IInd	IIIrd	IVth	Mean ME
1	31	1.24	0.97	31	1.21	1.22	1.23	0.97
15	35	1.13	0.97	35	1.17	1.17	1.19	0.96
45	39	0.98	0.98	39	1.15	1.15	1.21	0.98
87	46	0.83	0.98	46	1.00	1.00	1.15	0.99

* Root mean square error ; ** Modelling efficiency.

Table 4. Number of observations N, RMSE and ME in drawdown during calibration and validation in pressure heads

| | Calibration | | | Validation | | | | |
| | | | | | RMSE (m) | | | |
Time (h)	N	RMSE* (m)	ME**	N	IInd	IIIrd	IVth	Mean ME
1	34	3.57	0.87	300	3.50	3.80	3.82	0.85
16	44	3.20	0.88	300	3.10	3.37	3.51	0.86
34	47	3.05	0.89	300	2.95	3.10	3.21	0.87
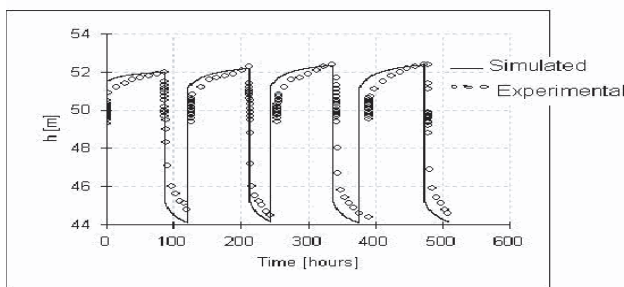
Figure 3. Simulated and experimental drawup/drawdown of piezometric head as given by model during successive recovery and recovery cycles

Validation

2nd, 3rd and 4th ASR cycles were used for the validation of HYDRUS 2D model. Goodness-of-fit and statistical measures as modeling efficiency (ME) (eqn. 6) and RMSE (root mean square error) (eqn. 8) as function of time revealed the similar results as observed during the calibration. There was decrease in RMSE in all the cycles (Table 3) from 1 hr recharge to 87 hrs recharge i.e. during 2nd cycle RMSE decreased from 1.21 m to 1.00 m, during 3rd cycle from 1.22 m to 1.12 m whereas, in 3rd cycle from 1.23 m to 1.15 m. Whereas, mean modeling efficiency increased from 0.97 to 0.99 from 1 hr recharge to 87 hrs recharge. Similarly, there was decrease in RMSE between experimental and simulated heads with increase in recovery time (Table 4). During 2nd cycle RMSE decreased from 3.50 m to 2.98 m, in 3rd cycle from 3.80 m to 3.10 m and in 4th cycle 3.82 m to 3.21 m with a recovery time of 1 hr to 34 hrs. Mean ME increased from 0.85 to 0.87 from 1 hr recovery to 34 hrs recovery.

Overall matching between observed and simulated data points was quite satisfactory except at start of recharge and recovery in each cycle (Fig. 3). It might have been due to the facts: 1) that hydrus 2D model does not take into account the storage coefficient in the aquifer and/or 2) the entry points resistances of piezometers might have delayed their response to drawup and drawdown pressure heads under actual field conditions. This may be reason for Slightly higher RMSE values and low ME during recovery (Table 3, 4).

Spatial influence

There was no appreciable drawup/drawdown in pressure head after a distance of 200 m (Fig. 4) in the 1st cycle. So the spatial influencing of recharge/recovery is upto 200 m. Spatial Influence means the length upto which there is 5% of the maximum drawup/drawdown during recharge and recovery of each cycle. Similarly during the IInd, IIIrd and IVth cycle the influencing zone was 210 m, 220 m and 230m during recharge and recovery in successive cycles. Spatial study also suggest that next tubewell should be installed atleast 230 m away from the existing ASR well. The study suggest that hydrus 2D may be adopted for simulations in ground water recharge studies in ASR well.
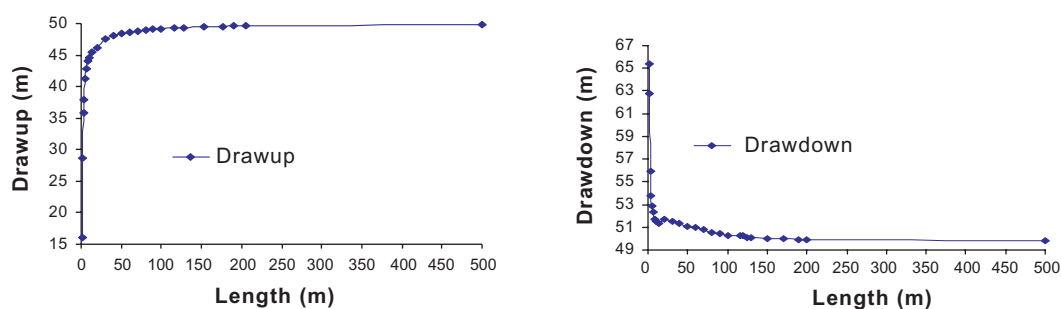
Figure 4. Radial Influence of the Ist ASR cycles during recharge and recovery as given by Hydrus 2D

Development of pressure isolines during recovery and recharge times as taken from 2 dimensional output graphs as presented in Fig. 5 showed that there was mound height rise building during recharge and mound height decrease during recovery near the cavity. The radial influence of recharge and recovery on ground water drawup/drawdown was not more than 325 m.
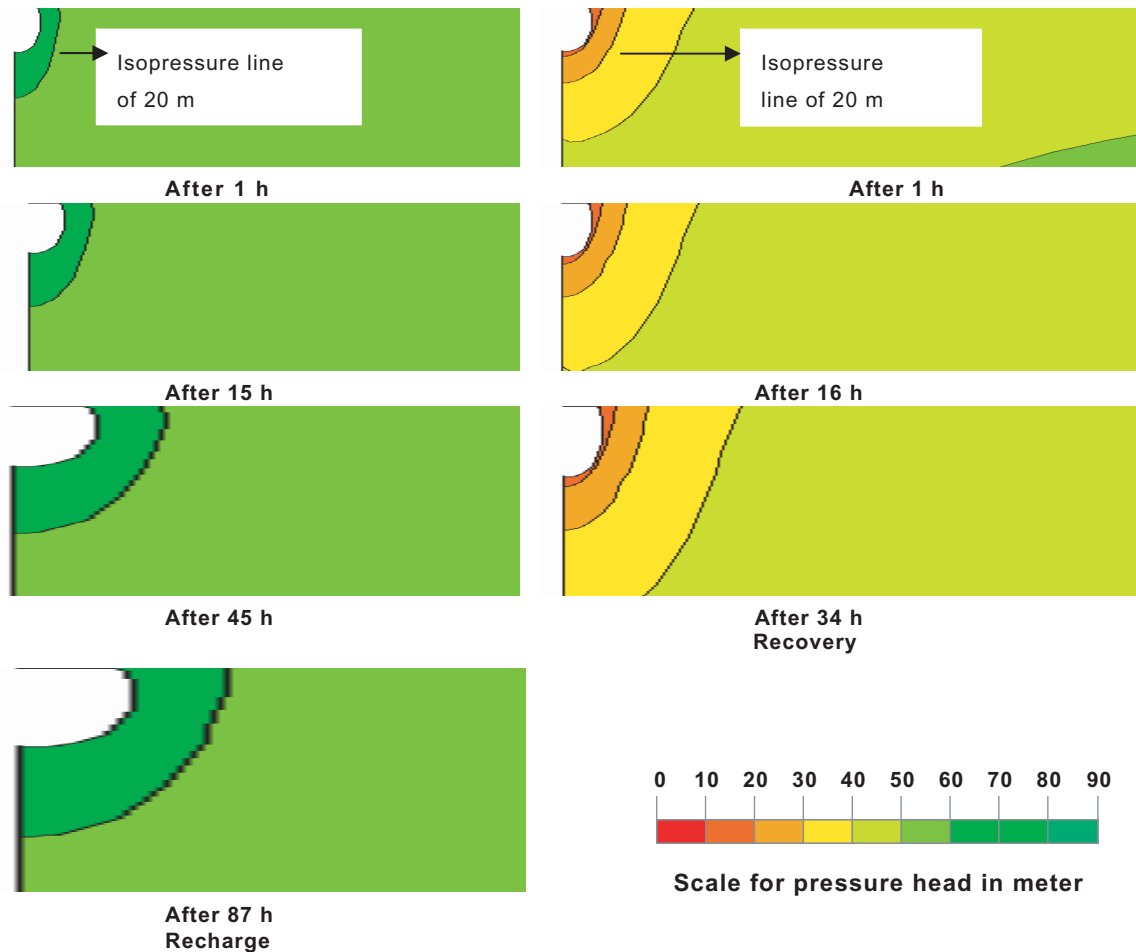
Figure 5. Development of pressure isolines during first validation ASR cycle as a function of time

CONCLUSIONS

- Hydrus 2D simulated the drawup/drawdown of pressure head quite well except at the start of recharge and discharge in each cycle.
- There was little change in pressure head after a distance of 325 m distance in the aquifer.

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Groundwater mathematical modeling of Sujas Basin Zanjan Province, Iran

A. Taheri Tizro and Sh. Lahooni

Abstract

Groundwater Models are simple representation of actual physical processes and suitably constrained differential equation that describes groundwater behavior in a system of interest. This model enables us to test hypothesis to predict the relative effects to management strategies and to perform sensivity analysis. A three-dimensional, numerical groundwater model of the Sujas aquifer in the Zanjan province of NW Iran was developed to help estimate groundwater availability and water levels in response to pumping and potential future droughts. The model includes historical information on the aquifer and incorporates results of new studies on water levels, structure, hydraulic properties, and recharge rates.

Keywords

Groundwater, mathematical modeling, Sujas Basin, modflow, calibration.

INTRODUCTION

With the development of finite difference and finite element techniques and digital computers, solution of complex mathematical models can be achieved. The governing differential equation is approximated by difference from of equation that is solved at the nodes that comprise a two or three dimensional grid (Franklin et al., 2003). The reliability of prediction using a groundwater model depends on the how well the model approximates field situations (Anderson 1992). One of the most important points is to select groundwater modeling software which most possess selecting index like its capability, popularity and its friendly use. The MODFLOW has emerged as the standard code simulating software was established in 1984 and later 1988 (McDonald and Harbaugh, 1988). The major revisions were released with addition of precondition conjugate gradient procedure PCG (Hill,1992) and etc.

AREA OF STUDY

Attempt has been made to perform groundwater mathematical modeling of Sujas basin located in NW of Iran. The area of study lies in between latitudes 36°7' to 36°30' north and longitudes 48°16' to 48°50' east. The total area is of 1,298 km.sq, which experiences average temperature of 9.68 c and average annual rainfall is about 369 mm. The area lies in between the tectonic zone of Alborz and central Iran. Geologically the area characterized by thick sequences of metamorphic and sedimentary formation of Mesozoic and Cenozoic age. The highest elevation in the basin is of to 2,812 m (Qaidar mountain) located in southern part and lowest elevation is 1,650 m .

GOAL OF MODELING

The aim of modeling is predicting the effect of pumping or recharge on groundwater levels. To understand groundwater hydrodynamic properties of aquifers to find better solution with regards to groundwater management strategies of the basin. Fig. 1 shows steps followed in groundwater modeling of Sujas basin.
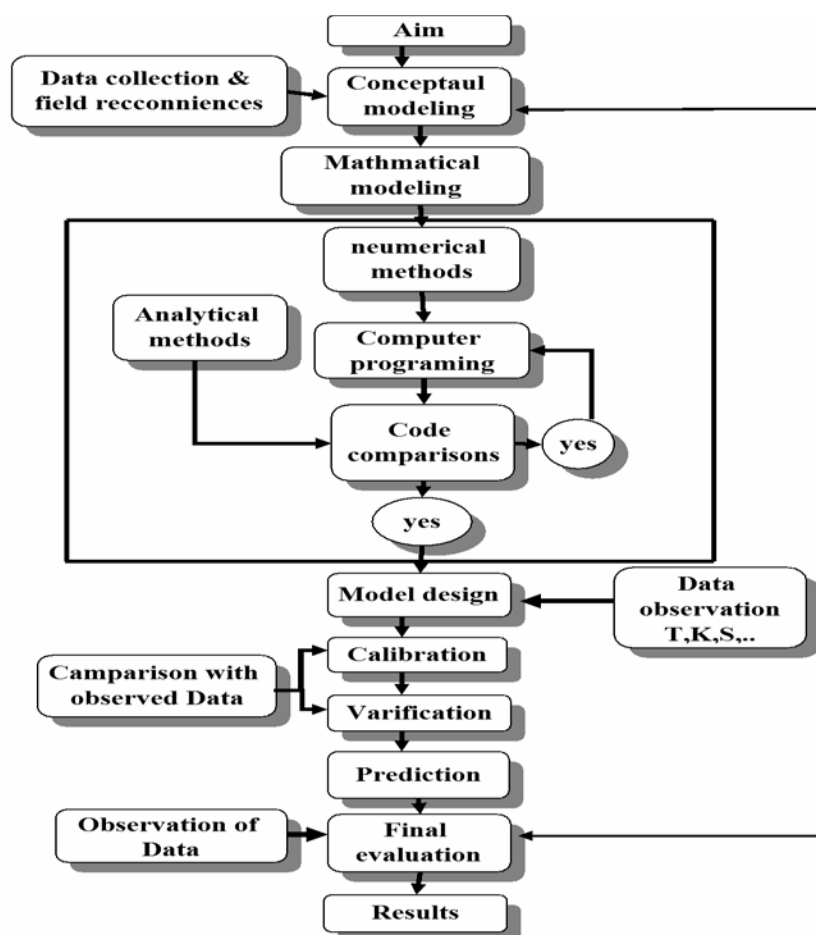
Figure 1. Steps followed in groundwater modeling

CONCEPTUAL MODELING

The conceptual model is our best understanding of groundwater flow in the aquifer. Generally more valid data were made available to construct the model. The data were used to understand: (1) the extent of the aquifer, its location and its boundaries,(2) the flow of water into and out of the aquifer, (3) behavior o the water levels, (4) variation in the thickness and the depth of the aquifer and any confining strata,(5) the spatial variation of the coefficient of the transmissivity and storage, (6) pumping test data and discharge of wells and its drawdown, (7) water level fluctuations in the aquifer over a number of years, (8) the possible river base flows, (9) spring location and spring flows, (10) information related to basic hydrogeology of region such as areas of interconnections between surface and groundwater.

In order to show three dimensional picture and groundwater conditions a fence diagram (Fig. 2) from lithological data has been prepared for the area.

For choosing the size and external boundaries, it was decided to consider the major recharge and discharge areas.

The central parts of the basin is effected by a major fault, which strikes parallel to Sujas river so that boundary condition assigned to these nodes considered to be constant at primary stages of calibration (Dirichlet type of boundary condition), but later it was decided to consider the influence of river and fault in the parameters. Fig. 3 illustrates surface two dimensional conceptual model and conditions of input and output areas.
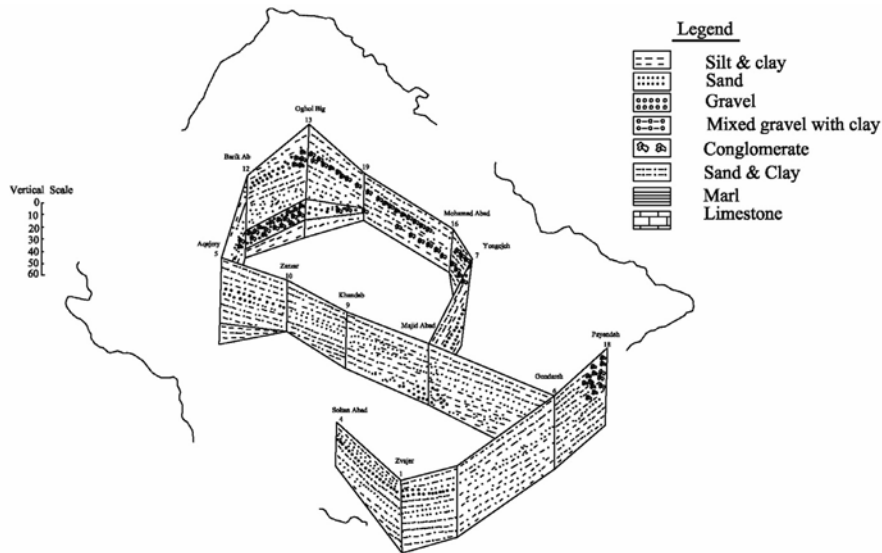
Figure 2. Fence diagram of study area
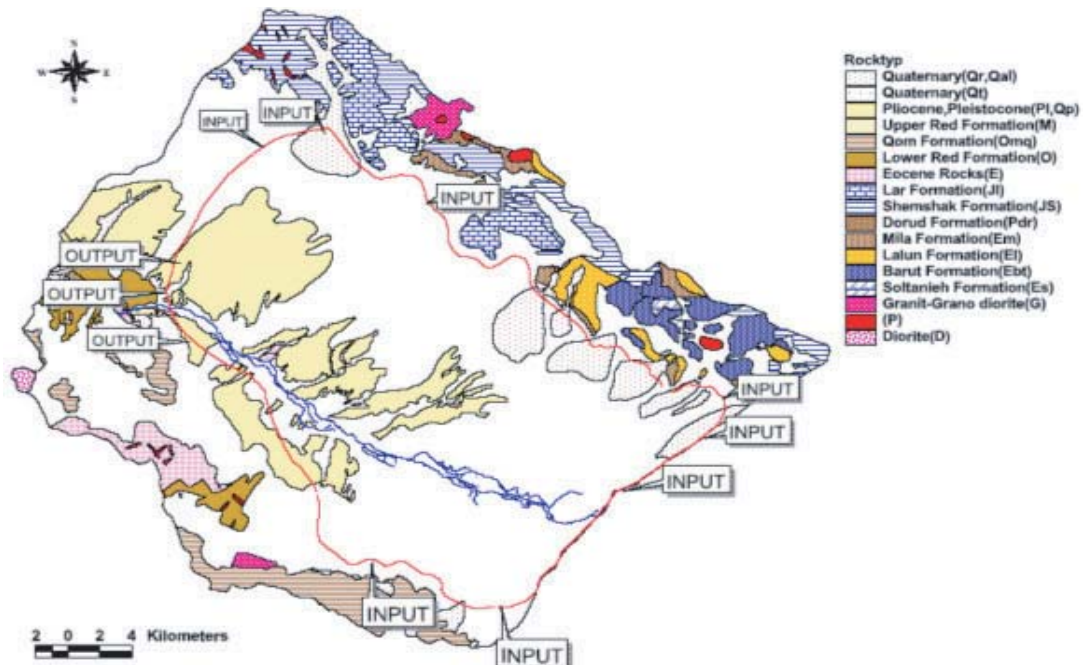
Figure 3. Two dimensional conceptual model of the area

MODEL DESIGN

The design of the model includes the choice of code and processor, the discrimination of the aquifer into layers and cells, and the assignment of model parameters. The model was designed to match well as much as possible with the conceptual model of groundwater flow in the aquifer.

Code and processor

The MODFLOW-96 (Harbaugh and McDonald, 1996), a widely-used modular finite-difference groundwater flow code written by the USGS was used. To obtain loading information into the model and observing model results, Processing MODFLOW for Windows (PMWIN) version 5.1 was used (Chiang and Kinzelbach, 1998).

Layers and grid

The lateral extent of the model corresponds to natural hydrologic boundaries, such as erosional limits, rivers, and the structural, and hydraulic boundaries to the west that coincide with groundwater divides. According to the hydrostratigraphy and conceptual model, the model was designed to have one layer. IBOUND defined by establishing the lateral extent of the formations in each layer using the geologic map. A cell assigned l as active if the formation covered more than 50 percent of the cell area. The model domain was discredited into grid dimensions of 1 x 1 km and sq. cells with smaller cell assigned to the active model area where rivers, springs and abstractions are simulated. In total, the model contains 38 columns and 34 rows (Fig. 4).
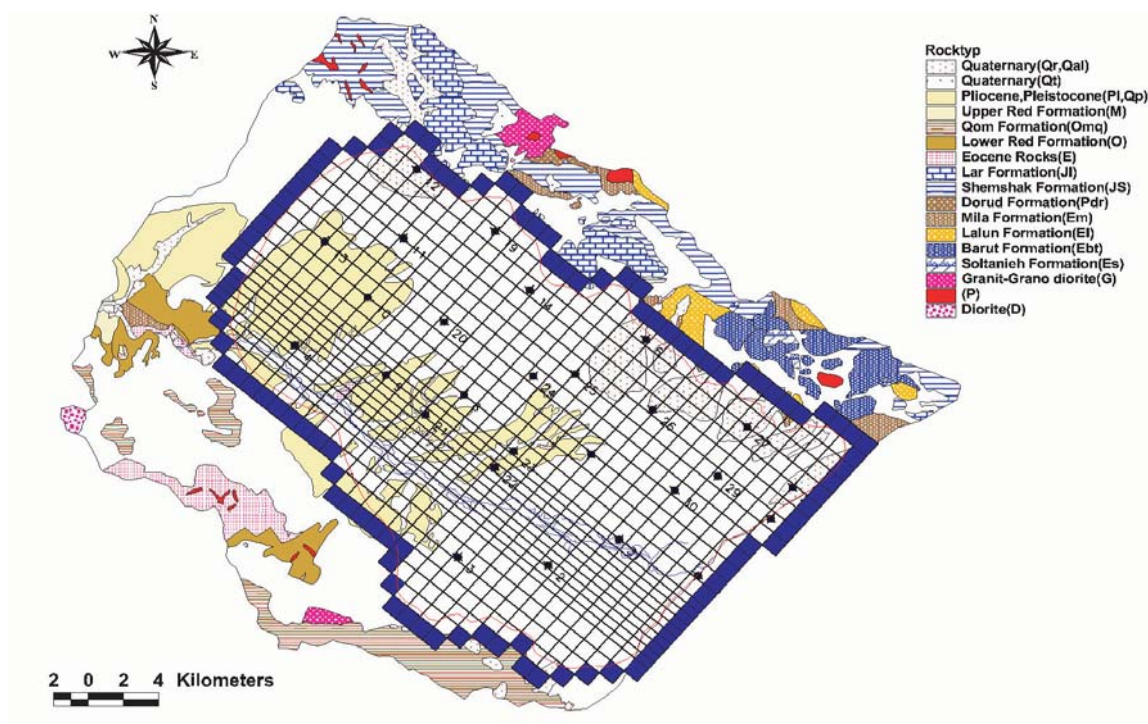
Figure 4. Nodal networks of the model and distribution of observatory wells

Model parameters

The distributed model parameters, including (1) elevations of the top and bottom of each layer, (2) horizontal and vertical hydraulic conductivity, (3) specific storage, and (4) specific yield using both Surfer and Arcview.

Model boundaries

The model assigned boundaries for the parameters including, (1) recharge, (2) pumping, (3) rivers, (4) springs, (5) outer boundaries, and (7) initial conditions. The initial values assigned of recharge according to Arcview analysis described in the recharge section. The interpretation of water levels at the October of 2000 as initial heads for the steady-state mode considered. The River Package of MODFLOW was used to represent rivers and streams in the model.

The constant-head and General Head Boundary (GHB) Package of MODFLOW used for nominating outer boundaries The GHB Package requires values for hydraulic-head and conductance. The hydraulic head according to the interpreted water in the area of the GHB cells and the GHB conductance according to the hydraulic conductivity and geometry of the cell were assigned.

The interpreted water-level maps as initial heads for the steady-state model was used.

Calibration of model in steady and unsteady conditions

The effectiveness of groundwater model will depend on how accurately it has been calibrated. The gap in data information like aquifer parameters should be determined first. The modeling strategy used in this was to calibrate the model in steady and unsteady conditions. Calibration

The calibration was attempted in steady-state model to measured water levels in the Sujas aquifer for October, 2002 when pumping expected to be lowest. To calibrate the model adjustment of the different model parameters was made to determine which parameters had the most effect on simulated water levels. Through this initial sensitivity analysis, it was observed that the water levels in the Sujas aquifer were most sensitive to the recharge rate, the horizontal hydraulic conductivity and the bottom layers. The final, calibrated model has good match of reproducing the spatial distribution of water levels in the Sujas aquifer for the October 2002 The model reproduces the interpreted direction of groundwater flow and approximates water levels in most parts of the study area.

 Period of calibration was considered as one year in unsteady state with 12 time steps. In the process of calibration a good match of selecting model parameters, between the predicted and measured static water level is obtained. The calibration was accomplished by a trial and error adjustment of model parameters. In case of any disorder in calibration was observed, the system of conceptualization was repeated by additional data collection studies and appropriate parameters values after many iteration processes. Judicial adjustment of parameters in the calibration process was done; therefore primary condition was considered with regard to water levels in each node. Fig. 4 shows the distribution observatory wells in the area.

Once the model was calibrated in steady state condition for the October, 2002 the calibration of the model for transient condition for 2002 and 2003 was done. Using monthly stress periods, the water-level fluctuations according to recharge and pumping variations in 2002 and 2003 was simulated. To calibrate the model, the adjusted specific-storage values until the model approximately reproduced the range of water-level fluctuations observed in wells in the model area. The model does a good matching of observed water-level fluctuations in some areas and no proper matching of water-level fluctuations in other areas. Differences may be due to the influence of local-scale conditions not represented in the regional model or errors in our parameterization of the aquifer data. Although there are limitations, the model does a good job of reproducing seasonal and year-to-year water-level variations in most wells and accurately representing areas where water levels respond quickly to variations in recharge and areas where the water-level response is much more subdued. Fig. 5 shows a compare between observed and simulated condition of model in the steady state condition.
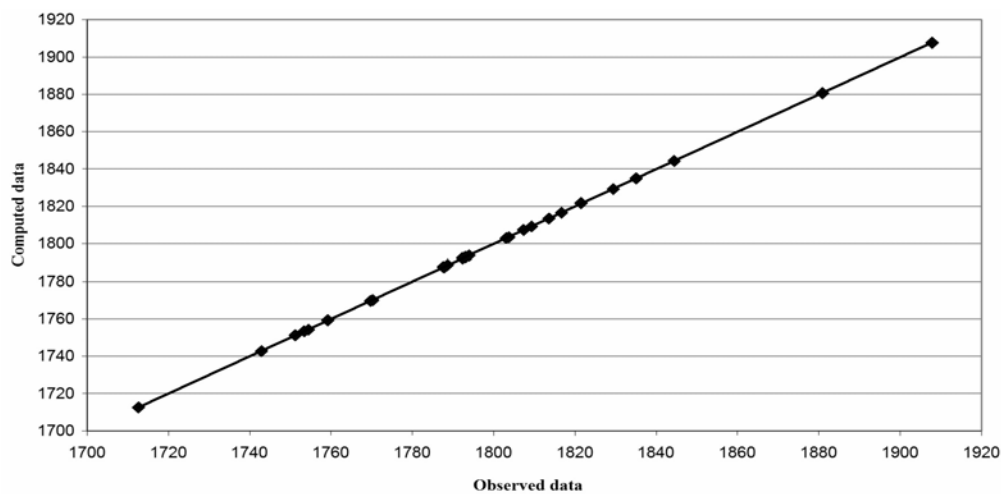
Figure 5. Correlation between computed values of water levels and observed in steady state

Verification for model in steady in and unsteady conditions

Model verification involves using the calibrated model to simulate a hyrologic system that is know. In this stage care must be taken to better underastand the parameters to represent valid hydrogeologic system. The basic of verification in sujas basin is to examine the computed static water levels with the observed field data.

The errors between the observed and simulated hydraulic heads were quantified in steady state condition and judicial adjustment of parameters in values of hydraulic conductivity, depth of bed rock and boundary condition were made.

The correlation between computed values of static water levels and observed values were simulated. In unsteady condition the parameter like storage coefficient were computed for each cell and separate maps were prepared after adjustment of its values. The simulated values of static water levels in model were compared with observed field condition for different periodical in the area. A selected Fig .6 illustrates the correlation of observed static water levels for well No. 2 and simulated in unsteady condition.
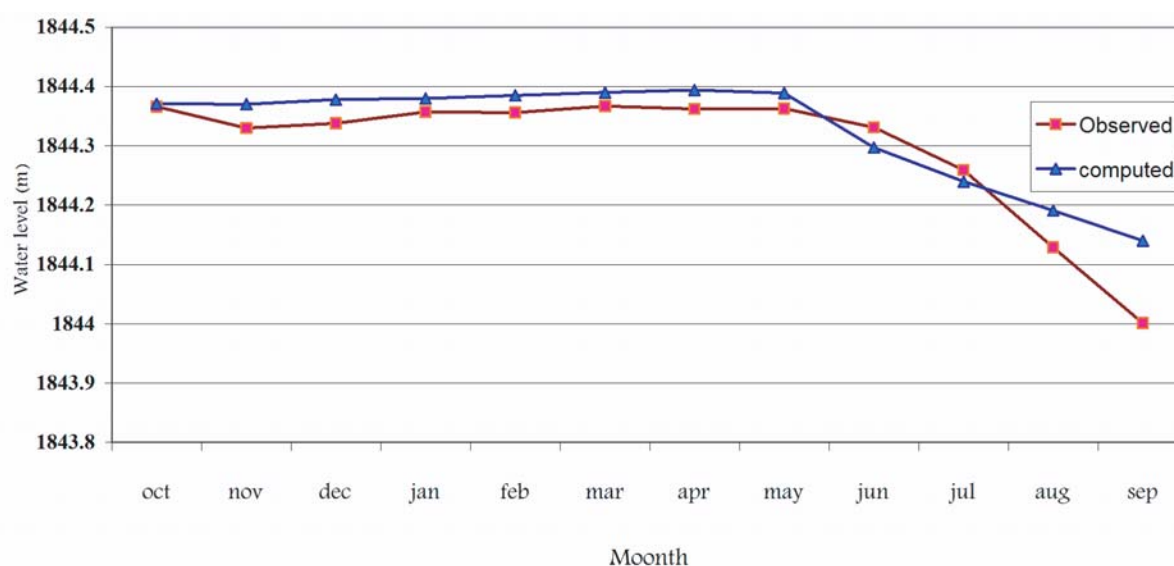
Figure 6. Correlation between computed and observed values in transient (well No. 2)

Predictions

To assess the future availability of groundwater in the Sujas aquifer, the calibrated model was used to predict future water levels for three scenarios which includes as, dry period, wet period and normal period. The time period of each scenario was considered as six months with one month time steps (30 days) was selected. The model was run to predict the future condition of the aquifer for the period of five years. For the future prediction of aquifer calibrated parameters in steady and unsteady condition were implemented. The primary condition of each scenario was considered to be similar and simulated condition was used in this stage.

RESULT AND CONCLUSIONS

As central parts of the basin is effected by a major fault, which strikes parallel to Sujas river the influence of river and fault in the hydrologic balance were clearly observed.

Through initial sensitivity analysis, it was observed that the water levels in the Sujas aquifer were most sensitive to the recharge rate, the horizontal hydraulic conductivity and the bottom layers.

In case of any disorder in calibration was observed, the system of conceptualization was repeated by additional data collection studies and appropriate parameters values after many iteration processes. Judicial adjustment of parameters in the calibration process was done therefore primary condition was considered with regard to water levels in each node. In unsteady condition the parameter like storage coefficient were computed for each cell and separate maps were prepared after adjustment of its values. The model does a good matching of observed water-level fluctuations in some areas and no proper matching of water-level fluctuations in other areas. Differences may be due to the influence of local-scale conditions not represented in the regional model or errors in parameterization of the aquifer data.

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Transient flow modelling of an overexploitedaquifer and simulation of artificial recharge measures

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Umberto Puppini, Maria Elena Scardazzi and Mike Streetly

Abstract

The Prato aquifer, mostly formed by alluvial fan deposits of the river Bisenzio, is the major source of water supply of the whole urban area of the Medio Valdarno, both for drinkable and industrial purposes. A significant depletion of groundwater resources due to strong overexploitation has been recorded over the past 40 years.

A numerical groundwater flow model using the MODFLOW code was built to increase the understanding of the groundwater flow system and to provide decision support for water resources management.

Continuously monitored groundwater levels and surface water levels of the Bisenzio River for a number of stations in the area of the alluvial fan as well as estimated and measured data for abstractions from municipal and industrial wells were available since the late 1950s.

The model was developed as a transient model covering the period from 1960 to 2001. The transient model was calibrated using groundwater elevation data in different years within the modelling period and was able to simulate the extensive groundwater depletion in the 1970s and the partial recovery in recent years.

To evaluate the effects of artificial recharge measures on the depleted groundwater levels, different preliminary predictive models were built investigating two different scenarios such as an increased hydraulic gradient between river and groundwater and of recharge wells placed in different locations of the Prato aquifer. The results of these simulations demonstrated a strong dependence of artificial recharge efficiency on the locations of the wells.

Keywords

Transient groundwater modelling; artificial recharge; predictive model.

INTRODUCTION

The Prato aquifer represents the major source of water supply in the Medio Valdarno region. It supplies about 65% of the water distributed by the public water supplier and in addition constitutes the main water resource for the local textile industry. The groundwater levels of the Prato aquifer show an almost continuous decrease since the late 1950s to date. The main reason for this development has been a growing demand for good quality groundwater, in particular by the industry (Consiag, 2001) but also for domestic purposes (Cozzi, 1999). The groundwater flow model discussed in this paper was constructed upon request of the public water supplier in order to augment the understanding of the main components and processes of the hydrogeological system and their interrelations. The final objective was to estimate the response of the groundwater system to variations in the abstraction rates of public wells and to the possible development of a system for artificial recharge of the aquifer.

CONCEPTUAL MODEL

The Prato aquifer is unconfined and is formed by deposits of the alluvial fan of the river Bisenzio, consisting mainly of sands and gravels of high hydraulic conductivities with intermittent layers of clay. It has a maximum

thickness of 55 m. The form of the alluvial fan has been defined using existing borehole log information. The main aquifer is underlain by a second aquifer characterised by prevailing layers of clay and loam and significantly lower hydraulic conductivities. The underlying aquifer could serve both as a source of water in times of low groundwater levels or as a drain of the major aquifer. However, measurements of the hydraulic conductivity revealed that the storage coefficient of the lower aquifer was several orders of magnitude lower than in the upper aquifer. Therefore it was concluded that an in- or outflow from this formation would be insignificant to the long-term water balance of the overall water resource. North of the alluvial fan lie the formations of Monte Morello and Monte della Calvana that are part of the mountain range of the Alberese. The geology of this formation is characterised by Eocene flysch with an alternating structure of limestones and marls and intermittent clay layers. The River Ombrone southwest of the city of Prato and the River Bisenzio both flow into the River Arno to the south. (Figure 1).
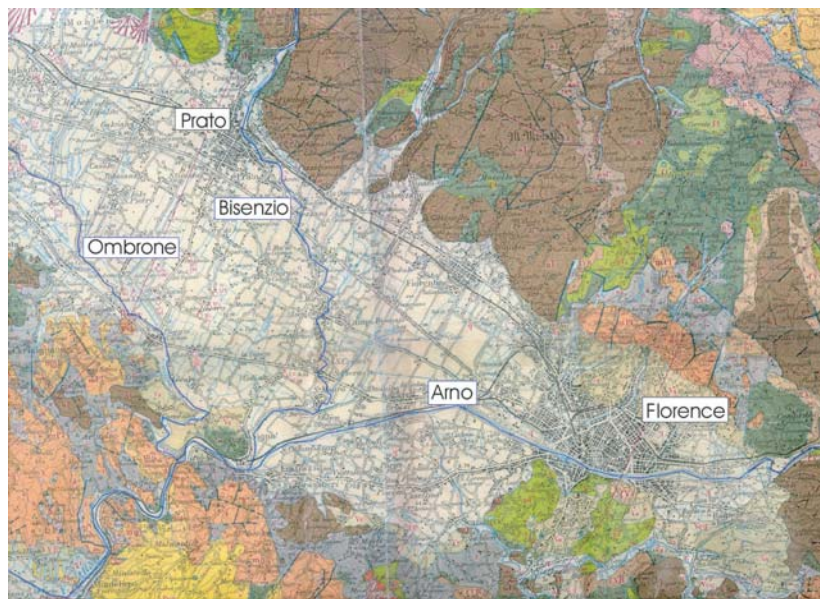
Figure 1. Geographic and geological setting of the model (SGI, 1967, modified)

Water balance

Inflows to the Prato aquifer occur through groundwater inflow from the mountain range in the north, effective precipitation, and infiltration from the riverbeds of the Rivers Bisenzio and Ombrone. Inflow from the northern mountain range occurs mainly as runoff and interflow from the mountain slopes. In the central part of the mountain range, north east of the city of Prato, the aquifer receives also a subsurface inflow of a part of the precipitation in the mountains due to underlying permeable formations. In the urbanised area of Prato leakage of the sewerage system constitutes an additional source of aquifer recharge. Outflows from the aquifer occur south of Prato as drainage to the rivers Bisenzio and Ombrone but mainly due to numerous wells that serve as the public and private water supply for industry, agriculture, and drinking water purposes.

NUMERICAL MODEL

The numerical model was built using the MODFLOW code (McDonald and Harbaugh, 1988). The model presented here was constructed as a transient model covering the period between 1960 and 2001 divided into 504 monthly stress periods. The extension of the model is from the Apennine mountain range in the north and northeast to the rivers Arno in the southeast and Ombrone in the west. The 15,840 cells of the finite difference grid have a size of 200 by 200 metres. The model consists of one layer and has a uniform thickness. The aquifer base was set to

0 m asl according to available borehole information. The top of the aquifer was fixed at 300 m asl to allow for a free oscillation of water levels.

Boundary conditions

The inflows from the northern mountain range are represented by a constant flow boundary. The volumes of flow from this boundary into the model where calculated using available precipitation data from the rain gauge of the town Vaiano. This boundary was subdivided into three reaches according to the geological settings. The central reach is characterised by a permeable calcareous subsurface while the external reaches are underlain by impermeable flysch material. Based on the study undertaken by Cicali and Pranzini (1987) it was calculated that 8.1% of the total precipitation arrive at the plain as runoff or interflow from the external reaches and eventually infiltrate into the aquifer. A higher percentage of 12.5% was calculated for the middle reach due to a subsurface inflow from the calcareous formation. The south-eastern and south-western boundaries are represented as no flow boundaries with river boundaries on top representing the Rivers Arno and Ombrone respectively. Also the River Bisenzio, passing the alluvial fan to the east was inserted as a river boundary condition. The elevations of the riverbeds, riverbed thicknesses and stages were determined on the basis of available measurements and were adjusted during the calibration process. Hydraulic conductivities of between 2×10^{-6} and 3.8×10^{-6} m/s were applied for the riverbeds.

Initial conditions

Using an available map of groundwater levels from 1960, a steady state model was calibrated and used to determine the initial heads of the simulation.

Model properties

Aquifer geometry, hydraulic conductivity and storage. According to previous studies, the hydraulic conductivities did not differ significantly between the area of the alluvial fan and adjacent areas (Adrenelli and Baldini, 1996). However, the thickness of the aquifer increased significantly towards the centre of the alluvial fan. To represent this increase in a single layer model of constant thickness, four zones of hydraulic conductivity have been defined reconstructing the 3-dimensional form of the alluvial fan (Figure 2). These zones do not represent existing zones of varying hydraulic conductivities but have been introduced to simulate the increase of transmissivity of the aquifer. During transient modelling the aquifer base had to be lowered to –10 m asl to avoid drying up of the aquifer. The problem of dry falling was also the reason why a constant aquifer thickness was chosen. The specific yields of the calibrated model were 0.11 for the area of the alluvial fan and 0.1 for the rest of the modelled area. These values are smaller than might be expected for sands and gravels. This is probably due to the intermittent layers of clay.

Recharge. The recharge of the aquifer by precipitation was determined for two different types of land use: the urban area covered by the city of Prato including its industrial zones and the surrounding areas characterised by predominant agricultural use. The urban area was extended during the simulation period taking into account the growth of the city. The recharge to the aquifer through the sealed surfaces of the urbanised area was estimated to be about 20% of the effective precipitation. In the non-sealed areas, a soil store model was applied to calculate the recharge to the aquifer. The model simulates the requirement of the soil zone for moisture for evaporation and plant uptake. The version of the Penman store model used consists of an upper and a lower soil store. The depth of the upper store is the depth up to which roots are able to draw as much water as required. At greater depths of store, water is only available to plants at a reduced rate. A bypass mechanism allowing direct percolation to the unsaturated zone via e.g. macropores or root channels was also included. The model defines all water leaving the soil zone as percolation, whether as direct (bypass) percolation or as release from the soil zone

store. Potential evaporation rates were calculated applying the Thorntwaite formula to data from the rain gauge 'Prato City'.
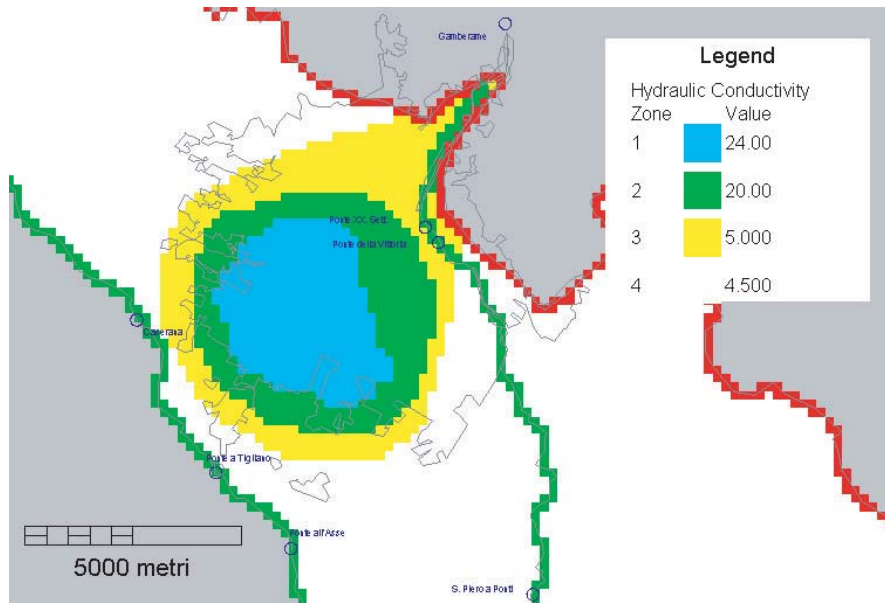
Figure 2. Zones of hydraulic conductivity representing the morphology of the alluvial fan of the River Bisenzio (values in metres per day). Grey line: 1994 extension of the city of Prato.

Well abstractions

Abstractions from public wells. 91 wells abstracting water for public water supply from the Prato aquifer were identified. Using available data on well activation and dismantling, the number and location of active wells was established for each stress period. For the period from 1960 to 1994, only the total annual abstracted volumes were available. The annual volumes were converted into monthly abstraction rates for each well dividing the total volume by the number of wells and applying the same abstraction rate for each month. Only since 1995 monthly abstraction rates had been documented for each well and could be inserted directly into the model.

Abstraction from private and industrial wells. 169 wells used for abstractions of private and industrial purpose were applied in the model. No information was available on the individual abstraction rates of the wells. It was also unknown which wells were used for industrial purposes and from which water was abstracted for private purposes (e.g. irrigation) only. Also the total volume abstracted from these wells was unknown for parts of the modelled period. From 1986 on, the industrial and private abstraction rates were reconstructed from available sewerage volumes. After 1986 the abstraction rates remained fairly constant at about 28 Mm^3/yr. After consultation of local experts of the public water supplier it was decided to apply an annual abstraction volume of 35 Mm^3 for the period between 1960 and 1973. Between 1974–1976 and 1990–1991 significant inflections of the Italian economic growth occurred (Bianchi, 2004). It was assumed that these inflections involved a reduction of industrial production and water consumption and therefore lower abstraction rates of 10 Mm^3/yr were applied. The extension of the industrialised area of the city of Prato at different times of the simulation period was defined by the public water supplier and was used to distinguish between industrial and private wells. During the simulation period, the industrial zone expanded and moved from the town centre to its southern margin. Abstractions from private wells have been estimated by Bendini (1963) as 3 Mm^3/yr and by Landini et al. (1990) as 2.5 Mm^3/yr since 1987. The abstraction volumes were divided by the respective number of industrial and private wells and an equal abstraction volume was applied to each month of the year.

CALIBRATION

The model has been calibrated following two criteria:
1. Comparison of measured and calculated groundwater levels in 1986, 1994, 2000 and 2001
2. The patterns of continuously measured groundwater elevations at the piezometers Le Badie, and Fossi.

Groundwater levels measured in 79 piezometers were used as targets for the calibration of the transient model. All targets were located in the area of the alluvial fan or in close vicinity. Table 6 gives an overview of the number of targets available for calibration on each of the four dates.

Table 6. Number of targets available on the four calibration dates

Year	Number of targets available for calibration
1986	8
1994	60
2000	12
2001	12

A calibration relying only on the comparison of measured and calculated groundwater elevations on the four dates was considered insufficient to assure a correct modelling of aquifer behaviour during the 42 years of simulation. For this reason the groundwater elevations of the two continuously measured piezometers Le Badie and Fossi were compared to the elevations calculated by the model to verify if the simulation was capable of reproducing the observed fluctuations of groundwater levels. Combining the two criteria it was not possible to calibrate the model working on statistic parameters alone as the reduction of calibration residuals often resulted in a decreased congruence between the observed and the calculated long term behaviour of groundwater elevations.

The calibration parameters defined by sensitivity analysis were:
*   hydraulic conductivity of zones 1 to 4,
*   specific yield of zones 1 and 2,
*   conductance of river beds,
*   abstractions from industrial and private wells.

The results of the calibration of the model using the targets of the years 1986, 1994, 2000 and 2001 are shown in Figure 3. While a general correlation of measured and calculated groundwater levels can be observed, the residuals are still relatively high especially for the last two years of calibration, 2000 and 2001. This is explained by the missing information on individual abstraction rates of the industrial and private wells. This might lead to high calibration residuals especially for targets that are close to industrial wells abstracting high quantities of water.
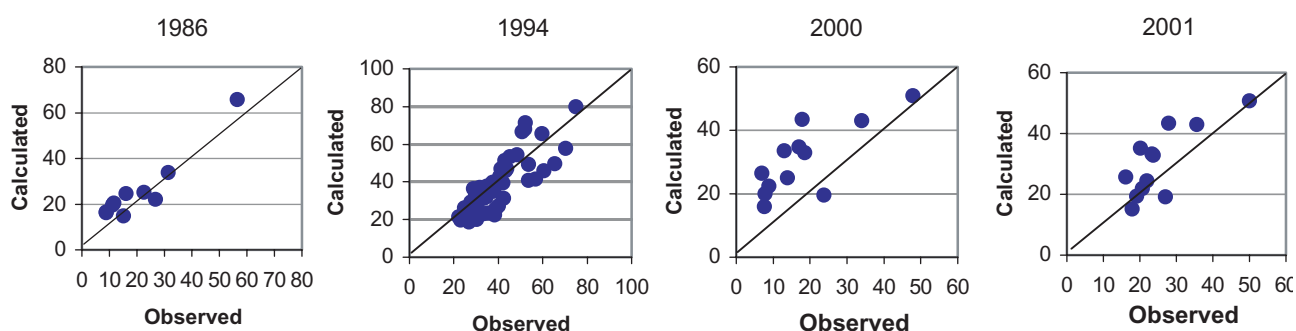
*Figure 3. Comparison of observed and calculated groundwater levels [m asl]
for the dates of calibration in 1986, 1994, 2000 and 2001*

The congruence between the measured and calculated fluctuations of groundwater levels is an additional criterion for the quality of the simulation. At the beginning of 1977 after a long period of continuous decrease of ground-

water levels, a strong and sudden rise of about 25 m occurred. A similar rise could be observed between 1991 and 1994. These rises could be partially explained by reduced abstraction rates during periods of economic inflection. Only a small proportion could be explained by an increased recharge of the aquifer due to periods of extreme precipitation and additional recharge from the northern boundary and no flooding had been recorded for these periods. Figure 4 compares the groundwater elevations measured at the piezometers Le Badie and Fossi with the values calculated by the model.
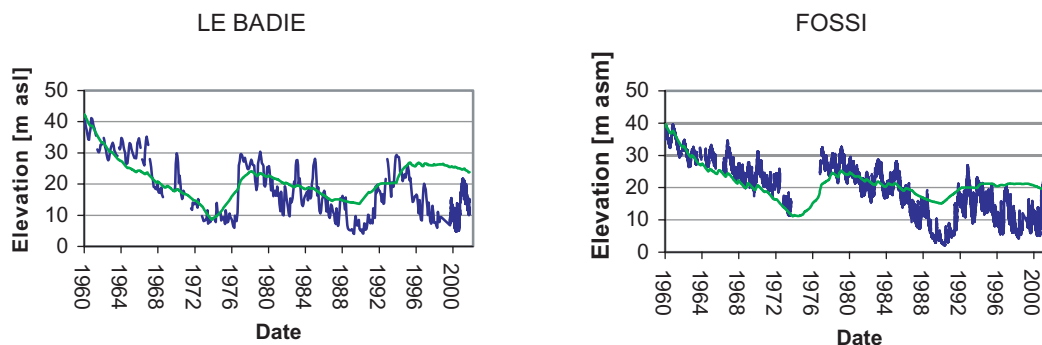
Figure 4. Comparison between measured (blue) and calculated (green) groundwater levels at the piezometres Le Badie and Fossi

While the first two major oscillations of groundwater levels in 1977 and 1991 are in general simulated by the model, the third decline reaching its minimum in 2000 is not calculated by the simulation. An explanation is that the location of the area defined as industrial zone in this period is located south of the city of Prato. In the area of the piezometers, only minor abstractions from private wells are simulated. This might be erroneous if single production sites remained in the town centre or water for industrial purposes is still abstracted from wells in this area. The model fails to simulate the short term fluctuation of groundwater levels. These fluctuations are most likely the effect of the operation patterns of the wells and could not be explained by correlation with precipitation hydrographs. Considering the low resolution of the available abstraction data and the need to use mean monthly data calculated from yearly abstraction rates together with the fact that the exact locations of well abstraction were not known it is not surprising that the model was not able to reproduce the short term fluctuations of groundwater levels.

PREDICTIVE MODELS

Due to the high abstractions from the Prato aquifer, periods of extreme drawdown and partial shortages of water have occurred, especially during periods of low precipitation. The last two crises were observed at the beginning and the end of the 1990s. Several scenarios have been developed by the public water supplier to augment the water quantity stored in the aquifer. To investigate the effects of these measures of artificial recharge, three predictive models have been developed that, starting in 2002, simulate the behaviour of groundwater levels during 12 years until 2013.

The first model was named 'Status Quo' and investigated the further development of the groundwater resource under constant conditions. The parameters used were those that represented the end of the simulation period of the transient model. Recharge rates and inflows from the northern boundary were applied as the mean values of the last three years of the transient model. In a second model, 'Ricart', the effect of an artificial recharge from a well field in the north of the alluvial fan was simulated. This scenario represents the intention of the public water supplier to divert water from the Bisenzio River and inject it into the aquifer. An injection of 400 l/s of water during 8 months of the year, from October to May, was simulated. The third model, 'Rivermod' investigated the effect of increasing the stage of the River Bisenzio by 1 m representing the constructing of a dam in the upper part of the river.

The results of the predictive models are presented in Figure 5. Already the 'Status Quo' model foresees a recovery of the groundwater level of between 1 and 4 metres during the 12 years of simulation, depending on the point of observation. The most significant rises of groundwater levels can be observed in the central part of the cone of drawdown. Close to the River Bisenzio only a minor recovery of less than 1 m was simulated. The influence of the river is evident in the scenario 'Rivermod' where the river stage has been fixed to 1 m above the currently measured water levels. Close to the river the groundwater levels start to rise almost immediately after the start of the simulation and show an increase of about 2 m after 12 years of simulation compared to the 'Status Quo' model. In the centre of the alluvial fan the effect starts to be visible about three years after the beginning of the simulation. The resulting increase of groundwater levels amounts to less than 1 m when compared to the 'Status Quo' model. The biggest effect is caused by the artificial recharge from a well field in the north of the alluvial fan. The effects of this measure start to be visible after 2.5 years and result in an increase of groundwater levels of about 3 m above the ones calculated for the 'Status Quo' model. A displacement of the well field to the centre of the alluvial fan and therefore to the centre of the observed cone of drawdown could further enhance the effect of the artificial recharge and would, according to the simulations, result in a rise of groundwater levels of between 6 and 8 m above the levels predicted by the model 'Status Quo'.
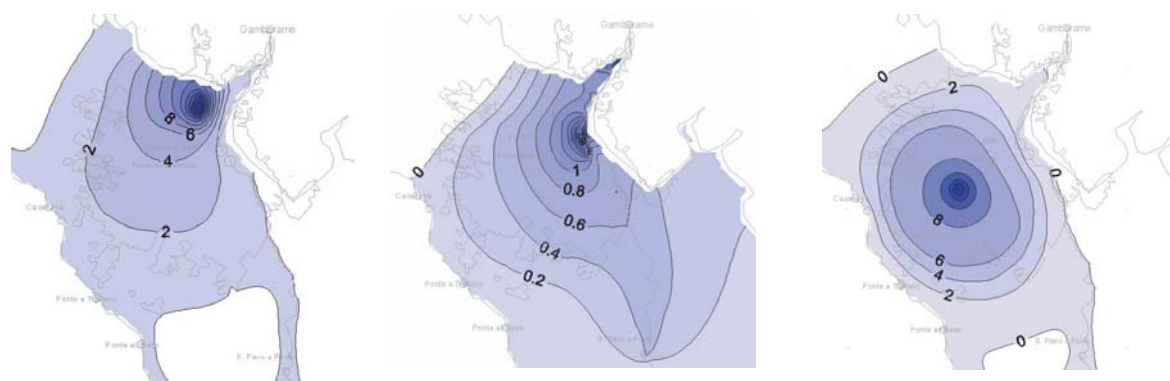
Figure 5. Resulting rise in groundwater levels of predictive models in metres after a simulation period of 12 years in comparison to the 'Status Quo' simulation. Left: Artificial recharge from injection wells; Middle: Effect of enhanced river stage; Right: Artificial recharge from the well field displaced in the centre of the alluvial fan.

The injected water fluxes of 400 l/s would have to be abstracted from the river Bisenzio. This is less than 10% of the average river flow of 5.1 m^3/s reported for the river Bisenzio at the station Gamberame upstream of the city of Prato. Nevertheless, the abstraction could potentially lower the water levels in the river and reduce the effects of infiltration from the river bed, an effect that has not been considered in the model. However, considering the effects calculated for an increase of water levels by 1 m, the reduction in infiltration is considered to be negligible.

CONCLUSIONS

The drawdown cone observed for the aquifer of the city of Prato, caused primarily by high abstractions from industrial wells, and the fluctuations of groundwater levels have been simulated by a transient groundwater flow model covering the period from 1960 to 2001. For the Prato aquifer of a maximum thickness of 55 m a change of groundwater levels of up to 35 m was observed in this period. With the presented model it was possible to simulate the primary patterns of the behaviour of groundwater levels. The major obstacle for a better quantitative calibration of the model was the fact, that the individual abstraction rates for industrial and private wells were not known. The model should therefore be used to estimate the behaviour of the aquifer as a whole and will not be suitable to undertake detailed studies on the effects of single abstractions. Despite of this shortcoming, the model is considered

a valuable tool for the management of the groundwater resource of the Prato aquifer. It is also considered representative for many real case studies as very often more detailed or continuous data are not available.

Three predictive models have been developed to investigate the effects of different measures of artificial recharge on groundwater levels. Continuation of the recharge and abstraction conditions that prevailed at the end of the transient simulation results in an increase in groundwater levels of about 3 m within the 12 years of simulation. The construction of a dam in the upper part of the river Bisenzio and the subsequent increase of the water column of 1 m only resulted in a recovery of the groundwater levels by 1 m above the simulation of the status quo. The biggest effect, consisting in a rise of groundwater levels of about 7 m at the centre of the cone of drawdown after 12 years of operation, has been simulated for artificial recharge from a well field north of the city of Prato. Displacing the well field to the centre of the Prato aquifer a rise of groundwater levels of about 11 m could be achieved.

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Modeling the size of microbiological protection zones around phreatic sandy aquifers in the Netherlands

Paul W. J. J. van der Wielen, M. Blokker and G.J. Medema

Abstract

The aim of the current study was to calculate the size of protection zones around (sub)oxic and anoxic sandy aquifers without confining layers using a virus infection and transport model. The maximum allowable virus infection risk was 10^{-4} per person per year at the 95%-confidence level. In addition, implications of model results for aquifer recharge were discussed. Model results demonstrated that phreatic (sub)oxic sandy aquifers in the Netherlands need protection areas with a residence time of 43 to 117 days to prevent that the maximum virus infection risk will be exceeded. This is 0.7 to 2 times the current guideline of 60 days. In contrast, phreatic anoxic sandy aquifers without confining layers need protection zones of 555 to 898 days to stay below the maximum virus infection risk. This is 9.5 to 15 times the current guideline. A sensitivity analysis of the model demonstrated that the calculated protection zone was most sensitive for virus inactivation rate and collision efficiency. Values of both parameters were predicted from values obtained in two field studies. At present, it is unknown if these values can be used at other locations as well. Therefore, model results should be interpreted with care.

Keywords

Aquifer recharge; groundwater well systems; modeling; protection zone; redox state; removal of viruses.

INTRODUCTION

A third of the drinking water in the Netherlands is produced from surface water. Surface water is commonly contaminated by fecal material from effluent of waste water treatment plants, manure run-off or by wildlife. An effective way to reduce pathogenic microorganisms from surface water is by soil passage. The removal of microorganisms from surface water during soil passage in the Netherlands has been studied in two recharge systems (Schijven et al., 1999; Schijven et al., 2000). One is dune recharge of surface water from infiltration ponds, the other being aquifer recharge by deep well injection of surface water. The two studied recharge systems are different in their redox state; dune recharge was performed in an oxic aquifer, whereas deep well recharge was performed in an anoxic aquifer. The inactivation rate of viruses is higher under oxic than under anoxic conditions (Gordon and Toze, 2003). If metal ions are available (sub)oxic conditions will also result in the formation of metal hydroxides, which provide attachment sites for microorganisms (Ryan et al., 1999; Schijven et al., 2000). Therefore, it has been suggested that removal of viruses is higher during soil passage in a (sub)oxic aquifer compared to an anoxic aquifer.

Recently, a model has been developed to calculate the protection zones around anoxic groundwater aquifers under a worst-case scenario of a leaking sewer and a 9 log reduction of viruses by soil passage (Schijven and Hassanizadeh, 2002). These authors concluded that under a worst-case scenario protection zones with a residence time of 400–800 days are needed to obtain sufficient removal of viruses. This is a significant increase compared to the currently used guideline for protection zones in the Netherlands, which is based on a residence time of 60 days. However, protection zones based on (sub)oxic groundwater aquifers were not calculated. In addition, the new

inspection guideline in the Netherlands states that the infection risk for viruses by drinking water should not exceed 1 per 10,000 persons per year (de Roda Husman et al., 2004).

The aim of the current study was to calculate protection zones around (sub)oxic and anoxic sandy groundwater aquifers without confining layers based on an infection risk of 10^{-4} persons per year (de Roda Husman et al., 2004). In addition, results were discussed in relation to artificial recharge systems as well.

METHODS

The protection zones around 3 anoxic and 5 (sub)oxic sandy groundwater aquifers in the Netherlands were calculated. Anoxic groundwater well systems had oxygen and nitrate concentrations below 0.5 mg l^{-1}, whereas wells with nitrate or oxygen concentrations above 0.5 mg l^{-1} were considered to be (sub)oxic (Schijven and Hassanizadeh, 2002). The overall equation to calculate the virus concentration at the abstraction well was given by:

$$C_A = \frac{q}{Q} C_0 e^{-\left(\frac{3}{5}\alpha k_1 R^{\frac{5}{3}} + \frac{1}{2}\mu_{in} k_2 R^2\right)} \tag{1}$$

Where C_A is the virus concentration at the abstraction well; q is the leaking rate of a sewer; Q is the abstraction rate of the groundwater well system; k_1 and k_2 are constants; α is the collision efficiency and μ_{in} is the virus inactivation rate.

The reader is referred to Schijven and Hassanizadeh (2000; 2002) for detailed information about factors affecting removal of microorganisms through soil passage and for information about equation 1 and about equations to calculate both constants k_1 and k_2.

The infection risk (p_{inf}) was modeled with a dose-response model for the infection of rotavirus (Teunis et al., 1996):

$$p_{inf} = 1 - \left(1 + \left(\frac{D}{0.422}\right)\right)^{-0.253} \tag{2}$$

with D being the doses:

$$D = \frac{C_A}{E} V \tag{3}$$

where V is the volume of non-boiled drinking water intake per person per year and E is the recovery of the virus method.

Combination of equation 1 and 2 describes the infection risk as a function of the size of the protection zone in meters: $P_{inf} = f(R)$. The size of the protection zone based on residence time was calculated using the following equation:

$$t = \frac{\pi n h R^2}{Q} \tag{4}$$

where t is the residence time; n is the porosity; h is the aquifer size and R is the size of the protection zone in meters.

All variables used in the modeling were either a constant value or were described by a normal or a lognormal distribution. A Monte Carlo analysis with 300,000 simulations from the distribution of the different model parameters was used to calculate the size of the protection zone. The maximum allowable virus infection risk used was 10^{-4} persons per year at the 50%- and 95% confidence level (de Roda Husman et al., 2004).

In addition, a sensitivity analysis was performed for the anoxic aquifer with the longest calculated protection zone. The sensitivity of the model was studied by changing each model parameter over a certain fixed range, while the distribution of the other model parameters was kept the same.

RESULTS AND DISCUSSION

The size of the protection zone was calculated for eight different abstraction wells by using the average abstraction rate over a year. The calculated size of the protection zone around anoxic aquifer 1 was the largest, and this distance was more than twice the distance around the other sandy groundwater aquifers (Table 1).

Table 1. The predicted size of the protection zone, expressed in distance and residence time, at a 50% (average) and 95%-confidence level that the infection risk of 10^{-4} persons per year will not be exceeded

Aquifer	Distance (meters)		Residence time (days)	
	Average	95% Percentile	Average	95% Percentile
Oxic 1	31	49	27	66
Oxic 2	42	64	51	117
Oxic 3	48	77	21	55
Oxic 4	55	90	16	43
Oxic 5	34	54	23	58
Anoxic 1	178	276	231	555
Anoxic 2	55	84	368	857
Anoxic 3	54	82	390	898

The protection zone expressed as residence time of viruses was much larger for anoxic aquifers than for (sub)oxic aquifers (Table 1). The maximum allowable virus infection risk at the 95%-confidence level was achieved when the protection zone around anoxic aquifers was based on a residence time of 555 to 898 days. This is 9.5 to 15 times the currently used guideline of 60 days around aquifers. The size of the protection zones was in the same range as the protection zones calculated for anoxic aquifers in a previous study (Schijven and Hassanizadeh, 2002). However, the methodology used to calculate the protection zone differed considerably between the two studies. In our study, the protection zone was calculated as a function of the infection risk, based on a dose-response model of rotavirus. In the study of Schijven and Hassanizadeh (2002) the protection zone was calculated based on a 9 log removal of viruses. Furthermore, model parameters used in our study were described by a normal or lognormal distribution from which parameter values were achieved using Monte Carlo simulations. In the study of Schijven and Hassanizadeh (2002) fixed average values were used for all model parameters. As a result, the average size of the protection zones obtained in our study should be compared with the values obtained in the study of Schijven and Hassanizadeh (2002). The average values for residence time in our study are approximately a factor two lower than in the other study (Schijven and Hassanizadeh, 2002). The use of different methodology to calculate the protection zones are responsible for the observed difference between the two studies.

In contrast to anoxic aquifers, the protection zone around (sub)oxic aquifers was between 43 and 117 days, 0.7 to 2 times the 60-days-guideline. In the Netherlands, only 3 phreatic anoxic sandy aquifers without confining layers are in use for abstraction of groundwater. The rest of the phreatic sandy aquifers in the Netherlands are either (sub)oxic or contain a significant confining layer. This implies that the currently used protection zone of 60 days

around most Dutch phreatic sandy aquifers is microbiologically safe. Only the protection zone around the three Dutch anoxic phreatic sandy aquifers without confining layer should be reconsidered and studied in more detail.

The difference in length of the protection zone between (sub)oxic and anoxic aquifers is caused by the use of different parameter values that describe inactivation and attachment of viruses. The inactivation rate of viruses is lower under anoxic compared to oxic conditions (Gordon and Toze, 2003). In the model an inactivation rate with a mean of 0.149 ln day^{-1} was used for wells with (sub)oxic groundwater and a mean of 0.024 ln day^{-1} was used for wells with anoxic groundwater. The collision efficiency, a parameter that describes attachment of viruses, is lower under anoxic conditions as well (Schijven and Hassanizadeh, 2000; Schijven et al., 2000). A 2.6 times lower collision efficiency was used in the model for wells with anoxic groundwater compared to wells with (sub)oxic groundwater.

A sensitivity analysis of the model was performed for each parameter value and the change in the length of the protection zone (in meters) for the three most sensitive model parameters are visualized in Figure 1. The length of the protection zone was most sensitive for the model parameters inactivation rate, grain size and collision efficiency. The inactivation rate and collision efficiency were not specifically measured for each groundwater well system. Instead, they were estimated from field studies at other abstraction sites: one oxic aquifer (Schijven et al., 1999) and one deep anoxic aquifer (Schijven et al., 2000). The reliability of using these values at other locations is unknown. In addition, most sandy phreatic groundwater well systems in the Netherlands are suboxic, with a redox status between oxic and deep anoxic. At present, the values for inactivation rate and collision efficiency under suboxic conditions are unknown.
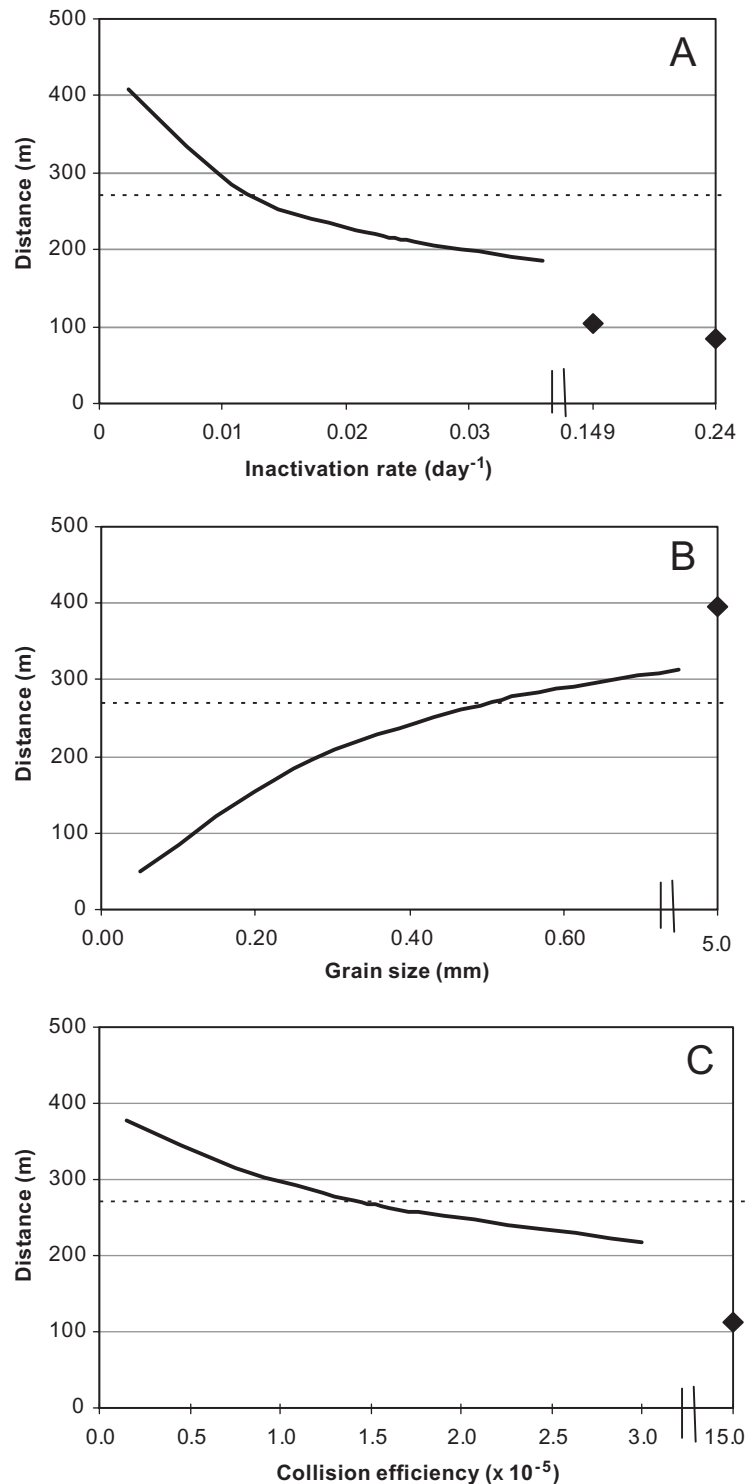
Figure 1. Distance of the protection zone
around aquifer' anoxic 1' at different values
of inactivation rate (A);
grain size (B)
and collision efficiency (C).

As a result, the predictive value of the model results is unknown. Therefore, it is recommended that field studies are performed to validate (and calibrate) the model. Until the reliability of the model is known, model results should be interpreted with care.

The parameter values used in the model were based upon field experiments under oxic (Schijven et al., 1999) and anoxic (Schijven et al., 2000) conditions and is to our knowledge the first study where the removal of viruses were modeled under both redox conditions. The observation that the modeled removal of viruses under anoxic conditions is lower than under (sub)oxic conditions has implications for aquifer recharge systems as well. In the Netherlands, artificial aquifer recharge of surface water is performed by dune recharge and deep well injection. The first is characterized by oxic conditions (Schijven et al., 1999), whereas deep well injection is characterized by anoxic conditions (Schijven et al., 2000; Saaltink et al., 2003). Consequently, the removal of microorganisms in the latter case will be much lower and care should be taken when determining the distance between the infiltration and abstraction site.

A field study at a deep well injection site in the Netherlands demonstrated that the removal of viruses in the first few meters was large, because the infiltrated oxic surface water changed the redox status in the first 10 meters of infiltration (Schijven et al., 2000). However, the change in redox depends on the rate of surface water infiltration, quality of the surface water and composition of the soil (Saaltink et al., 2003). Surface water with a high organic content will turn anoxic fast, whereas surface water with a low organic content will remain oxic for a longer period during soil passage. Similarly, in soils with a high content of pyrite or other reducing compounds oxygen from the infiltrated water will be reduced fast, whereas soils with little pyrite will allow oxygen to travel deeper into the aquifer.

Finally, it should be noted that the amount of viruses introduced by surface water will be much lower than by waste water. On the opposite, rate of infiltration is much higher than infiltration from a leaking sewer. As a consequence, viruses in leaking sewer water will be diluted to a great extend by surrounding groundwater, whereas infiltration of surface water will not result in high dilution rates. It is presently unknown how this dilution effect affects the length of the protection zone. However, the minimal distance between the infiltration and abstraction well can be calculated by the same model given that the concentration of viruses in the surface water is known.

CONCLUSIONS

- Model results demonstrated that the predicted length of the protection zone around phreatic shallow (sub)oxic sandy aquifers is in the same order of the currently used guideline of 60 days.

- In contrast, the predicted length of the protection zone around phreatic shallow anoxic sandy aquifers is 9.5 to 15 times the guideline of 60 days.

- In the Netherlands, only three phreatic shallow aquifers are anoxic and the length of their protection zone should be reconsidered.

- The removal of microorganisms in anoxic deep well recharge systems is low and care should be taken by determining the distance between infiltration and abstraction.

- The model used in this study to calculate the protection zone around groundwater well systems can be used to calculate the minimum distance between infiltration and abstraction in an aquifer recharge system as well.

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Hydraulic and transport modelling of bank filtration at Lake Tegel (Berlin)

B. Wiese and G. Nuetzmann

Abstract

A hydraulic and physically based transport model for the catchment of a well field was set up. With the study area situated in a region strongly influenced by surrounding well galleries the boundary conditions had to be worked out during modelling and partially had to be transient. Two important processes were clarified: Bank filtrate extracted at the investigated transect is composed of 3 water qualities from horizontal layers, each with a different age and infiltration area. Sampled wells containing the different water types were identified, providing information for correct chemical interpretation. Secondly, the lake sediments show a pronounced seasonal fluctuation in their leakage coefficient, with its winter values doubling in summer, and lagging 2–4 months behind water temperature.

Keywords

Bank filtration; layered flow; MODFLOW; seasonal leakance; temperature.

INTRODUCTION

The purpose of this study was to develop a model capable of clarifying the hydraulic properties at the transect of the bank filtration field site and yielding a physically based numerical reproduction of flow and transport, in order to properly interpret physico-chemical parameters obtained during the NASRI project. The origin of the groundwater had to be determined, in particular the contributions of old and young bank filtrate and autochthonous groundwater from the hinterland. Additional important parameters to be quantified were the travel time of infiltrated surface water to the abstraction and sampling wells and parameters controlling the hydraulics of bank filtration. As previous models set up by Eichhorn (2000), WASY (2003) and Wiese et al. (2004) did not fulfil these requirements, the modelling presented here was carried out.

MODEL STRUCTURE

The modelling was carried out using Modflow under PMWin (Chiang et al. 1994) advective transport by PMpath (Chiang 1998).

Geology

Lake Tegel and the aquifers in the region were formed during the quarternary Saale ice-age. Lake Tegel has maximum depth of 17 m, the lake bottom of the deeper regions is formed by impermeable thick sediments and mud, towards the shore they gradually diminish and colmated sand forms the lake bottom. The two upper aquifers (Fig. 2) are important for West well field, sealed to the bottom by mighty Pleistocene mud and silt layers (Parchur et al. 1977). The main aquifer, where the wells have their screens, is about 30 m thick and is covered by 4 m of glacial till. The upper aquifer is about 10 m thick, half of which is saturated on average. The upper aquifer is about 10 m thick, half of which is saturated on average. The aquifers consist of glaciofluviatile fine to coarse sands

resulting in a range of k values between 10^{-4} to 5 x 10^{-4} m/s (about 10 to 40 m/d, respectively) (Fugro 2003). The airport lake situated about 1,200 m east of the transect (Fig. 1) has a depth of 40 m and thus penetrates both upper aquifers. For further information please refer to HSM Tegel (Hydrogeologic Structural Model), Fugro 2003.

Hydraulic situation

The aquifers around Lake Tegel are highly affected by the Waterworks Tegel. Eight Galleries extract groundwater which is recharged by Lake Tegel, precipitation and artificial recharge. Puming and recharge are transient. The identification of correct boundary conditions, while avoiding a too large model area, was a process requiring the development of several models.

No flow boundaries

North of the model area another well field is pumping 60% of West well field with the ratio is roughly maintained for monthly sums, so that a no flow boundary is testified. No flow is also assigned in the north below Lake Tegel, as the aquifer is very thin there, due to a deep position of the lake sediments on top.
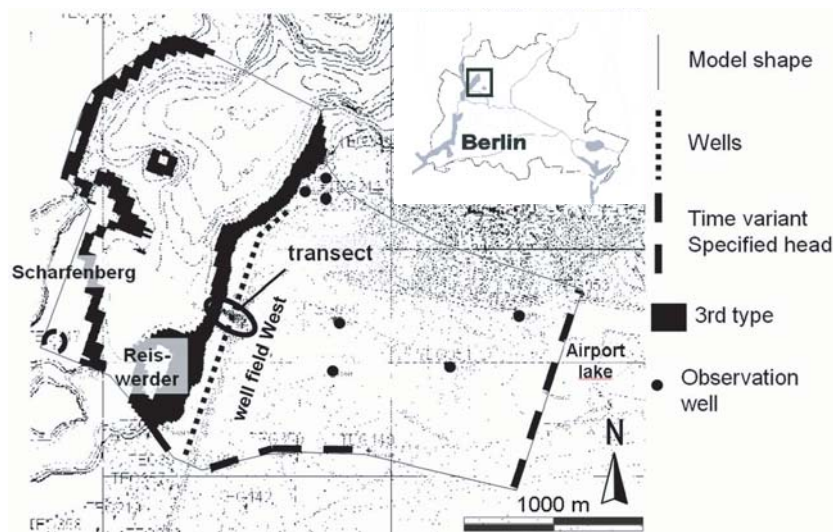
*Figure 1. Top view of the boundary conditions of the model area, and its location in Berlin.
Wells of the West well field are marked by a tightly dotted line, Lake Tegel introduced as 3rd type boundary
is marked as black area the other part of the lake is marked by bathymetric isolines with 1 m distance.
Time variant specified head boundaries are marked with a dashed line. Please note that boundaries
are not assigned to every layer. All borders with no value assigned are 'no flow' boundaries.*

Specified head boundaries (1st type)

Starting from the western boundary at the location of the Airport Lake and continuing clockwise the choice of boundary conditions is based of the following situation:

Eastern boundary: Piezometers in the model area screened in different aquifers show that at the airport lake the upper and lower aquifer have the same head, inducing horizontal groundwater flow. So to both aquifers the same boundary condition is assigned. As this region still is under influence of the waterworks, the boundary has to be transient.

Southern boundaries: The ratio of pumped volumes by West well field and adjacent well field Saatwinkel varies by factor 3, creating highly variable flow directions and water levels. This is respected with time variant specified head

boundaries. Unfortunately it is not possible to move it further away from the well field to have at least one control piezometer in between, because in the airport area (Fig. 1, bottom-right) data are lacking.

Scharfenberg: A piezometer exists at the horizontal well Scharfenberg. To avoid setting an time invariant groundwater divide the hydraulic head is introduced as a function of time.

Western boundary: This boundary is formed by well field North. It is set as a constant head of 29 m asl (above sea level). Due to Lake Tegel sediments reaching deeply into the aquifer and due to recharge above the boundary, at Lindwerder and the northern region of Scharfenberg modelling shows the head is a rather insensitive parameter. Changing it by one meter only induces a few cm change in water level at the transect.

Wells

Using daily well operation time and pumpage, pumping rates were calculated. Amounts were checked using the sums from the water meters and hours of well operation, both registered during the well capacity measurements which take place between 2 and 4 times a year. Wells are only screened in the second aquifer. Different layerwise pumping rates were assigned using the filter depth for wells 10 to 16 as vertical flow might affect the conditions at the transect; due to the distance to the transect the Dupuit assumption is used: for the other wells one average extraction rate is assigned for the lower aquifer.

Recharge

Groundwater recharge was fixed to one third of annual precipitation, of which three quarters take place during winter. However, as its contribution to the total water balance is only 10% its parameterisation is insensitive to the model outcome.

RESULTS

Two major impact types on the bank filtration site are identified: Three types of bank filtrate showing layered flow in the transect, secondly the leakage factor of infiltration from the lake to the groundwater is found to show a pronounced seasonal cycle.

One well – three types of bank filtrate

Bank filtrate extracted by well 13 has 3 different infiltration areas, each with different travel times (see Fig. 2). The different waters are extracted at different depths in the well which explains three observation phenomena:

1) Observed concentrations

Observation well 3301 shows a water composition different from the rest of the transect (NASRI 2005). At the beginning of the study period, namely in summer 2002, the latter can be seen from the concentrations of chloride and boron being significantly higher than concentrations in Lake Tegel. Assuming a travel time of a few months, these concentrations cannot be explained. However, assuming a travel time of 15 to 20 month, the concentrations match the lake concentrations (Fig. 3). Temporal variations of physico-chemical parameters in observation well TEG374, situated 2 m above the lower boundary of the aquifer, can be assigned to surface water infiltrated north of Reiswerder about 20 month ago, mixing with water showing concentrations of anthropogenic substances (Phenazone derivates) higher than surface water concentrations during the last 15 years, which can be assigned to water which infiltrated at the western shore of Scharfenberg. The travel time cannot be determined as the model period covers only 4 years, but extrapolation of a three month duration stationary flow field are with 15 years in the same order of magnitude.
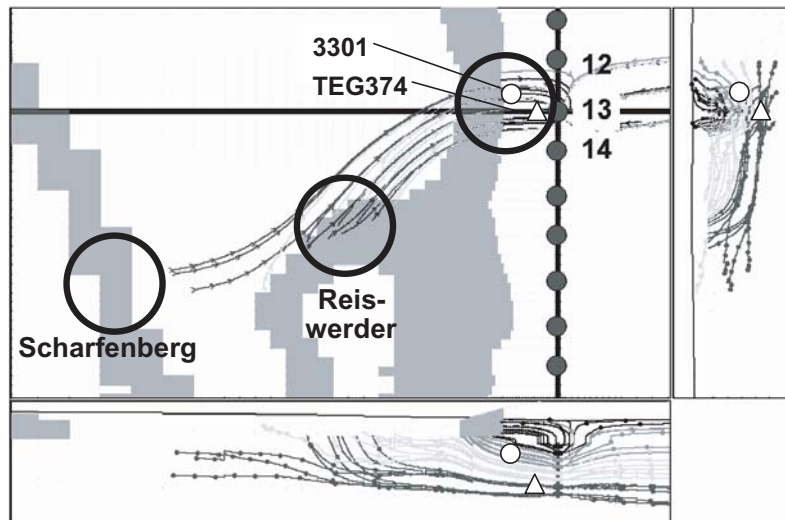
Figure 2. Backward flow lines from well 13. Total simulation length is 10 years; tick labels mark an interval of 1 year. The lower and right cross sections are projections along the axes of the black lines. The stationary flow field between January and March 2001 is extrapolated to the duration of 10 years. Different grey tones are used to mark particles arriving at different depths of well 13, big black circles indicate their infiltration area. Small black circles indicate wells, the white circle indicates observation well 3301, the white triangle observation well TEG374. The grey areas indicate areas of leakage from Lake Tegel into the groundwater. The top view is a zoom of Fig.1, well 13 lies within the transect.
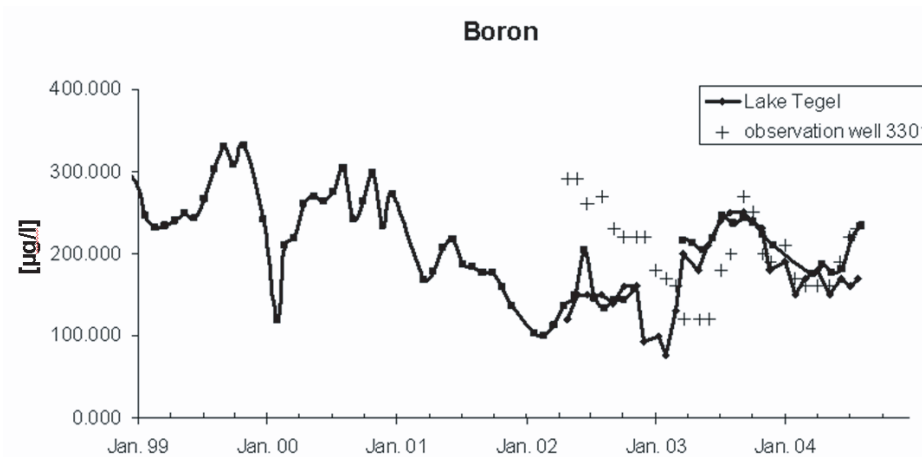
*Figure 3. Concentration of Boron in Lake Tegel and observation well 3301.
Concentrations in summer 2002 in 3301 match boron concentrations of Lake Tegel at least 2 years ago.
However, as boron may be slightly retarded in the subsurface, travel times can not be assessed from this graphic.*

2) Age dating

Age dating of water extracted from 3301 in spring 2003 showed an age of 20 month (NASRI 2004). This coincided quite well with the modelled travel times of 20 month and is higher than the minimum age boundary of 15 months obtained by chloride data. The model reveals that the age of 20 month is the real age of the water and not an artefact by mixing of old and young waters.

3) Anthropogenic chemical substances

Phenazone derivates are detected in observation well TEG374 in high concentrations (NASRI 2004). It was postulated that these substances infiltrated into the subsurface many years ago, when surface water concentrations

were much higher than nowadays. As the model starts in 2001, modelling of this infiltration is not directly possible, but extrapolation of the hydraulic conditions to the past results in travel times of 12 to 15 years.

Time variant leakage

The leakance is connected to the k value by

$$L \: [1/d] = k \: [m/d] \: / \: M \: [m]$$

where L is the leakance, k the hydraulic conductivity and M the thickness of the lake bed. The hydraulic conductivity k is a lumped parameter including the difference in viscosity of water at different temperatures.

It can be seen that leakance in summer is about 2 times the one in winter, roughly following the surface water temperature. However, a phase shift between the two curves can be observed. Looking at the period after January 2003, where dense data are available, the minimal leakage in winter and spring 2003 shows a lag of 2 months, in spring 2004 a lag of 4.5 months, the maximum in 2003 shows a lag of 1 month. The rising and falling limbs of the leakage always show a lag of a few months to the water temperature. Besides, the rising limb of the leakage curve always has a steeper slope than the falling.

Several mechanisms could contribute to this behaviour. The viscosity of the water by different temperature is very likely to have a contribution, theoretically with a factor of about 2 between 0 and 25°C. Column Experiments of the group Hydrogeo (FU) showed a difference of factor 4 with temperature differences of about 18°C (Taute 2004), probably due to degassing or biological effects. The group of Prof. Gunkel in accordance with Bouwer (2002) suggests algal and bacterial exopolymeres, particular organic matter or calcite precipitation as possible factors of influence.

The leakance shown in Fig. 4 has to be regarded as result of interaction of these factors, but it is not possible to asses each single contribution. Viscosity effects by temperature fluctuation have strong influence, but the lag of the annual leakance cycle to the annual temperature cycle postulates other influences existing which significantly affect the leakage factor and thus the infiltration rates. No satisfying transient model fit could be achieved with only using the viscosity effect for changing the leakance.
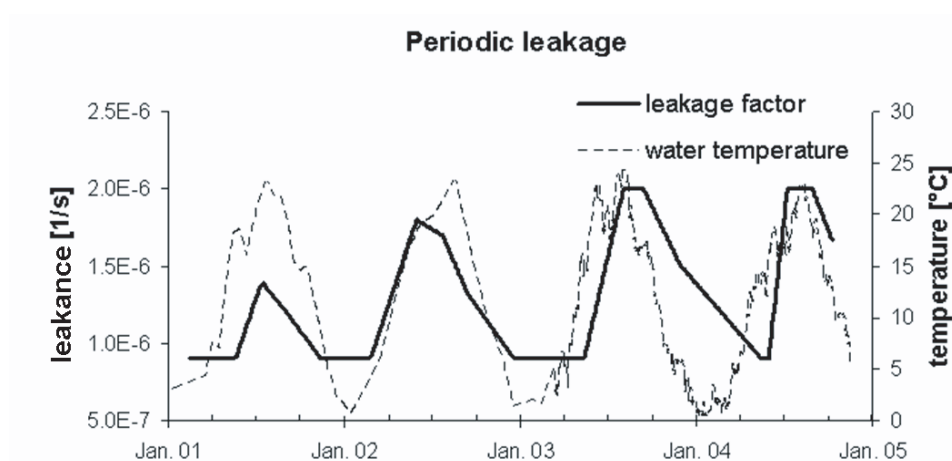
Figure 4. Periodic leakance of the lake bottom compared with lake water temperature.
Before 2003 data are sparse, so the leakage factor only indicates a trend. A high data density starts
from the beginning of January 2003, here magnitude and phase of the leakance are ensured.

CONCLUSIONS

Hydraulic and transport modelling of the study area revealed two aspects of major importance for the interpretation of the bank filtration process and data acquired: Bank filtrate extracted at well 13 is stratified horizontally in three water types of different age and infiltration areas. Sampling wells may contain therefore different water types. Sampling well 3301 shows a medium aged water, which was not expected to be present in the transect. Chemical indication of old water in sampling well TEG374 is confirmed, the presence of medium aged water being part of the content is identified by model based temperature interpretation. The lake sediments are found to show pronounced seasonal cyclic leakance, during summer two times higher than in winter and showing a 2 to 4 months lag to the water temperature. As result, amounts of water infiltrated in summer are higher than in winter, making the surface water quality in summer having higher impact on the quality of extracted bank filtrate.

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High resolution digital geological 3D models as a base of hydrodynamic calculation in heterogeneous aquifers

P. Wycisk, W. Gossel, A. Wollmann, H. Fabritius and T. Hubert

Abstract

Detailed 3D geological modelling was carried out in the contaminated mega-site Bitterfeld, enabling the structural analysis of the heterogeneous aquifers and the calculation of specific volumes. This was used for volumetric calculation of residuals of lignite seams left by the former open pit mining. The lignite is now serving as an adsorbens for organic groundwater contaminants. The geological model allows detailed analyses of the hydrogeological structure and thus the outline of contacts of aquifers. Furthermore it serves as a basis for the subsequent 3D numerical groundwater modelling. Two models have been built at local and at regional scale. Induced by the flooding event of the river Mulde in August 2002 and related rising of the water level of the Goitzsche Lake, the subsequent change of the groundwater flow direction could be modelled. Flow and transport direction changed completely by this event and show a strong influence by the preferential hydraulic conductivity of subsurface Quaternary channel-fill structures.

Keywords

Aquifer heterogeneity, Bitterfeld/Germany, 3D-geological modelling, groundwater modelling.

INTRODUCTION

Large-scale contaminated mega-sites, such as Bitterfeld, are characterized by regional pollution of groundwater due to the long and varied history of the former chemical industry in this region. Due to former industrial, as well as lignite mining activities in the region for more than 100 years, the groundwater was significantly contaminated by different sources of dump sites of the former chemical industry, and industrial areas. The situation is characterized by a complex mixture of organic compounds (chlorinated aliphatic and aromatic hydrocarbons) comprising a high variability of individual substances (Wycisk et al. 2003; Heidrich et al. 2004; Weiß et al. 2004)

The flooding event of the river Mulde in August 2002 in the Bitterfeld region led to a filling up of the former lignite open pit mine 'Goitzsche' and thereby to a rising groundwater level in the surrounding aquifers of several meters (Geller et al. 2004; Wycisk et al. in press). The rising water level caused changes in the groundwater flow directions. To calculate the effects of contacts between groundwater and layers that were not affected by contaminated groundwater before, a high resolution 3D geological model was built. The digital 3D geological model helps to identify these layers which are very important for adsorption and desorption processes and the transport of dense non-aqueous phase liquids (DNAPL) (Wycisk et al. 2002; Wycisk et al. 2003). It shows the 3D distribution of aquifers, adsorption materials such as clay and lignite, aquicludes and the distribution of leakage between aquifers. It thus helps to set up 3D structures and parameters for numerical groundwater modelling. A predictive calculation about hydrodynamic conditions, pathlines and solute transport could only be done by a numerical groundwater model. Thus the numerical groundwater flow model is of crucial importance for the adjacent industrial sites and the transport model is a basis for risk assessment in this area.

METHODS

Up to now, true 3D modelling of geological structures is not state of the art in regional and local environmental assessment except in the field of economic geology. The required geological information e.g. drilling information is not available on a local to regional scale. In case of former lignite open pit mining activities, the heterogeneity and complexity of the aquifer system is specifically increased by the disturbance of the primary aquifer situation and the re-filling activities of the former open pit mining. An additional problem for the 3D structural modelling to be solved are the very sharp edges and steep flanks of the open pits that were filled up by the overburden of the seam while mining was going on.

The techniques used in most cases (software tools) for geological 3D modelling are based on statistical or geo-statistical interpolation between stratified boreholes. These methods are inadequate in this specific field because it leads to a reduced heterogeneity and an inadequate loss of the 'real world' setting of the lithostratigraphic layers, as well as the artificial mining dump sediments and remaining lignite in the subsurface. Even the complex structural setting of the Quaternary sediments could not be represented correctly following a geostatistical approach. Therefore a first 3D geological model of about 16 km^2 was built for the most interesting sites in the southeastern part of the whole industrial area. The modelling system GSI3D (Geological Surveying and Investigation in 3 Dimensions, H.-G. Sobisch, Insight Ltd., Köln) was used. GSI3D is based on a 'constructive method' and allows the implementation of different additional geological information. Beside this 'expert-driven' approach, the 3D modelling of the Bitterfeld South Model comprises 16 km^2 and is based on a construction of 30 networked cross-sections which are based on 125 borehole records. Beside the drilling information, the following 2D and point information were used for the 3D-database: a) historical and recent maps of the mining areas, b) distribution of mining dumps, c) digital elevation model (DEM), d) geophysical logging information, e) and sediment distribution maps of the Quaternary as well as f) the geological maps. This first sub-model of the region was modelled in detail with a grid resolution of 10x10 m² and 31 lithostratigraphic units of Quaternary and Tertiary sediments.

Following this first sub-model, the modelled area was enlarged to about 50 km^2 for the entire region of the mega-site and their downstream areas. The structural model was generated by combining point information of about 250 boreholes which were implemented in about 62 cross sections. This model allows – beyond visualization purposes – volumetric calculations of partial or distinct sedimentary units, which are relevant for an assessment of retardation processes in the remaining lignites.

In a third approach the modelling system MVS/EVS (CTech Ltd) was used for the same model area mentioned before. MVS/EVS is strictly based on the geostatistical interpolation between boreholes. To get comparable results, the same boreholes as in the system GSI3D were given as input data. The pure interpolation was in parts very poor concerning the distribution of Quaternary sediments and the open cast mining dumps. Good results were reached by using cutting surfaces e.g. for the base of the mining dumps and the base of a Quaternary channel.

Based on the first geological sub-model, a numerical groundwater flow and transport model was set up. The database structure was converted to the GIS ArcView so that data exchange with the groundwater modelling systems was guaranteed. Structures and a first approach of parameter-settings (hydraulic conductivities and porosities) were taken from the geological model and implemented in a first local groundwater model. Due to the restricted area of the first geological model, the boundary conditions could hardly be defined. A new enlarged model of about 330457 km^2 was built to solve this problem. This regional model serves as principal study model for the calculation of the large scale hydrodynamical situation.

RESULTS AND DISCUSSION

The use of true 3D geological structural models allows different types of visualisation, calculation and predictions as well as the subsequent operation within hydraulic models. The provided information by the digital subsurface 3D model is of specific need in the field of environmental risk and impact assessment of contamination sites and also on a broader view in the context of hydraulic and transport modelling of complex and heterogeneous aquifer systems. The following examples outline the different options in using the true 3D geological models answering specific questions from the case study Bitterfeld.

One of the major questions is the aquifer/aquifer contact in heterogeneous systems and their regional distribution. The GIS-based information of any hydraulic relevant layer within the model can be provided. Figure 1 shows the example of the 'Hydrogeological Window' of the Quaternary and Tertiary aquifer contact and the related distribution. Additional information of land-use and point as well as non-point sources can be integrated in the GIS spatial model for further plausibility studies and risk assessment. In addition, the volumetric calculation of individual lithostratigraphic layers can be done. In the studied area the potentiality of high adsorption and also desorption capacities of the aquifer sediments is of high importants, e.g. remaining lignite in the aquifer system.
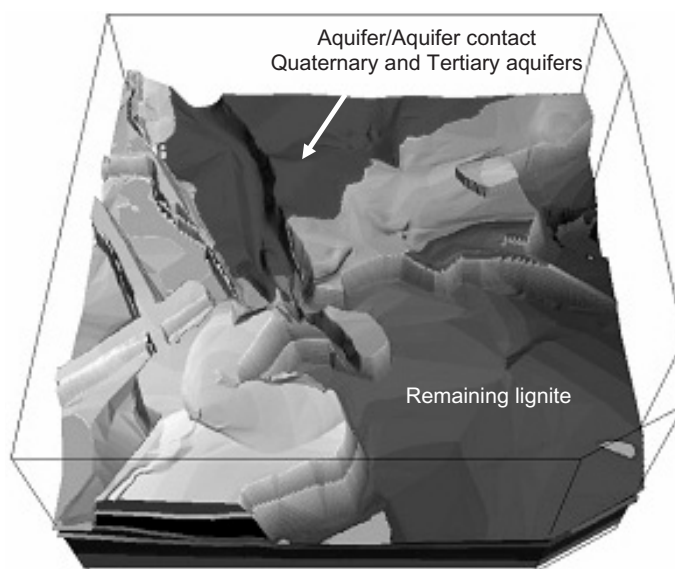
Figure 1. Hydrogeological contacts between Quaternary and Tertiary aquifers. Quaternary sediments and the lignite mining dump sediments are removed. Note the erosional NW striking structure of the channel, which is cutting into the underlying lignite seam and the Bitterfelder Glimmersand unit.

The calculation of the volume of the additionally saturated zone by rising water level affected by the flooding event in August 2002 is calculated and summarizes to 25.25 Mill. m^3 for the open pit mining dumps. Figure 2 shows the relation of saturated and unsaturated zone of the aquifer structures and the additional volume which is saturated by the flood. This calculation allows further hydrochemical calculation of the internal geochemical and hydrochemical induced processes. For a better understanding of the anisotropic structures of the open cast mining dumps orthophotos and a high resolution DEM were interpreted to locate internal structures of the anthropogenic layers. Ongoing evaluation tools implemented in the modelling software tools GeoObject 2 and GSI3D are the distribution and thickness in terms of equal area projection, cell-projection and contour lines for distinct sedimentary layers.

The influence of the detailed structural setting within the aquifer system and the related effects by increasing groundwater level is given in Figure 3. The groundwater models show the hydraulic effects of a subsurface Quaternary channel-fill which is characterized by a higher hydraulic conductivity. Before the flooding event 2002, the path
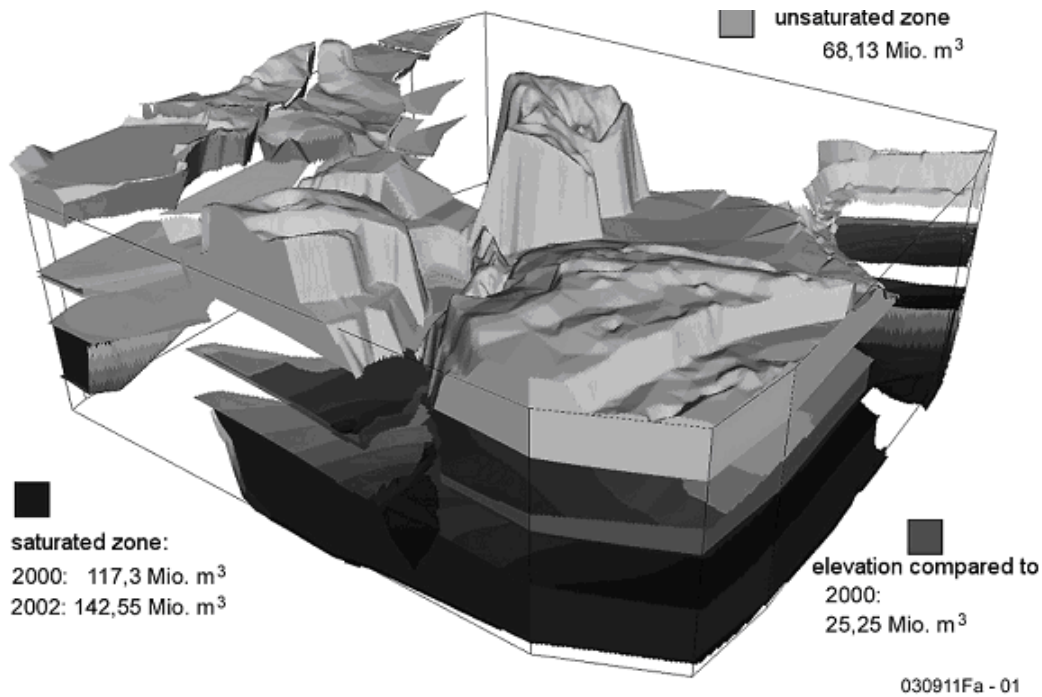
Figure 2. Volumetric calculation of the increasing amount of the water saturated zone
within the lignite mining dump sediments after the flood in August 2002, provided
for geochemical process modelling and prediction of hydro-chemical balance predictions.
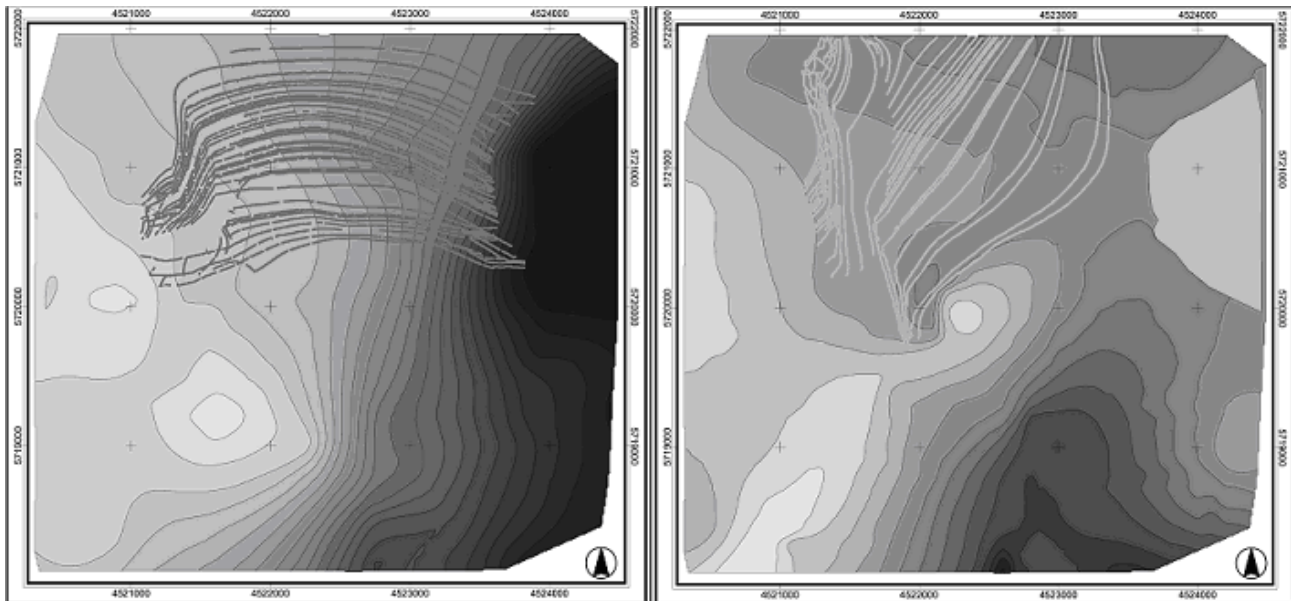
Figure 3. Path lines before and after the flooding event in August 2002 in the Quaternary aquifer.
The left fig. indicates the pre-flooding stage and the right, the post-flooding situation of the modelled area
(4 x 4 km). The increasing hydraulic effect of the subsurface Quaternary channel-fill
and the shift of path lines towards NW are remarkably indicated.

lines are directed toward the East, the Goitzsche Lake. After the rising water level induced by the flooding event 2002, a general shift towards the N and NE is modelled. The hydraulic effects of the Quaternary channel-fill in the subsurface are of conspicuous importance in terms of risk and exposure assessment as well as fate and pathway relations of the groundwater contaminants. The regional groundwater model (Fig. 4) shows in addition, that most

of the path lines lead to the North recently, towards the alluvial plain of the River Mulde. This calculation does not take into account, that distinct groundwater exploitation wells are used with varying extraction rates within the modelled area. Therefore, the local patterns of the path lines can be deviating in detail.
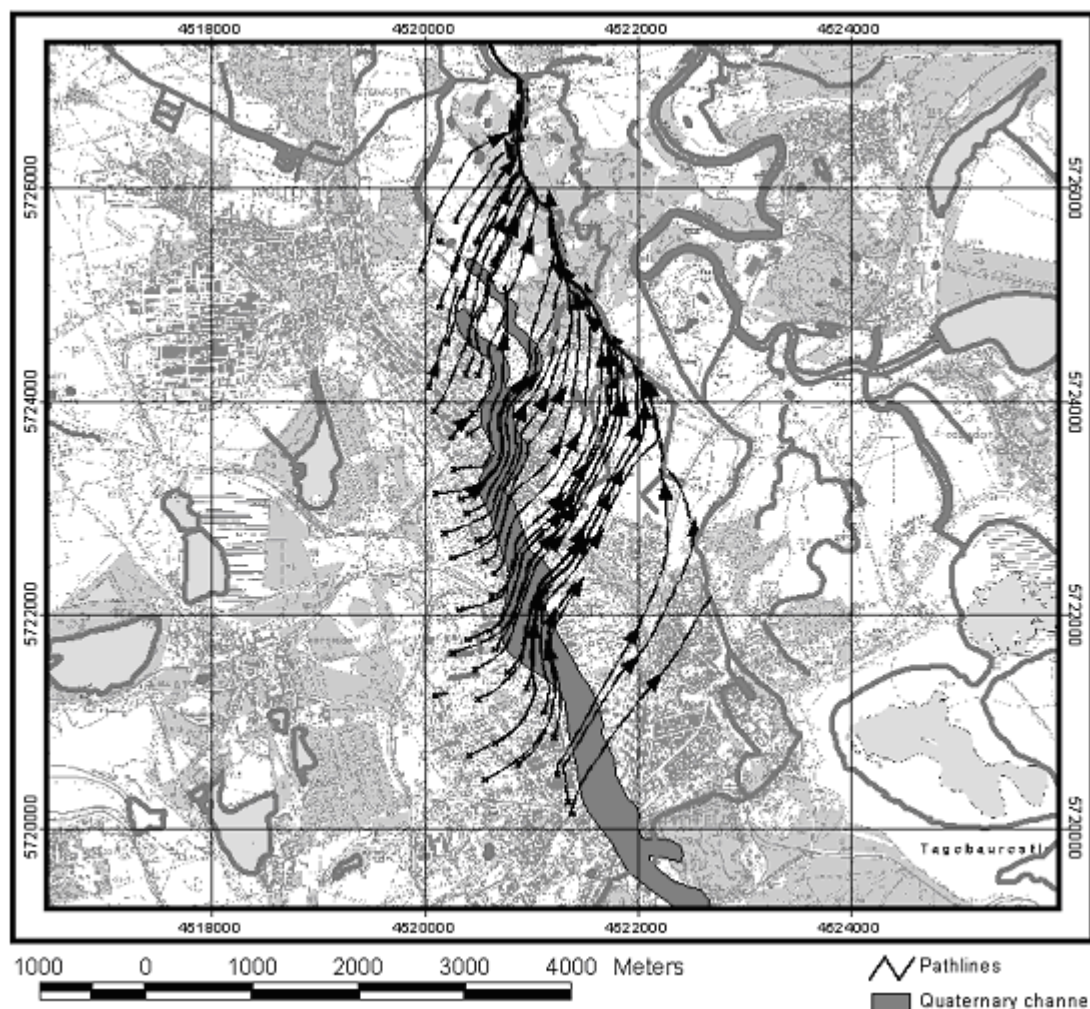
Figure 4. Calculated general path lines direction and distribution of the fluvial system of the regional groundwater model. The different influences of the subsurface Quaternary channel-fill are indicated. Local groundwater extraction and changing groundwater flow directions are not taken into account.

CONCLUSION

The detailed digital geological 3D model is an important need of subsequent numerical groundwater models in the field of fate and pathway prediction of regional contaminated groundwater aquifers. The step forward in getting detailed results which are based on true 3D structural models which are coming close to a 'real world scenario' seems to be convincing and a 'must' in the field of heterogeneous aquifers and related environmental risk and exposure assessment. As demonstrated in this paper, the detailed 3D geological modelling of the Bitterfeld area has shown that already the most detailed and realistic geological model enables the analysis of sorption processes and especially hydraulic effects of rising groundwater levels. The lignites are the most contaminant bearing layers due to their high adsorption capabilities for organic substances. Thus their distribution is of high importance for long-term transport processes of the groundwater contaminants. The GIS database was used for linking the geological model and numerical groundwater model. The two groundwater models were built, representing a local scale, and a

regional scale, respectively. The groundwater modelling includes aspects of groundwater balances, flow and solute transport prediction. The results show the importance of hydraulic relevant structures within the 3D geological model, as shown by the Quaternary channel-fill structure. The GIS environment of ArcView allows combining several different thematic layers to get valuable answers to environmental questions, e.g risk assessment of groundwater contamination, fate and pathway prediction, and land use planning concepts. Parts of this work were funded by the SAFIRA project (E1.1 FKZ 02WT0023) and the Ac-Hoc Project (FKZ PTJ0330492) by the BMBF, German Federal Ministry of Education and Research.

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Modelling parameter estimation
by tracer experiments in the porous aquifer
of Ljubljansko polje

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Abstract

Human activities in the area of the Hrastje waterworks of Ljubljana threaten groundwater quality. A numerical groundwater flow model was established for the wider area of the Ljubljansko polje aquifer. A lack of experimental data on solute transport leads to unreliability in the transport model and its predictions of pollution scenarios. The transport model needs to calculate reliable scenarios of pollution dispersion, which can only be achieved, with the application of real transport parameters. These could be provided from tracer experiments in the Ljubljansko polje aquifer. First, a small tracer test with potassium bromide was conducted in the Hrastje waterworks area, followed by a multi-tracer experiment (potassium bromide, uranine, microspheres and tinopal CBS-X) in the broader area of the Ljubljansko polje aquifer. Tracer test design considers differences between pollutant spreading in the unsaturated and saturated zones of the aquifer. Therefore, the tracer injection was performed as spreading on the surface (injection to the unsaturated zone), as well as injecting directly into the saturated zone through observation wells. Only potassium bromide and uranine gave successful results. Both tracers indicate the dominant groundwater velocity of about 20 m/d. The tracer experiment with uranine has shown sharp differentiation in the field and relatively long retardation times. The dispersivity depends on the length of the flow path and varies from 10 m at short distances to 100 m at long distances. Results of the multi-tracer experiment improved the flow and transport model. All together, this will enable better knowledge of the hydrodynamic conditions in the Ljubljansko polje aquifer, which will yield more effective measures for waterworks protection. Consistent implementation of these measures will improve the groundwater quality in the Hrastje waterworks.

Key words

Groundwater flow dynamics; Ljubljansko polje aquifer; modeling; porous aquifer; solute transport; tracer experiments.

INTRODUCTION

The gravel sandy aquifer of Ljubljansko polje is the drinking water source for nearly 300,000 inhabitants of Ljubljana city and vicinity. There are two main waterworks: Klece and Hrastje. The latter is momentarily only partly in use because of the environmental pollution. The plain area of Ljubljansko polje is a tectonic sink in the shape of a bowl and consists of river sediments that can reach thickness of more than 100 m in the deepest part. The bedrock is the impermeable permocarbonic shaly mudstone and sandstone. In the last million years, the river Sava filled up the basis thus forming a field. The lower part of the aquifer is composed of Pleistocene gravel and sand whilst in the upper part there are Holocene sand and gravel sediments. Among the sand and gravel deposits several conglomerate lenses appear. Above the conglomerate lenses, the clay is present and together with conglomerate represents the low permeable complex.

The hydraulic conductivity of Ljubljansko polje sediments is very good, from 10^{-2} m/s in the central part to 3.7×10^{-2} m/s on the borders of the plain. The existing lithological data are heterogeneous origin and very hardly comparable so the geological model of area is incomplete, special in the deepest part. The heterogeneousness

of Ljubljansko polje sediments has important impact on dispersion and tracers spreading velocity. A numerical groundwater flow model was established for the wider area of the Ljubljansko polje aquifer. However, a transport model was not established due to a lack of experimental data on solute transport. These could be provided from tracer experiments at the area of Ljubljansko polje aquifer.

The average groundwater level is 13 m under the surface. The groundwater level oscillation is about 3 m (from 273,18 m above see level to 276,01 m above see level). The general groundwater flow direction is from NW to SE, but on the tracing experiment area depends from lithological and thus hydrological conditions and it is variable in dry or wet season. The groundwater flow velocity cover a wide range of values according to the precipitation infiltration or Sava river infiltration and is from several meters to several tenths of meters per day.

EXPERIMENTAL SETUP

Tracer experiments were performed in the Hrastje waterworks area (Fig. 1). Because of a very poor knowledge about solute transport dynamics in the Ljubljansko polje aquifer, a preliminary tracer experiment with potassium bromide was conducted and followed by a multi-tracer experiment with potassium bromide, uranine, microspheres and tinopal CBS-X. Bromide was injected into an observation well PH-3 (Fig. 1), located very close to the waterworks. The water from the Hrastje waterworks is not treated (chlorination or ozonisation), therefore a usage of bromide did not endanger human health. Tracer amount and sampling schedule for the multi-tracer experiment were determined on the basis of the preliminary tracer experiment results.

A multi-tracer experiment was performed as spreading on the surface (injection to the unsaturated zone) near gravel pit Obrije (Fig. 1), as well as injecting directly into the saturated zone through observation wells PH-3, PAC-
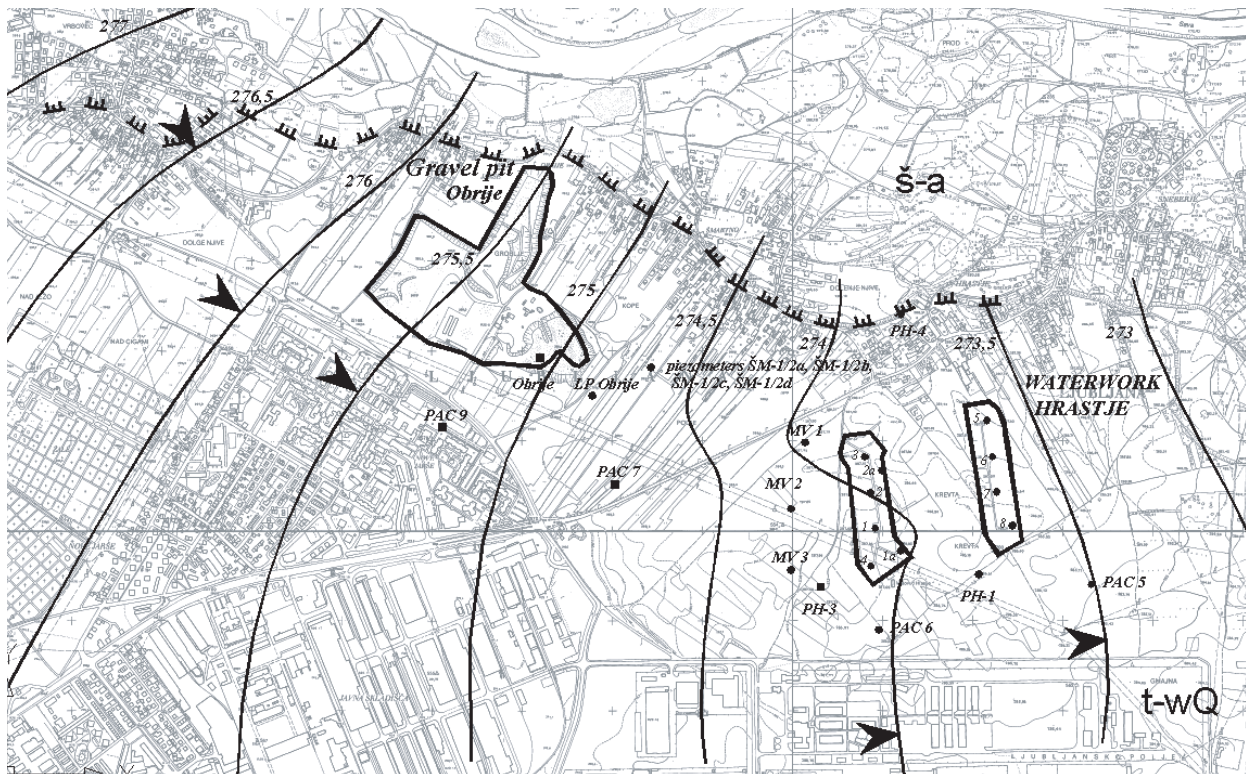
Figure 1. Location of the injection borholes and sampling boreholes with groundwater flow direction and geological features (š-a: Quaternary gravel, t-wQ: würm gravel deposit).

7 and PAC-9 (Fig. 1). Sampling was carried out in observation wells PAC-7, PAC-6, PH-1, LP Obrije, MV-1, MV-2, MV-3 and pumping wells of Hrastje waterworks (labeled as Hr-vd; Fig. 1). Allocations and tracer amounts are given in Table 1. In precedent paper only results of the multi-tracer experiment are presented.

Table 1. Details on tracer injections

Injection site	Tracer	Tracer amount	Injection date	Injection depth	Groundwater level
PAC-9	Uranine	3 kg	21.11	17.05 m	16.20 m
PAC-7	Tinopal CBS-X	10 kg	10.02	14.70 m	13.90 m
PH-3	KBr	35 kg	21.11	15.00 m	13.03 m
Gravel pit Obrije	KBr and microspheres	60 kg and 10 ml	21.11	surface	–

RESULTS OF THE TRACER EXPERIMENTS

The bromide, injected into observation well PH-3 was transported very reproducibly with some interesting results. Very fast movement of the bromide to the waterworks Hrastje was detected also connected with fast spreading of traces of the bromide away from the waterworks towards the observation well PAC-6. This phenomenon indicates nearly the same transport time of bromide to the observation well PH-1 and pumping well Hr-vd4 (Fig. 2). PH-1 has a three time longer distance (440 m) from injection well compared to the Hr-vd4 (150 m).
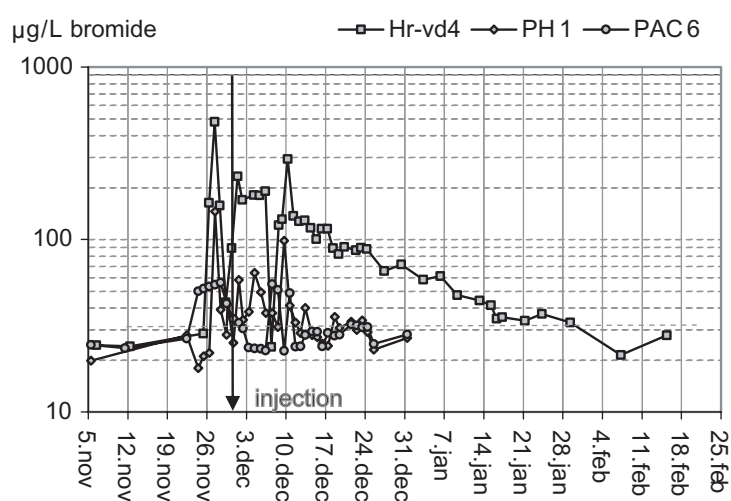
Figure 2. Concentration of bromide in observation wells PH-1 and PAC-6 and pumping well Hr-vd4 in dependency from time after the injection of KBr in PH-3. Bold horizontal line is average background of bromide app. 22 ± 2 μg/L in southern part of aquifer.

Uranine was injected in observation well PAC-9 (Fig. 1). Because of the possibility of cross-contamination at sampling of observation wells with a pump, a 'zero control' observation well (PH-5) was sampled during each sampling campaign. Transport of uranine, injected into observation well PAC-9 at northern side of the experimental area was determined up to a distance of 1,510 m. No difficulties with background concentration of uranine were observed. Long period of post-maximum relaxation was detected (Fig. 3). The uranine was transported to the west of the injection well towards LP-Obrije and than turned to observation well MV-1 and to the northern part of Hrastje waterworks (Fig. 1 and Fig. 2).
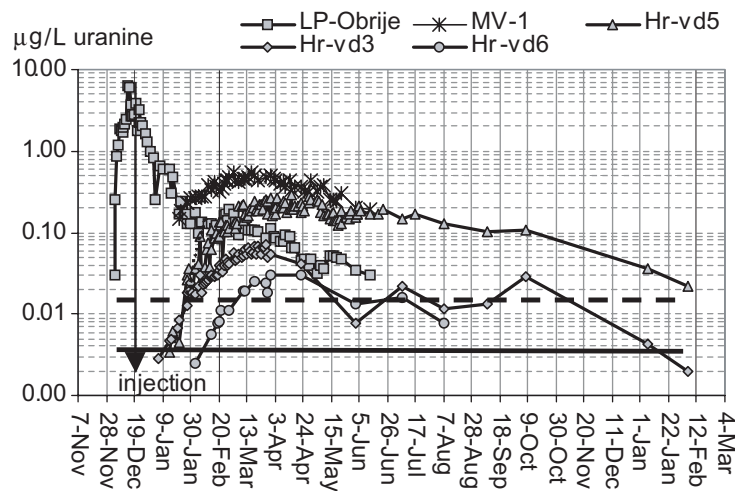
Figure 3. Concentration of uranine in observation wells LP-Obrije and MV-1 and pumping wells Hr-vd3, Hr-vd5 and Hr-vd6 in dependency from time after the injection of uranine in PAC-9. Continuous line: LOD of 0,002 μg/L for samples from pumping wells; dashed line: LOD of 0,01 μg/L for samples from observation wells. Pointing line: assumption from the other datasets because of delay at sampling of MV-1.

Tinopal CBS-X was injected into PAC-7 at the northern side of the Hrastje waterworks. During the sampling campaign, the tinopal was not detected in any of the wells. There are three possible reasons: the observation well PAC-7 has a poor conductivity and/or Tinopal CBS-X was adsorbed to the sediment an/or the injected amount was too small. Because of the detection of Tinopal CBS-X in observation well PAC-7 one and a half year after the injection we can conclude that the poor conductivity of PAC-7 has to be a main reason, but we cannot exclude even the influence of the adsorption.

TRANSPORT PARAMETERS AND MODELLING

Mathematical evaluation of tracer breakthrough curves was carried out with analytical (advection-dispersion model) and numerical (MIKE SHE groundwater model) methods.

Advection-dispersion model

Transport parameters were calculated with Maloszewski's advection-dispersion model (TRACI programe; Werner, 1998), which presumes uniform laminar flow and Fick's law (continuum). Simulated breakthrough curves were mostly calculated with multidispersion model, which enables calculation for more overlapping curves representing several flow paths.

The bromide breakthrough curves of Hrastje waterworks wells have shown dispersivities about 5,5 m (dispersion koefficient of 160 m^2/d) for preliminary tracer experiment and 20 m (dispersion coefficient of 160 m^2/d) for multi-tracer experiment. The most likely reason for these discrepancy is different pumping regime within the Hrastje waterworks in time of tracer experiment performance and different hydrological conditions.

The uranine breakthrough curves of LP-Obrije has shown a dispersivity of 6 m (dispersion coefficient of 100 m^2/d), Hrastje waterworks wells 50–90 m (dispersion koefficient of 400–700 m^2/d). From the results it is obvious that dispersion is increasing contemporary with flow path.

Numerical model

Numerical modelling significantly contributes to a better understanding of behaviour of water resource systems under normal conditions and in the case they are subjected to the natural changes or human interventions. The MIKE-SHE hydrological groundwater model of Ljubljansko polje aquifer was established a few years ago (Refsgaard and Gustavsson, 2000; Jamnik et al. 2001) and has served for solving everyday problems in managing water resource and designing long-term solutions. Its first objective was to establish as safe as possible structures for supplying drinking water to the population.

The groundwater model of Ljubljansko polje aquifer has initially been calibrated on a regional scale using 200 x 200 m-grid system. For the purposes of the simulation of tracer experiment the local model used 25 x 25 m-grid scale with 6 aquifer layers and time varying head boundary conditions simulated with the regional model. Several simulations were performed using advection-dispersion module in order to show the movement of the tracer with the water flow and dependence of the dispersion coefficient from several injection points. The cloud of tracer was simulated regarding time and space.

The simulations were based on water movement simulation during different groundwater level conditions in order to check the influence of the groundwater level on the velocity and direction of the tracer movement. The modelling results showed that the tracer cloud from different injection points travelled more southerly than the tracer cloud, which was determined by field experiment. The travel path depends on the groundwater level and moves towards north when the groundwater level is higher. The simulation showed that lower velocity and longer travel times of the cloud could be expected during the low groundwater level condition. The observed travel times of the tracer that was injected closest to the wells of the waterworks were unexpectedly higher than simulated travel times.

The changes in dispersion coefficient of course showed the influence on the travel times and widening of the pollution cloud, but not significantly. We can conclude that the considerable impact on the travelling direction of the tracer could be achieved by close examination of the hydraulic conductivity of the particular calculation layer, which is planned to be an important part of our future modelling work.

CONCLUSIONS

Groundwater tracing test in Ljubljansko polje area is the first field experiment regarding to this matter in the alluvial aquifers in Slovenia. The investigators recognise the realisation of the experiment as an important achievement for the particular branch of Slovenian groundwater research.

The benefits of the research are multiple. Despite of a great risk, the experiment was performed on the catchment area of the Hrastje waterworks, meanwhile used for drinking water supply. The research proved that the tracers could be used safely on sensitive area and that the researchers are capable and qualified to carry it out with a highest level of security.

The experiment has shown that the groundwater flow in the central part of the Ljubljansko polje aquifer on the catchment zone of Hrastje waterworks is generally well known, but in the local scale unpredictable and even faster then had ever been expected. The water supply was proved to be extremely vulnerable. From the water management point of view this means that in the case of existence of potential risk on the catchment zone area the remedial measures could be effective only if they were done fast.

The results of the experiments will serve as a basis for the verification of the existing groundwater protection zones.

It was the first time that the travel times and the catchment areas of the particular part of the Ljubljansko polje aquifer (around the Hrastje waterworks) were determined by field experiment and not just theoretically.

It has been confirmed that despite of numerous researches of the Ljubljansko polje aquifer the tracer experiments are requisite to give us some additional information that cannot be acquired in any other ways.

On the basis on our positive experience it is reasonable to believe that the tracer experiments will be increasingly used for the purposes of the water management in Slovenia. They should become a recommended tool for water management support in connection to the long-term decision and everyday, still unsolved questions.

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Applications of most recent borehole geophysical logging for aquifer characterization

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Abstract

The purpose of this article is to illustrate the applications of the most recent borehole geophysical measurements to the evaluation of the aquifer characteristics and its suitability for an Aquifer Storage and Recovery (ASR) project.

Amongst the many properties required for aquifer characterization are total porosity, lithology, grain size and grain sorting, fracture density, aperture and directions, free and capillary bound waters, saturated and unsaturated sands, and rock hydraulic conductivity.

In this article, we will present the application of lithodensity, Neutron, Array Induction, Elemental Capture Spectroscopy, Nuclear Magnetic Resonance, and Formation Micro Imager to the deduction of aquifer properties.

We will also present methodology that estimates the formation Elemental Analysis through a global inversion integrating all the above mentioned measurements.

Step by step analysis of the individual measurements will be presented, as well as the integration of these individual measurements into the final solution.

INTRODUCTION

Our objective is to utilize the advanced geophysical borehole logging in the planning and feasibility phase of an ASR project to predict the recovery efficiency and rates with reasonable confidence. This can be done ahead of the trials by increasing our confidence in the accuracy of rock property values that determine those rates. We will explain and illustrate the use of some advanced geophysical logging tools that, to our knowledge, are very seldom used in the ground water industry, if at all.

Methodology

We will examine the various porosity measurement techniques, together with a newly developed rock lithology measurement device; we will then combine all these measurements using a global solver to calculate an accurate value for aquifer porosity; we will then compare this value to different individual porosity values obtained from stand alone measurements.

We will illustrate the use of resistivity measurements with different depth of investigation to discriminate, in real time, those aquifer intervals that are permeable. Finally, we will illustrate the use of Nuclear Magnetic Resonance and its calibration to estimate aquifer rock conductivity.

AQUIFER POROSITY

Porosity is the ratio of pore volume to the total volume of the formation, and can be devided into two types: primary or original porosity which existed at the time the rock was formed, and secondary porosity such as vugs and fractures resulting from tectonic stress and dissolution channels.

All porosity measuring devices are measuring porosity in an indirect fashion, and in the prosess, are affected by few factors, the most important of which are: lithology, pore fluids, and clay contents. We will attempt to show the effect of each one of those factors on the traditional porosity measurements, namely Neutron Porosity, Formation Density Porosity, and Accoustic Porosity.

Figure 1 below summarizes different porosity measurements values for saturated and unsaturated layers, together with percentage errors as compared to the actual computed porosity for our subject well under study.

	Saturated Zone				Unsaturated Zone		
	Porosity	Standard Deviation	% Error		Porosity	Standard Deviation	% Error
Actual Porosity	0.342	0.021			0.335	0.02	
CMR Porosity	0.337	0.017	1.46%		0.242	0.022	27.76%
Density Porosity	0.371	0.015	-8.48%		0.453	0.017	-35.22%
Neutron Porosity	0.336	0.024	-7.02%		0.152	0.038	54.63%

Figure 1. Porosity values from different geophysical logging sensors
with the associated errors compared to the true aquifer porosity

Elemental Capture Spectroscopy

We have seen from the above discussion the Neutron and Density porosity dependency on lithology which needs to be known prior to utilizing the two measurements in a simultaneous solver. The most direct lithology measurement available for the logging devices is the Elemental Capture Spectroscopy (ECS*).

ECS* relies on measuring the total Gamma Ray spectrum resulting from the different mineral compositional elements in the formation after have been exited with fast neutrons, knowing the spectra for the individual elements, the total Gamma Ray spectrum can be decomposed according to proportion of the specific element in the formation. The process is schematically shown below.

Figure 2 shows the results of the spectral processing and dry weight calculation for our subject well. Here we are showing only the elemental relative yield for Aluminum Silicon, Calcium, and Iron, from which the dry weight mineral fractions are calculated for Q-F-M (Quartz, Feldspar, and Mica), Carbonate, and Clay. From the dry weight mineral fractions, the grain density is calculated and shown in red. We notice the relatively clean formation, with clay content less than 10% down to 235 ft, while the aquifer matrix is primarily sandstone with calcite cementation. Below 275 ft, we get into the marl formation where the clay percentage goes up to 30%, and carbonate to 40%.

From the Gamma Ray curve shown in green in depth track, we notice the lack of correlation between the amount of clay content and GR reading indicating that the GR measurement would not be a good clay/shale indicator.
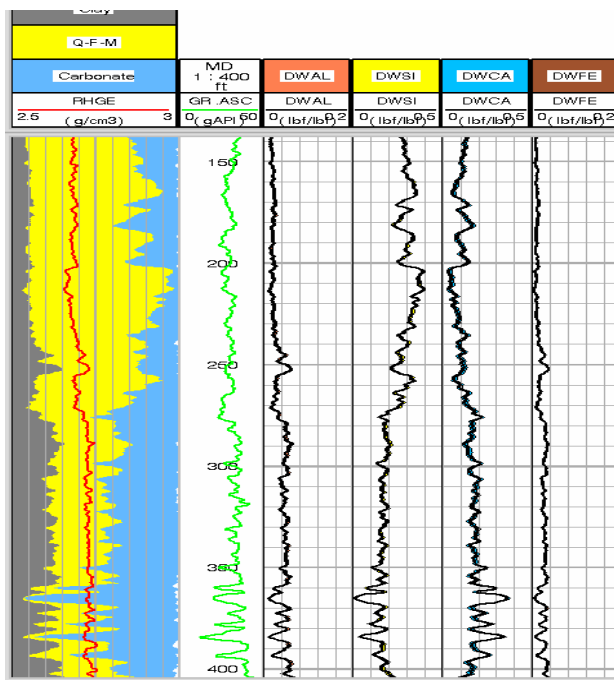
Figure 2. ECS outputs dry weights of the different elements in the formation rocks, which in turn is used to calculate rock percentage in formation

Elemental Analysis (ELAN*)

ELAN* software is an inversion program that evaluates geological formations assuming they conform to a specific petrophysical model with three primary mechanisms: tool measurements, formation volumes, and response parameters. The inversion process minimizes (in a least square sense) the difference between a synthetic computation of the measurements and the actual log values. The inversion is parametrized using the formation components such as volume of Quartz, Clay, Water, and Air. The reconstructed logs are compared to the measured logs and used as a diagnostic tool by the log analyst.

	Clay Percentage	Quartz Percentage	Calcite Percentage	Total Porosity	Effective Porosity	Free Fluid Porosity
Unsaturated zone	7.5%	42.5%	16.8%	34.7%	33.4%	8.2%
Saturated Zone	15.3%	47.6%	10.1%	36.2%	34.7%	16.8%
Transition Zone	21.5%	36.7%	15.3%	37.8%	35.7%	10.4%
Lower Layers	31.9%	24.7%	23.5%	32.3%	27.5%	0.8%

Figure 3. Results across different zones

Invasion profile

The mud filtrate invasion phenomena is utilized to predict the permeable intervals, and to calculate the unaltered formation resistivity which is used to determine the aquifer water salinity.
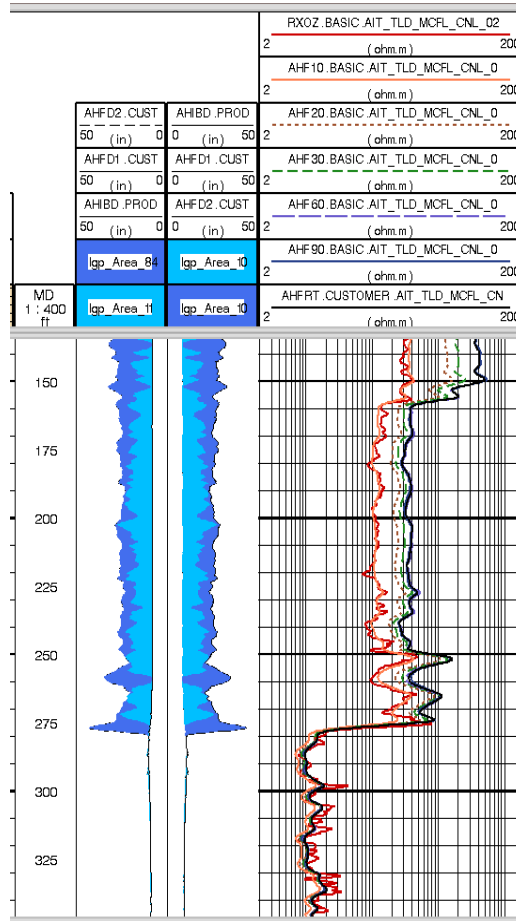
Figure 4. Invasion profile from the different depth
of investigation resistivity measurements indicate conductive intervals

Hydraulic conductivity

Nuclear Magnetic Resonance logging

Nuclear Magnetic Resonance logging (CMR)* relies on a strong permanent magnetic field that is applied to the formation and aligns the hydrogen nuclei parellel to its direction; a radio frequency antenna is then used to measure the time it takes for the nuclei to return back to their initial condition.

This relaxation time constant is decomposed in to its individual components and the distribution of the time constant T2 is plotted. The strength of the signal depends on the amount of hydrogen nuclei in the formation, and since hydrogen is present primarily in water, the strength of the signal, or the area under the T2 distribution curve is a representation of the total porosity.

Aquifer permeability/Hydraulic conductivity
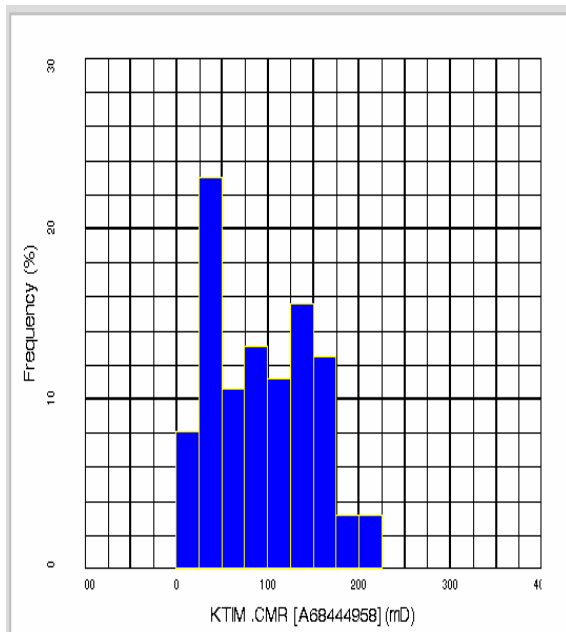
*Figure 5. Permeability distribution
over the aquifer interval*

	Median Value (MD)	Standard Deviation
Timur-Coates		
Permeability	94.69	53.79

Figure 6. Average Permeability in MD
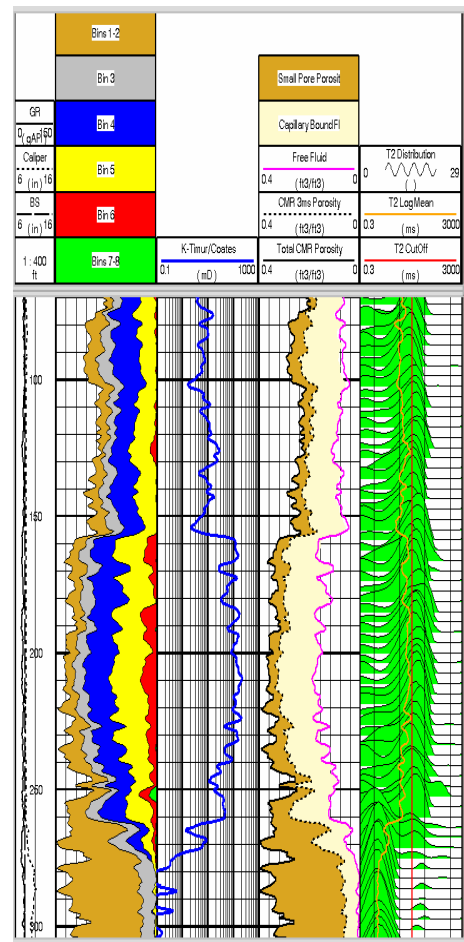
*Figure 7. Pore size distribution permeability
free and capillary bound water*

The calculated permeability in MD on a logarithmic scale from 0.1 to 1,000, we see that across the aquifer it is reading around 100 MD, below the aquifer it goes down to almost zero, and above the aquifer it reads in the range of 10 MD, but this reading above the aquifer is an artifact due to lack of hydrogen as the layers are unsaturated. We see clearly the correlation between the permeability on one side and the pore size distribution and amount of free water on the other side.

The histogram shown above is for the Permeability in MD across the aquifer interval, it has an average value of 94.7 MD.

Pumping test resulted in a total transmissivity of 800 mm/day for 35 mt interval, yielding aquifer conductivity of 23 mt/day. *Thus aquifer conductivity in mt/day = CMR pearmeability in MD X 0.25.*

Aquifer vertical heterogeneity/Homogeneity fracture analysis

The Formation Micro Imager (FMI)* allows continuous observation of detailed vertical and lateral variations in formation properties, one really sees the formation (Figure 8).

The measuring device comprises 8 pads mounted on 4 arms, each pad contains 24 micro electrodes for a total

of 192. The processing of the electrical current recorded by these microelectrodes provides images which look like core photographs with a high vertical and lateral resolution of 5 mm. These high resolution images are used in structural, stratigraphic geology and in and reservoir characterization applications.

Figure 8. Measuring device for Formation Micro Imaging (FMI)

Fracture analysis

Open fractures constitute permeability path which must be taken into account when modeling an aquifer. The presence of fractures also affects the value of the cementation factor, used in computing water salinity from formation resistivity.

Though in our study we were concerned with the upper unconfined layers, for the sake of illustration we will show an interval of the lower layers which are found to be highly fractured.

Figure 9 is an expanded scale log is for an interval deeper than our target aquifer, it illustrates an open fracture dipping at an angle of 68 deg towards south east.
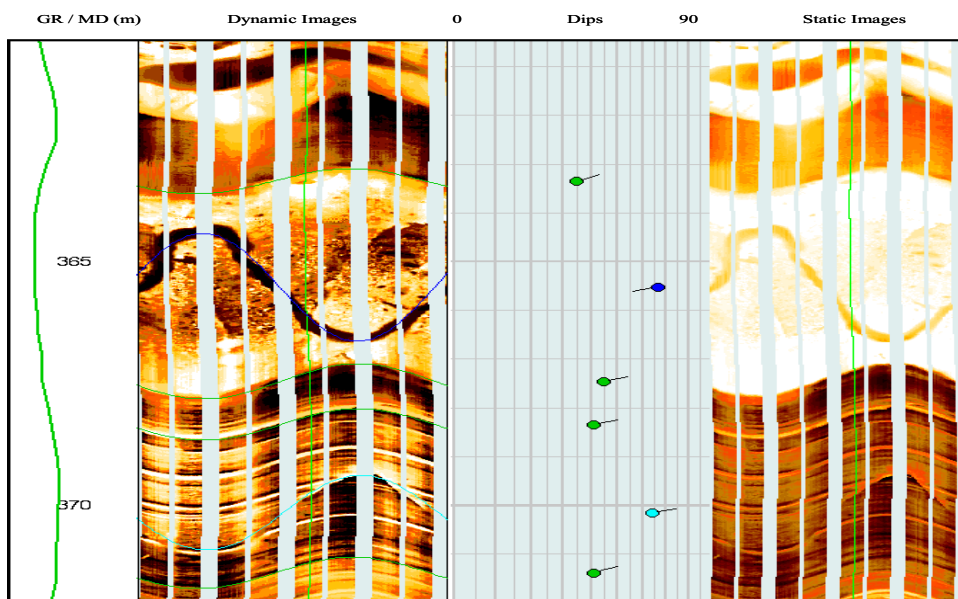
Figure 9. Open Fracture at 365 MT

CONCLUSIONS

- A holistic approach to the determination of aquifer porosity was introduced utilizing the newly developed lithology measurement technique (ECS*).
- Multi depth of investigation resistivity measuring tool (AIT*) was used to predict, in real time, which layers of the aquifer are permeable, as well as estimating an accurate aquifer resistivity.
- The Nuclear Magnetic Resonance (CMR*) was introduced to differentiate between free water,capillary bound water, and clay bound water. A calibration coefficient from measured permeability to aquifer conductivity was also derived.
- The Formation Micro Imager (FMI*) was used to qualitatively identify which layers in the aquifer are vertically homogenous, as well as detecting fractured layers.
- Modern state-of-the-art logging tools allow measuring the physical properties that control the behaviour and performance of an aquifer. It allows to build a quantitative model of the aquifer that can be used in a predictive manner to locate additional production wells or to monitor an ASR project.

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Health aspects

Pathogens and micro pollutants

Occurrence, fate and behaviour of pharmaceutical active compounds (PAC) and endocrine disrupting compounds (EDC)

Persistent organic substances

Nomogram to predict
water quality improvement
for managed recharge of aquifers

Peter Dillon, Paul Pavelic, Karen Barry,
Susanne Fildebrandt and Notoadmodjo Prawoto

Abstract

A simple nomogram was developed to describe the number of log-removals of pathogens and biodegradable organics between injection and recovery wells for ASR with nearby production wells, for simple Aquifer Storage Transfer and Recovery (ASTR) projects, and between the river bank and production wells in bank filtration projects. The method assumes a homogeneous isotropic aquifer of uniform thickness and porosity with uniform ambient hydraulic gradient, uniform rate of pumping and extraction, a constant exponential rate of biodegradation or pathogen inactivation, and linear adsorption isotherm. Only two non-dimensional parameters were needed to define the number of $\log_{10}$ reductions or biodegradation during transport through the aquifer to the recovery well. These describe advective transport due to pumping wells and due to the regional hydraulic gradient respectively. These parameters uniquely defined the ratio of minimum travel time to the time for one-$\log_{10}$ reduction, and therefore defined the number of log10 reductions of the contaminant reaching neighbouring wells.

The method was applied to a case study in the Bandung Basin, Indonesia to derive safe distances between stormwater injection wells and downgradient pumping wells for nine hydrogeological zones in the basin. Recharge enhancement was being considered to help address the groundwater overdraft in the basin. While the approach is very simple, it was adequate to demonstrate that in each of these zones the rate of groundwater flow was high in relation to rate of pathogen inactivation, so that runoff from the ground surface, where stormwater and sewage systems merged, should not be admitted into wells. However roof-runoff, piped directly to a well, was likely to yield water suitable for injecting into the aquifer without adverse impacts on the quality of neighbouring wells. Although the model is very simple, it may be useful as a planning tool or to assist in designing MAR projects.

Keywords

Groundwater, solute transport, analytical models, pathogens, ASR, Indonesia.

INTRODUCTION

A simple screening tool was developed to evaluate the potential for viable pathogens to be transported from an injection well to a down-gradient pumping well. Through the use of common simplifying assumptions, such as steady-state flow and a constant exponential rate of pathogen attenuation, the model could be reduced to a nomogram to represent a wide array of situations. The method was applied to the Bandung Basin to assess the water quality impacts of injecting stormwater and roof-runoff into the aquifer.

METHOD

Firstly it is assumed that the pathogens (or other trace organic contaminants) undergo first-order exponential decay with respect to residence time in the aquifer.

$$C_t = C_0 \; 10^{-t/\tau} \qquad\qquad \text{Eqn. (1)}$$

where C_t is the concentration or number of viable pathogens per unit volume after storage time t, C_o is the concentration or number in water recharged via the injection well, and τ is the time required for the initial concentration or number to be reduced to 10% of its original value, often called the one-$\log_{10}$ removal time.

Single well systems (ASR)

The minimum residence time of injected water in the aquifer in ASR systems is simply the storage period between injection and recovery, defined here as t_s. Hence the worst case scenario for biodegradation or inactivation is when $t = t_s$ in equation (1), neglecting the potential effect of dilution with native groundwater.

Dual well systems

In Aquifer Storage Transfer and Recovery (ASTR) or with ASR and a nearby pumping well which needs water quality to be protected, the worst-case scenario considers the water that has travelled to the recovery well along the shortest flow path when the injection and recovery wells are operating continuously (at the same rate). This gives the minimum travel time (t_{min}) over which biodegradation can occur. If the ambient groundwater velocity is small with respect to the gradients induced by the injection and recovery wells, and assuming injection and pumping rates are equal;

$$t_{\min} = \frac{\pi D n_e L^2}{3 Q} \qquad\qquad \text{Eqn. (2)}$$

where: D is the aquifer thickness (m);
   L is the distance between injection and recovery wells (m);
   n_e is the porosity of the aquifer (–);
   Q is the rate of steady-state pumping (in and out) $(m^3 d^{-1})$.

The shortest travel time occurs when there is a regional hydraulic gradient in the aquifer and the recovery well is situated directly downgradient of the injection well. Dilution with ambient groundwater is neglected as this effect leads to enhanced contaminant attenuation. In such a two well system, t_{min} is given by Rhebergen and Dillon, (1999) as:

$$t_{\min} = \frac{n_e L}{\dfrac{3 Q}{\pi D L} + v_{do}} \qquad\qquad \text{Eqn. (3)}$$

where v_{do} is the Darcian velocity component between the injection well to the recovery well (md^{-1}). As continuous concurrent injection and recovery rarely occur, this equation is likely to underestimate travel time because when wells are operated intermittently the average hydraulic effective gradient over the time of travel will be less than the value which has been assumed in this equation (worst-case scenario).

Assuming that the contaminant or pathogen of interest is also sorbed onto the aquifer matrix with a linear adsorp-

tion isotherm, then contaminant transport is slowed by a constant retardation factor, R, with respect to conservative transport of the water molecules. Thus the minimum travel time, t $_{min\ i}$ of a contaminant, i, is a factor R times the travel time of conservative solutes that move at the same rate as the water, ie:

$$t_{min\ i} = R\ t_{min} \qquad\qquad \text{Eqn. (4)}$$

where

$$R = 1 + K_d\ \rho\ /\ n_e \qquad\qquad \text{Eqn. (5)}$$

and

$$K_d = f_{oc}\ K_{oc} \qquad\qquad \text{Eqn. (6)}$$

where K_d is the distribution coefficient for a linear adsorption isotherm [m^3/kg];

ρ is the dry bulk density of the porous media [kg m^{-3}];

f_{oc} is the weight fraction of organic carbon in the porous media [–];

K_{oc} is the adsorption coefficient related to organic carbon content [m^3/kg OC].

In this model, sorption acts only to extend the travel time during which biodegradation takes place, and by itself is not regarded as a sustainable attenuation process. It is also assumed that all of the water from the pumping well originates from the injection well.

Bank filtration

From image well theory, constant head along a stream can be approximated by an injection and recovery well pair bisected by the stream, as in equation (2), but substituting the distance, a, between the stream and the pumping well (a = L/2) and noting that the water derived from the stream has only half the travel distance of the water travelling from the image well to the pumping well. This results in equation (7) (Dillon et al., 2002).

$$t_{min}\ =\ \frac{2\,\pi\,D\,n_e\,a^2}{3\,Q} \qquad\qquad \text{Eqn. (7)}$$

In this instance the rate of streambed infiltration, q, induced by pumping from the well at a rate, Q, at any time, t, since the commencement of pumping, is approximated by Glover and Balmer (1954) as:

$$\frac{q}{Q}\ =\ erfc\left(\frac{a}{\sqrt{4\,\alpha\,t}}\right) \qquad\qquad \text{Eqn. (8)}$$

Where α is aquifer diffusivity (transmissivity/storage coefficient) and here in an unconfined system of approximately constant saturated thickness, D, and hydraulic conductivity, K, is KD/n_e. For steady state pumping, the value of q/Q is one for a semi-infinite aquifer with an initially horizontal free-surface.

A contaminant originating from the stream at a steady concentration, C_0, reaches a concentration, C_t in the well at time t is given by Equation (9) which is a restatement of equation (1) accounting for equations, (4), (7) and (8). Dillon et al. (2002) also account for the effect of conservative contaminants, e.g. salinity, in groundwater for bank filtration design.

$$C_t\ =\ C_0\ \frac{q}{Q}\ 10^{-R\,t_{min}\,/\,\tau} \qquad\qquad \text{Eqn. (9)}$$

Other situations

Wherever the travel time between a constant sole source of contaminant and a point of groundwater discharge can be calculated, along with the relative contribution of that source to the discharge, and retardation and degradation or inactivation rates are known, then the concentration remaining at the point of discharge may be calculated. ASRRI (Aquifer Storage and Recovery Risk Index) is a computer program recently developed by CSIRO that, using the principles outlined above, calculates the risk of contamination for a range of single and dual well ASR systems (Miller et al., 2002). The risk indexes calculate whether specific trace organic or microbial contaminants in the recovered water would reach its target attenuation ratio (log removal) or its guideline value for a given scenario.

NOMOGRAM FOR DUAL WELL SYSTEM

Using three nondimensional terms, each with a physical meaning, a nomogram based on equations 1, 3 and 4 was produced to define the $\log_{10}$ removal under a range of scenarios for dual well systems. This allows separation distances or pumping rates to be determined that will meet any required number of log-removals for a given set of aquifer characteristics (depth, porosity, Darcian velocity and retardation factor (organic carbon fraction). The non-dimensional terms are as follows:

Term 1: Advective transport expression due to pumping wells $\dfrac{n_e\, L^2\, D\, R}{Q\, \tau}$;

Term 2: Advective transport due to regional hydraulic gradient $\dfrac{n_e\, L\, R}{v_{do}\, \tau}$;

Term 3: The ratio of contaminant travel time to one $\log_{10}$ removal time $\dfrac{t_{\min}}{\tau}$.

The nomogram, shown in Figure 1, can be used in various ways. Knowing existing or planned well locations and aquifer properties, and an approximate one-$\log_{10}$ removal time, e.g. from Dillon et al. (2005), a value for term 1 can be found on the x axis and a curve value for term 2 can be used to interpolate a y coordinate value on the nomogram. This immediately defines the value of term 3 which also defines the $\log_{10}$ reduction expected for that
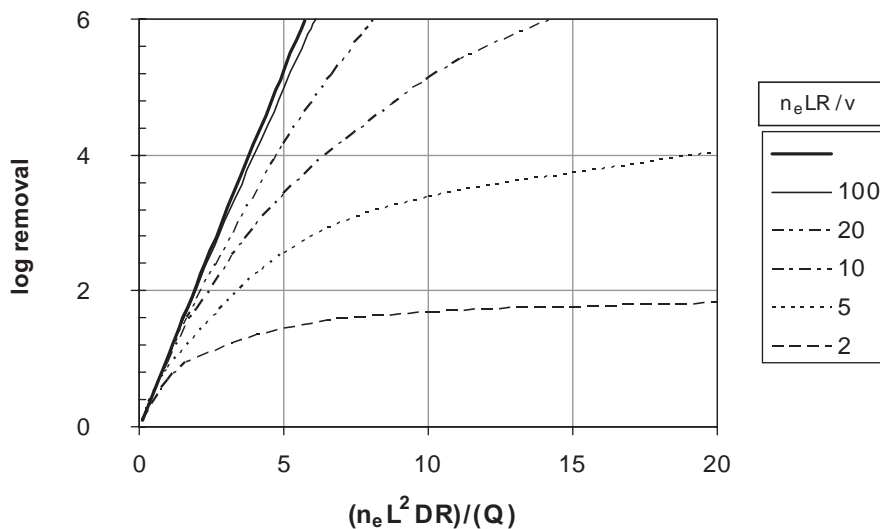
Figure 1. Nomogram for dual well ASR system (recovery well down-gradient of injection well)

contaminant. Alternatively, if the aim is to achieve a given log removal (say 4-log) then the curve parameter (term 2) cannot be less than 5. The curve parameter (term 2) is composed of variables that are intrinsic to the aquifer and the contaminant or pathogen, and the sole control variable is the separation distance, L. Hence L can, in principle, be increased so that it is sufficiently large that the curve rises above the number of log removals required. Term 1 contains two control variables, and is proportional to L squared and inversely proportional to Q. In this case if Term 2 had a value of 10, then Term 1 would need a value exceeding 7 to achieve the required 4 log removals. This may require reducing the pumping rate.

The nomogram shows that there are some combinations of characteristics where no realistic system design will produce sufficient log removal for satisfactory performance. In those areas ASR should be prevented unless the injected water is adequately pre-treated. Conversely, higher n_e, L and R, values increase terms 1 and 2 to give more time available for biodegradation, and result in a higher log removal. Note that the curve parameter tends towards infinity as the ambient gradient approaches zero. Fildebrandt et al. (2003) also provide nomograms for recovery wells positioned upgradient of the injection well and at various angles, however normally the down gradient direction is expected to be the most critical case for decision making concerning water quality impacts on the aquifer.

APPLICATION TO BANDUNG BASIN, INDONESIA

The city of Bandung in western Java, Indonesia, is facing increasing stress on the quantity and quality of its groundwater resources due to the rising population, increasing water demand, and changes of land use. Groundwater abstraction in the Bandung Basin increased from 47 Mm^3 to 61 Mm^3 per year between 1990 and 1994, while the number of deep wells increased from 1,000 to 4,700 (Suryantoro, 1999) and groundwater levels continued to fall by 2−4 m per year (Soetrisno 1998). In recent years the Indonesian government has become more active in groundwater protection and has considered developing methods to enhance groundwater recharge (Bukit 1995; Supriyo et al., 1999), including use of excess stormwater runoff.

The Bandung Basin is surrounded by a number of volcanic mountains in the north and south. The Citarum River originates in the south and flows northwest through a central 1,000 km^2 plain where most of the urban and industrial areas are located. The climate is tropical with an annual rainfall of 1,900−2,200 mm with distinct wet and dry seasons (Soetrisno, 1998). The central basin is composed of various Quaternary fluvial sediments of volcanic origin. The general direction of groundwater flow is from the northern and southern mountain ranges towards the Citarum River. There are two hydrogeological systems: the *shallow aquifers* and the *deep aquifers. The shallow* Kosambi aquifer occurs within the upper 40 m. and has low (< 2 L/s) or low to moderate well yields (2−10 L/s) and was originally of potable quality (Suhari and Siebenhuner, 1993). The *deep aquifers* are semi-confined to confined and are present at depths of between 40 m and 150 m. The two uppermost deep aquifers are the Cibeurum aquifer and the underlying Cikapundung aquifer. The Cibeurum aquifer, composed of young volcanic deposits from which most water is pumped, has moderate to high well yields (2−10 L/s to >10 L/s) whilst the Cikapundung aquifer, which consists of old volcanic deposits, has moderate well yields (2−10 L/s) (Soetrisno, 1998).

The fate of pathogens in the groundwater presents the greatest single threat to public health from recharge enhancement using waters of impaired quality. Therefore this study aimed to find separation distances between injection wells and pumping wells down-gradient that would achieve specified levels of microbial attenuation in different zones of the Bandung Basin. For illustration purposes here, a 4-log removal is suggested for injected surface water whilst for inherently better quality roof-runoff a 1-log removal may be enough.

The shallow aquifers of the Bandung Basin were classified into regions with different hydraulic gradients (<1%, 1−10% and 10−17%) and well yield (<2, 2−10, and >10 L/s) (Suhari and Siebenhuner, 1993). These maps were

overlain to form zones to allow the assignment of L, the distance between the injection and recovery well necessary for a certain log removal. There were nine possible combinations (3 hydraulic gradients x 3 well yields). Zone 1 produced the smallest separation distance and zone 9 the largest. Two of the nine different zones (7 and 9) did not occur in the basin (Figure 2). Zones 1, 2 and 4 occupy the largest the area.
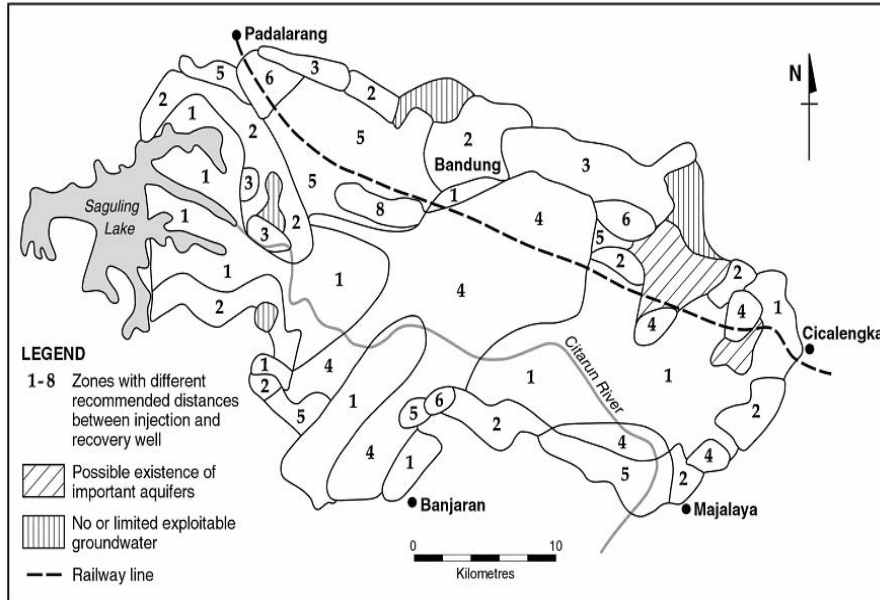
Figure 2. Aquifer zones, based on piezometric gradient and well yield,
for which well separation distances have been calculated

To calculate L values for each zone the following parameter values were assigned: K = 1 m/d, n_e = 0.1, D = 10 m, R = 1 (i.e. no retardation) and assuming τ = 10 days (Toze 2004). Sensitivity analyses are reported in Fildebrandt et al. (2003). The calculated separation distances (L) for 1-log removal are 45 to 150 m, and for 4-log removal are 92 to 326 m (Table 1).

Table 1. Recommended separation distances (m) between injection and recovery wells for different zones
for 1-log removal and 4-log removal, τ = 10 days

τ = 10 d	Estd. yield and Darcian flow		D = 10 m, n_e = 0.1	
Zone	Q [l/s]	V_{do} [m/d]	1-log	4-log
1	1	0.01	45	92
2		0.10	56	144
3		0.17	67	195
4	5	0.01	99	200
5		0.10	110	246
6		0.17	119	288
7	10	0.01	139	280
8		0.10	150	326
9		0.17	159	365

CONCLUSIONS

Conservative simplifying assumptions about solute transport and degradation in aquifers have been used to determine distances over which a preconditioned minimum level of pathogen inactivation occurs in aquifers during ASR. Maps of hydraulic gradient and well yield have been used to characterize regions in the shallow aquifer system of the Bandung Basin within which the spatial extent of pathogen inactivation is expected to be consistent. The results obtained indicate that 1-log removal distances vary from 45 m to 150 m between the zones, and for 4-log removals from 92 m to 326 m. This suggests that individual domestic scale ASR with water containing pathogens is quite likely to influence groundwater quality at nearby wells. Hence at domestic scale, roof runoff only should be admitted to injection wells, to reduce the level of dependence on the aquifer for water treatment. The well separation required for 1-log removal is approximately half that required for 4-log removal. The smallest acceptable separation distances occur in flat low yielding areas that occur in the central part of the Bandung Basin. The more problematic areas, requiring larger separation distances between wells occur on the margins of the Bandung Basin where the aquifer is higher yielding and hydraulic gradients are steeper. Given these distances are larger than the average size of land holdings, any landholder injecting water of poor quality could adversely impact other groundwater users. The techniques developed and utilised in this study may have widespread application in the design of ASR systems.

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Are there limits
to cyanobacterial toxin (microcystin) elimination by sand passage?

G. Grützmacher, G. Wessel, I. Chorus and H. Bartel

Abstract

Cyanobacterial toxins are substances produced by cyanobacteria that occur in surface waters world wide. The most common group of cyanobacterial toxins is the group of structurally similar microcystins (MCYST).

Sand passage as used in slow sand filtration, artificial recharge and bank filtration has shown to be effective in eliminating microcystins in many cases. For secure drinking water production from surface waters infested by microcystins removal has to be ensured in a wide variety of cases met in the field.

It was therefore the aim of experiments in technical and semi-technical scale on the UBA's experimental field in Berlin to test some worst case scenarios for the reliability of microcystin elimination during sand passage. Experiments were conducted with virgin sand (no previous contact to MCYST) and high filtration rates as well as under anaerobic conditions. The results show that the greatest problem for MCYST elimination can be found under anaerobic conditions as degradation is not complete and may lead to harmful residual concentrations.

Keywords

Microcystins, underground passage, worst case scenarios, field scale experiments.

INTRODUCTION

Cyanobacterial toxins are substances with high acute and chronic toxicity produced by cyanobacteria or 'blue-green-algae' that can be observed in surface waters world wide. In case of cyanobacterial blooms that often develop in eutrophic water bodies, high concentrations of cyanobacterial toxins occur, with microcystins (MCYST) being the group of substances most frequently found. Under normal conditions over 90% of the toxins are contained within the cells. There have however been reports on high extra-cellular concentrations e.g. in case of aging populations and/or sudden cell lysis.

Passage through biologically active strata, mostly sand, has shown to be effective in eliminating cyanobacterial cells as well as extracellular microcystins from surface water in many cases (Lahti and Hiisvirta, 1989; Sherman et al., 1995; Grützmacher et al., 2002). As aging populations accumulating on sediments may be assumed to show increased lysis and toxin release, an assessment of the reliability of the elimination of dissolved toxin for drinking water production is important. Elimination has to be secure under a variety of conditions met in the field. For this reason different worst case scenarios were simulated within the interdisciplinary NASRI (Natural and Artificial Systems for Recharge and Infiltration) research project dealing with river bank filtration processes. The aim was to identify the basic conditions under which sand and sediment passage can securely eliminate microcystins from surface water so that no further drinking water treatment has to take place.

The aim of a first series of experiments conducted in 2003 on a slow sand filter was to test whether a combination

of 3 worst-case conditions (SSF2: virgin sand that had no previous contact to MCYST, missing colmation layer and with 2.4 m/d high filtration velocities) leads to lower removal rates than experiments carried out with lower filtration velocities (SSF5: 1.2 m/d and SSF6: 0.6 m/d) and clogging. The second series of experiments conducted in 2003 and 2004 on semi-technical scale enclosures were carried out under aerobic and anaerobic conditions and had the aim to show the influence of redox conditions on MCYST degradation.

METHODS

Experimental methods

- *SSF experiments*

The experiments were conducted on a technical scale slow sand filter (SSF) on the UBA's (German Federal Environmental Agency) experimental field for simulation of underground passage (Bartel and Grützmacher, 2002).

The MCYST applied was extracted from a mass culture of *Planktothrix agardhii* HUB 076 by centrifugation and freeze thawing in order to release the mainly cell-bound, highly water soluble microcystins. The freeze thawed extract was homogenized and then centrifuged to remove the cell debris and stored frozen.

In preparation of the experiments the flow rate through the SSF was adjusted to the desired amount, corresponding to filtration velocities of 2.4 m/d, 1.2 m/d and 0.6 m/d. The tracer and the MCYST were applied by spraying them evenly across the water reservoir with a hose from a barrel containing the concentrated substances diluted with 100 L of tap water. The tracer applied was sodium chloride (NaCl) so that the sampling intensity in the different sampling points could be addapted by observing the electrical conductivity (EC). Care was taken, not to raise the electrical conductivity by more than 10% (Grützmacher et al., in press (a)). The resulting MCYST concentrations in the water reservoir amounted to 10 µg/L ± 2 µg/L.

- *Enclosure experiments*

Enclosure experiments were conducted in semi-technical scale columns for simulation of underground passage with an overlying water reservoir and a filter area of 1 m^2 (see Grützmacher et al. in press (a) for details). Samples were taken from the water reservoir, from 40 cm and 80 cm depth and from the effluent (after 100 cm of sand passage).

One experiment (E5) was conducted under strictly aerobic conditions (oxygen was detected in the effluent at concentrations of 11.7 mg/L ± 0.3 mg/L or 98% ± 2%). For the second experiment (E9) anaerobic conditions were induced by adding biodegradable DOC (acetic acid) continuously with a resulting concentration of 0.3 mmol/L additional DOC (resulting DOC concentration in water reservoir: 9.6 mg/L). After 4 days of continuous dosing oxygen could not be detected in 40 cm depth, after 9 days, no more oxygen was found in the effluent. After 3 weeks the redoxpotential (E_H) had dropped to values < 0 mV in all 3 sampling points and iron and manganese reduction was observed, thus implying strictly anaerobic conditions after 40 cm of sand passage. Due to nitrate concentrations below detection limit in the inlet (< 0.1 mg/L) denitrification not was observed.

The enclosure experiments were carried out similar to those on the slow sand filter, by applying a pulse of MCYST obtained from the mass culture of *Planktothrix agardhii* together with NaCl as a tracer to the water reservoir. Samples were taken in regular intervals and analysed for MCYST by ELISA.

Analytical methods

Microcystin analysis was carried out by ELISA (Enzyme-Linked ImmunoSorbent Assay) and HPLC (High Performance Liquid Chromatography) with a photodiode array detector. Usually samples were first tested for their microcystin content using the ELISA and selected ones subsequently analyzed by HPLC to verify the results and distinguish the different microcystin variants.

- *Sample preparation*

The water samples for analysis of extracellular microcystins were filtered (RC 55, pore size 0.45 µm) and stored deep frozen (−20°C). After thawing they were either analyzed directly (ELISA) or enriched by solid phase extraction (SPE) over C18-cartriges.

- *ELISA*

The ELISA used was a generic microcystin immunoassay based on monoclonal antibodies against the unusual Adda group characteristic for microcystins, nodularins and certain peptide fragments (Zeck et al., 2002). The measuring range lies between 0.1 µg/L and 1.0 µg/L. Samples from the water reservoir during the first phase of the experiments had to be diluted with deionized water. Each value was determined as the average of two parallels taken from the same sample.

- *HPLC*

After C18-SPE (see above) the microcystin variants were analysed by HPLC/photodiode array detection and identified by means of their characteristic UV-spectra (Lawton et al., 1994). Depending on the water quality the detection limits ranged from > 1 µg/L to 0.05 µg/L.

RESULTS AND DISCUSSION

SSF experiment with high filtration rate and virgin sand

The relative concentrations of the main MCYST variant (demethylated MCYST-RR) applied during the SSF-experiment related to the initial maximum of 10.5 µg/L as well as the calculated dilution curve (confirmed by two tracer tests) are shown in Figure 1. This data confirms that this MCYST-variant is not degraded in the water reservoir during the mean residence time (half life of exponential input curve) of about 3 hours.

In total the MCYST recovered in the effluent amounted up to 24% of the total MCYST applied (356 mg). The curve shown in Figure 2 describes the modelled breakthrough (VCXTFIT, Nützmann et al., in press) with a retardation coefficient of 2.6 and a degradation rate of 0.17 h^{-1} (which corresponds to a half life of 4,2 h).

Table 1 shows the retardation coefficients and degradation rates observed during all three SSF experiments. Surprisingly experiment SSF2 with the highest filtration rate, virgin sand and virtually no clogging shows the highest retardation coefficient and degradation rate. This might be due to reactions (e.g. irreversible sorption) that take place only during first contact with MCYST.
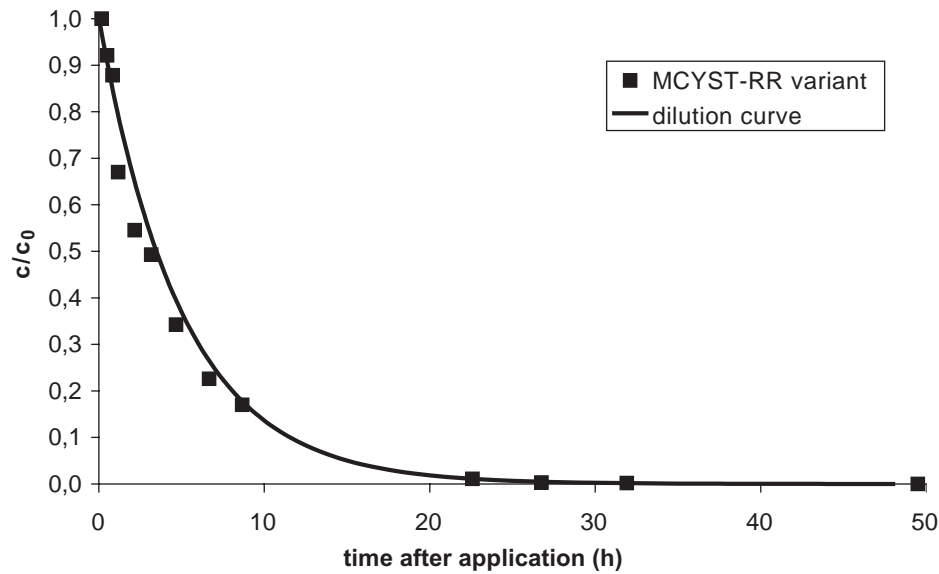
Figure 1. Concentration of demethylated MCYST-RR (main MCYST variant of mass culture) in the water reservoir during the slow sand filter experiment
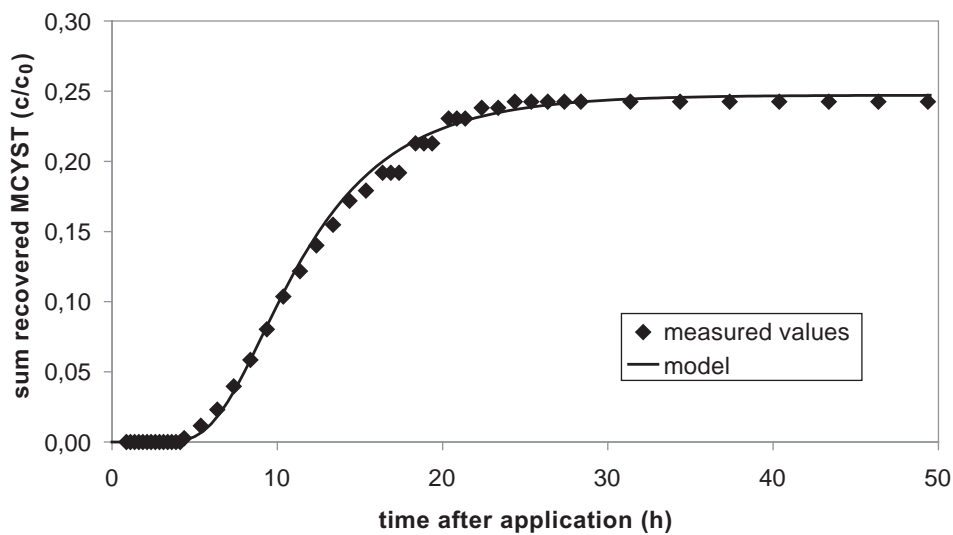
Figure 2. Sum of recovered MCYST (all variants) in the effluent during the SSF-experiment (model: R = 2,6; λ = 0,17 h^{-1})

Table 1. Retardation coefficients (R) and degradation rates (λ) obtained by modelling slow sand filter experiments with different filtration velocities and clogging situations

Experiment #	Filtration velocity [m/d]	R	λ [h^{-1}]	Remarks
SSF2	2.4	2.6	0.17	virgin sand, no clogging
SSF5	1.2	1.3	0.04	some clogging
SSF6	0.6	1.5	0.05	clogged

Enclosure experiment under aerobic and anaerobic conditions

The maximum MCYST concentrations measured by ELISA in the four sampling points during the aerobic (E5) and the anaerobic (E9) experiment in relation to the maximum input concentration (E5: 28.6 µg/L and E9: 10.3 µg/L) are shown in Figure 3. Due to clogging during the anaerobic experiment, the residence times for 80 cm and the effluent were much higher than during the aerobic experiment. However MCYST reduction in the anaerobic experiment reached only 60% maximum in 80 cm depth with no further reduction in the following 20 cm of filter material. The degradation rates calculated by fitting exponential decay curves to the measured values amount to 0.235 h^{-1} (aerobic) and 0.22 h^{-1} (anaerobic), thus showing no substantial difference. The main difference is the residual of 3.8 µg/L (35%) under anaerobic conditions that does not seem to be degraded any further after 12 h contact time.
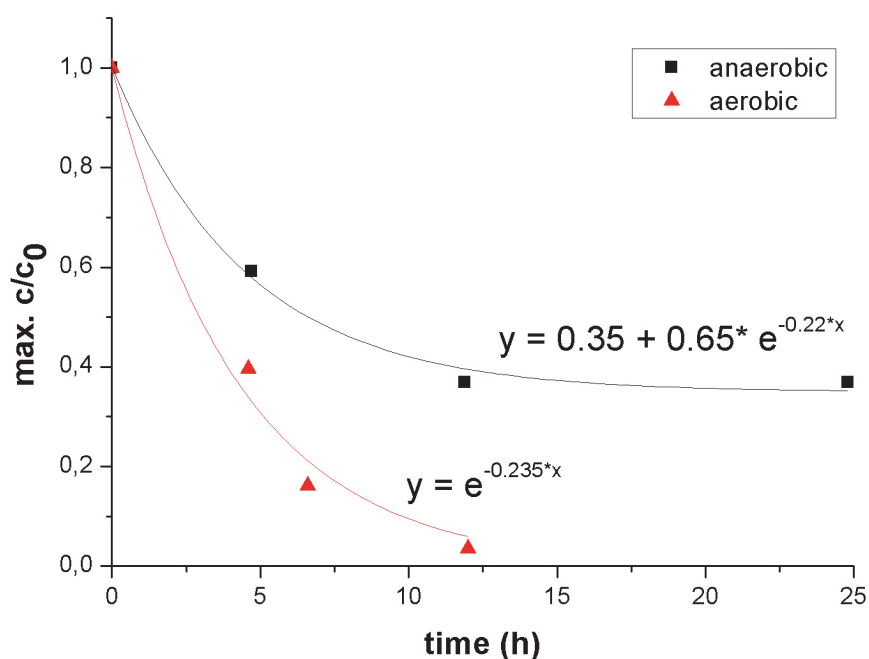
Figure 3. Maximum MCYST concentrations (measured by ELISA)
during the aerobic and anaerobic enclosure experiment

Indications of reduced biodegradation of MCYST und anaerobic conditions have so far been reported largely from laboratory experiments (Grützmacher et al., in press (b); Holst et al., 2002). It was therefore important to verify this in nearly natural settings and to show that in spite of an aerobic zone (which is present in most cases where surface water infiltrates into the aquifer), MCYST is not degraded completely, thus leading to a residual concentration that remains in the water.

CONCLUSIONS

Two worst case scenarios for MCYST elimination during underground passage were tested in technical and semi-technical scale experiments. The first scenario with conditions assumed to be quite unfavourable for degradation, i.e. virgin sand and high flow velocity, did not show reduced elimination of MCYST compared with the other scenarios tested (preconditioned sand, lower flow velocities). In contrast, under the presumably unfavorable conditions, higher retardation and degradation was observed. For the setting simulated (aerobic sand passage) only a few days (< 5) are sufficient to reduce maximum extracellular MCYST concentrations of 100 µg/L to values lower than the provisional WHO guideline value of 1 µg/L MCYST-LR (WHO 1998).

The second scenario tested was the simulation of anaerobic conditions that are likely to be met in the field during and after a cyanobacterial bloom, due to high organic carbon concentrations in the surface water. After an initial reduction of about 65% of the maximum input concentration no further degradation was observed under anaerobic conditions between 12 and 25 h contact time. Under aerobic conditions, in contrast, MCYST was reduced to values near the detection limit after 12 h contact time. Holst et al. (2003) carried out laboratory batch experiments on MCYST degradation under anaerobic conditions with addition of nitrate and glucose to stimulate denitrifying bacteria. The results showed that MCYST was not degraded in the sterile experiment and only little degradation was observed in the series without additional nutrients (to about 70% of the initial value within one day and then no further decline). In the experiments with additional nutrients, MCYST was degraded to less than 20% of the initial value within one day. This implies that under anoxic conditions MCYST degradation is limited to denitrifying, high nutrient conditions. Thus in our experiments the lack of nitrate might have been the reason for incomplete MCYST degradation as concentrations measured in the water reservoir were already below the detection limit.

ACKNOWLEDGEMENTS

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On the behaviour of microcystins in saturated porous medium

G. Grützmacher, G. Wessel, H. Bartel,
I. Chorus and E. Holzbecher

Abstract

Microcystins (MCYST) are a group of toxic substances produced by cyanobacteria ('blue-green-algae'). In case of cyanobacterial blooms microcystin concentrations in surface waters may reach values far above the value proposed as provisional guideline for drinking water by the WHO of 1 µg/L for MCYST-LR. For drinking water production via underground passage it is therefore necessary to ensure removal to a large extent.

For this reason experiments with extracellular microcystins were conducted in the laboratory as well as in a natural setting on the UBA's (German Federal Environmental Agency) experimental field for simulation of underground passage.

Laboratory batch experiments showed that adsorption of microcystins can be neglected in sandy material ($k_d < 1$ cm³/g). Batch and column experiments identified biodegradation as the predominant elimination process in these sediments. The degradation rates derived from laboratory column experiments as well as semi-technical scale enclosure experiments varied between 0.2 d^{-1} and 18 d^{-1}. In the worst case this means a half life of 2.8 days, so that under aerobic conditions contact times of several days should be sufficient to eliminate MCYST to an extent safe for use as drinking water.

Keywords

Degradation rates, field scale experiments, k_d-values, laboratory experiments, microcystins, underground passage.

INTRODUCTION

Microcystins (MCYST) are a group of structurally similar toxic substances produced by cyanobacteria or 'blue-green-algae'. They occur frequently in cyanobacterial blooms which can be observed in surface waters world wide (Bartram et al. 1999). When using these surface waters for drinking water production via bank filtration or artificial recharge elimination of microcystins by underground passage has to be ensured as the WHO has proposed a provisional guideline value for drinking water of 1 µg/L for MCYST-LR (WHO 1998).

Microcystins are subject to different processes in the porous medium, such as degradation, sorption and desorption. The relevance of the different processes depends on various conditions: on the properties of the porous medium (grain size distribution, organic content), as well as on the redox state (aerobic/anoxic/anaerobic), the pH, temperature, etc..

Within different interdisciplinary research projects dealing with river bank filtration processes several experiments were conducted. The aim of the paper is to present the outcome of the different measurements and to compare the main sorption and degradation parameters.

METHODS

Experimental methods

Experiments were carried out in laboratory batch- and column systems (Grützmacher et al. in prep.), as well as in technical scale enclosures and slow sand filters under different clogging conditions (Grützmacher et al. in press). Enclosures are large scale columns (diameter: 1.13 m) embedded in a slow sand filter with a water reservoir of 40 cm and a sand filter of 1 m depth. This arrangement was chosen in order to expose the column to the most close-to-real conditions, the only difference being the smaller size of the filter bed. Extra-cellular microcystins were applied in realistic concentrations of about 1 to 10 µg/L. The microcystins were either prepared from commercially available standards (MCYST-LR, -YR and -RR) or obtained by freeze-thawing a concentrate (aqueous extract) of a mass culture of *Planktothrix agardhii* HUB 076 that is currently run by the UBA in Berlin. The concentrate was subsequently centrifuged and the supernatant deep frozen for further conservation. The microcystin concentrations in the undiluted extract reached 50 mg/L, the DOC amounted up to 4 g/L. For the experiments the extract was diluted about 1:5,000. MCYST concentrations in the samples from the laboratory experiments were usually determined by ELISA, those taken during enclosure experiments by HPLC (see below).

Analytical methods

Microcystin analyses were carried out by ELISA (Enzyme-Linked ImmunoSorbent Assay) and HPLC (High Performance Liquid Chromatography) with a photodiode array detector. Usually samples were first tested on their microcystin content using the ELISA and selected ones subsequently analyzed by HPLC to verify the results and distinguish the different microcystin variants.

• *Sample preparation*

The water samples for analysis of total microcystins were deep frozen and thawed in order to release cell-bound microcystins, filtered by membrane filters (RC 55, pore size 0.45 µm) and either analyzed directly (ELISA) or enriched by solid phase extraction (SPE) over C18-cartridges and additional silica clean-up according to Tsuji et al. (1994). For analysing the dissolved microcystins samples were filtered directly after sampling and then deep frozen for subsequent analysis by ELISA or HPLC.

• *ELISA*

The ELISA used was the EnviroGard Plate Kit (offered by Coring System Diagnostix GmbH, Germany) with a measuring range between 0.1 µg/L and 1.6 µg/L. Samples from the water reservoir during the first phase of the experiments had to be diluted with deionized water. Each value was determined as the average of two parallels taken from the same sample. The preparation of the plates was done according to the instructions by the producer.

• *HPLC*

After C18-SPE (see above) and additional silica cleanup the microcystin variants were analysed by HPLC/photodiode array detection and identified by means of their characteristic UV-spectra (Lawton et al. 1994). Depending on the water quality the detection limits ranged from 0.05 µg/L to > 1 µg/L.

Numerical methods

Parameter estimation for tracers was performed using Visual CXTFIT (Nützmann et al. in press), a graphical user interface for the CXTFIT code, developed by Toride et al. (1995). Retardations and degradation rates were

estimated using basic MATLAB (2002) package in connection with the optimization toolbox. The numerical *pdepe*-solver, included in MATLAB, was used for direct modelling within the estimation procedure.

RESULTS

Adsorption isotherms

Batch experiments for determination of adsorption parameters were carried out with different sediments (material from a slow sand filter, aquifer sediment as well as pure quartz sand) and MCYST-variants (MCYST-LR, -YR and -RR). The resulting k_d-values are presented in Table 1. Although slight differences in adsorption can be observed depending on the MCYST-variant (MCYST-RR adsorbing more readily than MCYST-YR and -LR), the most important parameter seems to be the texture of the sediment as the aquifer sediment contains a relatively high portion of organic substance as well as clay and silt.

Table 1. k_d-values determined in batch experiments and important sediment properties (LOI (loss on ignition), silt and clay-content)

Sediment	LOI (%)	Silt and clay (%)	MCYST-variant	k_d-value (cm³/g)
			MCYST-LR	0.25
Slow sand filter material	0.2	< 0.1	MCYST-YR	0.22
			MCYST-RR	0.85
Aquifer sediment	0.8	max. 3	MCYST-LR	11.6
Quartz sand	< 0.05	0	MCYST-LR	0.08

Degradation experiments

Parallel sterile (autoclaved) and non sterile batch experiments carried out with natural filter sand (SSF-material) and surface water showed virtually no decrease of MCYST-concentration in the sterile experiment. Biodegradation is therefore the most important elimination process for MCYST in contact with sandy sediment (Figure 1).

For further quantification of degradation rates closed loop column experiments were conducted with the same material (SSF-material) and different MCYST-variants under aerobic conditions (Figure 2). The results showed similar degradation rates between 1.33 d^{-1} and 1.74 d^{-1} (the half lives correspondingly amount to around 10 h).

In order to test MCYST degradation under field conditions an aqueous extract from the mass culture of *Planktothrix agardhii* was applied to the water reservoir of an enclosure on the UBA's experimental field. Here sediment passage can be simulated in a semi-technical scale (filter area: 1 m², filter depth: 1 m). Samples were taken from the water reservoir, from 20 cm, 40 cm, 60 cm, 80 cm depth and from the effluent after 100 cm sand passage. Simultaneously applied tracer (NaCl) gave values for pore velocity and dispersion coefficients (Grützmacher et al. in press). On the basis of these values retardation factors and degradation rates were calculated, that took the measured input concentrations of MCYST into account. An example for a modelled curve as well as the measured input and output values is given in Figure 3.
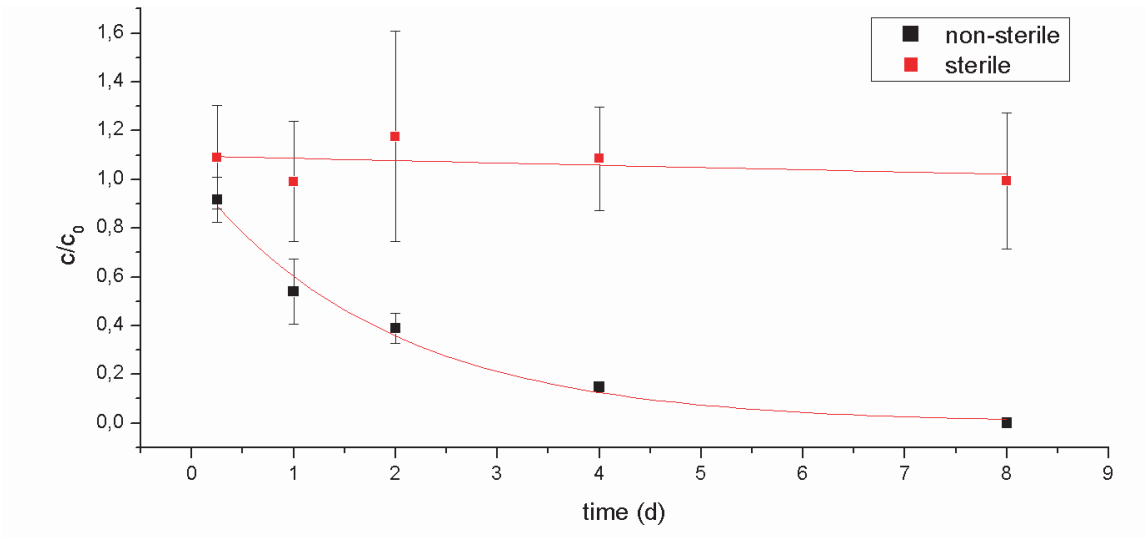
*Figure 1. Batch experiments with MCYST (in surface water with filter sand)
under sterile and non-sterile conditions (average and rage of 2 parallels)*
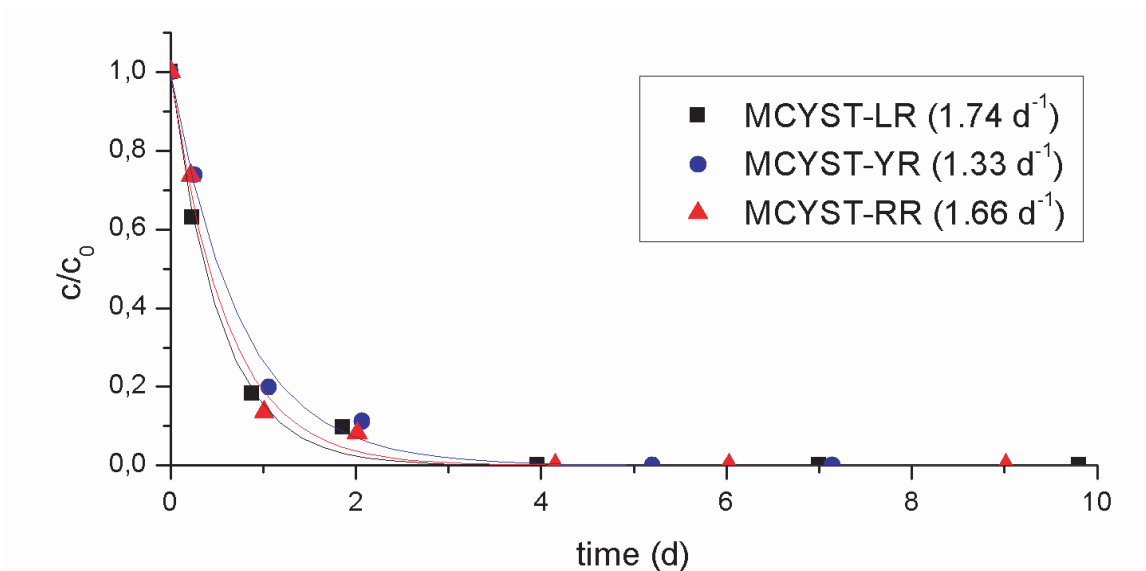
*Figure 2. Results of non-sterile closed loop column experiments
with different MCYST-variants (degradation rates in brackets)*

Two enclosure experiments were conducted on the same enclosure with an interval of one month, allowing some clogging to take place in the meantime. The hydraulic conductivity decreased about 11% during this time (from 1.8×10^{-5} m/s to 1.6×10^{-5} m/s, Grützmacher et al. in press). The results of the modelling of the two experiments with MCYST are given in Table 2. The highest retardation and degradation rates were observed in the uppermost centimetres in both experiments. Both retardation and degradation tend to decrease with depth. There is, however, a zone of lower retardation (E3) and lower degradation (E2) between 20 cm and 40 cm. This might be due to varying saturation, as unsaturated conditions were observed with proceeding clogging. The last zone between 80 cm and the outlet also shows surprisingly high retardation coefficients. This may be due to column-end-effects arising from the gravel drainage layer.
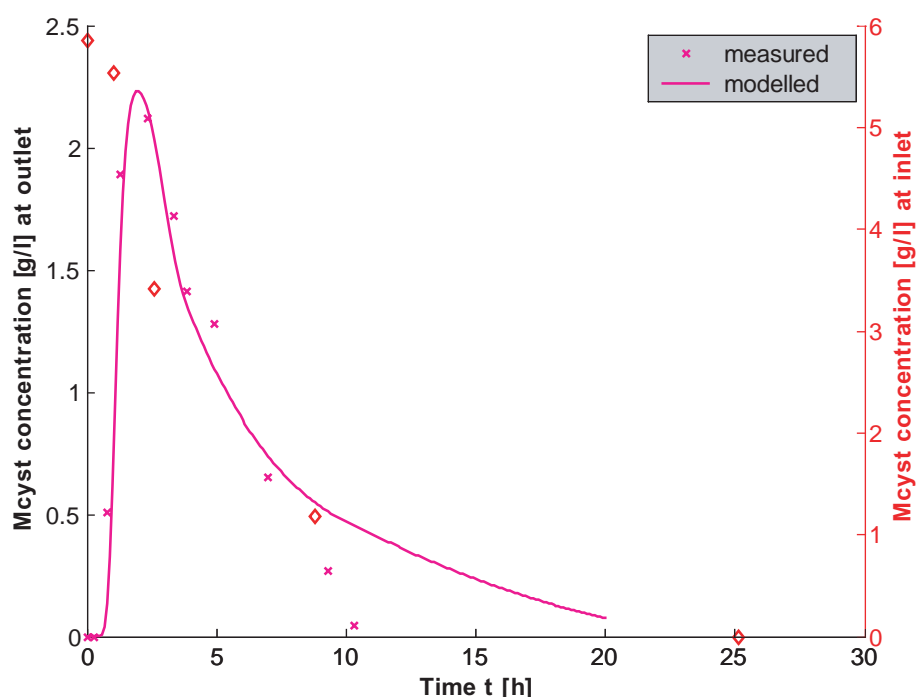
*Figure 3. Measured and modelled MCYST-concentrations in 20 cm depth
during enclosure experiment E2 (diamonds: MCYST-concentration in the inlet);
parameters for the given situation are: v = 0.32 m/h, α_L = 0.0063 m, R = 1.97, λ = 0.74 l/h*

*Table 2. Retardation coefficients and degradation rates calculated for extracellular MCYST
during two enclosure experiments E2 and E3*

Experiment	retardation coefficient (R)		degradation rate (λ) [1/d]	
	E2	E3	E2	E3
water reservoir -> 20 cm	1.97	1.76	17.8	13.0
20 cm -> 40 cm	1.9	1.25	5.76	7.44
40 cm -> 60 cm	1.0	1.36	18	4.56
60 cm -> 80 cm	1.0	1.14	13.0	0.24
80 cm -> outlet	1.58	1.3	1.68	1.68

In order to compare adsorption parameters obtained from laboratory batch experiments to those calculated on the basis of field enclosure experiments k_d-values were calculated on the basis of the retardation coefficients according to (1), with n_e ranging from 0.16 to 0.4 and ρ_s amounting to an average of 1.65 g/cm³.

$$k_d = \left(\frac{n_e}{\rho_s} \right) * \left(R - 1 \right)$$
Eqn. (1)

The resulting average k_d-value for experiments E2 and E3 amounted to 0,14 cm³/g and 0.07 cm³/g, respectively. This is less than 50% of the values obtained in the batch experiments. This might be due to the different ratio of water to sediment (batch experiment 1:1, enclosure experiment about 3:1).

The degradation rates observed during the laboratory experiments also differ from those calculated from the results of the enclosure experiments. They are, however, in the same order of magnitude and in some depths of the enclosure (e.g. 80 cm to outlet) they correspond well.

CONCLUSIONS

Under aerobic conditions biological degradation during sand passage was shown to be effective in eliminating extracellular microcystins. Contact times of only a few hours seem to be sufficient to reduce concentrations typical for cyanobacterial blooms in the water body to values below the WHO guideline value. Similar degradation rates were determined by Christoffersen et al. (2002), however for surface water without sediment contact. Batch experiments conducted by Holst et al. (2001) showed much slower degradation (half lives around 25 d). These experiments were however carried out with aquifer material without previous contact to MCYST and at very low temperatures (4°C).

Adsorption is only relevant in material with clay or silt content, which is supported by the findings of Miller (2000). If adsorption takes place it can lead to longer contact times which supports elimination by biological degradation. There are however indications of reduced degradation rates under anoxic and anaerobic conditions. These conditions are subject to further investigations that are currently being carried out.

Laboratory experiments can give an impression of which processes take place under certain, controlled conditions. However k_d-values and degradation rates calculated on the basis of laboratory experiments only have to be treated with caution. Before transferring these to the field it is suggested to conduct field observations or technical scale experiments for further verification.

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Simulating bank filtration and artificial recharge on a technical scale

Gesche Grützmacher, Hartmut Bartel and Bernd Wiese

Abstract

The UBA's experimental field on the outskirts of Berlin offers a unique possibility of simulating bank filtration, artificial recharge and slow sand filtration on a technical scale.

The site consists of a storage reservoir (pond) with an adjacent artificial aquifer consisting of sand and gravel. Additionally the surface water can be conducted into 4 infiltration basins (two slow sand filters and two aquifer infiltration ponds). Three enclosures as well as large scale columns can be used for shorter and longer term simulation of groundwater transport. The whole site is separated from the surrounding aquifer by a layer of clay. A variety of physico-chemical parameters can be measured continuously and observed online.

The travel times for the bank filtration passage determined by tracer experiments range from a few days to a maximum of 3 weeks. In the enclosures, infiltration ponds and large scale columns contact time can be varied between a few hours up to 3 months.

Keywords

Artificial recharge, bank filtration, experiments, slow sand filtration, technical scale.

INTRODUCTION

The UBA (Umweltbundesamt or Federal Environmental Agency of Germany) is running an experimental field on the outskirts of Berlin. Part of the site consists of a storage pond with adjacent artificial aquifer and infiltration ponds that offer a unique possibility of simulating bank filtration, artificial recharge and slow sand filtration on a technical scale.

In order to characterize groundwater flow in the system many tracer tests were conducted revealing the hydraulic properties of the facilities as well as temporal variations in hydraulic conductivity due to clogging.

METHODS

Experimental facilites

The site consists of a storage reservoir (pond) with a water volume of about 3,500 m^3 and an adjacent artificial aquifer consisting of fine and coarse gravel (Figures 1 and 2). Additionally the surface water of the pond can be conducted into four infiltration basins each with a square basal surface of 72 m^2. Two of these basins are sealed by concrete at the bottom with a sand depth of 0.8 m (here water is collected by 3 parallel drainages at the bottom), two have direct contact to the underlying sediments and can be operated as aquifer infiltration ponds. For smaller scale experiments three enclosures with a surface area of 1 m^2 and medium grained sand filling are installed in one of the infiltration basins (Figure 3). The whole site is separated from the surrounding aquifer by a layer of clay thus

forming an independently operable hydraulic system so that experiments with toxic substances can be carried out without adverse effects on the environment.

A variety of physico-chemical parameters (e.g. pH, temperature, oxygen, redoxpotential, electrical conductivity (EC), TOC, total bound nitrogen (TN_b) as well as fluorescence for determination of algal biomass) can be measured continuously and observed online.
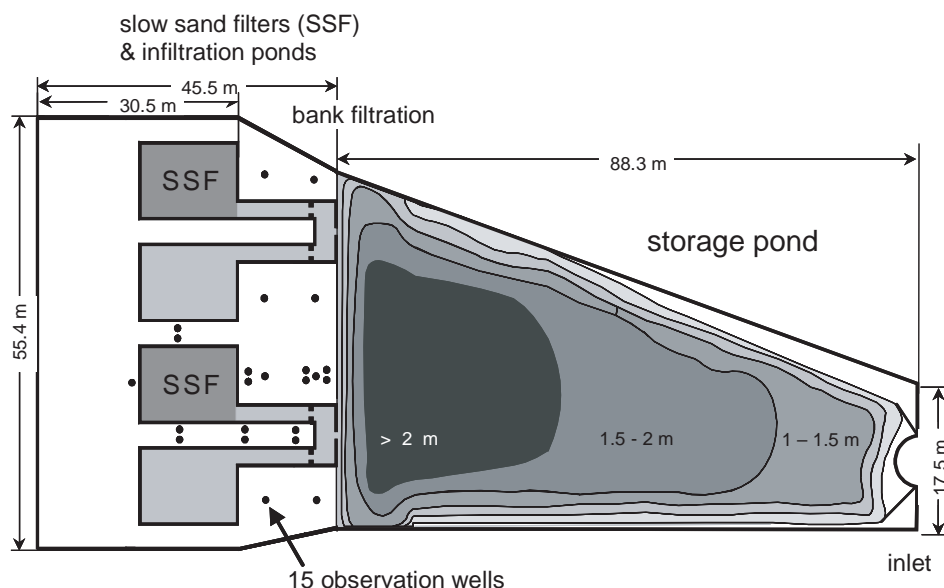
Figure 1. Map of the storage pond system (modified from Bartel & Grützmacher 2002)
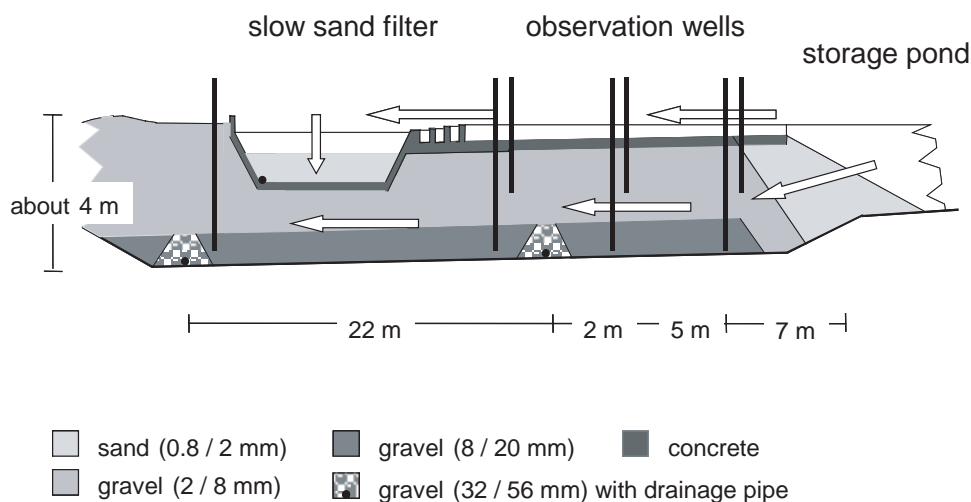
Figure 2. Cross-section of the artificial aquifer with the overlying storage pond

The sand filling of the slow sand filter and the enclosures was replaced in autumn 2002. The k_f-value calculated on the basis of the grain size distribution according to Beyer (1964) is $7 * 10^{-4}$ m/s for the middle sand inside the slow sand filters and enclosures and $1*10^{-1}$ m/s for the gravel drainage layer. In March 2003 the basins were flooded again and experiments were conducted until November 2003 (Table 1). In the slow sand filter the clogging layer was not removed throughout 2003 thus giving experimental results for growing clogging conditions. In April 2003, however, between experiment SSF1 and SSF3 bacterial slide holders were installed in one part of the filter with substantial disturbance of the filter sand up to a depth of about 30 cm.

The hydraulic conductivity (k_f) of the SSF was calculated according to (1) with v_f (filtration velocity = average flow rate during the experiment divided by the average filter area of 60 m^2) and i (gradient = surface water reservoir minus height of outlet divided by 0.8 m filter depth).

$$k_f \ = \ v_f / i \qquad (1)$$

The flow in the enclosures had to be interrupted several times due to technical problems. During the experiments hydraulic conductivity was determined according to (1) with an average filter area of 1 m^2 and i = p (bar) $*$ 10.197 (m/bar) (p measured in front of the pump at the effluent). Pumping at Enclosure III was commenced shortly before the first experiment in July 2003 after a stagnant period of about 3 months. Enclosure II was used only in November 2003 after having remained flooded with interrupted flow for 8 months. After experiment No. E4 the upper layer of 0.5 cm of algae and debris was removed in order to observe the effect of this layer on substance removal.
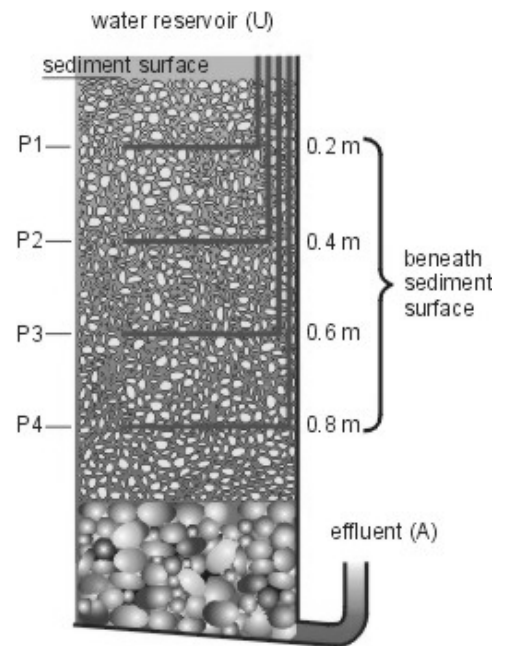
Figure 3. Schematical cross-section of an enclosure

Table 1. Summary of parameters for the tracer tests in 2003

Experiment No.	Location	Date	Filtration velocity [m/d]
SSF1	Slow sand filter	19.03.03	2.4
SSF3	Slow sand filter	23.04.03	2.58
SSF5	Slow sand filter	17.06.03	1.43
SSF6	Slow sand filter	19.11.03	0.59
E2	Enclosure III	05.08.03	1.25
E3	Enclosure III	09.09.03	1.12
E4	Enclosure II	11.11.03	1.09
E5	Enclosure II	25.11.03	1.22

Tracer tests

Parallel to each experiment simulating substance removal during underground passage on the artificial aquifer system tracer tests were conducted for characterization of groundwater flow. Usually chloride was used as a tracer by adding sodium chloride to the infiltrating water and measuring the electrical conductivity in the different sampling ports and effluents. Care was taken to raise the electrical conductivity not more than 10% above the background level (950 µS/cm in average) to avoid induction of hydrochemical reactions. The hydrochemical analyses showed this was successful with exception to a slight cation exchange reaction during tracer passage.

The tracer was added to the water reservoir in a pulsed application and pumps were installed in the water body throughout the experiment in order to reach a homogenous distribution. Subsequently inflowing water diluted the maximum tracer concentration in a rate depending on the adjusted flow rate thus giving an exponentially decreasing input concentration. The fitted first order exponential function was used as input for modelling using the pro-

gram Visual CXTFIT (Nützmann et al., in press). By inverse modelling the values of dispersion and pore velocity could be determined serving as a basis for modelling the behaviour of reactive substances added. The sampling ports in the enclosures and slow sand filters were connected to peristaltic pumps and run continuously with low flow rates (60 to 600 ml/h).

The artificial aquifer as a whole was subject to one tracer experiment using bromide and a Gd-DTPA complex (Wiese et al., submitted). The tracers were applied to the storage pond by pumping a highly concentrated solution through a perforated hose that was arranged parallel to the aquifer's bank. A current induced by air injection through a perforated hose next to the tracer hose provided mixing.

Samples from the 15 observation wells were taken with a peristaltic pump, ensuring the water in the tube was exchanged at least once. The data was modelled with Modflow-MT3DMS to obtain hydraulic properties of the 3-D structure and Visual CXTFIT to obtain transport properties of each borehole.

Analytical methods

Electrical conductivity was determined using a WTW conductometer (measuring range 1 to 100,000 µS/cm, ± 1 %). Bromide concentrations were measured with an Ion chromatograph DX 500 (Dionex Coop.) according to DIN EN ISO 10304-1/2. Gadolinium was analysed with a method described by Bau and Dulski (1996).

RESULTS AND DISCUSSION

Tracer tests on slow sand filters and enclosures

Figure 4 gives an example of the conductivities measured in the effluent as well as the curves modelled by using Visual CXTFIT. Table 1 lists the modelled pore velocities, dispersion lengths (α_L) as well as the filter resistance for each of the experiments conducted on the slow sand filters and enclosures.

The slow sand filter experiments were conducted with three different filtration velocities (about 2.5 m/d, 1.4 m/d and 0.6 m/d). Therefore pore velocities changed correspondingly. Porosities are between 0.357 and 0.385. The
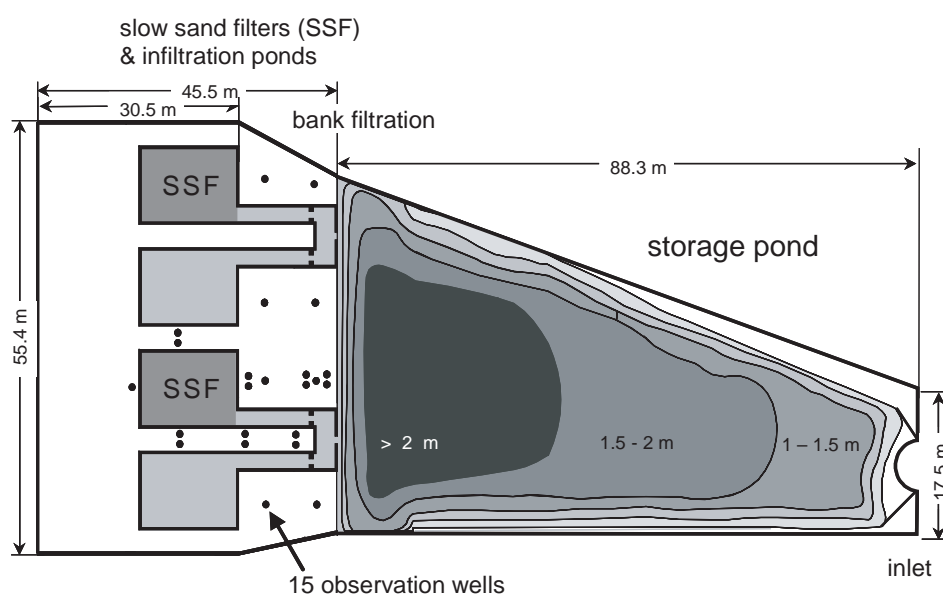
Figure 4. Measured electrical conductivity and modelled output function for tracer experiment No. E2 (v_f = 1.2 m/d)

Table 2. Pore velocities, dispersion lengths, effective pore volumes and hydraulic conductivities during the tracer experiments on the enclosures and slow sand filters

Experiment No.	Filtration velocity [m/d]	Pore velocity [m/d]	Dispersion length (a_L) [m]	Eff. Pore-volume (n_e)	Hydraulic conductivity [m/s]
SSF1	2.4	6.24	0.012	0.385	n.d.
SSF3	2.58	6.48	0.05	0.37	n.d.
SSF5	1.43	3.12	0.023	0.385	$1.4*10^{-4}$
SSF6	0.59	1.68	0.01	0.357	$7.4*10^{-6}$
E2	1.25	3.36	0.011	0.37	$1.8*10^{-5}$
E3	1.12	2.76	0.013	0.40	$1.6*10^{-5}$
E4	1.09	2.88	0.016	0.379	$1.5*10^{-5}$
E5	1.22	3.07	0.023	0.398	$1.7*10^{-5}$

n.d. not determined.

reduction of the dispersion length (α_L) between SSF3, SSF5 and SSF 6 can also be explained by the decline of flow velocities. A substantial disturbance of one part of the filter due to installation of microbial slide holders before SSF3 may be the reason for the rise in dispersion length between SSF1 and SSF3 with otherwise similar conditions. As to be expected average hydraulic conductivities dropped during 2003 due to clogging.

In the enclosure experiments changes in hydraulic conductivities were not as obvious. This may be due to the fact that there had been periods of little or no flow before the experiments so there was little time for a clogging layer to develop. The overall values are, however, distinctively smaller than those observed during the starting phase in the slow sand filter, which also may be due to phases with little or no flow, during which anaerobic conditions are likely to occur. Subsequent flushing with aerobic water may have lead to precipitation e.g. of ferrous oxides thus leading to clogging inside the filter bed. Nevertheless a slight reduction of hydraulic conductivity can be observed in those cases in which a clogging layer had been given time to develop (E3 and E4).

Tracer test on the bank filtration area

The known subsurface structure (Figure 2) could be confirmed and limits for the parameters hydraulic conductivity and porosity for the different structural areas could be determined (see Table 3) using data obtained for smaller scale by 15 sampling wells and for the whole system by the two drainages. Average aquifer parameters were deter-

Table 3. Hydraulic conductivity, eff. porosity (n_e) and dispersivity (α_L) for different structural areas of the bank filtration site

	Whole system			Sampling wells		
	k_f [m/s]	n_e	α_L [m]	k_f [m/s]	n_e	α_L [m]
Bank	$7.00e^{-4}$	~ 0.3	n.d.	~ $7e^{-4}$	~ 0.3	n.d.
Upper layer	> 0.01	~ 0.3	~ 0.5	$1.5e^{-3}$ to $3e^{-2}$	~ 0.3	< 0.7
Lower layer	> 0.2	0.4	< 0.5	> 0.2	0.4	< 0.1

n.d.: not determined.

mined using the data from the drainage pipes. K_f values could be determined quite well, porosities however have a much smaller band of variability and due to cross correlation with the k_f -values literature values are taken for the bank and the upper layer. As the flow of the lower layer is known, here porosities could be determined well. Structural dispersivity of the aquifer is smaller than dispersion induced by sampling

For each sampling well pore velocity, residence time and dispersivitiy were determined, (not presented here) which enables to separate hydraulic and chemical effects of transport of reactive substances.

CONCLUSIONS

Tracer experiments were carried out in different experimental settings on the storage pond system of the UBA's experimental field. Observed hydraulic conductivities range from 0.2 m/s to $7 *10^{-6}$ m/s. Clogging lead to a substantial reduction of conductivities only after continuous flushing of the sand for more than one month.

The storage pond system therefore offers the possibility to simulate sand passage as used for bank filtration and artificial recharge under realistic conditions (hydraulic conductivities are similar to those met in the field and different clogging situations can also be simulated). Tracer experiments, however, need to be carried out regularly for accurate definition of hydraulic flow as conditions may change depending on the clogging situation.

ACKNOWLEDGEMENTS

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Separation of *Cryptosporidium* oocysts in different filter sands during slow sand filtration

Ulrike Hütter and Gudrun Preuss

Abstract

Cryptosporidium is a waterborne parasitic protozoan which can contaminate surface water. To guarantee the quality of drinking water produced during the treatment of surface water via artificial groundwater recharge, adequate removal of human pathogenic parasites has to be ensured. The aim of this study was to gain information about the performance of a double stage filtration (gravel pre-filter and slow sand filter) for *Cryptosporidium* removal. At a semi-technical test plant different operating stages close to practice were investigated with three different filter sands. In the applied surface water varying, but rather low oocysts concentrations were found. The semi-technical investigations proved a high efficiency of the gravel pre-filtration. The filter sands with a smaller effective particle size achieved a better reduction efficiency. Especially in the first filtrates in restart stages with higher filtration rates oocysts concentrations increased. The presence of *Clostridium perfringens* was no indicator for the presence of oocysts and vice versa. Suitable operating conditions of water works in combination with a comprehensive protection of the surface water against the entry of pathogens belong to the concept of a multi-barrier-system and guarantee the quality of drinking water.

Keywords

Clostridium perfringens, *Cryptosporidium*, pathogens, slow sand filtration, surface water.

INTRODUCTION

Cryptosporidium (diameter 4 to 6 µm) is a waterborne parasitic protozoan, which exists in either the free-swimming (trophozoite) form or the oocysts (dormant) form. *Cryptosporidium parvum* is recognized as a human pathogen which can cause severe diarrheal illness. The oocysts are shed in large quantities with the faeces of infected individuals. Via this route they are spread into the environment and can contaminate surface waters and groundwater (Gornik and Exner, 1997; Moulton-Hancock et al., 2000). The oocysts – as an environmentally stable lifeform – can survive for several months in a watery milieu. This form is resistant to the drinking water disinfection with chlorine. To guarantee the quality of drinking water produced during the treatment of surface water via artificial groundwater recharge, adequate removal of human pathogenic parasites has to be ensured. The experiments presented in this paper were carried out to investigate the *Cryptosporidium* oocysts reduction efficiency of different filter sands during drinking water production via gravel pre-filtration and slow sand filtration.

MATERIALS AND METHODS

In autumn 2000 the method of artificial groundwater recharge via gravel pre-filtration and slow sand filtration without subsequent underground passage was reproduced at a semi-technical test plant (Figure 1). The gravel pre-filter received the surface water (river Ruhr) with its natural inconstant oocysts concentrations; the created pre-filtrate was used as influent to the sand filters. The double stage filtration at the test plant simulated realistic field conditions. Only the thickness of the sand filter was low in order to investigate the possibilities of optimization of

operations. Three filter sands with different effective particle sizes (d_{10}) and particle-size distributions were used (Table 1). During three operating stages with defined conditions close to practice (start-up stage, supernatant stage, restart stage) sampling of surface water, pre-filtrate and filtrate (5 consecutive samples) was carried out. After start-up of the sand filter they were intermittently operated (2-day-rhythm). The supernatant phase was investigated after a filter running time of 3 to 4 weeks, and another 3 weeks later the restart phase. During the experiments microbiological parameters (e.g. *Cryptosporidium*, *C. perfringens*, total coliforms) and relevant hydrochemical parameters (e.g. pH factor, turbidity) were measured.
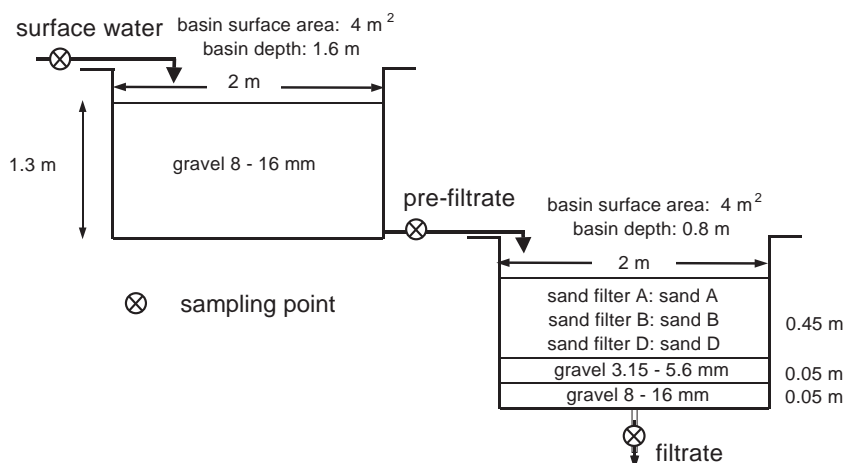
Figure 1. Scheme of the double stage filtration (cross section)

Table 1. Characteristics of the applied filter sands

	Sand A	*Sand B*	*Sand D*
Particle size at 10% sieve pass d_{10} [mm]	0,28	0,12	0,15
Particle size at 60% sieve pass d_{60} [mm]	0,68	0,42	0,40
Unconformity coefficient (d_{60}/d_{10}) [–]	2,43	3,50	2,67
Porosity [%]	37,5	32	34
Coefficient of hydraulic conductivity k_f [m/s]	$9,1 \times 10^{-4}$	$1,7 \times 10^{-4}$	$2,6 \times 10^{-4}$

Sampling procedures and determination of microbiological and chemical parameters were accomplished according to German standard methods (Drinking Water Ordinance, 2001). In addition, *E. coli* and total coliforms were detected on ENDO-Agar (DIFCO) after membrane filtration and incubation at 36°C (± 1°C) for 20 hours. The protozoan parasites *Cryptosporidium* and *Giardia* were determined according to the USEPA method 1623 (US-EPA, 1999) by using filter capsules (Envirochek®), immunomagnatic separation technique (IMS), fluorescent monoclonal antibodies (FITC) and DAPI staining. The fluorescent oocysts and cysts were enumerated by a laser scan cytometer (Chemunex) and confirmed by epifluorescence microscopy. In surface water the detection limit was 70,4% for *Cryptosporidium* oocysts and 94,1% for *Giardia* cysts. Results were calculated without the consideration of detection limits. This method does not distinguish between viable and non viable parasites.

MICROBIOLOGICAL CHARACTERIZATION OF THE SURFACE WATER

The chemical and microbiological conditions in the river Ruhr depend on regional and seasonal effects like e.g. rainfall and effluents from waste water treatment plants or agricultural areas. Investigations in the last years showed the permanent presence of protozoan parasites in the Ruhr. The results of some microbiological investigations between 2000 and 2003 are listed in Table 2. *Giardia* cysts were found in concentrations between 3.92

and 80.15 per L. The concentrations of *Cryptosporidium* oocysts were generally lower and ranged from 0.09 to 19.92 oocysts per L. The number of bacteria (DEV-Agar, 20°C) ranged from 650 to 112,000 CFU/mL. Coliforms were detected in concentrations between 2,500 and 38,000 CFU/100 mL. These results demonstrate the wide range of microbiological conditions in the river although DOC was rather constant with concentrations between 1.8 and 3.0 mg/L. However, for statistical correlations between the different microbiological parameters and the assessment of seasonal effects more data are needed. These sporadic investigations allow only a general view on the average water quality in the river Ruhr.

Table 2. The microbiological condition of surface water at various sampling locations in the Ruhr at Schwerte

Sample point	Oocysts /L	Cysts /L	CFU (20°C) /mL	Coliforms /100 mL	DOC mg/L
Ruhr Hengsen, 20.01.00	0.22	–	4,000	18,000	–
Ruhr Lake Hengsen, 17.04.01	19.92	80.15	4,900	26,000	3.0
Ruhr Lake Hengsen, 24.04.01	1.13	19.87	1,290	4,200	2.0
Ruhr Lake Hengsen, 06.01.03	0.09	16.18	3,100	2,500	–
Ruhr Villigst, 20.01.00	0.84	–	–	–	–
Ruhr Villigst, 03.09.01	2.71	3.92	650	6,800	3.0
Ruhr Villigst, 28.04.03	0.20	17.78	112,000	17,000	–
Ruhr Westhofen, 07.05.01	1.15	43.46	12,000	38,000	2.6
Ruhr Westhofen, 27.11.01	6.73	70.94	8,200	30,000	1.8
Ruhr Westhofen, 08.05.03	0.38	8.12	880	1,400	–

– = not investigated.

RESULTS AND DISCUSSION

During the investigations at the semi-technical test plant oocysts concentrations from 0.29 to 1.28 per L (mean value 0.70 per L) were detected in the surface water (river Ruhr). The measured concentrations in the **pre-filtrate** (= sand filter influent) laid between 0.03 and 0.28 oocysts per L (mean value 0.14 oocysts per L). These results confirm an obvious removal of oocysts in the gravel pre-filter (particle size 8 – 16 mm). The concentrations of *Cryptosporidium* oocysts, *Clostridium perfringens* and coliforms in the sampling points of sand filter A, B and D during the different operating stages are shown in the Figures 2, 3 and 4.

In the **start-up stage** (filtration rate 3.2 m/d) only in one filtrate of sand filter A a small amount of oocysts could be detected. The high amount of *Clostridium perfringens* in the filtrates of sand filter A exceeds in all samples the concentration in the influent. This effect is probably caused by the leaching of organisms which were brought in during the sand transport and storage. The samplings in the **supernatant stage** (filtration rate 2.4 m/d) were carried out after a filter running time of 12 to 15 days. In 10 of 15 filtrates oocysts concentrations between 0.01 to 0.03 per L were detected. Sand filter D (d_{10} = 0.15 mm) showed the fewest positive results.

After a filter running time of 22 to 25 days samplings were carried out in a **restart stage** (filtration rate 3.3 m/d). Positive results were observed in 7 of 15 filtrates. Whereas in sand filter B (d_{10} = 0.12 mm) no oocysts were

sand filter A (d_{10} = 0.28 mm)
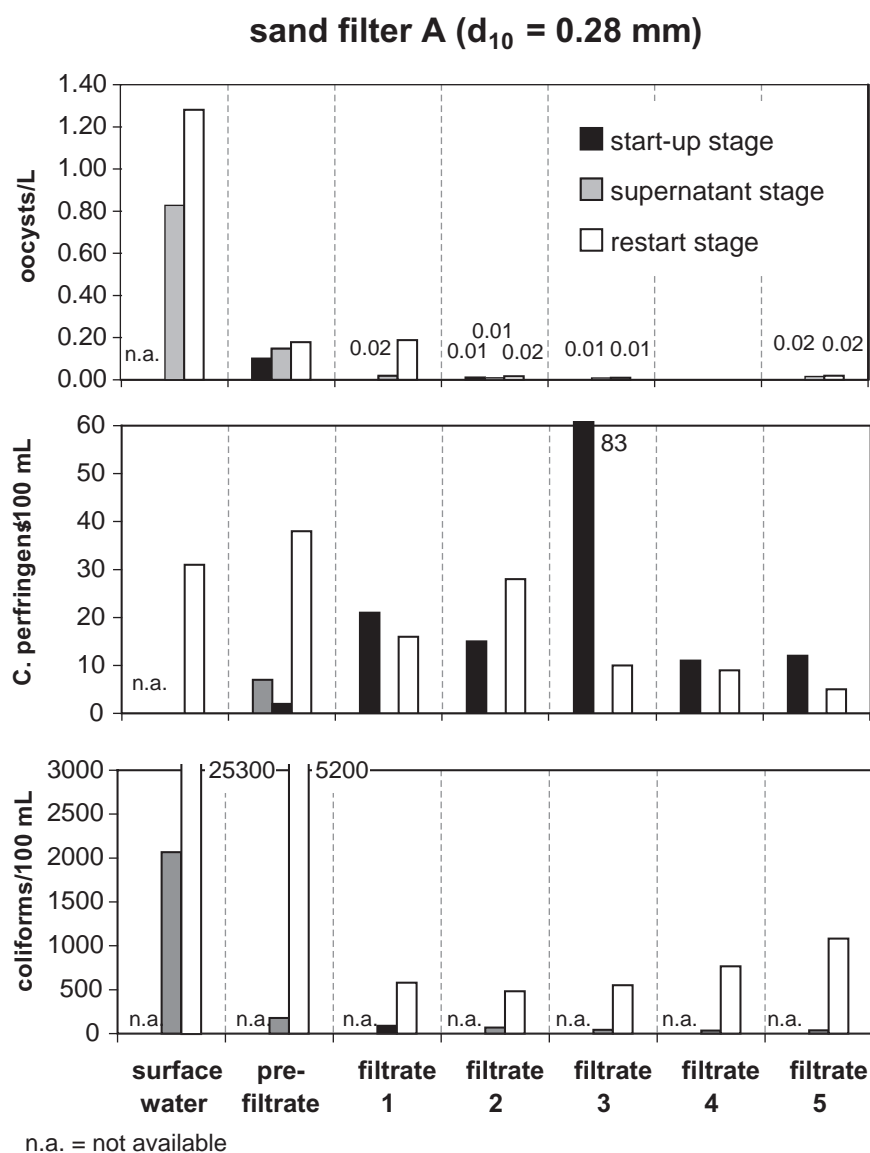
Figure 2. Microbiological results: Filter A

detected in the filtrate samples, sand filter A (d_{10} = 0.28 mm) showed the highest concentrations. In filtrate 1 0.19 oocysts per L were determined. This concentration decreased rapidly in the following filtrate samples. Sand filter D (d_{10} = 0.15 mm) showed the same effect. These results prove that in restart stages with higher filtration rates oocysts concentrations especially in the first filtrates can increase. *C. perfringens* and coliforms also showed higher levels in the first filtrates. The relationship between the numbers of oocysts and *C. perfringens* was not statistically significant.

Table 3 shows the *Cryptosporidium* oocysts reduction efficiency of the different filter materials investigated in this study. Also listed in the table are the influent concentrations in the pre-filtrate respectively filtrate. The calculated separation efficiencies confirm a removal of oocysts in the gravel filter between 62 to 87%. Sand filter B is the only filter which achieved an oocysts removal of 100% in the restart stage with higher filtration rate. The filter sands with a smaller effective particle size (d_{10} – value) achieved a better reduction efficiency but this is not an absolute safety against a pathogen breakthrough. Important elimination processes of oocysts in the filter material are filtration, sedimentation, irreversible adsorption, destruction of oocysts caused by transport and a time-related degradation. However, the study of Harter et al. (2000) indicated that a significant part of the initial oocysts separation is reversible probably because of a change of surface charge.

sand filter B (d$_{10}$ = 0.12 mm)

n.a. = not available
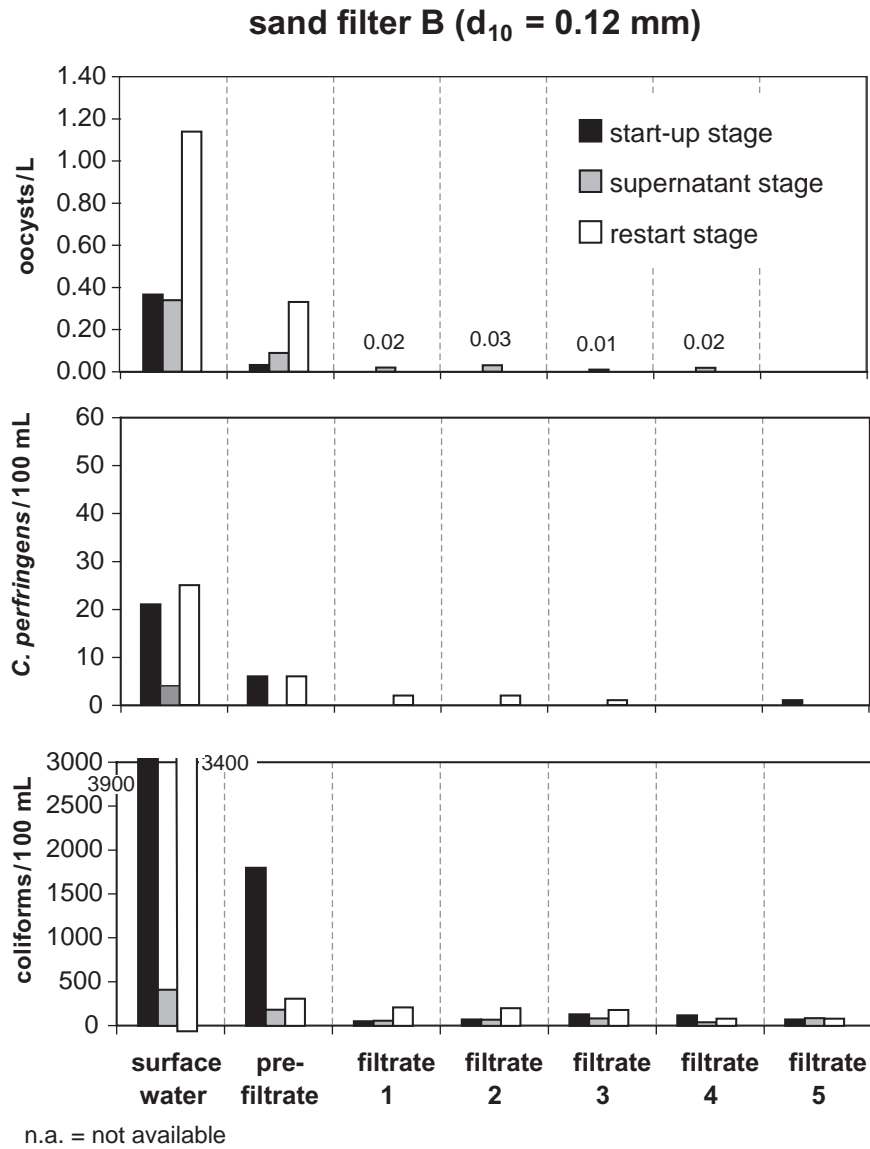
Figure 3. Microbiological results: sand filter B

Table 3. **Cryptosporidium** *oocysts reduction efficiency of gravel and sand filters in various stages of operation*

	Gravel filter 8–16 mm	Sand filter A Sand A d$_{10}$ = 0.28 mm	Gravel filter 8–16 mm	Sand filter B Sand B d$_{10}$ = 0.12 mm	Gravel filter 8–16 mm	Sand filter D Sand D d$_{10}$ = 0.15 mm
Start-up phase	n. a.	97%	n. a.	100%	87%	100%
		(0.10**)		(0,03**)	(0.69*)	(0.09**)
Supernatant phase	82%	93%	74%	83%	n. a.	n. a.
	(0.83*)	(0.15**)	(0.29*)	(0.08**)		
Restart phase	86%	80%	71%	100%	62%	93%
	(1.28*)	(0.18**)	(0.98*)	(0.28**)	(0.50*)	(0.19**)

* input concentration *Cryptosporidium* oocysts per L surface water,
** input concentration *Cryptosporidium* oocysts per L pre-filtraten.
n.a. = not available.

sand filter D (d_{10} = 0.15 mm)
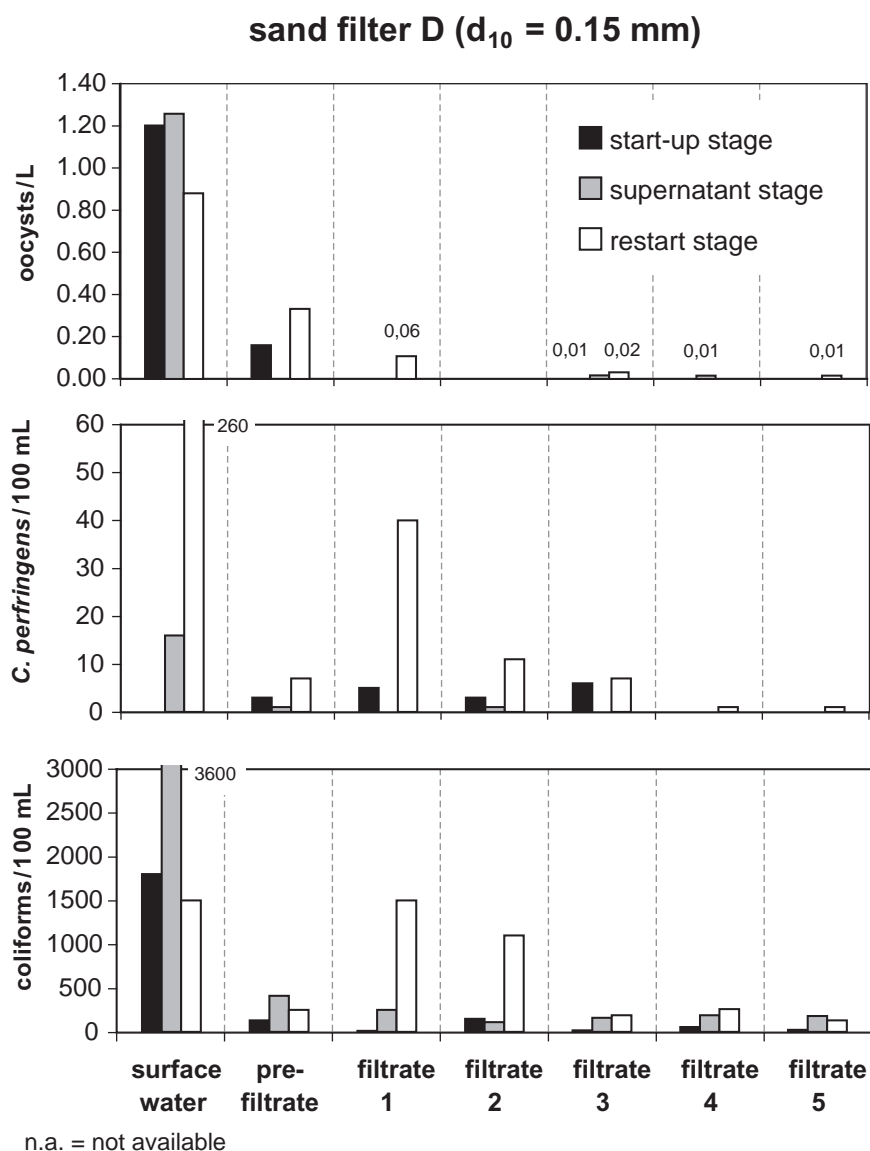
Figure 4. Microbiological results: sand filter D

CONCLUSIONS

In the river Ruhr, varying but rather low oocysts concentrations can be found. Continuous investigations over long-time periods which would allow load calculations and the consideration of seasonal effects, are still missing.

The semi-technical investigations proved a high efficiency of the gravel pre-filtration. During the second filtration step the three tested filter sands were not able to retain the *Cryptosporidium* oocysts completely in all operating stages. Filter sands with a small effective particle size are more suitable for avoiding oocysts break through. Increased filtration rates e.g. in restart stages could cause a remobilisation of retained oocysts which leads to increased concentrations in the initial filtrates.

The presence of *Clostridium perfringens* was no indicator for the presence of oocysts and vice versa.

Artificial groundwater recharge via gravel pre-filter and slow sand filter and a subsequent underground passage causes a sufficient reduction of oocysts. Suitable operating conditions of water works (multi-stage treatment

facilities with adequate filter capacity, sufficient filter depth, little void space of the filter material, filter operation as steady as possible with low filtration rate) in combination with a comprehensive protection of the surface water against the entry of pathogens belong to the concept of a multi-barrier-system and guarantee the quality of drinking water.

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ACKNOWLEDGEMENTS

Influence of groundwater redox conditions on decay of enteric viruses and *Cryptosporidium*

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Abstract

The managed recharge of recycled water to aquifers is gaining popularity in Australia and worldwide. A significant advantage of MAR is a potential improvement in water quality of the recharged water during storage. One of the major water quality issues associated with the recharge of aquifers with reclaimed water is the potential presence of pathogens such as enteric viruses in the recharged water. The processes affecting the decay of enteric viruses and other pathogens in groundwater are still not clearly understood. In this study the survival of adenovirus, coxsackievirus, bacteriophage MS2 and *Cryptosporidium* in groundwater incubated under different redox conditions was monitored. Decay rates of pathogens were found to be lower under anaerobic conditions as compared to aerobic conditions. *Cryptosporidium* had the highest observed decay rate under both nitrate and sulphate reducing conditions followed by MS2 and then the enteric viruses. Differences in the decay rate appeared to be more influenced by the type of pathogen rather than the redox condition under which they are incubated.

Keywords

Virus, *Cryptosporidium,* redox, groundwater, managed aquifer recharge.

INTRODUCTION

Water shortages affect more than 2 billion people worldwide in over 40 countries (WHO/UNICEF, 2000). Population growth, climate change, frequent droughts, contamination of surface water and uneven distribution of water resources are major reasons of water shortages (Asano and Cotruvo, 2004). At the same time, water is used inefficiently and wasted in many human population centres. Water reuse has the potential to alleviate water shortages and aid in reducing wastage and inefficiencies. Reclaimed water can be used to recharge selected aquifers and the stored water can be subsequently recovered for a variety of uses. Managed aquifer recharge (MAR) is a water reuse method that is gaining popularity. During MAR, a number of processes can collectively improve water quality during recharge and during storage in the aquifer (Dillon et al., 2005). Microbial pathogens, particularly enteric viruses, protozoa and bacteria have been shown to be present in treated reclaimed waters and wastewater (Gennaccaro et al., 2003; Abbaszadegan 1997; Aulicino et al., 1996). Gantzer et al. (1998) reported that infectious enteric viruses can survive secondary and tertiary wastewater treatment in low numbers. The presence of infectious *Cryptosporidium parvum* oocysts in secondary treated wastewater has also been reported (Gennaccaro et al., 2003). Consequently, one of the major issues associated with MAR using reclaimed water is the potential introduction of enteric pathogens into the groundwater. Microbial pathogens such as enteric viruses are known to lose viability in groundwater (Toze and Hanna 2002). A range of factors that have been implicated in influencing the inactivation of pathogens in groundwater include temperature (Yates et al., 1990), pathogen type (Toze and Hanna 2002; Sobsey et al., 1995), dissolved oxygen (Jansons et al., 1989), and organic carbon concentration (Feng et al., 2003). Gordon and Toze (2003) showed that the presence of metabolically active indigenous groundwater microorganisms had the major influence on the inactivation of pathogens during MAR and that other factors such as temperature and

oxygen were secondary influences. A range of factors such as redox potential, influence of degradable carbon and a greater analysis of pathogen type require further analysis. The purpose of this study was to compare the effect of different redox conditions on the ability of indigenous groundwater microorganisms to influence the inactivation of selected enteric viruses and the protozoa *Cryptosporidium*.

MATERIALS AND METHODS

Groundwater source and collection

Groundwater was obtained from two bores located in the superficial aquifer on the Swan Coastal Plain, Perth, Western Australia. One bore was located at the CSIRO Laboratory in Floreat Perth while the other was located at the Subiaco Wastewater Treatment Plant (WWTP). The bore at the CSIRO Laboratory was used to source unimpacted native groundwater, while the second bore was used to collect groundwater that has been impacted by long term diffuse pollution from the activities of the Wastewater Treatment Plant. The bores were purged by pumping for 30 minutes using a submersible pump prior to the collection of groundwater samples to prevent the collection of stagnant water. Groundwater was collected into sterile, nitrogen flushed 1 L borosilicate glass bottles which were then sealed with a sterile butyl rubber septum and immediately stored at 4°C until processed.

Pathogen source and storage

The enteric viruses Coxsackievirus B3 and Adenovirus strain 41 were initially cultured in cell lines (African Green Monkey Kidney cells) by the Pathology Centre, WA. The viruses were then harvested from the lawns and frozen as a crude cell extract at -80 °C until needed. The number of infective viral particles in the viral suspensions had been determined by the Pathology Centre using the MPN method in fresh cell culture lawns. The titre for each virus was determined to be 10^9 pfu ml^{-1} for coxsackievirus and 10^7 pfu.mL^{-1} for adenovirus. Before use in the survival experiments, the crude viral suspensions were cleaned and resuspended in sterile groundwater to the approximate concentration of the original crude cell extract using the methods outlined in Gordon and Toze (2003).

Cryptosporidium was purified from faecal samples obtained from the PathCentre WA and stored in sterile PBS at 4°C prior to use. Viable *Cryptosporidium* oocysts were quantified by the vital stain method used by Campbell et al., (1992). Bacteriophage MS2 (ATCC 15597-B1) was grown using *E. coli* host HS (pFamp)R (ATCC 700881 in Tryptone broth. Bacteriophage suspension was washed in P-buffer using JumboSep^{TM} filters.

Bioreactors

Experiments were under taken in 2.5 L bioreactors each filled with 2 L of a groundwater mixture consisting of 90% native groundwater from the CSIRO site bore and 10% impacted groundwater from the treatment plant bore. The chemistry of the groundwater mixture is given in Table 1.

Table 1. Groundwater chemistry of the groundwater mixture

pH	EC	Fe	PO$_4$	NO$_3$	SO$_4$
7.30	144 mS/cm	0.23 mg/L	<0.01 mg/L	0.79 mg/L	96 mg/L

Nitrate-reducing conditions were created in one of the bioreactors by feeding 6 mL of sterile CSIRO site groundwater containing sodium nitrate (100 µg/L) and ethanol (110 µg/L) every 24 hours. Similarly, sulphate-reducing conditions were created in a second bioreactor by feeding 6 mL of sterile CSIRO site groundwater containing sodium sulphate (100 µg/L) and sodium acetate (60 µg/L) every 24 hours. A control bioreactor was set up with

groundwater under anoxic conditions with no additional electron acceptor added. Bioreactors for the experiment on *Cryptosporidium* were set up similarly except for the use of a reduced carbon source mixture consisting of ethanol (0.25 µg/L) and acetate (0.6 µg/L) respectively for the nitrate reducing reactor and sulphate reducing reactor. The lower carbon source concentrations were used to reduce the excessive growth of the indigenous groundwater microorganisms. All of the bioreactors were incubated at 28 °C. Additionally, *Cryptosporidium* decay was tested in two reactors (under nitrate- or sulphate-reducing conditions) which contained groundwater that had been filtered to remove the indigenous groundwater microorganisms as a control to determine the influence of active indigenous groundwater microorganisms on the decay of *Cryptosporidium*.

Viruses and bacteriophage were placed in nylon chambers fitted with 100K MWCO filters (Pall, Australia) on both ends. These chambers were a modification of the chambers used by Pavelic et al. (1998). Thin polypropylene tubes were fitted in the sides of these chambers to enable samples to be collected on a regular basis and three chambers were suspended in each bioreactor. The sampling tubes for each of the chambers were passed through a rubber bung which sealed one of the sampling ports in the lid of the bioreactors. Seven mL of viral and bacteriophage suspension (containing approximately 10^6 coxsackievirus, 10^5 Adenovirus and 10^6 MS2) was delivered in each chamber suspended in each bioreactor. The chambers used for *Cryptosporidium* were made out of modified Nanoseps (Pall, Australia) with 100K MWCO membranes inserted at both ends. A volume of 600 µL of groundwater seeded with *Cryptosporidium* was added to each chamber. The chambers were then were suspended on a string in each reactor. On each sampling occasion, two chambers containing *Cryptosporidium* were removed from each of the bioreactors via a second sampling port in the bioreactor lid. Duplicates of all samples were collected and analysed on each sampling occasion.

Viral quantification

Viral RNA/DNA was extracted from the samples using a BD biosciences Clontech NucleoSpin® Virus nucleic acid extraction and purification kit. Viral RNA or DNA was extracted from 150 µL of sample according to the manufacturer's instructions. Final elution (50 µL) was collected in sterile RNase free centrifuge tubes and stored at −80°C prior to analysis. All analysis for virus quantification was performed in triplicate using quantitated PCR or reverse transcriptase PCR (RT-PCR). Quantitative RT-PCR and PCR reactions were run on a Biorad iCycler, using the iScript one step RT-PCR kit and iScript PCR Supermix (Bio-Rad) and quantified using the methodology detailed by Gordon and Toze (2003). Primers used for coxsackievirus were the Ent-up and Ent-down primers (Abbaszadegan et al., 1993) while adenovirus was detected using the primers HexAA 1885 and HexAA 1913 (Allard et al., 1990). Detection limits for the viruses had previously been determined to be 10 viral nucleic acid units or less per (RT-) PCR reaction (Gordon and Toze 2003).

Quantification of *Cryptosporidium* and Bacteriophage MS2

Cryptosporidium in the water samples taken form the chamber was quantified by using the vital dye staining method outlined by Campbell et al., (1992). Enumeration of MS2 was carried out by using the standard double layer agar method used previously by Toze and Hanna (2002).

Data analysis

One $\log_{10}$ removal time (τ) was determined from each plot using equation (1):

$$\tau = t/\log_{10}(C_t/C_0) \text{ (days)} \tag{1}$$

Where C_t = the final copy number at day t, C_0 = the copy number at day 0. The average one $\log_{10}$ reduction time (T_{90}) on each sampling occasion was determined as described by Gordon and Toze (2003).

Where the microorganisms tested had a linear decay a single T_{90} value was obtained. If the decay was observed to be non-linear two or more T_{90} values were obtained which related to each of the dominant decay times for the microorganism.

RESULTS

T_{90} Decay times for the viruses and *Cryptosporidium* oocysts under the different redox conditions were determined by plotting changes in numbers against time (Figures 1and 2). The time for a one $\log_{10}$ removal for each of the viruses tested is presented in Table 2 and in Table 3 for *Cryptosporidium*. As can be seen in Figures 1 and 2, differences were observed in the T_{90} times of the pathogens at the same redox condition (e.g, nitrate- or sulphate-reducing conditions). It was also observed that, each of the viruses had a different one $\log_{10}$ removal time under nitrate and sulphate reducing conditions (Table 2). *Cryptosporidium* oocysts were observed to have a very similar removal time regardless of the redox conditions.

Table 2. One $\log_{10}$ removal time (days) of enteric viruses and bacteriophage
          under different redox potential conditions

Virus	Nitrate reducing[1]	Sulphate reducing[1]	Control[1]
Coxsackievirus	5.0, 80.8[2]	4.5, 203.3[2]	4.7, 395[2]
Adenovirus	10.5	40.5	4.5, 65.7[2]
MS2	46.8	7.4	ND[3]

[1] Average redox potential was −76 µS/cm for nitrate-reducing, −226 µS/cm for sulphate-reducing, and 160 µS/cm for un-amended control; [2] two separate log reduction times given as observed decay was non-linear; [3]ND = not determined.

Table 3. One $\log_{10}$ removal time (days) of Cryptosporidium under different redox potential conditions

Bioreactors	Redox potential in bio-reactors	T_{90} removal times
Filtered NO_3-reducing	87	72.5, 8.0[1]
Filtered SO_4-reducing	−137	128.4
Unfiltered NO_3-reducing	+ 119	6.2
Unfiltered SO_4-reducing	−149	6.3

[1] two separate log reduction times given as observed decay was non-linear.

Cryptosporidium had the fastest overall decay rate with 1 $\log_{10}$ removal times of less than 10 days. MS2 and the enteric viruses varied in their removal times depending on the redox conditions. MS2 exhibited a quicker decay rate under sulphate-reducing conditions than under nitrate-reducing conditions. In comparison, adenovirus had a quicker decay under nitrate-reducing conditions. Coxsackievirus had a non-linear decay under all the redox conditions with an initial rapid decay followed by an extended, much slower decay rate under both nitrate- and sulphate-reducing conditions. The decay of coxsackievirus and adenovirus was lower under nitrate- and sulphate-reducing conditions when compared to the decay in the unamended groundwater water incubated under anoxic conditions (Figure 1). Another significant observation was that the inactivation rates of *Cryptosporidium* were significantly less in the absence of active indigenous groundwater microorganisms (filtered) when compared to the inactivation rate in the presence of active groundwater microorganisms
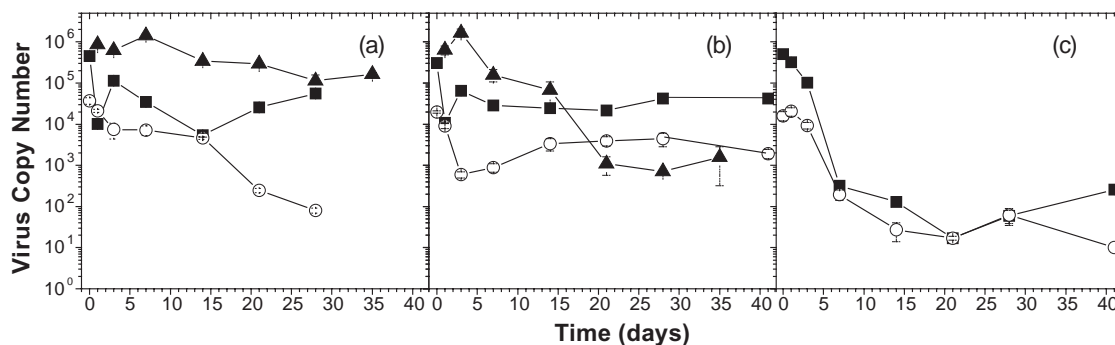
*Figure 1. Effect of redox conditions (a) nitrate reducing, (b) sulphate reducing
and (c) anoxic control on virus inactivation where ■ = coxsackievirus, ○ = adenovirus, and ▲ = MS2*
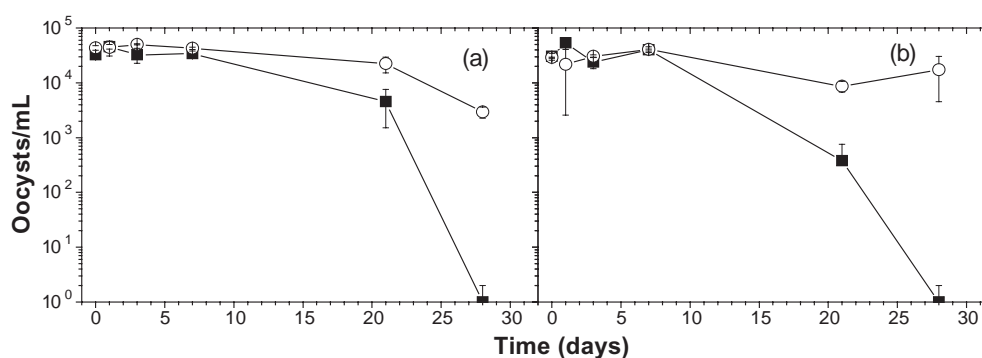
*Figure 2. Inactivation of Cryptosporidium oocysts under (a) nitrate-reducing conditions
and (b) sulphate-reducing conditions where ■ = presence of active groundwater microorganisms,
○ = absence of active groundwater microorganisms*

DISCUSSION

Enteric pathogens are known to decay naturally in groundwater and the activity of indigenous microorganisms has been identified as the single most important factor influencing this decay (Gordon and Toze 2003). The research presented in this study focused on obtaining a better understanding of the influence of different redox conditions (nitrate and sulphate reduction) on the decay rate of enteric viruses and *Cryptosporidium* in groundwater. The results indicate that different enteric pathogens behave differently under similar redox conditions. Generally the decay of coxsackievirus and adenovirus was observed to be slower than that for *Cryptosporidium* oocysts. Gordon and Toze (2003) reported that the inactivation of poliovirus and coxsackievirus was significantly lower in groundwater under anoxic conditions than under aerobic groundwater. The removal times of enteric virus and MS2 incubated under nitrate- and sulphate-reducing conditions in this study was lower compared to the previously reported results (Gordon and Toze 2003; Toze and Hanna 2002) where incubation was carried out under anoxic conditions in the presence of indigenous microorganisms. In those studies the anoxic conditions were only an absence of oxygen while in this study redox conditions were established through the addition of the electron acceptors nitrate and sulphate. The present study, as with these previous studies, also showed that the rate of decay under a given groundwater condition differed between the types of microorganisms. Little research has been undertaken on the persistence of *Cryptosporidium* oocysts in groundwater. Like the decay observed for other enteric pathogens in other studies, *Cryptosporidium* oocysts were observed to decay faster in the presence of active indigenous groundwater microorganisms under both nitrate- and sulphate-reducing conditions than occurred in groundwater not containing active groundwater microorganisms.

CONCLUSIONS

The results of this study indicate that the decay of enteric viruses and protozoa can occur under specific anaerobic redox conditions but is slower than has been previously observed under aerobic conditions. Schijven et al., (2000) reported that *clostridium bifermentans* spores were more mobile in aquifer under anaerobic conditions. The results of the current study suggest that higher mobility observed under anaerobic conditions could be due to prolonged survival of pathogens under anaerobic conditions. It is also evident that the type of pathogen is as important as the redox condition of groundwater in determining decay rate. Further work is needed to investigate the influence redox potential and other groundwater conditions have on these and other enteric pathogens to enable an effective prediction and understanding on the fate of pathogens during MAR.

ACKNOWLEDGEMENTS

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Interactions of indigenous groundwater bacteria with enteric viruses during water quality improvement by aquifer storage and recovery (ASR)

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Abstract

Aquifer Storage and Recovery (ASR) is a method of water storage and reuse, and also a method of water quality improvement. Reclaimed water, recharged to aquifers can be extracted with a higher level of water quality due to the action of indigenous groundwater microbes. It has been shown that interactions between indigenous groundwater microbes and enteric viruses have enhanced viral decay in groundwater. Investigations of this interaction have shown a small percentage of groundwater bacteria may be responsible for this observed influence. Each isolated bacterium impacted the decay of poliovirus type 1, Coxsackievirus B3 and adenovirus B41. Viral decay rates varied between the groundwater bacteria, and also between the different viruses. When indigenous groundwater bacteria were present as a whole groundwater microcosm, viral decay rates were observed to be faster than occurred with the individual groundwater isolates. Investigations of the mechanisms of decay for each active bacteria and the interaction of all active bacteria as a whole microcosm will give insight into what is happening *in situ* during ASR. Results may be useful for risk prediction for water reclamation to determine required residence times of reclaimed waters in the aquifer to ensure that there is no public health risk once the water is recovered from storage.

Keywords

ASR; decay rates; enteric viruses; indigenous groundwater bacteria; water quality.

INTRODUCTION

With water supplies under pressure throughout the world, aquifer storage and recovery (ASR) presents a viable water reuse method. The health aspects of water reuse have been identified as an issue of concern, in particular the potential presence of enteric viruses (Asano and Levine, 1996). ASR has been shown not only to be a low cost method of water storage, but also a method of water treatment (Dillon et al., in press). A number of studies into viral decay in groundwater have demonstrated the importance of the indigenous groundwater microorganisms (Gordon and Toze, 2003; Toze and Hanna, 2002; Jansons et al., 1989; Yates et al., 1985; Keswick and Gerba, 1980). It is recognised that viral decay in water is greater in the presence of other microorganisms, however, the mechanisms by which this occurs are still not well understood. It is still not clear whether viral decay by indigenous microorganisms is a result of direct predation, the production of virucidal compounds or other unknown activities. The aim of this study was to determine the influence of individual groundwater bacteria as compared to the groundwater microcosm as a whole. Results may give insight into viral decay *in situ* during ASR.

METHODS

Groundwater source and collection

Groundwater was obtained from bores located in the superficial aquifer on the Swan Coastal Plain, Perth, Western Australia. One bore was located at the CSIRO Laboratory in Floreat Perth while the other two bores were located at the Halls Head wastewater treatment plant in Mandurah, a small holiday township 70 km South of Perth. The bore at the CSIRO Laboratory and one of the bores at the Halls Head WWTP were used to source unimpacted native groundwater, while the second Halls Head WWTP bore was used to source recharged treated effluent (Toze et al., 2004). The bore was purged by pumping for 30 minutes using a submersible pump to prevent the collection of stagnant water prior to the collection of aerobic groundwater samples. Groundwater was collected into sterile 1 L borosilicate glass bottles and immediately stored on ice until processed.

Indigenous groundwater bacteria isolation

Duplicate spread plates were done using 100μL of fresh groundwater from CSIRO, Halls Head production bore and Halls Head background bore onto R2A (Oxoid) and GSP (Merck) media. Plates were incubated aerobically at 28°C from 1 to 7 days depending on growth. Plates were checked at 24 hours, 4 and 7 days for the growth of representative bacterial colonies. Colonies with a variety of morphologies were selected and subcultured onto R2A and GSP media. Plates were incubated aerobically at 28°C for 24 hours, or until growth was achieved. Subcultures were re-streaked at least twice to obtain pure isolates. All isolates were preliminarily characterised using colony and cell morphology, Gram-stain reaction, and oxidase and catalase reactions. Pure cultures were frozen in duplicate at $-80°C$ in a mixture of 20% glycerol and nutrient broth.

Viruses

Poliovirus type 1, Coxsackievirus B3 and Adenovirus B41 were initially cultured in cell lines (African Green Monkey Kidney cells) by the Pathology Centre, WA. The viruses were then harvested from the lawns and frozen as a crude cell extract at -80 °C until needed. The number of infective viral particles in the viral suspensions had been determined by the Pathology Centre using the MPN method in fresh cell culture lawns. The titre for each virus was determined to be 10^8 pfu ml^{-1} for poliovirus, 10^9 pfu ml^{-1} for coxsackievirus and 10^7 pfu.mL^{-1} for adenovirus. Before use in the survival experiments, the crude viral suspensions were cleaned and resuspended in sterile groundwater to the approximate concentration of the original crude suspension using the methods outlined in Gordon and Toze (2003).

Characterisation of bacterial isolate activity versus groundwater microcosm

A one tenth strength R2A broth was inoculated with selected groundwater isolates in sterile polypropylene tubes and incubated aerobically at 28°C for 24 hours. A 9.0mL aliquot of inoculated broth was then further inoculated with 500 μL of poliovirus, coxsackievirus or adenovirus. Viral decay by groundwater microcosms was investigated by inoculating 500 μL virus into 9.5 mL non-sterile groundwater. Tubes were then incubated aerobically at 28°C and samples were taken at time 0 and day 35. Samples were stored at $-80°C$ until nucleic acid extraction and further analysis was carried out.

Negative controls and blanks were also carried out following the same procedure by inoculating viruses into sterile, one tenth strength R2A broth and groundwater. Presence of groundwater bacteria or sterility of un-inoculated broth or groundwater was monitored on each sampling occasion by inoculating 20μL of sample in 180μL nutrient broth and incubated at 28°C for 24 hours.

Virus quantification

Changes in viral numbers were determined using quantitative RT-PCR or PCR. Viral RNA/DNA was extracted from 150 µL of each collected samples using the Viral Nucleic Acid Extraction Kit (Borhinger-Mannien) using supplied instructions. All extracted nucleic acid samples were stored at −80°C until analysed by RT-PCR or PCR.

Quantitative RT-PCR and PCR reactions were run on a Biorad iCycler, using the Titan™ RT-PCR Enzyme System (Roche Diagnostics) and quantified using the methodology as detailed by Gordon and Toze (2003). Primers used for poliovirus and coxsackievirus were the Ent-up and Ent-down primers (Abbaszadegan et al., 1993) while adenovirus was detected using the primers HexAA 1885 and HexAA 1913 (Allard et al., 1992; Allard et al., 1990).

The detection limit for viruses using Reverse Transcriptase Polymerase Chain Reaction (RT-PCR), or PCR, was determined by making serial 10-fold dilutions of extracted viral RNA/DNA and determining the lowest detectable dilution. The lowest detectable dilution was 10^{-6} for both poliovirus and coxsackievirus and 10^{-4} for adenovirus, which equated to a detection limit of approximately 10 or less viral RNA/DNA molecules per PCR reaction (based on original MPN titre).

Determination of decay rates

The viral copy numbers determined at each time interval were plotted as a semi-logarithmic function. A 1 $\log_{10}$ removal time T90 was determined from each plot using equation (1)

$$\text{Decay Rate (T90)} = t / \log_{10} (c_t/c_0) \text{ (days)} \tag{1}$$

Where T90 = number of days to a 1 log removal, t = incubation time, C_t = the final copy number at day t, C_0 = the copy number at day 0 (Toze and Hanna, 2002).

RESULTS

Of all indigenous groundwater bacteria isolated, 27% showed the ability to reduced mean viral copy numbers of the enteroviruses poliovirus type 1 and Coxsackievirus B3 over time. The top ten isolates were also tested against adenovirus B41. Decay rates were similar between poliovirus and adenovirus, whereas decay rates varied for coxsackievirus (Table 1). The decay of poliovirus, coxsackievirus and adenovirus over time in groundwater and in the presence of two indigenous groundwater isolates can be seen in Figure 1, with corresponding mean decay rates in Table 2. Characterisation of the two groundwater isolates tested found that they were both Gram-negative bacilli and were oxidase and catalase positive indicating that they could be initially characterised as being *Pseudomonas*-like bacteria.

Poliovirus showed a large decay in non-sterile groundwater with a 1 $\log_{10}$ removal time of 11 days. Isolate 1 and isolate 2 also caused large T90 decay time of 17.9 and 13.9 days respectively. Poliovirus was relatively stable in the negative control with a decay time of 469.4 days.

Coxsackievirus had a similar trend of decay time in the non-sterile groundwater with a T90 time of 9.3 days. For the individual groundwater isolates, coxsackievirus had a slower decay time of 62.1 days in the presence of Isolate 1 than in the presence of Isolate 2 (26.1 days). Coxsackievirus remained stable in the negative control with a decay rate of 1159.9 days. Adenovirus showed a similar decay time in the presence of non-sterile groundwater and in the presence of both isolate 1 and 2 with decay times of 35.4, 27.1 and 16.2 days respectively. Adenovirus remained relatively stable in the negative control with a decay time of 228.2 days.

Table 1. T90 decay times for top ten groundwater isolates on enteric viruses

Groundwater Isolate	T90 decay times (days)		
	Poliovirus	Coxsackievirus	Adenovirus
1	17.9	62.1	27.1
2	13.9	26.1	16.2
3	38.1	3.91	no decay
4	70.7	19.1	27.5
5	59.2	24.9	no decay
6	no decay	49.2	17.7
7	no decay	4.5	101.6
8	9.4	10.4	4.7
9	62.0	88.0	190.7
10	97.0	88.9	76.6
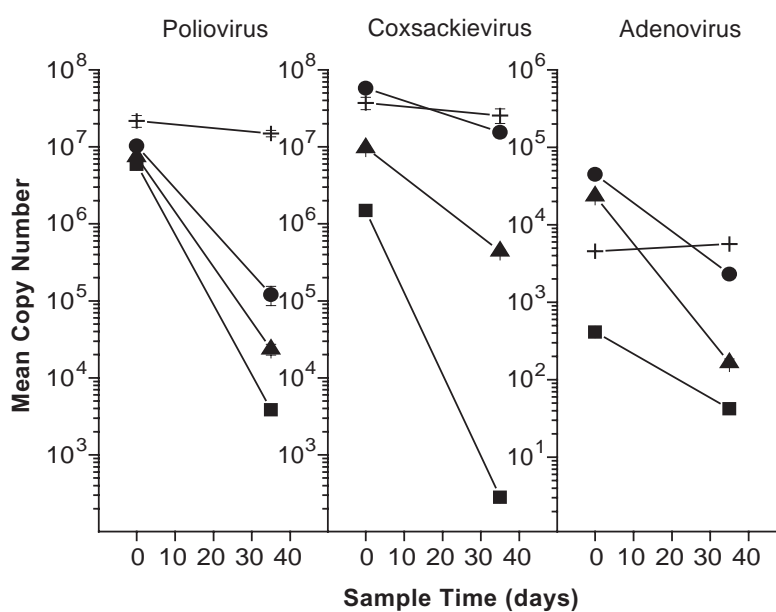
Figure 1. Decay of poliovirus, coxsackievirus and adenovirus over 35 days in
■*: groundwater,* ▲*: 1:10 R2A with isolate 1,* ●*: 1:10 R2A with isolate 2,*
and +: double filter sterilised groundwater (negative control)

Table 2. T90 decay rates (days) of poliovirus, coxsackievirus and adenovirus over a 35 day period in the presence of non-sterile groundwater, isolate 1, isolate 2 and sterile 1/10 strength R2A (negative control)

	Poliovirus	*Coxsackievirus*	*Adenovirus*
Groundwater	11	9.3	354
Isolate 1	17.9	62.1	27.1
Isolate 2	13.9	26.1	16.2
Negative Control	469.4	1159.9	228.2

DISCUSSION

The aim of this study was to determine the impact of individual and groups of groundwater microorganisms on the decay of enteric viruses. In addition, variations that may exists amongst individuals in a microbial population as compared to the complete groundwater microcosm was determined. The isolation of a range of indigenous groundwater bacteria and testing each individually could give insight into the dynamics involved in viral decay in groundwater during aquifer storage and recovery (ASR).

It has long been known that virus decay in ground and surface water is greater in the presence of other microorganisms, but the mechanisms by which this occurs is not well studied. Cliver and Herrmann (1972) investigated the decay of both poliovirus type 1 and coxsackievirus A9, A7 and B1. Viral decay was monitored in bacterial cultures and decay was found to be a result of proteolytic bacteria, most notably, *Pseudomonas aeruginosa. Bacillus subtilis* was also shown to have virucidal activity. Ward et al. (1986) also investigated the decay of poliovirus and coxsackievirus, and also echovirus and rotavirus in filtered and non-filtered riverine and lake water. It was also found that any treatment that removed or inactivated microorganisms resulted in loss of virucidal activity. Nobel and Fuhrman (1997) have also suggested that extracellular enzymes produced by marine microbes had an important role in viral decay. Studies of the decay of poliovirus and coxsackievirus in groundwater have indicated the production of virucidal compounds, as poliovirus showed decay when separated from the indigenous groundwater microorganisms contained in groundwater (Wall et al., 2004). Coxsackievirus required direct contact with groundwater to achieve significant decay. These results suggest the production of numerous virucidal compounds that may be virus type specific, of varying sizes, some of which may be larger than 250,000 molecular weight, or closely associated with the indigenous groundwater microorganisms.

Of the aerobic bacteria isolated in the current study, 27% exhibited an ability to reduce the mean copy number of poliovirus and coxsackievirus over time when compared to the degradation observed in the sterile control. The top ten bacteria were then tested against adenovirus and results were similar to the influence on poliovirus. T90 decay rates of the top ten bacteria varied from as little as 4 days for a 90 % reduction in mean copy number through to no decay (a positive T90). As poliovirus and coxsackievirus are very closely related as opposed to adenovirus, it would be expected that the influence of active bacteria would be similar against poliovirus and coxsackievirus, but this was not observed to be the case. The results obtained for coxsackievirus were quite different from those for poliovirus and adenovirus.

Survival of poliovirus, coxsackievirus and adenovirus in the presence of two active groundwater isolates was investigated and compared to decay in non-sterile groundwater. The results obtained supported the hypothesis that there may be a number of compounds produced by a small number of active indigenous groundwater microorganisms as the influence of the different isolates varied with the pathogens tested. In addition none of the isolates had the same

influence on all three enteric viruses. These influences also seem to act together to result in a much greater decay of the viruses when groundwater microorganisms were present as a total microbial population in non-sterile groundwater. A greater variation and reduction in decay times were obtained for the three viruses when in the presence of the individual bacteria compared to the decay times obtained in the presence of the non-sterile groundwater. This suggests that the influence of the active groundwater microorganisms may be working in conjunction when combined in a groundwater microcosm.

CONCLUSIONS

Indigenous groundwater bacteria are important in the decay of enteric viruses in groundwater with a large loss of viral copy numbers in non-sterile groundwater containing indigenous bacteria. Not all indigenous groundwater bacteria had the potential to influence in the decay of viral copy numbers. The 27% of indigenous groundwater bacteria that showed potential for viral decay varied in their influence. Decay times for the three viruses varied amongst the groundwater isolates and also between the viruses. This suggest the presence of different decay factors, such as protease enzymes, that may work in conjunction to produce the consistently large decay times observed for the viruses in the presence of the entire groundwater microcosm.

These results presented provide important information regarding the fate of enteric viruses during aquifer storage and recovery (ASR). As viral decay is not the result of one individual groundwater bacterium, significant virus decay during ASR can be expected even if there are changes in indigenous groundwater bacterial population structure. This means that reclaimed water recharged to an aquifer via ASR can be expected to have a greatly reduced health risk after sufficient storage time.

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Transport and attenuation of antibiotic residues during river bank filtration in Berlin, Germany

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Abstract

This paper presents the results on the occurrence and fate of antibiotic residues during bankfiltration obtained from a study carried out in terms of an interdisciplinary project at three transects in Berlin, Germany. Six anti-biotic compounds and two metabolites were detected at ng/L concentrations in water samples from the lakes or in the monitoring wells of the transects. Clarithromycin, roxithromycin (macrolide), trimethoprim (synergist for sulfonamides) and acetyl-sulfamethoxazole (metabolite) are efficiently removed by bank filtration. Residues of clindamycin (lincosamid) and dehydro-erythromycin (metabolite) were completely attenuated during the soil passage. For sulfamethoxazole (sulfonamide), a significant but not complete removal during bank filtration was observed. It was the only compound that could be detected at tracelevels in samples collected from water-supply wells.

Keywords

Antibiotics; bank filtration; LC-MS/MS; water.

INTRODUCTION

In Germany, approximately 270 tonnes of antibiotic substances are annually prescribed for human use. Residues of antibiotic drugs are frequently found in the aquatic environment [Alexy and Kümmerer, 2005; Batt and Aga, 2005; Heberer, 2002; Hirsch et al., 1998; Sacher et al., 2001]. They can reach the aquatic system due to an incomplete metabolism in target organisms (humans or animals), by improper disposal, or by production spills. For some of these compounds only partial elimination was observed during wastewater treatment, and residues of veterinary drugs occurring in manure may reach surface or groundwaters by leaching or agricultural runoff [Alexy and Kümmerer, 2005; Heberer, 2002].

In Berlin, Germany, water suppliers produce approximately 70 percent of the drinking water by bank filtration and artificial recharge. Thus, knowledge about the behavior of drug residues during groundwater recharge is highly important for public water-supply. In terms of an interdisciplinary project, entitled 'Natural and Artificial Systems for Recharge and Infiltration' (NASRI), the fate and transport of antibiotics during bank filtration were investigated at Lake Tegel and Lake Wannsee in Berlin.

METHODS

From May 2003 to August 2004, the occurrence and fate of 19 environmentally relevant antibiotics and two of their metabolites were investigated at three transects located at lakes Wannsee and Tegel (Figure 1). Samples were

analyzed monthly for antibiotic substances from various prescription classes (fluoroquinolones, macrolide anti-biotics, sulfonamides, tetracyclines, and penicillins). The samples were pre-concentrated by solid-phase extraction and identified and quantified using high-performance liquid chromatography-electrospray tandem-mass spectro-metry (LC/ESI-MS/MS) [Fanck and Heberer, in prep.].
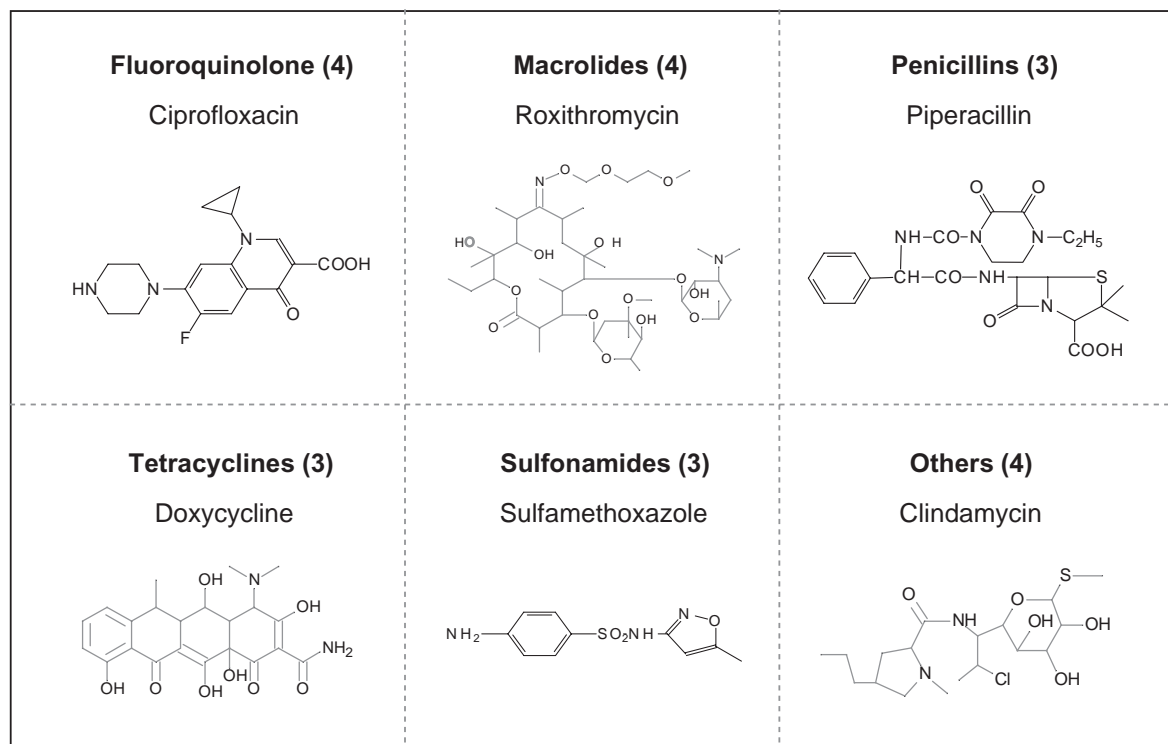
Figure 1. Examples of chemical structures of compounds from each of the investigated compound classes. In brackets: investigated number of compounds from each class

RESULTS AND DISCUSSION

Only six of the 19 investigated compounds, the macrolides clarithromycin and roxithromycin, the sulfonamides sulfamethoxazole and sulfadimidine, the sulfonamide synergist trimethoprim, and the lincosamide clindamycin were detected at ng/L concentrations in water samples from the lakes or in the monitoring wells of the transects. Additionally, the drug metabolite dehydro-erythromycin, formed by hydrolysis from the macrolide erythromycin, and acetyl-sulfamethoxazole, the main human metabolite of sulfamethoxazole, were found. As expected from results of previous studies, no tetracyclines and penicillins were found. This is in agreement with the results of other studies [Christian et al., 2003; Hirsch et al., 1998] and most likely caused by metabolism (penicillins) and/or chelating and sorption effects in the sewage treatment plants or distribution system.

The investigations of the field sites showed a different behavior of the six antibiotic residues during infiltration. Trimethoprim, Clarithromycin, and roxithromycin are efficiently removed by the bank filtration. They are found in the lakes and showed a similar time-dependent concentration trend (Figure 2). During winter, antibiotic residues were detected at higher concentrations than in the summer. Seasonal variations were also observed in an additional survey on the occurrence of antibiotic residues in municipal sewage. In winter, such residues were found at increased concentrations both in samples collected from the influents and the effluents of the municipal sewage treatment plant in Berlin-Ruhleben. The elevated concentrations and loads were assigned to higher seasonal con-sumption of antibiotics [Fanck and Heberer, in prep.]. In the same study, better removal of residues of macrolide antibiotics was observed during summer.
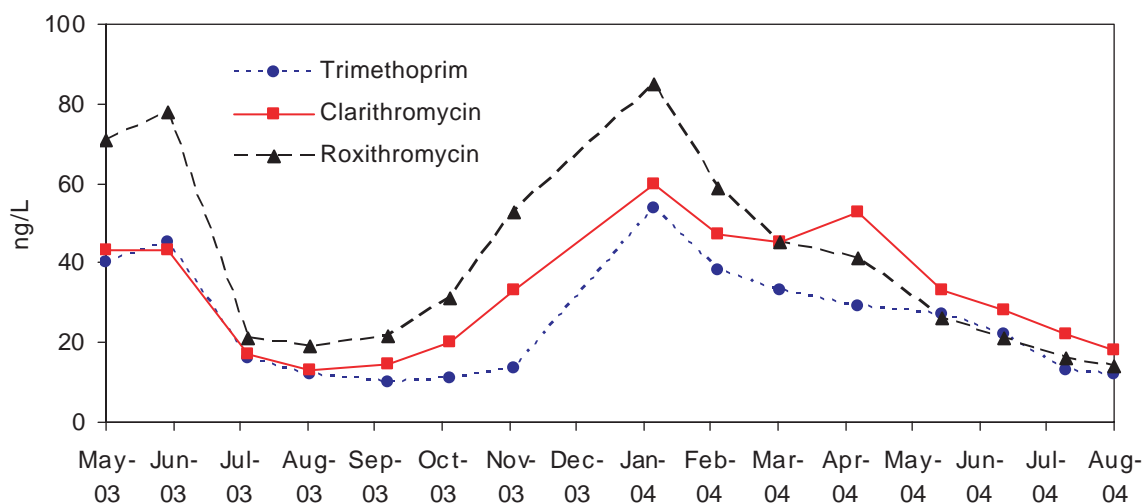
Figure 2. Concentrations of Trimethoprim, Clarithromycin and Roxithomycin measured in Lake Tegel

The concentrations of clindamycin and dehydro-erythromycin are also significantly decreasing during the soil passage. In most cases, they were not detected beyond the first monitoring wells. Only sulfamethoxazole occurred in most of the monitoring wells at concentrations higher than those of all other compounds. It is the only antibiotic that was also found in measurable concentrations (mean value of 5 ng/L) in water-supply wells.

Surprisingly, sulfadimidine was observed (albeit at low concentrations, 3–9 ng/L) in some of the deeper wells even though this compound is only used for veterinary purposes in Germany. This observation can be explained by applications of large quantities of sulfadimidine in the past, when it was used in livestock farms north of Berlin.

The pharmacologically inactive human metabolite of sulfamethoxazole, acetyl-sulfamethoxazole, occurred only in the lake and with very low concentrations, between 5 and 15 ng/L. For acetyl-sulfamethoxazole, a removal rate of more than 99% was found in the sewage treatment plant of Ruhleben [Fanck and Heberer, in prep.].

CONCLUSIONS

The studies within the NASRI project have shown that bank filtration, as operated in Berlin, is a useful tool for the pre-treatment of contaminated surface water used for drinking-water supply. With one exception, antibiotic residues are not found in the water-supply wells. Sulfamethoxazole was detected at trace-levels in samples collected from monitoring and water-supply wells. These quantities are, however, way too low to cause any toxic effects in humans. Thus, bank filtration has proven as being an efficient method for the removal of antibiotic residues by natural attenuation.

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Occurrence, transport, attenuation and removal of pharmaceutical residues in the aquatic environment and their relevance for drinking water supply in urban areas

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Abstract

Residues of pharmaceuticals used in human medical care have recently been detected as important trace contaminants of sewage, surface and groundwater. This paper compiles the recent state of knowledge on the occurrence and fate of pharmaceutical residues in the aquatic environment of urban areas. Findings in sewage effluents, surface, ground, and drinking water at concentrations up to the µg/L-level have been reported and will be discussed to demonstrate the impact of pharmaceutical residues on the aquatic environment and on public water supply. The efficiency of natural and technological processes such as bank filtration or membrane filtration for the removal of pharmaceutical residues including estrogenic steroids, analgesics, antibiotics, anti-epileptic drugs, blood lipid regulators, and several drug metabolites will be presented and discussed.

Keywords

Pharmaceuticals, EDCs, endocrine disrupters, bank filtration, risk assessment, drinking water.

BACKGROUND

To date, residues of more than 100 different pharmaceuticals have world-wide been detected in municipal sewage and in several samples of surface, ground, and in a few cases even in drinking water (Stan and Heberer, 1997; Halling-Sørensen et al., 1998; Heberer et al., 1998; Daughton and Ternes, 1999; Sedlak et al., 2000; Kuemmerer, 2001; Sacher et al., 2001; Ternes, 2001; Heberer, 2002a; Kolpin et al., 2002). For the most part, such residues are entering the receiving surface waters by discharges from sewage treatment plants but that are also several other potential sources (Ternes 1998; Heberer 2002a,b). The observed concentrations of pharmaceutical residues in the aquatic environment range from less than 1 ng/L to more than 10 µg/L. As shown in Table 1, many of the pharmaceuticals found in wastewater effluent have also been detected in surface waters. However, the number and the concentrations of pharmaceutical residues are decreasing from sewage/surface water to ground and drinking water. The compounds that were detected at trace-level concentrations in drinking water are analgesics, anti-epileptic drugs, blood lipid regulators and X-ray contrast media.

Under recharge conditions, polar pharmaceutical residues can be leaching through the subsoil into the groundwater aquifers. Infiltration of sewage effluent and effluent-impacted waters are the most common sources of pharmaceutical residues in groundwater. However, releases from industrial sources or leaking landfills may also be the sources for the occurrence of such residues in groundwater (Eckel et al. 1993; Holm et al. 1995). Recently, Reddersen et al. (2002) reported concentrations of phenazone-type drug residues and their metabolites as high as 4 µg/L for the individual compound in groundwater near a Superfund site that discharged to a small stream upstream from the bank filtration area. The mere occurrence of such residues at trace-level concentrations in groundwater or even in drinking water does not necessarily imply that groundwater recharge techniques such as bank filtration should in

general be avoided. Depending on various parameters such as the composition of the colmation layer or the individual traveling times several pharmaceuticals such as antibiotics, estrogenic steroids and some other compounds can efficiently be removed during bank filtration or other natural recharge processes. Thus, there is an essential need for and a lack in information on the fate and transport of pharmaceutical residues and related compounds in the aquatic environment and during groundwater recharge that enables to understand and to enhance the efficacy of natural attenuation processes and of water management strategies. Moreover, the occurrence of such residues in groundwater resources should not only be approached as a potential thread to drinking-water supplies. Such residues might also be seen and used as indicators to monitor and improve the efficiency of natural and artificial treatment techniques.

Table 1. World-wide findings of pharmaceutical residues in sewage, surface, ground-, and drinking water. Reported number of compounds in different environmental compartments according to their prescription class type (supplemented as cited in Heberer, 2002a)

Analytes (prescription classes)	Number of compounds detected in		
	Sewage and surface water	Groundwater	Drinking water
Analgesics/anti-inflammatory drugs + metabolites	26	15	8
Antibiotics	31	3 (+ 5 traces)	–
Antiepileptics	2	2	2
Lipid regulators + metabolites	7	3	3
Beta-blockers	7	1	–
Contrast media + metabolites	8	6	3
Cytostatic drugs	2 (traces)	–	–
Oral contraceptives (EE2 and mestranol)	2 (almost < 2/0.5 ng/L)	(1) validity in doubt*	(1) validity in doubt *
Other pharmaceuticals	21	4	–
Total	106	34 (39)	16

* Validity of positive detections in doubt for analytical reasons and thus not considered as inevitable positive detection.

REMOVAL BY NATURAL ATTENUATION

The fate of pharmaceutical residues has been investigated for several processes including bank filtration (Heberer and Stan 1997, Heberer et al. 1997, 2001, 2002a, Brauch et al. 2000, Kühn and Müller 2000, Reddersen et al. 2002, Verstraeten et al. 2002), artificial groundwater replenishment (ponds) (Heberer and Stan 1997, Heberer and Adam, 2004), soil-aquifer treatment (SAT) (Drewes et al. 2002), and slow-sand filtration (Preuß et al. 2001, Ternes et al. 2002). Especially, bank filtration has been shown to be an efficient tool for minimizing the concentrations of pharmaceuticals (Verstraeten et al, 2002). Non-polar pharmaceuticals, such as the blood-lipid regulator bezafibrate or indomethacine, as well as most antibiotics are efficiently removed during bank filtration but also by artificial groundwater replenishment (Heberer and Mechlinski, 2003; Heberer and Adam, 2004; Heberer et al., 2004). Likewise, the analgesic drug ibuprofen was degraded in soil-column experiments (Mersmann et al., 2002). Other pharmaceuticals including the anti-epileptic drug carbamazepine and the analgesics diclofenac and propyphenazone were retarded in soil-column experiments (Mersmann et al., 2002) but only partly removed during bank filtration and other artificial groundwater recharge methods (Heberer and Mechlinski, 2003; Heberer and Adam, 2004; Heberer et al., 2004). The same compounds were also detected at low concentrations in water-supply wells in Berlin together with the anti-epileptic drug primidone and the drug metabolites clofibric acid and AMDOPH (1-acetyl-1-methyl-2-dimethyl-oxamoyl-2-phenylhydrazide) which were reported to pass bank filtration and other groundwater

recharge sites without or only little significant retardation (Heberer and Mechlinski, 2003; Heberer and Adam, 2004; Heberer et al., 2004).

Steroid hormones such as 17β-estradiol (E2) and 17α-ethinyl estradiol (EE2) have received considerable attention because of their potential in causing feminization in freshwater fish. These compounds are difficult to study because they only appear at very low but still environmentally important concentrations of between 0.1 and 10 ng/L in wastewater effluent (Ternes et al., 1999; Baronti et al., 2000; Huang and Sedlak, 2001). On the basis of their hydrophobicity (log Kow~4) and taking into account their potential for biotransformation, these hormones would not be expected to persist in the subsurface. However, Kuch and Ballschmiter (2001) and Adler et al. (2001) reported several detections of estrogens including EE2 in ground and drinking water in Germany. However, other investigations of steroid hormones in sewage, surface, and bank-filtered water do not indicate the presence of such compounds for ground or drinking water (Zühlke et al., 2005). Thus, even short distances between the river or lake banks and monitoring wells can lead to dramatic decreases of estrone concentrations illustrating the potential of groundwater recharge systems for the retention of estrogenic steroids (Zühlke et al., 2005).

REMOVAL BY OTHER TECHNIQUES

Besides other purification techniques such as ozonation (Andreozzi et al. 2002, Ternes et al. 2002, 2003, Huber et al. 2003) or filtration applying granular activated carbon (Ternes et al. 2002), membrane filtration using nanofiltration (NF) or reverse osmosis (RO) membranes is one of the most promising techniques for the removal of pharmaceutical residues from contaminated raw water sources (Heberer and Feldmann, 2004).

Kimura et al. (2003) investigated the rejection of disinfection by-products, endocrine disrupting compounds and pharmaceuticals by polyamide NF/RO membranes. According to the results of their investigations, negatively charged compounds such as the analgesic drug diclofenac can be rejected to a great extent (i.e., >90%) regardless of other physical/chemical properties of the tested compounds due to electrostatic repulsion whereas rejection of non-charged compounds was influenced mainly by the size of the compounds. Adams et al. (2002) evaluated conventional drinking water treatment processes including RO under typical water treatment plant conditions demonstrating that RO was very efficient in removing all studied antibiotics. Heberer et al. (2002b) and Heberer and Feldmann (2004) successfully tested the performance of membrane-based mobile drinking water purification units for the removal of pharmaceuticals from highly contaminated raw-water sources in two extended field-trials. In investigations of large-scale operations in the U.S., Drewes et al. (2002) have shown that NF and RO might also be able to remove pharmaceutical residues from municipal sewage effluents.

CONCLUSIONS

Persistent residues of pharmaceuticals used in human medical care are discharged by municipal sewage treatment plants into the receiving surface waters. Under recharge conditions, especially some of the polar compounds are able to leach into groundwater aquifers and a few of them have also the potential for turning up into water-supply wells. Natural processes such as bank filtration or other groundwater recharge techniques are able to remove pharmaceutical residues or at least to decrease the concentrations of such compounds from contaminated raw waters. A prerequisite for obtaining maximum attenuation of such contaminants are both background knowledge of the potential contaminants and their fate during groundwater recharge at different hydrogeological conditions and a proper construction and operation of the recharge facilities. Nevertheless, even this will not guarantee a complete removal of all pharmaceutical residues occurring in raw waters that are under the influence of sewage effluents (Heberer and Verstraeten, 2002). Additionally, reliable methods such as membrane filtration are available and might be used for the removal of remaining traces of pharmaceutical residues in terms of a multi-barrier approach

(Heberer and Verstraeten, 2002). However, it might be questioned if an additional use of such techniques for the removal of trace residual amounts of contaminants is reasonable? From our current state of scientific knowledge, the trace-level concentrations of several specific pharmaceuticals that have (until today) been detected in drinking water do not have a toxicological impact on human health. On the other hand, the mere occurrence of pharmaceutical residues in human drinking water supply originating from discharges of municipal sewage treatment plants may by the public be recognized as being undesirable even at trace-level concentrations.

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The impact of alternating redox conditions on groundwater chemistry during artificial recharge in Berlin

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Abstract

The aim of the study was to evaluate the influence of variable redox conditions on a number of pharmaceutically active compounds, namely carbamazepine, phenazone and AMDOPH (1-acetyl-1-methyl-2-dimethyl-oxymoyl-2-phenylhydrazide) below an artificial recharge pond in Berlin. The redox conditions change seasonally, mainly as a result of temperature changes of 0 to 24°C in the infiltrate. Aerobic conditions prevail in winter, while manganese reducing conditions are reached below the pond in summer. Phenazone is redox sensitive and was generally fully degraded before reaching the first groundwater well as long as oxygen was present. When conditions turned anaerobic, phenazone was not fully eliminated. AMDOPH (1-acetyl-1-methyl-2-dimethyl-oxymoyl-2-phenylhydrazide) and carbamazepine are very persistant drug residues. However, results suggest that AMDOPH may be degradable under certain favourable conditions (i.e. aerobic conditions; relatively high temperatures, low recharge rates), but further studies will need to verify this statement.

Keywords

Artificial recharge, redox conditions, biodegradation, phenazone, pharmaceuticals.

INTRODUCTION

In view of growing populations with increasing water demands and potential water shortages, artificial recharge of groundwater is becoming increasingly more important all over the world as a sustainable method to save groundwater resources and improve their quality. The 3.4 million inhabitants of metropolitan Berlin rely on drinking water originating from local groundwater. Around 70 % of the abstracted groundwater originates from induced bank filtration and artificial groundwater recharge (Pekdeger and Sommer-von Jarmerstedt, 1998). Because the local waste water (WW) treatment plants discharge treated wastewater into the surface water while the natural discharge is low, the hydrological system can be considered a semi-closed water cycle relying partly on indirect WW reuse. A number of pharmaceutically active compounds (PhACs) such as clofibric acid, diclofenac, ibuprofen, phenazone, propyphenazone, primidone and carbamazepine are not eliminated completely during the WW treatment process and have been detected in the surface water (Heberer, 2002b; Reddersen et al., 2002).

Redox processes associated with the degradation of organic carbon by terminal electron acceptors (TEAs) such as oxygen (O_2), nitrate (NO_3^-), manganese (Mn) and iron (Fe)oxides and hydroxides and sulphate (SO_4^{2-}) typically proceed in a sequential order from the highest energy yield downwards (e.g. Berner, 1981). Redox processes change the hydrochemical conditions of a hydrochemical system; they also may affect the behaviour of various other water constituents such as organic substances, pharmaceutically active compounds (PhACs; Holm et al., 1995) or heavy metals, which is why it is important to understand the redox conditions of a groundwater recharge site.

This paper reports results from a field study derived from a monitoring transect of observation wells at an artificial recharge pond near Lake Tegel (Tegeler See) in Berlin, Germany, where surface water is infiltrated. The main objec-

tive of this study was to investigate the fate of a number of PhACs during infiltration and their dependence on the prevailing redox conditions below the pond.

FIELD SITE

The studied recharge pond lies within the catchment area of Tegel water works, which belongs to the Berlin Water Company (Berliner Wasserbetriebe, BWB). The pond, together with two other ponds, is surrounded by over 40 production wells of the Saatwinkel and Hohenzollern well galleries (Figure 1). Surface water of the adjacent Lake Tegel is pumped into the pond (pond area: 8,700 m²) after passing through a microstrainer for pre-filtration. During an operational cycle, the hydraulic conductivity of the pond decreases with time due to clogging, leading to a decrease of the infiltration capacity. To restore the infiltration capacity, the bottom of the pond is scraped off regularly.

A transect of groundwater observation wells was installed between pond 3 and production well 20 of the Saatwinkel production well gallery (Figure 1). Only results of two observation wells are discussed in this paper, both (Teg365, TEG366) are located directly next to the pond and are screened in 4.3–6.3 and 6.0–8.0 m depth below the pond base.
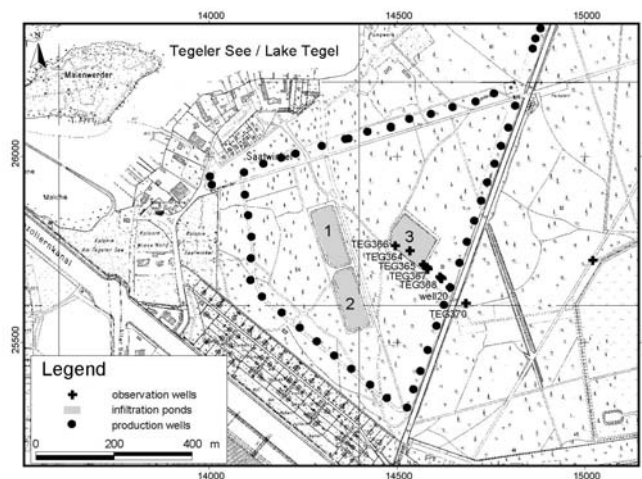
Figure 1. Location of infiltration ponds, observation wells and production wells (Hohenzollern and Saatwinkel galleries) of the Tegel artificial recharge site

The porous sediments in the area are of Quaternary age and consist of fine to coarse-grained sands with a hydraulic conductivity of $2-8*10^{-4}$ m/s. A more detailed description of the sediment properties is given in Massmann et al. (subm.).

METHODS

From November 2001 to April 2004, groundwater was sampled monthly. The sampling was later intensified and done once or twice a week at the pond and the two sampled observation wells. Samples for cation analysis were preserved with concentrated nitric acid (HNO_3) and stored at 4°C. Full water analysis was generally performed one day after sampling. Water levels and temperatures were recorded daily using data loggers. Anions in the water were measured with ion chromatography, DX 500 (Dionex Coop.) Cations in the water were determined with an ICP (OES) IRIS (Nicolet). The DOC measurements were carried out with a 'high TOC' TOC/DOC-analyzer (Elementar). From April 2004, anions and cations were measured by ion chromatography (IC, DX 100) and atomic adsorption spectrometry (AAS, Perkin Elmer 5000), while DOC was measured photometrically with a Technicon autoanalyser. The analgesic and antipyretic pharmaceuticals, namely dimethylaminophenazone, phenazone, propyphenazone, oxidative products of dimethylaminophenazone 1-acetyl-1-methyl-2-dimethyloxamoyl-2-phenylhydrazide (AMDOPH) and 1-acetyl-1-methyl-2-phenylhydrazid (AMPH) and the antiepileptic drug carbamazepine were analysed among other high polar pharmaceutical residues. The determination of the PhACs was performed by the laboratory of the BWB with a method based on a solid phase extraction of the analytes on RP-C18 materials using an automated extraction system (Zuehlke et al., submitted, a). Liquid chromatographic separation coupled with mass spectrometry was used for the detection of the analytes. To obtain both high sensitivity and selectivity,

tandem mass spectrometry, recording multiple reaction monitoring chromatograms, was applied. The limits of quantification (LOQs) are 50 ng/l for all relevant PhACs. Using a suitable surrogate for quantification, recoveries were always around 100% (Zuehlke et al., submitted a).

RESULTS AND DISCUSSION

Hydraulic situation

The pond is operated in cycles. An operational cycle starts after the pond has been emptied and the pond bottom sands (the upper ~0.1 m) have been washed and cleaned of finer grained material. The pond is refilled and the pond level kept constant. The groundwater level below the pond rises due to infiltration of water and the hydraulic condition below the pond changes from unsaturated to fully saturated (reached when the groundwater-table is ~1 m or less below the pond). The infiltration rate is largest after recovery, when saturated conditions prevail. With time, the pond bottom becomes increasingly more clogged until, at some stage, the infiltration capacity decreases to an extent that the conditions below the pond besome unsaturated due to decreasing groundwater levels. As soon as this happens, the infiltration capacity decreases even more. The pond has to be redeveloped. The hydraulic regime is described and illustrated in detail in Greskowiak et al. (in press) and Massmann et al., (this volume).

Redox conditions of the infiltrate

The redox processes observed are highly dependent on the groundwater temperature (Figure 2). While native groundwater temperatures are around 10°C (data not shown), the temperature variations during artificial recharge are large. The water temperatures of both surface and groundwater drop to nearly zero °C in winter. In summer, the surface and groundwater temperatures close to the pond are 25°C and greater, due to the short travel times. Because redox processes are microbially catalysed, a temperature decrease leads to lower microbial activity and less O_2, NO_3^- and Mn^{2+} turnover (e.g. David et al., 1997). Prommer and Stuyfzand (2005) discovered during deep well injection of aerobic surface water that oxygen was completely reduced by pyrite oxidation when temperatures rose above 14°C. The pond water is saturated with oxygen (data not shown). In the midst of summer, all O_2 is removed before reaching the first groundwater well, while in winter O_2 is only partly reduced (figure 2). Nitrate reduction only takes place occasionally in summer and particularly towards the end of a saturated phase (data not shown). Hence, the Mn^{2+} appearance commences before NO_3^- has been fully removed and Mn^{2+} appears in the groundwater as soon as O_2 is fully removed (Figure 2). Iron was never detected in TEG365 or TEG366 and SO_4^{2-} reduction does not occur. Times with temperatures of > 15°C are marked in grey in Figure 2. It becomes clear that anaerobic conditions largely correspond to temperatures above ~15°C. Some deviations to this observation are due to the fact that the cyclic hydraulic changes have a minor influence on the redox regime. Detailed investigations in the variably saturated zone below the pond have demonstrated that the transient hydraulic conditions during the recharge cycles in summer have a strong influence on the redox chemistry in the seepage water but only to a minor extent in the groundwater (Greskowiak et al., in press). The extent of the influence of the hydraulic conditions, i.e. the recharge cycles, on the groundwater chemistry is discussed in Massmann et al. (this volume). The dependency on the hydraulic condition is not as profound as the temperature dependency. However, it seems that at least in TEG366, the O_2 concentrations sometimes peak on top of the seasonal variation towards the end of the unsaturated phases (Figure 2).

Redox dependency of pharmaceutically active compounds (PhACs)

The pharmaceuticals phenazone (average 0.34 ± 0.17 µg/L), propyphenazone (average 0.08 ± 0.04 µg/L) and carbamazepine (average 0.48 ± 0.19 µg/L), as well as the oxidative metabolite of dimethylaminophenazone named AMDOPH (Reddersen et al., 2002; average 0.28 ± 0.11 µg/L) were detected in the pond water while
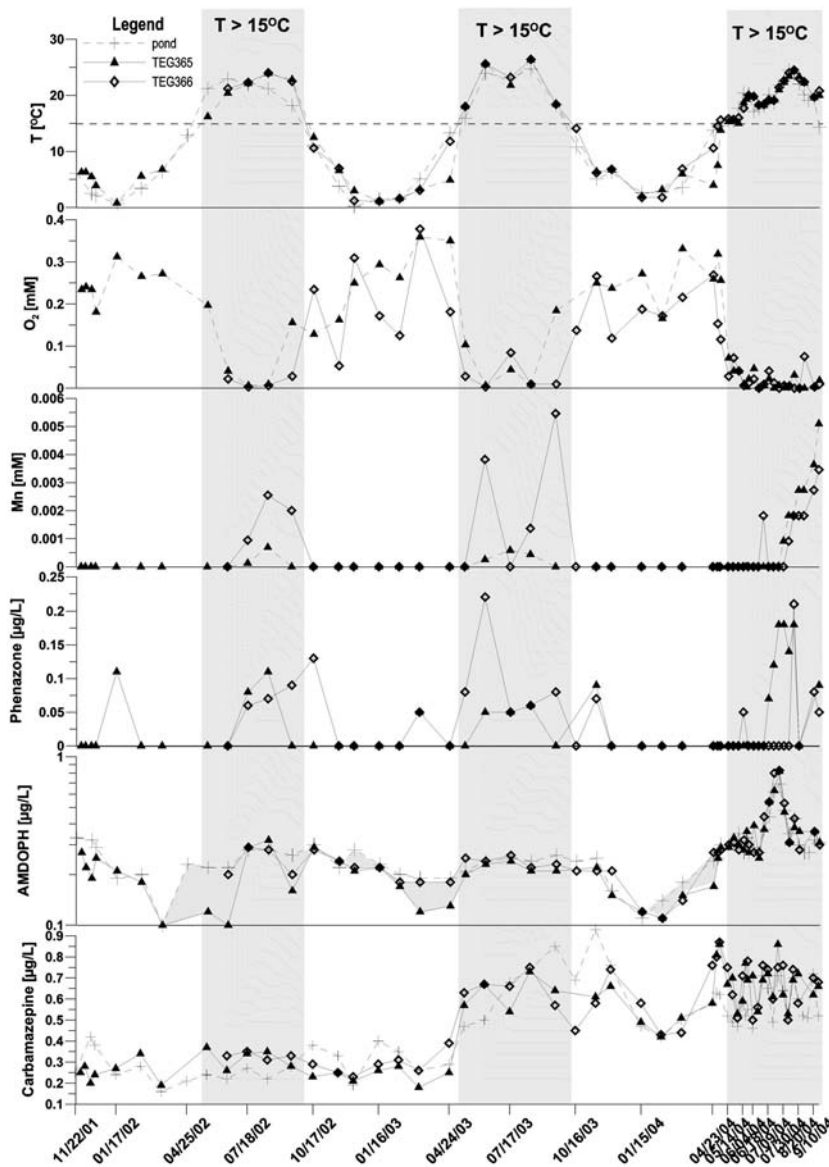
Figure 2. Temperatures and concentrations of O_2, Mn^{2+}, phenazone, AMDOPH (log scale) and carbamazepine in surface and groundwater.

Dimethylaminophenazone and its oxidative metabolite AMPH were not detected in any of the samples. Figure 2 shows the concentrations of the detectable PhACs phenazone, carbamazepine and AMDOPH in the infiltrate. Propyphenazone was not found in any of the observation wells at the transect and therefore has to be eliminated before reaching the first groundwater observation well (data not shown).

Phenazone appears to be a redox sensitive PhAC. The pond concentrations are generally fairly constant with the exception of a distinct peak of up to 1.2 µg/L in July 2004, whose origin remains unclear. Phenazone was fully degraded before reaching the first groundwater well as long as O_2 was present. When conditions turned anaerobic and traces of Mn^{2+} appeared in summer, phenazone was found to break through and could be detected in the groundwater wells (Figure 2).

Until a sudden increase of AMDOPH in the surface water of 0.69 µg/L in July 2004 (and consequently groundwater) of unclear origin AMDOPH concentrations in the surface water lay between 0.1 and 0.3 µg/L. So far, AMDOPH has been classified as a persistent and undegradable PhAC during sewage and surface water treatment (Zuehlke et al., submitted). It has been detected in observation and drinking water wells throughout Berlin where it is not removed efficiently during drinking water treatment in the presence of O_2 and an adapted microbiology (Reddersen et al., 2002). However, the time-series of AMDOPH in the pond and observation wells suggests that AMDOPH may be eliminated under certain favourable conditions. Areas with lower AMDOPH concentrations in groundwater than in the surface water are shaded in Figure 2. The presence of O_2 and high temperatures may have a positive effect on the removal of AMDOPH. Because O_2 is eliminated at higher temperatures, AMDOPH removal can only be observed in spring/early summer, when temperatures are reasonably high and O_2 is still present in the infiltrate. Another relevant factor could be the retention time within the clogging layer. At times with AMDOPH elimination, the amount of water recharged via the pond was low (data not shown: refer to Massmann et al., subm.). However, these assumptions merely indicate that under favourable conditions there could be a removal of AMDOPH and will need to be validated with appropriate laboratory experiments in future.

Carbamazepine is thought to be a relatively persistent PhAC. Ternes (1998) showed that it is not removed efficiently during drinking water treatment (<10%). During WW treatment in Berlin, Heberer and Reddersen (2001) reported a removal rate of 8%. Any elimination during bank filtration in Berlin has been ascribed mainly to dilution with background groundwater and only to a small degree to real removal (Heberer et al., 2004). In column studies performed by Mersmann et al. (2002), 15% of the carbamazepine was removed. Preuß et al. (2002) reported a high persistence of carbamazepine in artificial groundwater model systems under both aerobic and anaerobic groundwater conditions with elimination rates of <20%, which were ascribed to biodegradation rather than sorption processes. These results can be confirmed in the current study. Irrespective of the redox state, carbamazepine concentrations in the groundwater resembled those of the surface water. A removal during infiltration was not observed (Figure 2).

CONCLUSIONS

Redox conditions are highly transient during artificial recharge in Berlin and depend on a number of factors out of which seasonal temperature changes are most dominant. In winter, at low temperatures, aerobic conditions prevail while in summer, O_2 is rapidly consumed and nitrate and manganese reduction commences. The removal of phenazone is not complete when anaerobic conditions are present i.e. under water-saturated conditions during summer. AMDOPH may be degradable under favorable conditions (i.e. when aerobic conditions prevail; temperatures are relatively high while recharge rates are low). However, this will need to be verified in the future, since other studies showed that AMDOPH behaves conservatively. While numerous studies have reported the detection of PhACs in the groundwater (e.g. Stan and Linkenhägner, 1992; Heberer et al., 1998; Heberer, 2002a; Reddersen et al., 2002; Derksen et al., 2004), only few have linked the PhAC removal and behaviour to the hydrochemical conditions at a site. Daughton (2001) points out that the fate of PhACs in groundwater is poorly understood. On the basis of current knowledge, more attention should be paid in future to the interrelation of PhACs' occurrence with the hydrochemical conditions, particularly the redox conditions, in a natural environment.

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Fate and transport of pharmaceutical residues during bank filtration

Andy Mechlinski and Thomas Heberer

Abstract

The interest in natural surface-water treatment techniques such as bank filtration and artificial ground water replenishment has increased with the growing worldwide need for clean drinking water. After detecting a number of pharmaceutical residues in groundwater samples from a bank filtration site in Berlin, Germany, the research on these compounds has focused on investigating their transport behavior during the infiltration process. In the studies presented in this paper, the fate of six pharmaceutical residues detected at concentrations up to the µg/L-level in Berlin's surface waters was investigated. During bank filtration, the analgesic drugs diclofenac and propyphenazone, the antiepileptic drugs carbamazepine and primidone and the drug metabolites clofibric acid and 1-acetyl-1-methyl-2-dimethyl-oxamoyl-2-phenylhydrazide (AMDOPH) were found to leach from the surface water into the groundwater aquifers. They also occur at low ng/L-concentrations in the receiving water-supply wells. Other compounds namely the antiphlogistic drug indometacine and the blood regulating drug bezafibrate which are also detected at concentrations up 100 ng/L in the surface water are efficiently removed by bank filtration. Thus, they have not been detected downstream of the first two monitoring wells. In conclusion, bank filtration was found to decrease the concentrations of some drug residues (e.g. of diclofenac, carbamazepine) or even to remove selected compounds (e.g. bezafibrate, indometacine). However, a complete removal of all potential pharmaceutical residues by bank filtration cannot be guaranteed.

Keywords

NASRI, drug residues, bank filtration, GC/MS, surface water, ground water.

INTRODUCTION

Worldwide, pharmaceuticals are used in high amounts in human and animal medical care. The high production volume of pharmaceuticals and an often incomplete metabolism of administered drugs in the target organism results in often very polar and highly persistent residues. These residues of pharmaceuticals and their metabolites are excreted via urine or faeces and are discharged via municipal sewers to the receiving sewage treatment plants (STP's). Because of their polar and highly persistent properties, several of the compounds can pass the STP's and are finally released into the receiving surface waters. Since the mid 90s, when the occurrence of several PhAC's in different compartments of the aquatic environment has been reported by several authors (Heberer et al., 1997; Halling-Sørensen et al., 1998; Daughton et al., 1999), these compounds have now been recognized as an emerging issue in environmental chemistry but also as being relevant for public drinking water supply. Bank filtration, used for ground water recharge in drinking water production, is known as an important, effective, and cheap pre-treatment technique for the removal of microbes, inorganic and also some organic contaminants from contaminated surface waters. However, its purification capacity varies and is limited to selected contaminants. After detecting a number of pharmaceutical residues in groundwater samples from a bank filtration site in Berlin, Germany, the research on these compounds has focused on their transport behavior during the infiltration process. These investigations were part of the interdisciplinary research project entitled NASRI (Natural and artificial systems for recharge and infiltration) that was initiated in May 2002.

Figure 2 shows the scheme of the transect at lake Tegel. It consists of eight monitoring wells (marked as 3305–3311 and TEG372) reaching into the shallow aquifer, five monitoring wells screened in the deeper aquifer (marked as 3301–3304 and TEG374), one multi-level well (TEG371OP/UP) screened at different depths to monitor the water in both aquifers and the receiving water supply well no. 13. Additional information on hydrogeological and hydrogeochemical processes are available and will be published soon (Massmann, in prep.).

Surface and ground water from all transects were sampled between May 2002 and August 2004. The samples were analyzed using two multi methods applying solid-phase extraction, chemical derivatization with pentafluorbenzylbromide and gaschromatography-masspectrometry (GC/MS) using selected ion monitoring (SIM).In total, residues of nineteen pharmaceuticals were detectable when using both methods as described elsewhere (Reddersen et al., 2003).

RESULTS AND DISCUSSION

In terms of the investigations, residues of six pharmaceuticals or their metabolites including AMDOPH (1-acetyl-1-methyl-2-dimethyl-oxamoyl-2-phenylhydrazide), bezafibrate, carbamazepine, clofibric acid, diclofenac, indometacine, primidone and propyphenazone were found up to the µg/L-level in the surface water used as raw water resource for bank filtration.

AMDOPH, is a metabolite of dimethylaminophenazone, a formerly used antiphlogistic/analgesic drug that has in the meantime been prohibited from prescription. As shown in Table 1, AMDOPH was detected with median concentrations of 185 ng/L in surface water at transect Wannsee II. In the shallow monitoring wells near the lake (BEE205 and BEE206) median concentrations between 133 ng/L and 148 ng/L were observed. However, due to historical reasons (Reddersen et al., 2002), the deeper wells of the multi-level well BEE202OP/MP1/MP2/UP contained higher median concentrations of AMDOPH with concentrations of up to 870 ng/L in monitoring well BEE202MP2. In the receiving water supply well (no. 3) it was measured with a median concentration of 375 ng/L. The water in this well represents a mixture of background ground water, 'young' bank filtrate from the shallow monitoring wells and 'older' more contaminated bank filtrate as it was monitored at the deeper monitoring wells of the multi-level well.

Table 1. Median concentrations [ng/L] of AMDOPH and carbamazepine at transect Wannsee II. Sampling campaign from January 2003 until August 2004

median concentration [ng/L]	Surface water	BEE205	BEE206	BEE202 OP	BEE202 MP1	BEE202 MP2	BEE202 UP	BEE203	Well 3	BEE204 UP	BEE204 OP
	n=24	n=22	n=24	n=24	n=18	n=16	n=20	n=24	n=20	n=19	n=15
AMDOPH	185	133	148	49	480	870	750	105	375	220	15
Carbamazepine	380	328	340	218	163	35	18	260	65	10	n.d.

n.d.:not detectable

At transect Tegel, for AMDOPH a median concentration of 328 ng/L was detected in the surface water (Table 2). Similar values were also detected in water samples collected from the shallow monitoring wells.

Table 2. Compounds with positive findings and their median concentrations [ng/L] at transect lake Tegel. Sampling campaign from May 2002 until April 2004

median concentrations [ng/L]	Surface water	3311	3310	3308	TEG371 OP	TEG371 UP	TEG372	Well 13	3305
	n=26	n=16	n=11	n=16	n=17	n=18	n=18	n=22	n=6
AMDOPH	328	275	290	170	260	358	180	1165	78
Carbamazepine	583	493	520	510	440	183	345	80	3

n.d.:not detectable

Again, AMDPOH was found at higher concentrations in the deeper monitoring wells (3301-3303) (Table 3). Median concentrations between 308 ng/L and 490 ng/L were detected in these monitoring wells. The highest median concentration of 6.7 µg/L was detected in the deepest monitoring well (TEG374) screened in a depth around 38 m below the ground.

Table 3. Compounds with positive findings and their median concentrations [ng/L] at transect lake Tegel.
Sampling campaign from May 2002 until August 2004

median concentrations [ng/L]	Surface water	3301	3302	3303	TEG374	Well 13	3304
	n=26	n=27	n=26	n=21	n=8	n=22	n=27
AMDOPH	328	490	308	340	6710	1165	220
Carbamazepine	583	235	295	250	n.d.	80	n.d.

n.d.:not detectable

The antiepileptic drug carbamazepine, another compound which was identified as being both highly persistent and very mobile, was observed with a median concentration of 380 ng/L in the surface water at transect Wannsee II and median concentrations between 328 ng/L and 340 ng/L in the shallow monitoring wells BEE205 and BEE206. The median concentrations observed at multi-level well BEE202 are decreasing with depth starting with 218 ng/L at BEE202OP down to 18 ng/L at the deepest well (BEE202UP). At the water supply well no. 3, a median concentration of 65 ng/L was observed for carbamazepine. The water in well no. 3 represents a mixture of 'young' bank filtrate containing high concentrations of carbamazepine, of almost residue-free background ground water and of 'older' bank filtrate with almost non detectable concentrations of carbamazepine.

In front of transect Tegel, a median concentration of 583 ng/L of carbamazepine in the surface water was observed. At the shallow monitoring wells 3311, 3310 and 3308, it was found with median concentrations between 493 ng/L and 520 ng/L. As presented in Table 3, the deeper monitoring wells showed lower median concentrations down below the limit of detection (well TEG374). A median concentration of carbamazepine of 80 ng/L was observed at water supply well no. 13 that receives background ground water and both kinds of bank filtrate, younger bank filtrate as investigated in the shallow monitoring wells and "older" bank filtrate as observed in the deeper wells.
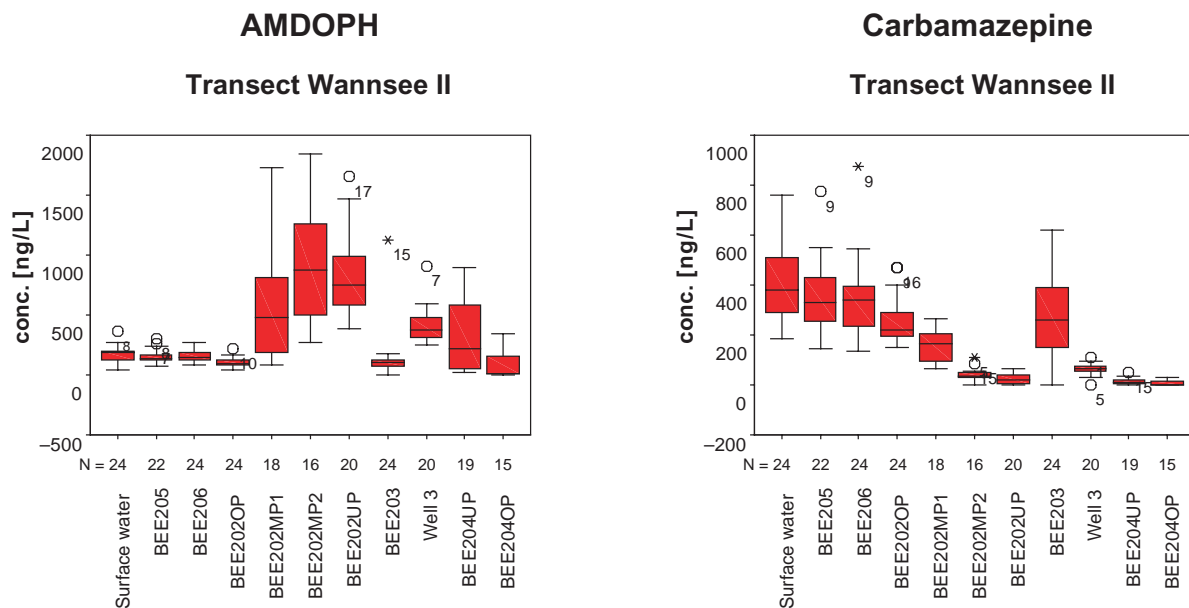
Figure 3. Boxplots showing the concentrations of AMDOPH and carbamazepine as measured at the different monitoring wells at transect Wannsee II

In Figure 3, the distribution of the detected AMDOPH and carbamazepine residues in the different monitoring wells and the multi-level well BEE202 at transect Wannsee II are presented by their concentrations. The figure shows that AMDOPH concentrations are increasing in the deeper wells whereas carbamazepine concentrations are decreasing. As shown in Figure 4, the median concentrations of carbamazepine indicate only a slightly attenuation of this compound of 11%–14% between the surface water and those monitoring wells (BEE205, BEE206) located underneath the lake. However, between the surface water and the receiving water supply well no. 3, an attenuation rate of 83% was observed for carbamazepine.
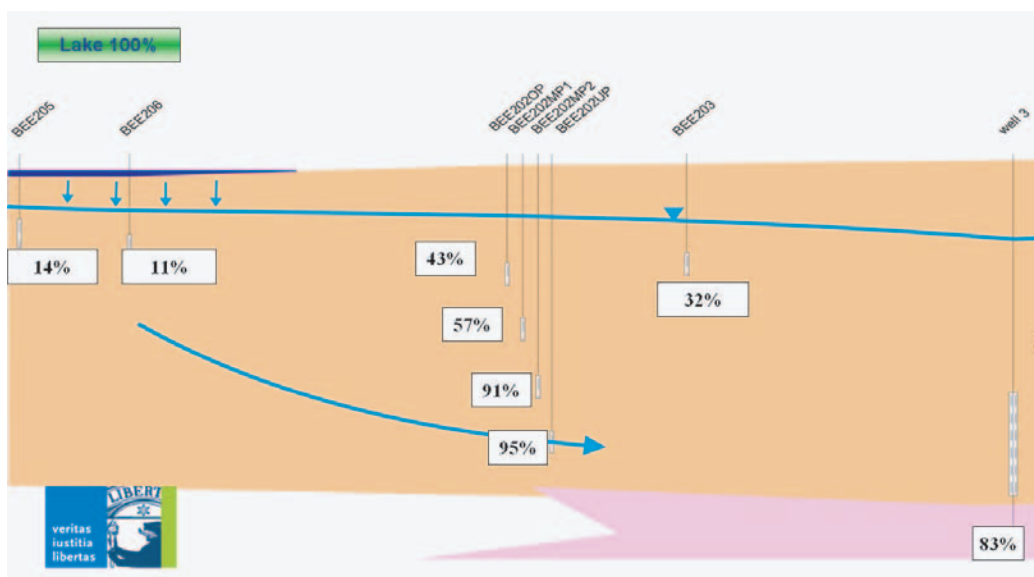
Figure 4. Attenuation rates [%] of carbamazepine median concentrations at transect Wannsee II
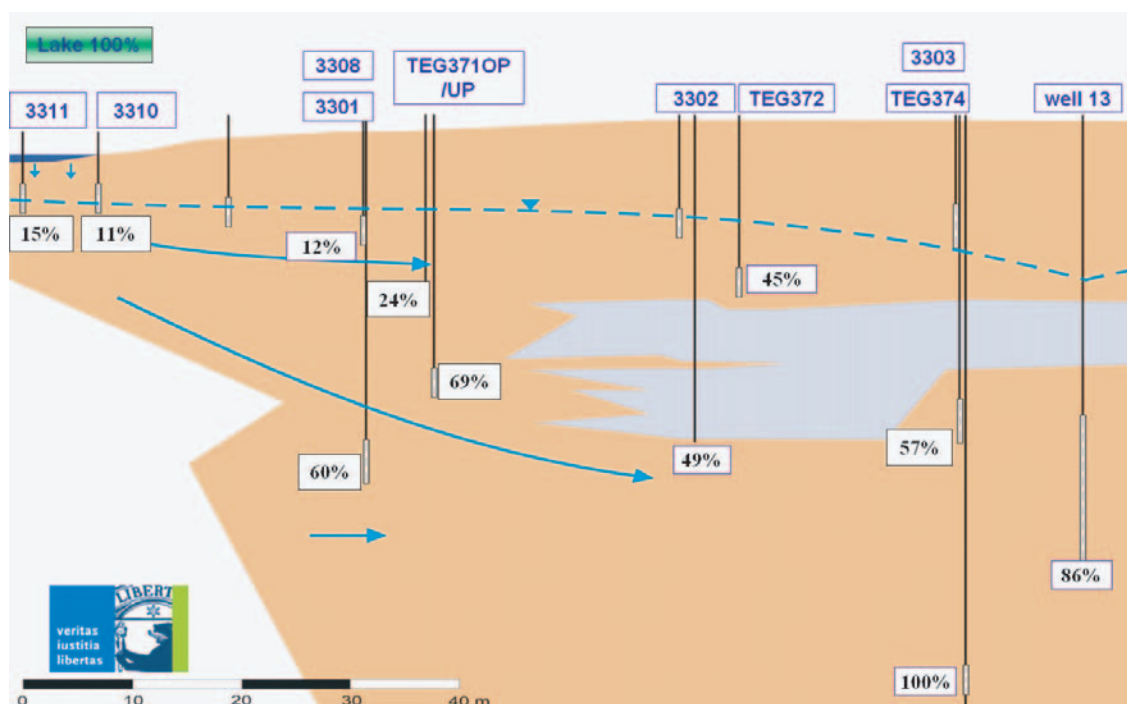
Figure 5. Box plots of AMDOPH and carbamazepine concentrations at the monitoring wells at transect Tegel. The box plot of AMDOPH does not include the deepest monitoring well TEG374 (median: 6,710 ng/L) because of increasing resolution for significantly lower concentrations at the other monitoring wells

As shown in Figure 6, attenuation rates between 11% and 15% were observed for carbamazepine by comparing the median concentrations measured in the surface water and the shallow monitoring wells 3308, 3310 and 3311. At the water supply well no. 13, carbamazepine was attenuated by more than 85%.
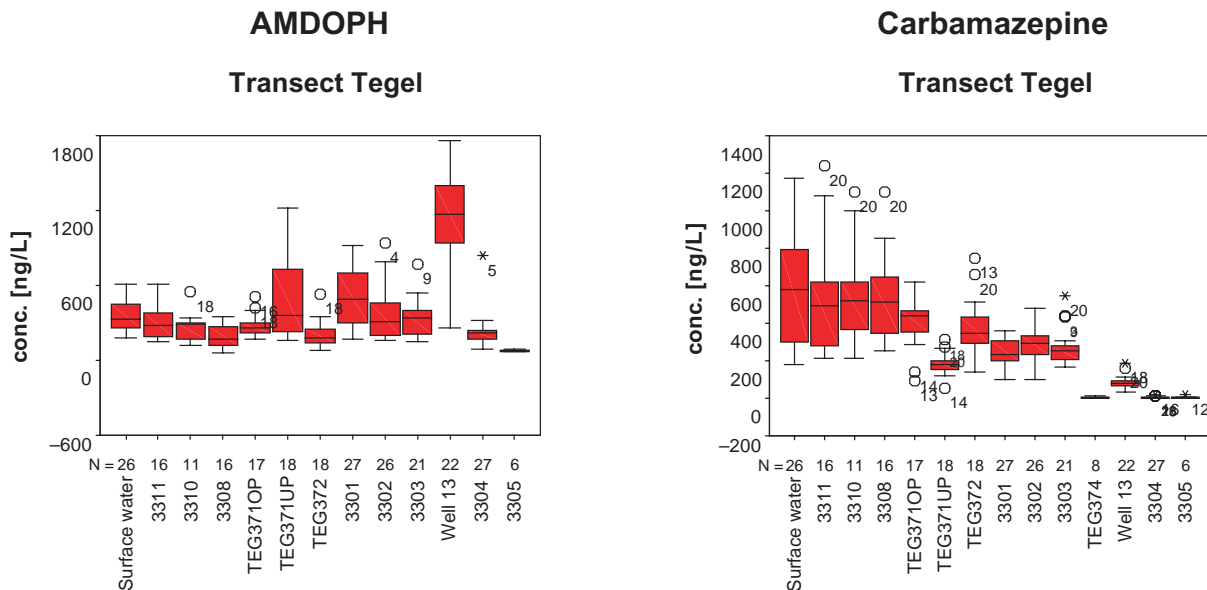
Figure 6. Attenuation rates [%] of carbamazepine median concentrations at transect Tegel

CONCLUSIONS

During bank filtration as conducted at lakes Wannsee and Tegel in Berlin, Germany, several drug residues including the analgesic drug diclofenac and the antiepileptic drugs carbamazepine and primidone are leaching from contaminated surface water into the ground water aquifers. They also occur at low concentrations in receiving water-supply wells. Indometacine (antiphlogistic drug) and bezafibrate (blood lipid regulator) are efficiently removed and were not detected downstream of the first two monitoring wells. However, they were detectable at concentrations up to 100 ng/L in the surface water. The drug metabolites AMDOPH and clofibric acid and the analgesic drug propyphenazone were measured with higher concentrations in some monitoring wells than those detected in surface water. As far as occurrence of AMDOPH and propyphenazone is concerned, this phenomenon could be explained by an old environmental plume (Reddersen, 2002). Additionally the detected concentrations of AMDOPH and propyphenazone at deeper parts of the aquifers allow the conclusion that these compounds are highly persistent under anoxic/anaerobic conditions. The higher concentrations of the drug metabolite clofibric acid in some monitoring wells compared to the surface water are the result of higher application rates of the original (precursor) pharmaceuticals in the past as well as the high persistence of its residues.

In conclusion, bank filtration has proven as being a useful technique for surface water pre-treatment and is also able to remove some of the pharmaceutical residues such as indometacine and bezafibrate. Diclofenac and carbamazepine were only partly removed during the infiltration process. For historical reasons, other highly persistent residues including AMDOPH, clofibric acid and propyphenazone appear at higher concentrations in some monitoring wells compared to those found in surface water.

ACKNOWLEDGMENTS

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Fate of bulk organics during bank filtration of wastewater-impacted surface waters

Steffen Grünheid and Martin Jekel

Abstract

More than two years of monitoring data from a bank filtration site in Berlin, Germany, and a long retention soil column system (30 m) were analyzed to study the influence of redox conditions on the degradation of bulk organics. Dissolved organic carbon (DOC), UV-absorption at 254 nm (UVA_{254}) and liquid chromatography with online carbon detection (LC-OCD) was employed to receive qualitative and quantitative information about the fate of different fractions of DOC. It was found that the kinetics of DOC-degradation depend significantly on the dominant redox conditions during infiltration. A faster mineralization of biodegradable DOC was observed during oxic soil passage (~1 month). Anoxic infiltration led to a comparable residual DOC-concentration, but 3-6 months were required for complete removal of biodegradable DOC (BDOC). LC-OCD measurements revealed that the fraction of polysaccharides (PS) is removed very fast during infiltration in the field. Under strictly anoxic conditions the PS were more stable. The fractions of humic substances, building blocks and low molecular weight acids were degraded partially, independently from the redox potential, while the change in aromaticity of the residual DOC was influenced by the dominant redox conditions.

Keywords

Bank filtration; soil columns; DOC; LC-OCD; SUVA.

INTRODUCTION AND OBJECTIVES

The issue of safe and stable drinking water supply is addressed around the world with different treatment technologies. One possibility of growing interest is the technology of bank filtration. In Europe, river- and lake- bank filtration sites have been operated for over a century to provide save drinking water to cities like Zurich, Düsseldorf and Berlin. In the city of Berlin 56% of the drinking water is derived from bank filtration and 14% from artificial recharge (BWB, 2003). The water utility is using well operated bank filtration and artificial recharge facilities to provide high quality water that needs little further treatment. With the growth of the city a semi-closed urban water cycle has been managed. At some bank filtration sites the surface water is strongly influenced by well-treated domestic waste water effluent (e.g. 15–30% in Lake Tegel) (Ziegler, 2001). Despite this indirect potable reuse, the bank filtration system is providing high quality water, which is distributed without chlorination. This unique situation in Berlin was considered an interesting field site for a research project of the Berlin Centre of Competence for Water. The objectives of the NASRI-project (Natural Systems for Recharge and Infiltration) were to investigate the fate of microbiological and chemical loads of the surface water during infiltration (NASRI, 2003). This paper will show results of the Department of Water Quality Control at the Technical University of Berlin (DWQC) on the removal of bulk organic matter during bank filtration. Additionally, the DWQC utilizes soil column systems to isolate important factors of influence for the degradation of bulk organics.

METHODS

The field studies on the fate of bulk organics were conducted at a bank filtration site in Berlin-Tegel. Additionally,

results of experiments on a long retention column system which was operated under oxic and anoxic conditions will be presented. The monthly analytical program which is discussed in this paper contained DOC, UVA and LC-OCD analysis. Additionally, all samples were analyzed for general hydrochemistry parameters to characterize the infiltration process more detailed.

- *Field site*

The examined field site is operated with water from Lake Tegel which is influenced by the discharge of the waste-water treatment plant Schönerlinde. Based on LC-OCD analysis, the DOC-composition of Lake Tegel is characterized by high values (~7.5 mg/l), of a mostly natural origin: fulvic acids associated with the background groundwater (4–5 mg/l DOC), effluent organic matter with a contribution of 1–2 mg/l DOC and algal cellular products. The dominant soil type at the Lake Tegel bank filtration site is sand. In the direct infiltration zone near the lake the hydraulic permeability of the clogged sand ($k = 5 * 10^{-6}$ m/s) (Fritz et al. 2002) is lower than in the deeper aquifer ($k = 2 - 8 * 10^{-4}$ m/s) (Pekdeger et al. 2004). The sampling points for the bulk organic analysis at the field site were selected to reproduce the flow path of the infiltrating water to the production well. The individual retention times for each monitoring well were determined by the shift of the seasonal variation of the ratio of oxygen isotopes ($\delta^{18}O$) in the surface water (Pekdeger et al. 2004). At the bank filtration site some dilution with deeper groundwater was observed and quantified with the help of boron data (Pekdeger et al. 2004). Data for retention times and dilution with deeper groundwater are included in Figure 1. A detailed description of the field site and a distribution of the used monitoring wells are available at Pekdeger et al. (2004).
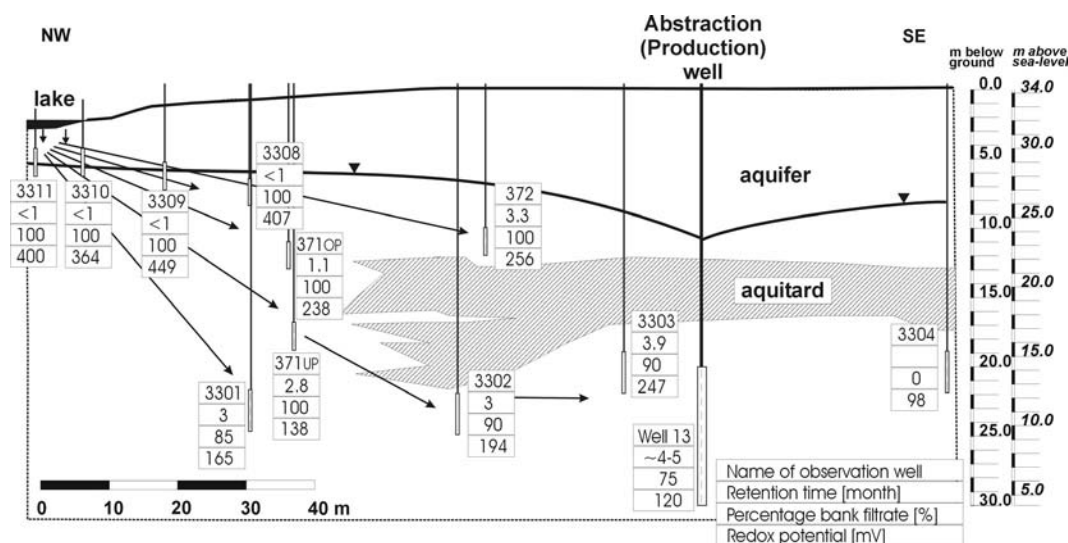
Figure 1. Cross section of bank filtration site at Lake Tegel (modified and reproduced from Pekdeger, 2004)

At the bank filtration site it was found that most of the lake water is initially infiltrating under oxic conditions. Because of mineralization of DOC and sedimentary bound POC (~0.5% w/w) during infiltration oxygen is used up quickly and most of the 4–5 month long infiltration (100 m) to the production well is usually taking place under anoxic and anaerobic conditions (iron and manganese reduction). The extension and position of the redox zones varies seasonally and is horizontally stratified (Pekdeger et al. 2004). The average nitrate concentration in Lake Tegel is 1.83 mg/l NO_3–N. During the first sixteen months (05/02 – 09/03) of the observation period the nitrate level was significantly reduced during infiltration (Average NO_3–N: 3302: 0.07 mg/l and 3303: < LOQ).

The summer of 2003 was very dry in Berlin and the combination of the low water level in Lake Tegel and the

heavy pumping of the production well pumps led to an expanded zone of unsaturated infiltration under the lake. After passage of the biologically very active water-sediment interface the bank filtrate was aerated again during a short unsaturated infiltration. Beginning in October 2003 until June 2004 elevated oxygen and nitrate concentrations were observed in the deeper monitoring wells and the production well ($\varnothing$-NO$_3$–N: 3302: 1.16 mg/l and 3303: 0.47 mg/l). During these nine month the dominant redox conditions at the field site changed from anoxic/anaerobic with Fe/Mn-reduction to oxic. The differences in the fate of the bulk organics between the two redox conditions are investigated.

• *Column system*

A long retention soil column system was used to simulate a one dimensional aquifer. The study was conducted on a 30 m column system, which was operated with spiked water from Lake Tegel and had a retention time of 30 days (Figure 2). The columns were filled with natural fine sand from the Berlin area. Flow and hydraulic loading rate were selected to fit the conditions at the field site. The columns were sampled at different depths to investigate the kinetics of the bulk organic degradation.
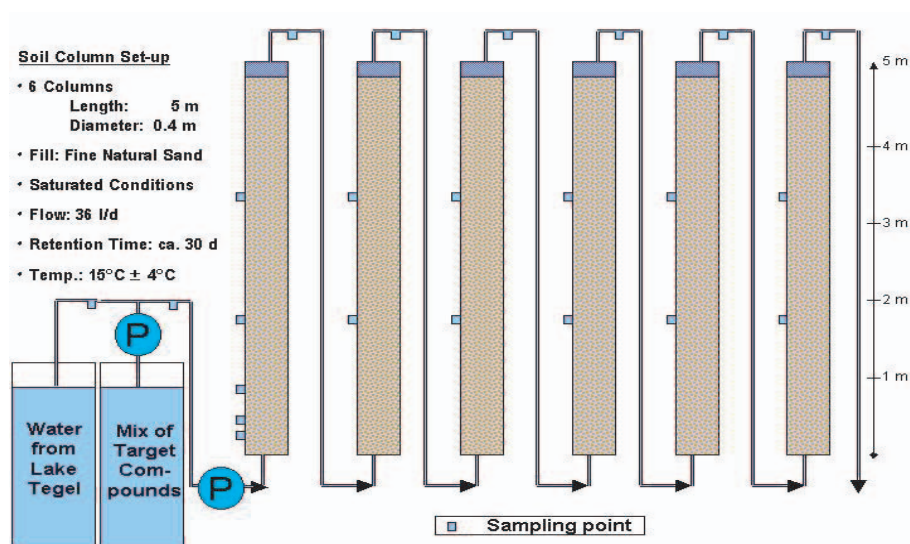
Figure 2 Experimental set-up of the long retention columns

During the first part of the experiment oxic conditions were established in the soil columns. The oxic period lasted twelve months until May 2004. During this period oxygen was present in the whole column and no denitrification was observed. From May 2004 the redox status of the column system was switched to anoxic (NO$_3$– reducing) by sparging the influent with nitrogen (O$_2$ in influent: ~1 mg/l). Denitrification was observed but nitrate was not completely depleted from the system. No oxygen was detectable in the sampling ports. Monitoring of the DOC, UVA and redox-conditions ensured stable conditions over the time. Results from both phases will be presented.

Analytics

At the field site the surface water, monitoring wells, and abstraction (production) well were sampled monthly over a period of 27 months for the bulk organic parameters. The wells were purged for 30 minutes at pumping rates of 12 m^3/h before sampling to ensure stable conditions (on line measurement of temperature, redox potential, dissolved O$_2$, pH, and conductivity). Afterwards the samples were stored in glass bottles at 4°C in the dark and if necessary filtered through 0.45 µm cellulose nitrate membrane filters. DOC was determined after 0.45 µm filtration

with an ELEMENTAR HighTOC analyzer (Hanau, Germany). UV absorbance at 254 nm (UVA_{254}) was measured using a PERKIN-ELMER Photometer Lambda 12 (Berlin, Germany). For the LC-OCD, a liquid chromatography method that quantitatively distinguishes between different fractions of DOC, a system of the DOC-Labor Dr. Huber (Karlsruhe, Germany), was used (Huber and Frimmel 1996). The system uses a macroporous resin to achieve a separation of different sized DOC-fractions. The fractions are detected by UVA and an online DOC-analyzer. SUVA was calculated as the quotient of UVA_{254} and DOC-concentration.

RESULTS AND DISCUSSION

Results of 2¼ years of field monitoring at Lake Tegel and more than two years of column studies are summarized in this paper. The results are presented with the focus on the influence of the redox conditions on the removal of bulk organics.

Dissolved organic carbon (DOC)

The monthly DOC-monitoring (05/2002–08/2004) for the bank filtration site confirmed stable surface water concentrations in Lake Tegel of 7–7.5 mg/l. The results of the monitoring well sampling are plotted in Figure 3a as box plots with the median value (horizontal centre line), the 25%- and 75%-quartiles (box) and the minimum and maximum values (vertical line). The removal rates are calculated from the arithmetic means of all measurements. Under oxic and anoxic/anaerobic conditions the DOC is reduced by 2.8 and 2.6 mg/l from the surface water to the monitoring well 3303, respectively. The degradation kinetic indicates a more rapid removal of DOC under oxic conditions and a slower but continuing removal under anoxic/anaerobic conditions. Under oxic conditions 35% of the DOC is degraded from the surface water to the first deep monitoring well 3301. During further infiltration the fraction of degraded DOC is increasing by only 5% (3303) or 9% (Well 13, includes some dilution). Under anoxic/anaerobic conditions 24% of the initial DOC is degraded in monitoring well 3301. The continuing removal of DOC leads to reduction rates of 34% (3303) and 42% in the production well (Figure 3). Comparing the concentrations of DOC during oxic and anoxic/anaerobic infiltration reveals that under both conditions similarly 3.1 mg/l DOC were removed between the surface water and the production well. The dilution in the production well is assumed to be constant (25% background groundwater) and the background monitoring well 3304 shows low DOC-concentration during the observation period (oxic: 2.5 mg/l; anoxic/anaerobic: 2.9 mg/l). Under anoxic/anaerobic conditions the slower process of DOC-mineralization demands the entire retention time, whereas under
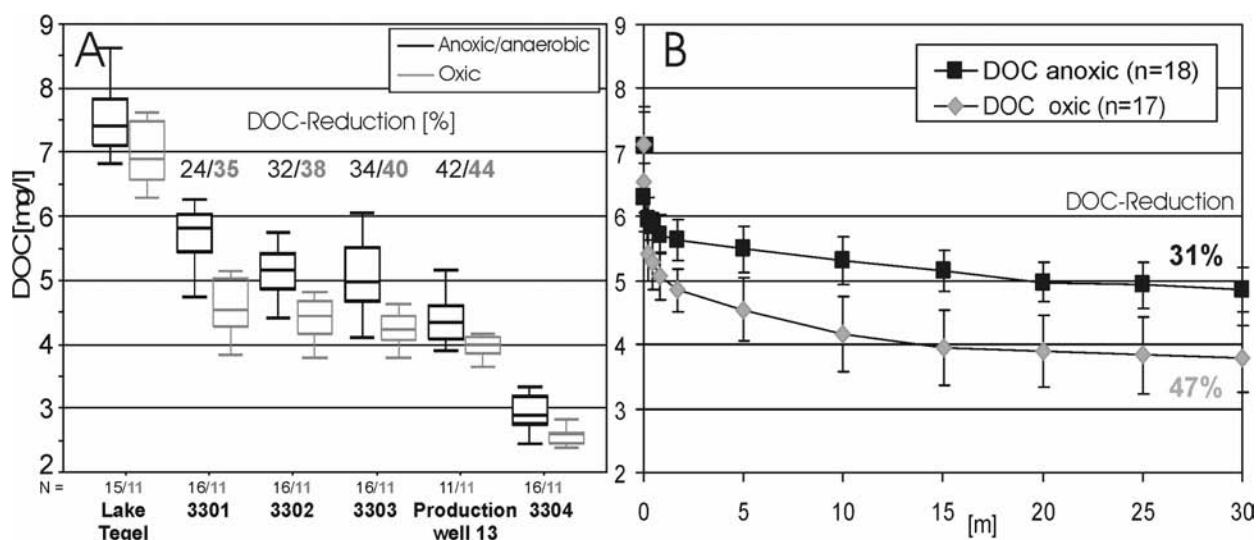
Figure 3 A. DOC at bank filtration site for two observed redox conditions / B: DOC of column experiment

oxic conditions the efficient removal during the initial infiltration is followed by a plateauing of the DOC levels and a slower removal. Under both conditions a similar residual DOC concentration can be achieved, if sufficient retention time is allowed under anoxic/anaerobic conditions.

These results are consistent with finding of the soil columns experiments (Figure 3b). The simulation experiments allowed a more precise investigation of the fate of DOC during the first month of infiltration depending on the dominant redox conditions. In the oxic soil columns 47% of the DOC was removed. Under anoxic conditions the initial oxic phase which occurs invariably in the field because of oxygen saturated surface water was reduced to the lowest practicable dimensions (<0.21 m). There only 31% of the source DOC was removed during 30 days of infiltration. The results support the field results that the DOC removal under oxic conditions is faster than under anoxic or anaerobic conditions. The removal rates are within the range of the rates observed in the field. A slightly higher removal can be explained by a higher average water temperature in the soil columns (15°C, field: 12°C). Because of their limited retention time the column experiments could not demonstrate that the residual not-degradable DOC-concentration is similar under oxic and anoxic/anaerobic conditions.

LC-OCD

Figure 4 shows LC-OCD chromatograms of samples from the bank filtration site, and indicates that the character of the DOC partly changed after infiltration. The first peak corresponds to the largest molecular weight fraction, interpreted as polysaccharides (PS; elution time at 35-45 min), the second peak corresponds to the humic substances (HS; elution time 52 min) and HS building blocks (secondary peak; 57 min), and the third to low molecular weight acids (LMA; 62 min). From the LC-OCD measurements of the field samples (Figure 4) no distinct differences between the oxic and the anoxic/anaerobic stage could be assessed. Under all conditions the fraction of polysaccharides was completely removed, whereas other fractions (humics, humic hydrolysates) exhibited only partial removal.
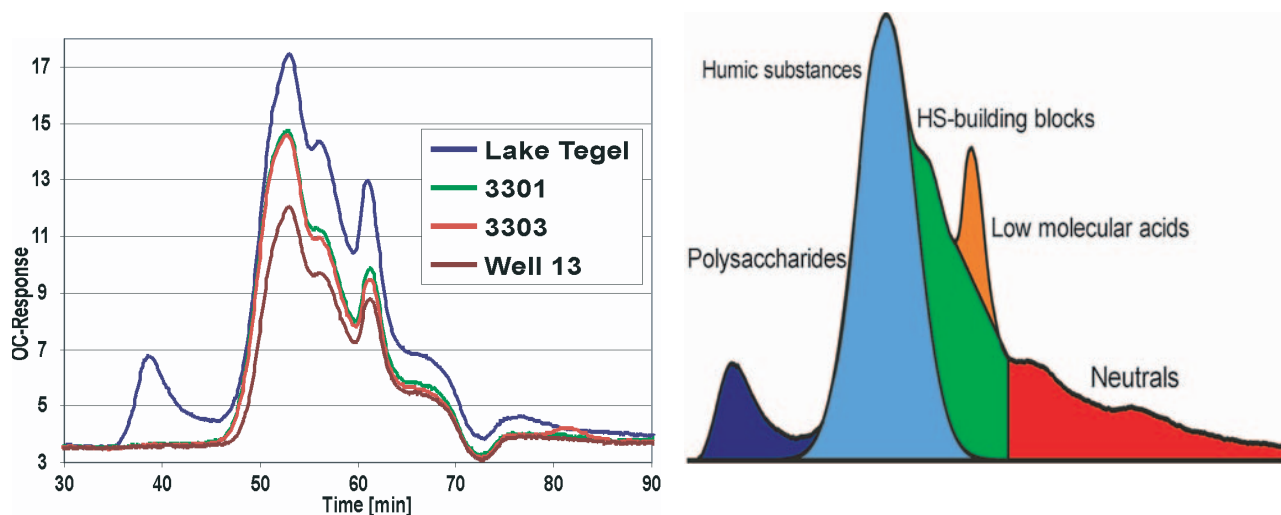
*Figure 4. LC-OCD chromatogram of bank filtration site (May 2003, anoxic period)
and explanation of LC-OCD chromatogram*

It is assumed that this fraction of polysaccharides is mineralized instantly in the oxic infiltration zone, because at no time PS were detected in the first observation well (3311). The results of the soil column experiments support the assumption of a rapid biodegradation for this DOC-fraction, because in an abiotic column only minor filtration effects were observed. Generally, LC-OCD chromatograms of the field site were comparable to the chromatograms
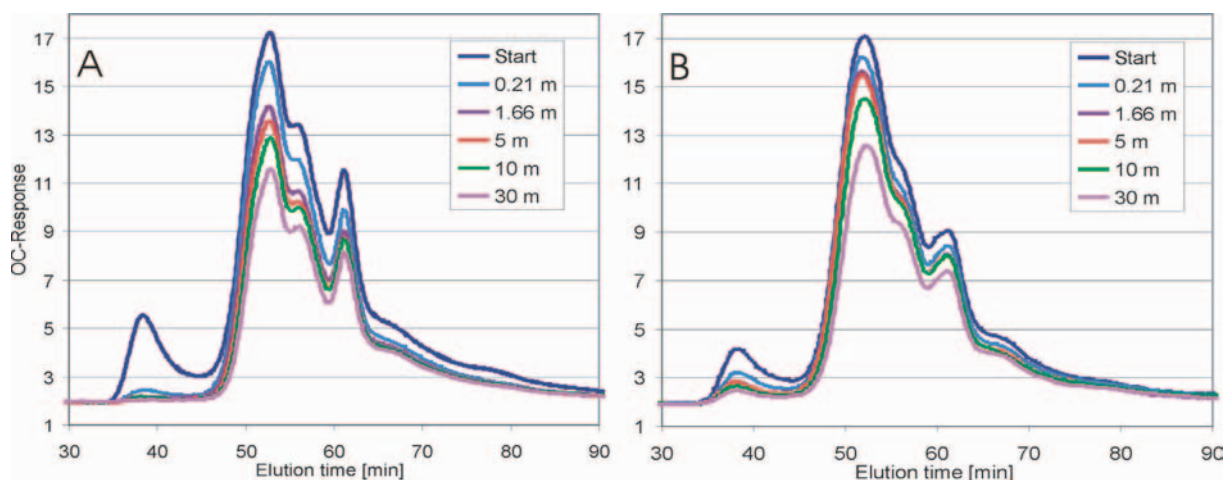
Figure 5a/b. LC-OCD chromatogram of soil column experiment – A: oxic / B: anoxic

of the oxic period of the long retention column (Figure 5a). Both, in the field and in the columns the most funda-mental change during infiltration was observed for the fraction of the polysaccharides. PS were efficiently removed in the first 0.21 m of oxic infiltration in the soil columns (Figure 5a). Furthermore, the fractions of HS, HS building blocks and LMA were only partially degraded. It is noteworthy that even a part of the HS peak was removed, gener-ally assumed to be mostly nondegradable.

During the anoxic period of the column system the portion of HS-building blocks and LMA on the total DOC of the source water decreased and the peak intensity of the corresponding peaks was reduced (Figure 5a/b). However, a slower degradation of the polysaccharide fraction was observed under anoxic conditions in the column system. PS were still detectable after 30 days of anoxic infiltration (Figure 5b). The fractions of HS, HS building blocks and LMA showed a similar partial removal as under oxic conditions. Because of the slower DOC degradation kinetics the chromatograms are more pooled together in the anoxic period. Despite of the PS-fraction, a difference in the degradability of the single fractions under oxic or anoxic/anaerobic conditions was not observed.

SUVA

The specific UV-absorbance (SUVA) was calculated using the DOC-results and the UVA_{254}-measurements. SUVA-data reveal more details about the mechanisms of DOC-mineralization because a change in SUVA during treatment or infiltration indicates a preferred removal of aliphatic (SUVA-increase) or aromatic (SUVA-decrease) carbon sources. The SUVA of the surface water in Lake Tegel is around 2.1 l/m ∗ mg. Seasonal variations of the PS-concentrations lead to slightly higher SUVA results during winter. During infiltration at the bank filtra-tion site an initial increase to 2.4 l/m ∗ mg of the SUVA was observed, but during subsequent soil passage the SUVA decreased again to 2.1 l/m ∗ mg. In the oxic soil column experiment the SUVA increased very rapidly to 2.38 l/m ∗ mg after 0.21 m infiltration (Figure 6). The results indicate that during initial aerobic soil passage
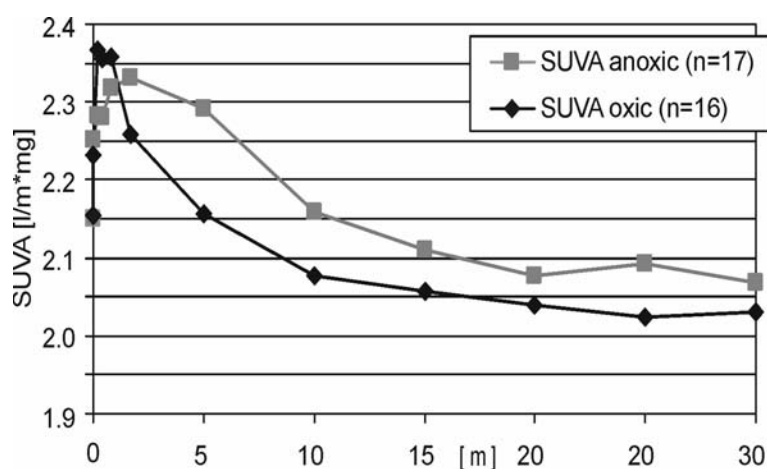
Figure 6 SUVA results of the soil column experiment

aliphatic carbon sources are preferentially used. LC-OCD measurements revealed that the changes in SUVA result from the fast removal of the PS-fraction and from a change of the aromaticity in the HS-fraction. In the online measurement the HS-fraction showed major changes in SUVA, while HS-building blocks and LMA were constant. It is assumed that in the HS-fraction aliphatic side-chains are very fast mineralized, but aromatic and double-bond structures remain longer unchanged. During further infiltration more aromatic structures in the HS fraction are degraded and the SUVA decreases. During anoxic infiltration the increase in SUVA is slower. LC-OCD measurements proved that under anoxic conditions the slower removal of PS is responsible for the slower rise of SUVA. In the HS-fraction no initial preferential removal of aliphatics was observed and the SUVA remained stable during the first 1.6 m of infiltration. Afterwards, the mineralization of aromatics from the HS-fraction leads to a decreasing SUVA. However, the results indicate that aliphatic compounds are initially degraded under oxic conditions and subsequently aromatics. Under anoxic conditions the fate of aromatics and aliphatics seemed more balanced, excepting polysaccharides.

CONCLUSIONS

The results of the field monitoring program give new insights into the mechanisms of bulk and trace organic removal. Aerobic and anoxic subsurface conditions during infiltration can lead to approximately the same residual DOC but the study indicates significant differences in the kinetics of DOC-removal depending on the redox conditions. Aerobic conditions result in a fast degradation of BDOC within a month, while anoxic conditions appear to require 3 – 6 months. LC-OCD analysis reveals that the change in character is comparable. Under oxic conditions the fraction of polysaccharides is removed more efficiently than under anoxic conditions. The fractions of humic substances, building blocks and low molecular weight acids are degraded partially. The SUVA-results indicated that aromaticity of the residual dissolved organic carbon is increasing fast during initial infiltration. Afterwards, aromatic compounds are mineralized leading to a lower SUVA. Under anoxic conditions the described processes are slower and more balanced.

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Fate of trace organic pollutants during bank filtration and groundwater recharge

Steffen Grünheid and Martin Jekel

Abstract

Investigations on the behaviour of different trace organic compounds at a bank filtration site at Lake Wannsee in the city of Berlin, Germany are reported. More than two years of monitoring for the bulk parameter differentiated adsorbable organic halogens (AOX) revealed a more efficient degradation of adsorbable organic iodine (AOI) and adsorbable organic bromine (AOBr) under anoxic/anaerobic conditions. 64% of AOI were removed under reducing condition, whereas under oxic conditions only ~35% were dehalogenated. One year of monitoring of the single organic pollutants Iopromide (X-ray contrast agent), Sulfamethoxazole (bacteriostatica) and naphthalene-sulfonic acid (industrial chemical) showed that the redox conditions have a strong influence on the degradation behaviour of some of the monitored compounds. Iopromide was efficiently removed under oxic conditions, but no evidence for a dehalogenation under oxic conditions was found. Sulfamethoxazole showed a better removal under anoxic/anaerobic conditions (97% in 0.5 month retention time). Oxic infiltration only led to a removal of 62%, even with longer retention times of 2.8 months. The very stable 1.5-naphthalenesulfonic acid was not removed under either redox conditions.

Keywords

Bank filtration; artificial recharge; trace pollutants; AOX (absorbable organic halogens).

INTRODUCTION AND OBJECTIVES

Bank filtration and groundwater recharge provide an important drinking water source to the city of Berlin. 56% of the drinking water is derived from bank filtration and 14% from artificial recharge (BWB, 2003). At most field sites, surface water contains portions of sewage treatment plant effluent (Ziegler, 2001). Due to water recycling, the introduction of effluent organic matter (EfOM) and persistent trace pollutants in the drinking water may be a concern. The project 'Organic Substances in Bank filtration and Groundwater Recharge - Process Studies' at the Department for Water Quality Control (DWQC) at the Technical University Berlin is part of the 'Natural and Artificial Systems for Recharge and Infiltration (NASRI)'-project of the Berlin Centre of Competence for Water (NASRI, 2003). One of the research objectives of the project is to study the removal of trace organics at three field sites with different characteristics. Besides the redox state, factors such as retention time, initial degradable carbon concentration, soil properties and hydrogeological conditions may affect the final concentration. The factors of influence are studied for a few model compounds that represent groups of trace pollutants.

METHODS

The paper highlights the fate of trace organics at a bank filtration site at Lake Wannsee in Berlin. The trace organic analysis is focusing on differentiated AOX (AOI, AOBr)-analysis (Oleksy-Frenzel, 2000) and three groups of trace pollutants. The groups are X-ray contrast media (Iopromide); bacteriostatica (Sulfamethoxazole) and naphthalene-disulfonates (1,5-NSA; 1,7-NSA; 2,7-NSA). Besides the trace organic analytics all samples were analyzed for general

hydrochemistry and bulk organic parameters to characterize the infiltration process more detailed. (Hydrochemistry data in Pekdeger et al. (2004))

Field site

The examined field site is operated with water from Lake Wannsee which is influenced by the discharge of different wastewater treatment plants of Berlin. The wastewater portion in front of the transect is highly variable from 5–50% depending on precipitation and discharge schedule of treatment plant Ruhleben. Highest wastewater proportions are observed in summer when the natural discharge is low and Ruhleben discharges into the Teltowcanal (Pekdeger et al., 2004). A more detailed description of the field site is available in Pekdeger et al. (2004). At the bank filtration site it was found that only in the upper part of the top groundwater layer is included into the actual bank filtration system receiving new bank filtrate. Deeper monitoring wells receive older bank filtrate and do not show the shifted seasonal variations of the surface water.

Therefore, the paper will highlight the processes in the shallow monitoring wells 205, 206, 202OP and 203. At the water-sediment interface in the lake the water infiltrates under oxic conditions. Depending on the position in the lake the infiltration takes place under more saturated conditions (monitoring well 205) or under unsaturated conditions (206). Under more saturated conditions the mineralization of aqueous DOC and sedimentary bound POC during infiltration leads to a depletion of oxygen and nitrate (205) and a decreasing redox potential. Under unsaturated conditions no depletion of O_2 and NO_3 occurs because of the close contact of the water to the interstitial air and the redox potential remains high. During further infiltration in the upper ground water layer oxygen is delivered to the bank filtrate continuously from the water - soil air interface. The infiltration along 202OP and 203 to production well 3 (70 m, 3 month) can be regarded as an oxic infiltration. The extension and position of the redox zones is horizontally stratified and varies seasonally. In the production well water from different aquifers is mixed and the portion of 3 month old bank filtrate is low (à~15%). Other fractions are older bank filtrate from the other side of the lake and deeper groundwater from different aquifers (Pekdeger et al., 2004).

The sampling points for the trace organic pollutants analysis at the field site were selected to reproduce the shortest flow path of the presently infiltrating water to the production well. The individual retention times for each monitoring well were determined by the shift of the seasonal variation of the ratio of oxygen isotopes ($\delta^{18}O$) in the surface water (Pekdeger et al., 2004). The dilution with deeper groundwater in the production well was quantified with the help of boron data (Pekdeger et al., 2004). Data for retention times and dilution with deeper groundwater are included in Figure 1. The investigated trace organic compounds were present in the surface water in detectable con-
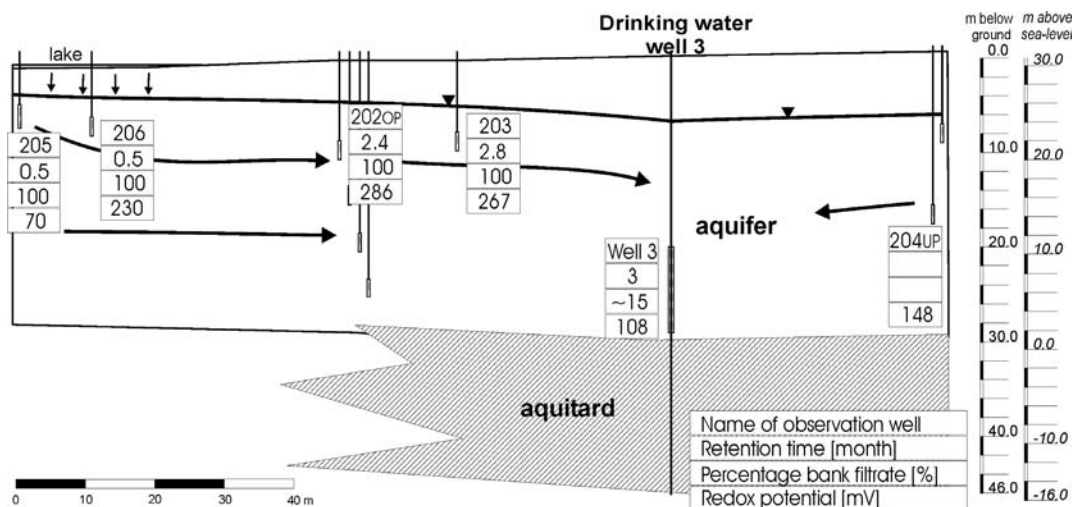
*Figure 1. Cross section of bank filtration site Berlin-Wannsee (**upper aquifer; modified from Pekdeger, 2004**)*

centrations. The concentration of AOX (AOI and AOBr) in the surface water followed a seasonal cycle with higher concentrations during summer.

Analytics

Besides the DOC and UVA$_{254}$-analysis (not a focus of this paper) the analytical program is comprised of differentiated AOX-analysis and trace organic analysis. Adsorbable organic halogens (AOX) are an issue for the aquatic environment for several years. Triiodinated X-ray agents, which are very stable and hydrophilic, contribute to large amounts (> 50%) of the total AOI in Berlins municipal wastewater and are introduced to the surface waters. The protocol for the differentiated analysis of the organohalogens consists of coupled combustion and ion chromatography and is described in Oleksy-Frenzel et al. (2000).

The trace organic compounds Iopromide, Sulfamethoxazole and the isomers of naphthalenesulfonic acids (Table 1) were all extracted by different solid phase extractions (SPE) and measured with standard addition in high performance liquid chromatography with MS/MS- and FLD-detectors. Table 2 shows more details on the trace compound analysis.

Table 1. Chemical structure of analyzed trace compounds
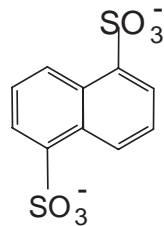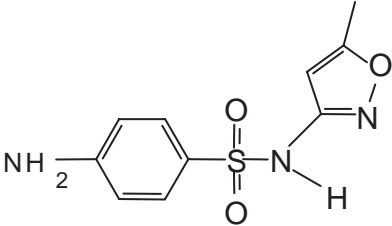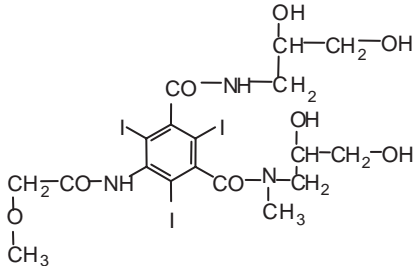
Table 2. Details on trace organic compound analysis

	Iopromide	*Sulfamethoxazole*	*Naphthalenesulfonic acids*
SPE-Cartridge	Lichrolute EN	Lichrolute EN	Lichrolute RP-18
Liquid chromatography	H$_2$O/MeOH with Gradient	H$_2$O/MeOH with Gradient	Ion pair chromatography
Detector	MS/MS (Micromass)	MS/MS (Micromass)	Fluorescence (Gynkotek)
Limit of detection	5 ng l^{-1}	5 ng l^{-1}	10 ng l^{-1}
Limit of quantification	20 ng l^{-1}	20 ng l^{-1}	30 ng l^{-1}
Method described in	Putschew et al. 2001	Hartig 2000	Storm et al. 1999

RESULTS AND DISCUSSION

Results of more than two years of field monitoring (AOX) and nearly one year for trace compounds are presented, with a special focus on the fate of the pollutants during infiltration under different redox conditions.

AOX

- *AOI*

Figure 2 illustrates that the AOI- concentration in the surface water of Lake Wannsee follows a seasonal trend with higher concentrations during summer. This is due to variation in the dilution of the sewage treatment plant effluents. The results show that the fate of the AOI during infiltration is different under aerobic (206, 202OP and 203) and anoxic/anaerobic (205) conditions. The surface water variation of AOI has a stronger effect on the aerobic monitoring wells, where the average AOI-reduction is only 30–40%. This seems not strongly dependent on the retention time because the monitoring well with retention times of less than a month (206; 39%) shows similar results as the monitoring well with retention times of 2.6 month (203; 31%). Monitoring well 205 has a complete different redox chemistry. Due to the saturated infiltration the redox potential is lower and nitrate reduction was observed (NO_3-N: surface water: 1.0 mg/l; 205: 0.03 mg/l). The lower redox potential causes a more efficient reduction of AOI during the recharge process (64% during 0.5 month retention time). In Berlin-Wannsee the AOI of the deeper and background groundwater at the field site is very low (~2 µg/l) and the mixing in the production well (Well 3) leads to an AOI-concentration in the extracted raw water of 2–2.5 µg/l.

These results are consistent with findings at other field sites in Berlin. At the predominantly anoxic/anaerobic bank filtration site at Lake Tegel AOI reduction rates of app. 60% were observed during a 4–5 months infiltration (100 m infiltration distance).
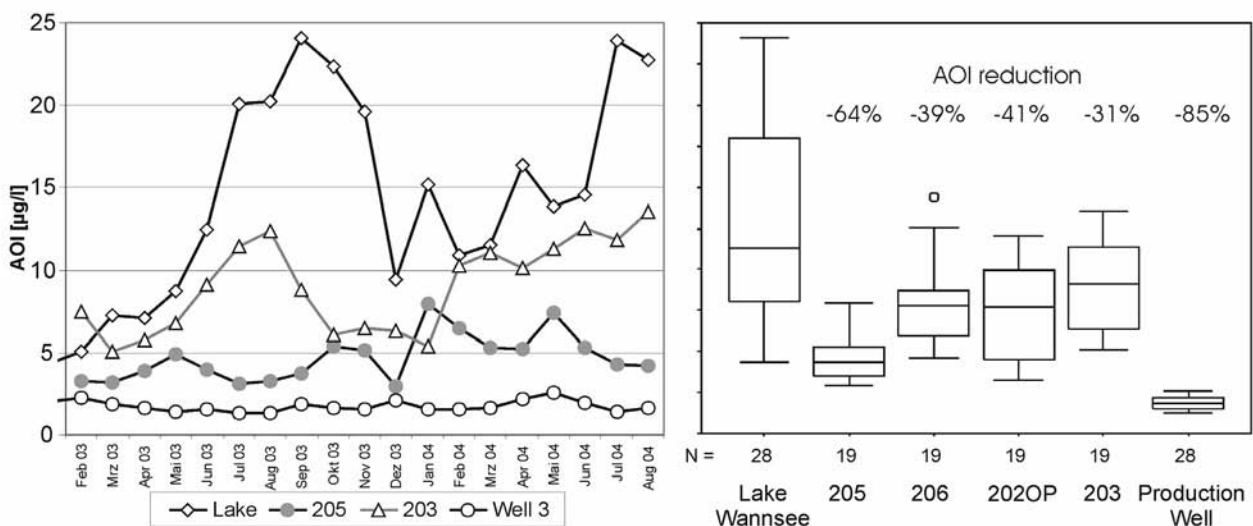
Figure 2a/b. AOI results for bank filtration site Wannsee

At the oxic artificial recharge facility in Berlin-Tegel the reduction rate was 30%. This inverse correlation between AOI-removal and redox potential most probably originates from the initial step of AOI-mineralization - reductive dehalogenation. It is known that the degradation of halogen substituted organics is more effective in soil passages with low redox potential (Mohn and Tiedje 1992). The results from the field site Wannsee show evidently that low redox potentials are essential for an efficient degradation of AOI and that even a duplication or triplication of retention times and infiltration distances can not compensate the absence of reducing conditions.

- *AOBr*

For AOBr a similar trend as for AOI was observed. The monitoring well with the lowest redox potential (205) showed a reduction of 81%. Under oxic conditions the average removal rates were clearly lower (206: 45%; 202OP: 58%; 203: 54%). The water of the production well contained only 14% of the surface water AOBr.

The overall better removal of AOBr than of AOI was due to high concentrations of algae associated AOBr during summer. This seasonal fraction of AOBr seems to be easily degradable under oxic and anoxic/anaerobic conditions, because the 500%-increase of AOBr-concentration during summer was not observed in any of the monitoring wells. There the AOBr-concentration remains year-round relatively stable at the mentioned levels. This indicates the existence of a second AOBr-fraction which is not algae associated and which is present all year. Only this fraction shows the preferential removal under anoxic/anaerobic conditions that we observed for AOI.

Trace pollutants

The single compound analysis for the trace pollutants Iopromide, Sulfamethoxazole and different naphthalene-sulfonic acids was started in spring 2003, providing data for the time period May 2003 to March 2004. The monitoring showed that these trace organic compounds which stand for different groups of persistent pollutants behave differently during infiltration. For some compounds an influence of the redox conditions on the degradability could be revealed. Figure 3 presents data on the fate of the trace compounds during infiltration. The results are presented as box plots with the 50%-quartile or median (horizontal centre line) and the 25%- and 75%-quartiles (box). The error bars denote the minimum and maximum values. For the calculation of the removal rates the arithmetic mean of all measurements was used.

Iopromide occurred in the highest concentrations in the surface water. Due to the fact that Iopromide as a tri-iodinated benzene derivate is part of the bulk parameter AOI (share in surface water ~5%), it was expected that the removal mechanisms for both parameters are similar.

This was not confirmed. Contrary to the fate of AOI Iopromide showed a very good removal (99% in 203) along the oxic infiltration pass to the production well. More details on the degradation characteristics of Iopromide were revealed by a comparison of the monitoring wells 205 and 206. Both monitoring wells are located under the lake and differ primarily in dominant redox conditions. During anoxic/ anaerobic infiltration the Iopromide concentration was reduced from the lake (1102 ng/l, n = 16, σ = 336 ng/l) to observation well 205 (386 ng/l, n = 12, σ = 118 ng/l) by 65%. This is consistent with the observed AOI reduction (205: 64%). But during oxic infiltration Iopromide showed a clearly better removal of 97% (206: 37 ng/l, n = 16, σ = 34 ng/l) than AOI (206: 39%). This confirms that Iopromide and AOI behave differently during oxic infiltration. One assumption that is presently being investigated is that the Iopromide molecule is rapidly metabolized, but not mineralized. The metabolite is not efficiently dehalogenated and remains detectable as AOI.

The effluent concentration of the bactriostatic Sulfamethoxazole in Berlin treatment plants varies between 370–1,200 ng/l. Because of its high stability it is also found in the surface waters at bank filtration
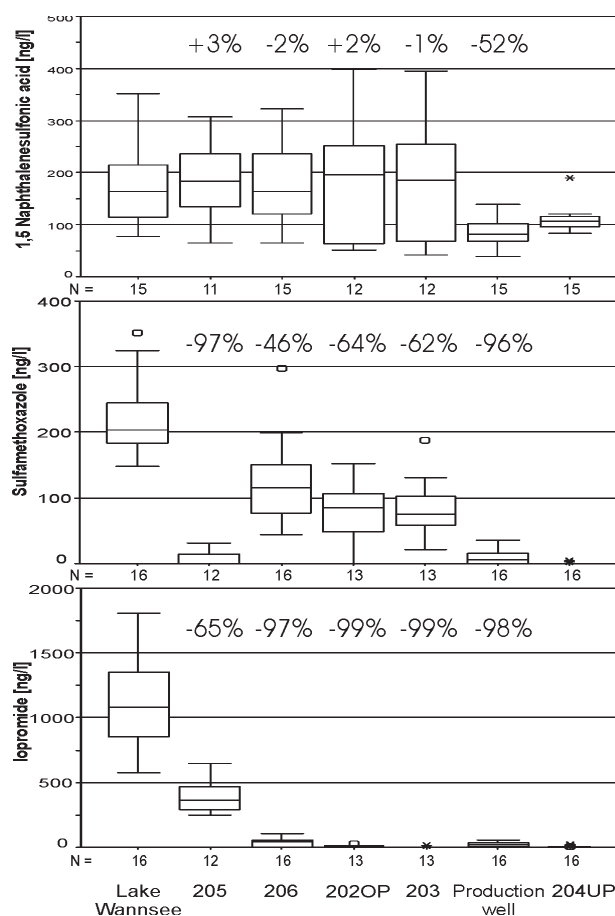
Figure 3. Trace compound results for bank filtration site Wannsee

sites. The degradation of Sulfamethoxazole seems to be redox dependent as well. In Berlin-Wannsee Sulfame-thoxazole shows a better removal under anoxic/anaerobic conditions. During half a month of infiltration to monitoring well 205 the surface water concentration (224 ng/l, n = 16, σ = 62 ng/l) was efficiently degraded by 97% (205: 4 measurements > LOQ; average: 7 ng/l, n = 12, σ = 12 ng/l). The oxic infiltration path along 206, 202OP and 203 displayed smaller degradation rates ranging from 46% to 64%. An influence of a longer retention time was observed because the monitoring well 202OP and 203 (retention times 2.4 and 2.8 month) showed higher removal rates than the 206 (0.5 month). The better removal of Sulfamethoxazole under anoxic/anaerobic conditions is consistent with findings of Schmidt et al. (2004) at other bank filtration sites in Germany. In the production well 3 the mixing with background and deeper groundwater leads to low concentrations.

Naphthalenedisulfonates are well known polar contaminants of treated wastewater and surface water. The different isomers behave nearly similar in the environment but show different degradation properties. In contrast to Iopromide and Sulfamethoxazole, 1,5-naphthalenesulfonic acid was not efficiently degraded under either redox conditions. This is consistent with findings of Stüber et al. (2002) who reported the 1.5 NSA as very stable in wastewater treatment plants. The concentration of the 1,5-naphthalenesulfonic acid remains nearly constant during oxic and anoxic/anaerobic infiltration and is only diluted in the production well. For the 1,7- and the 2,7 NSA-isomers, degradation rates of 13% and 3% were observed under anoxic/anaerobic conditions (205). The results for monitoring wells 206, 202OP and 203 indicated a better removal of these pollutants under aerobic conditions. The surface water concentration was reduced by 72% and 63%, respectively. These results show that some naphthalenesulfonic acids (1,5 NSA) are very poorly biodegradable and it is not to be expected to eliminate these isomers during bank filtration. Other biodegradable isomers (1,7- and 2,7 NSA) display a more efficient reduction during oxic infiltration.

CONCLUSIONS

The results from the field monitoring at the bank filtration site Berlin-Wannsee provided insight into the degradation characteristics of different trace organic pollutants under field conditions. It was found that for some compounds the dominating redox conditions are important for the observed reduction rates. A significant different removal between aerobic and anoxic/anaerobic soil passage was observed for AOX. AOI and AOBr were better degradable under anoxic/anaerobic conditions with removal rates of 64% and 81%, respectively.

The trace compound Iopromide, an x-ray contrast agent achieved higher removal rates under oxic conditions. But there were strong indications that the removal was based on an aerobic metabolization and not on dehalogenation and mineralization. Sulfamethoxazole exhibited an efficient removal under anoxic/anaerobic conditions (97%). Oxic infiltration led for this compound to a reduction of 46%, which was increasing with higher retention times (2.8 month: 64%). For the isomers of the naphthalenesulfonic acid different removal efficiencies were observed. The 1,5-NSA was not degradable under all redox conditions, but the 1,7- and 2,7-NSA isomers were preferably removed aerobically. The presented field data shows that depending on the redox conditions the removal of various trace organic compounds follows different patterns and kinetics. Apart from compounds that are not biodegradable in bank filtration and compounds that are already mineralized in the water or at the water/sediment interface different groups of trace pollutants are only efficiently degraded under favoured conditions. These favoured conditions differ from group to group. Any knowledge about preferential degradation conditions for problematic compounds might help to estimate the removal potential of a given field site. Within the limits the design parameters can be optimized to increase the removal of the problematic compound.

ACKNOWLEDGEMENTS

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Fate of pharmaceuticals during soil infiltration leading to artificial groundwater recharge

T. Rauch, J. Munoz, J.E. Drewes, G. Amy and H. Choi

Abstract

The fate of unregulated organic micropollutants is a key concern for aquatic life and human health in water reuse projects. In particular hydrophilic, small molecular sized organic contaminants are not fully retained during conventional and advanced wastewater treatment and may accumulate in local water cycles. In this study we investigated the fate of four pharmaceutically active residues during simulated soil infiltration. The target micropollutants represent compounds of different hydrophobicity and biodegradability. Laboratory column systems were used to simulate aquifer recharge in presence of different effluent-derived organic carbon substrates (hydrophilic carbon, hydrophobic acids, organic colloids, etc.). The aqueous organic carbon composition played a crucial role for the degradation of the selected pollutants, in that biodegradable organic carbon s timulated soil biomass growth and promoted the removal of certain compounds. Removal was, however, also promoted in extremely oligotrophic environments (e.g. infiltration of reverse osmosis treated effluent), where without exception the organic micropollunts were more effectively removed compared to copiotropic soil microcommunities. Results support the conclusion that efficient micropollutant removal is possible during groundwater recharge using advanced treated effluents that are deprived of biodegradable organic carbon.

Keywords
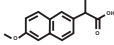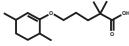
Pharmaceutically active residues, degradation, organic carbon, oligotrophic metabolism.

INTRODUCTION

One of the major concerns regarding the potable reuse of water sources of impaired quality, such as municipal wastewater effluents, is the survival and accumulation of organic micropollutants, such as endocrine disruptors (EDCs), pharmaceutically active compounds (PhACs), or carcinogenic disinfection by-products (DBPs). In particular polar (hydrophilic), small molecular size organic contaminants are likely to survive even advanced wastewater treatment processes such as activated carbon filtration or membrane treatment. Soil-aquifer treatment (SAT) and direct aquifer injection are common methods for replenishing groundwater resources. However, many hydrophilic micropollutants are not efficiently attenuated during soil passage by physical adsorption and have been detected in production wells recovering recharged groundwater.

The purpose of this study was to investigate the roles that biological metabolism plays in the removal of selected trace organic contaminants in artificial groundwater recharge systems. Specifically, we investigated how different source water qualities promote the microbial breakdown of trace organic contaminants. We investigated four organic micropollutants (pharmaceutically active compounds) that differed in hydrophobicity and biodegradability as identified by a literature review (Table 1). Ketoprofen, and naproxen are commonly used analgesics that are easily degradable during wastewater treatment and during soil infiltration (Sedlak and Pinkston 2001, Drewes et al. 2003). Phenacetine represents a low molecular weight and hydrophilic analgesic drug. Gemfibrozil is a commonly prescribed blood lipid regulator, commonly found in impaired groundwater in the lower to medium ppt range (Heberer et al. 1997, Drewes and Shore 2001).

Table 1. Physical-chemical properties of the target compounds used in this study

Compound CAS-No.	Expected degradability	pKa	log Kow/log D	Structure
Phenacetine (antipyretic) 62-44-2	Poor	n.a.	1.58 [a]	
Naproxen (analgesic/anti-inflammatory) 22204-53-1	Degradable	4.15[a]	0.41[b](pH 7)	
Ketoprofen (analgesic/anti-inflammatory) 22071-15-4	Degradable	4.45 [a]	0.07 [b] (pH 7)	
Gemfibrozil (blood lipid regulator) 25812-30-3	Poor	4.77	2.14[b](pH 7)	

a. SRC Physprop Database; b. Scifinder, calculated by software ACD.

The working hypothesis for this study was that the composition of biodegradable organic carbon in recharged water introduced into an aquifer has a major impact on establishing soil biomass activity and a soil microbial community that enables the metabolic breakdown of certain trace organic contaminants.

METHODS

Analytical methods

All organic micropollutants were analyzed by gas chromatography/mass spectrometry (GC/MS) using a HP 6890 gas chromotograph and a HP 5973 quadrupole mass spectrometer from Agilent Technologies (Waldbronn) Germany. The analytical procedure followed Reddersen and Heberer (2003). The following compounds were used for sample preparation and for GC-MS standard preparation: Gemfibrozil (Sigma), ketoprofen, naproxen, phenacetine (Aldrich Chemicals). Limits of detection and limits of quantification differed for each compound depending on the background water matrix and were in the range of 1–10 ng/L and 4–40 ng/L, respectively, for RO permeate samples, and 2–10 ng/L and 2–40 ng/L, respectively, for samples collected from column system 2.

A Sievers 800 Total Organic Carbon Analyzer was used for dissolved organic carbon (DOC) quantification after microfiltration (0.45 µm, Whatman) by the persulfate/ultra violet oxidation method (Standard Method 5310C). Soil biomass was determined as total viable biomass (i.e. viable, not necessarily active bacteria) using phospholipid extraction (PLE) as described in Rauch and Drewes (2005). Soil samples were collected in triplicates from the top soil (0–2 cm, infiltration zone) of the columns.

Column studies

The removal of the target compounds was studied in two column systems, simulating two different groundwater recharge strategies, soil-aquifer treatment (SAT) and direct aquifer injection (DI). Column system 1 (L = 60 cm,

i.d. = 15 cm, filled with silica sand, d_{50} = 0.65 mm, f_{oc} = 0.004%) was acclimated with a reverse osmosis (RO) treated effluent generated with a laboratory-scale RO unit (XLE, Dow/Filmtec), for a duration of 1.5 years. This column simulated direct injection of an advanced treated effluent low in organic carbon concentration. The column was operated under aerobic, saturated flow conditions in recycle mode (Figure 1). Column system 2 consisted of four parallel columns (i.d. 5 cm, L = 30 cm, filled with the same sand as column systems 1). These four columns were used to investigate the effects of different organic carbon matrices (hydrophilic carbon (HPI), hydrophobic acids (HPO-A), colloidal organic carbon, effluent organic matter (EfOM)) on micropollutant removal during saturated recharge. The organic carbon matrices were isolated from secondary treated effluent from the City of Boulder (Colorado) according to Rauch and Drewes (2004). HPI and HPO-A were isolated using XAD-8 fractionation (capacity factor k'= 4) according to Leenheer et al. (2000). Colloidal carbon was operationally defined as the organic carbon with a molecular size between 7,000 Dalton and 1 μm. EfOM refers to the total organic carbon of the effluent. The feed waters of all columns in column system 2 were adjusted to 3 mg/L as total organic carbon (TOC) by dilution with Type I water. All column influents were adjusted in their background composition with regards to macro-and micro-nutrients as described in Rauch and Drewes (2004).
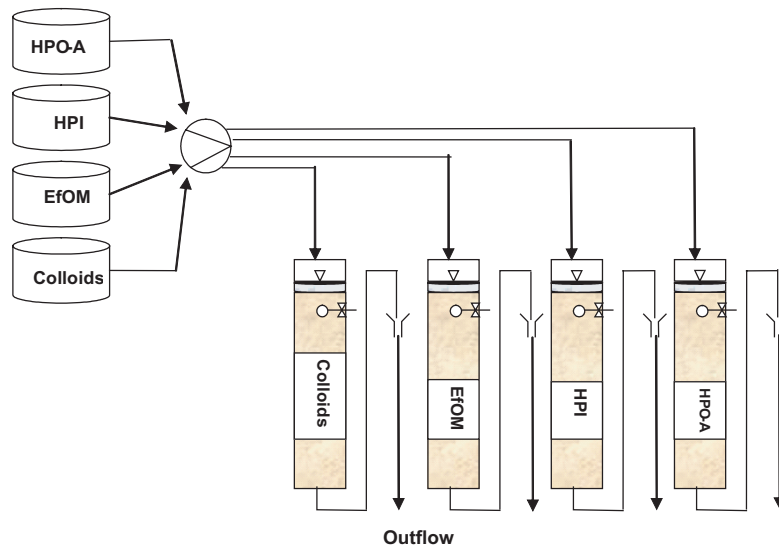
Figure 1. Schematic of the experimental set-up of column system 1 (left) and column system 2 (right)

Spike experiments

All four micropollutants, gemfibrozil, ketoprofen, naproxen, and phenacetine were spiked into the column feed waters at concentrations between 500 and 900 ng/L. The feed waters were adjusted to a pH of 7.5 to 8 in order to minimize adsorption of the acidic drugs onto the silica sand.

The feed water of column systems 1 was spiked with the four organic micropollutants after the column was acclimated to the infiltration of reverse osmosis treated effluent for over 1.5 years. Samples for micropollutant analysis were collected from the feed water, after single passage through the column (average detention time 7 hours) after complete breakthrough of a conservative tracer (sodium chloride) had occurred, and periodically thereafter during recycle operation.

After columns system 2 was biologically acclimated to the infiltration of the respective carbon fractions in the different feed waters (approximately 1.5 years) (Figure 1), the four organic micropollutants were spiked into the feed waters in three consecutive spike events. Spike event 2 and 3 followed 11 and 22 days after spike event one,

respectively. During each spike event samples were collected from the column influent and the column effluent (average detention time 19 hours), respectively, after complete breakthrough of a conservative tracer (sodium chloride) had been observed in the columns.

RESULTS AND DISCUSSION

Organic carbon concentrations in the reverse osmosis treated effluent ranged between 0.3 and 0.5 mg/L as TOC. No organic carbon removal was observed during recycle operation in column system 1. Nevertheless, a significant amount of soil biomass developed during the column acclimatization of over two years, compared to the sand not exposed to feed water infiltration (0.5 ± 0.3 nmol org-PO_4^{3-} g^{-1} sand) (Table 2). The different carbon substrates HPI, HPO-A, EfOM and organic colloides were able to support different soil biomass activities (Table 2).

Table 2. Average organic carbon removal and soil biomass in column systems 1 and 2

	Soil biomass (nmol org-PO_4^{3-} g^{-1})	Organic carbon (mg/L)	
		Column influent	Removal in column
Effluent, reverse-osmosis treated	5.7 ± 0.4	$0.3 - 0.5$	n.a.
HPO-A	3.3 ± 0.5	3.08 ± 0.94	0.42 ± 0.94
HPI	10.4 ± 5.3	2.80 ± 0.39	0.25 ± 0.41
EfOM	20.6 ± 2.5	3.11 ± 0.52	0.4 ± 0.44
org. colloids	27.2 ± 1.8	2.98 ± 0.37	0.75 ± 0.46

During the first spike event the columns fed with HPI, EfOM, and organic colloids (column system 2) showed a positive correlation between organic micropollutant removal and soil biomass activity (Figure 2). Additional abiotic column studies were conducted that showed that adsorption of the target micropollutants onto the experimental sand was negligible (results not shown). These results indicate that the three organic carbon substrates (HPI, EfOM, and organic colloids) promoted the establishment of different soil microbial community compositions and/or concentrations, which effected organic micropollutant removal.
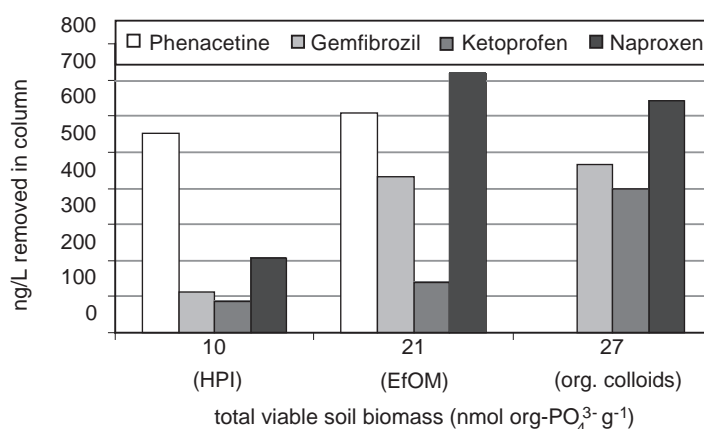
Figure 2. Removal of pharmaceuticals in columns fed with different effluent organic carbon substrates at different days after spiking target micropollutants

Gemfibrozil was the only compound for which this trend was repeatedly observed during the second and third spike event (11 and 22 days after the first spike event), indicating that biodegradable effluent-derived organic carbon may serve as a secondary carbon substrate for the metabolic decay of this micropollutant. The other three compounds (phenacetine, ketoprofen, and naproxen) were increasingly better removed with time during extended exposure of the columns to these trace pollutants. (The columns were acclimated to the organic carbon substrates HPI, EfOM, and organic colloids, but not to the organic micropollutants before feed water spiking). Three weeks after the column feed waters were started to be spiked with the four micropollutants, the pollutant were very efficiently removed in all columns (70−100%) within 30 cm of soil infiltration (19 hours detention time). It is likely that microbial adaptation to the micropollutants took place within the first three weeks of exposure.

The columns containing the lowest soil biomass concentrations (fed with HPO-A and reverse osmosis treated effluent) exhibited the most efficient removal of all organic target micropollutants. In all three spike events, the four micropollutants were removed to more than 90% in the HPO-A column. Likewise, column system 1 removed all compounds close to the detection limit during single passage (detention time 7 hours) (Figure 3).
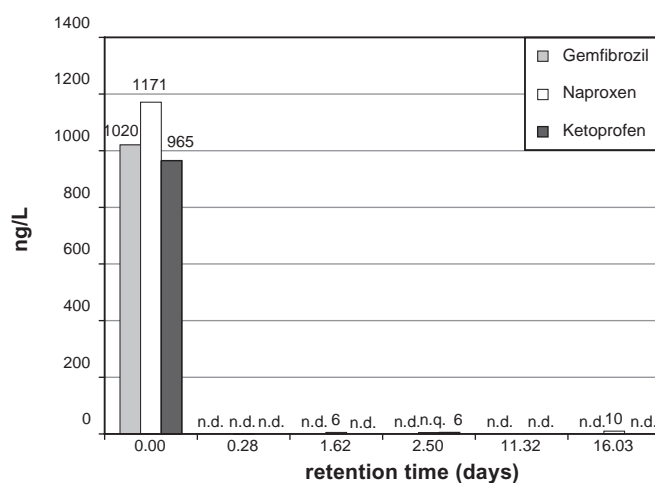
Figure 3. Removal of pharmaceuticals in the recycle column fed with reverse osmosis treated effluent

CONCLUSIONS

With this study we were able to identify the influence of the aqueous organic carbon quality on the removal of selected organic micropollutants during simulated soil infiltration. Very efficient removal of all target micro pollutants occurred within the first 30 to 60 cm of soil infiltration at environmentally relevant concentrations. Bioavailable organic carbon seems to promote the removal of certain trace compounds (e.g. gemfibrozil) by sustaining soil biomass growth. Oligotrophic recharge (infiltration of HPO-A and membrane treated effluent) conditions enhanced the removal of organic micropollutants, likely, by stimulating a slow-growing, diverse microbial community.

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Removal mechanisms of effluent organic matter during soil infiltration

T. Rauch and J. E. Drewes

Abstract

A novel approach was developed in this study to investigate the bioavailability of organic carbon being introduced into soil-aquifer treatment systems, by monitoring soil biomass growth response during the infiltration of effluent derived organic carbon substrates. The approach combined soil microbiological parameters (e.g. total viable biomass) with organic carbon fractionation. Using well acclimated biological column experiments, the degradation behavior of three bulk organic carbon fractions isolated from a secondary treated effluent was investigated: hydrophobic acids (HPO-A), hydrophilic carbon (HPI) and organic colloids. The relative removal of organic carbon in each of these fractions differed significantly and added up to the overall removal observed for total effluent organic matter (EfOM), which was studied in a parallel column set-up. An empirical equation was developed to predict *a priori* the amount of easily biodegradable organic carbon (BOC) in any conventionally treated domestic effluent. The empirical equation provided a good estimate of BOC concentrations in five different water types, which differed in organic carbon bulk composition and concentration. Modeled BOC concentrations corresponded closely with BOC concentrations determined through conventional laboratory-scale degradation tests. This study presents a new approach to predict the portion of easily degradable organic carbon in domestic effluents or impaired surface waters by employing established fractionation procedures.

Keywords

Soil-aquifer treatment, biodegradable organic carbon, soil infiltration, organic carbon fractionation.

INTRODUCTION

The fate of effluent organic matter (EfOM) during artificial groundwater recharge has been intensively studied during the last decades. The persistence of organic carbon during soil infiltration is of concern since it can potentially elevate dissolved organic carbon concentration in receiving aquifers, affect the transport of pollutants in the subsurface, or limit a co-metabolic removal of trace organics. Previous research has demonstrated that the fate and transport of organic carbon during soil infiltration is primarily affected by biological and physical processes (Quanrud et al. 1996). The portion of organic carbon in effluents that is degraded during groundwater recharge is commonly assessed using biodegradable dissolved organic carbon (BDOC) batch tests (Servais et al. 1987, Schnabel et al. 2002). These laboratory tests are relatively time consuming and not standardized in their design, procedure, and test interpretation. In this study we hypothesized that easily biodegradable organic carbon (BOC) concentrations in effluents can be predicted if the organic carbon composition in effluents is known in terms of three organic bulk fractions: hydrophilic carbon (HPI), hydrophobic acids (HPO-A), and organic colloids. Specific objectives of the study were: (1) to determine the importance of biodegradation for the removal of each fraction; (2) to quantify the portion of easily degradable organic carbon in each fraction in biological column experiments; and (3) to develop an empirical equation for an *a priori* assessment of BOC concentrations in effluents, for which the bulk organic carbon composition is known.

METHODS

Organic carbon fractionation

Organic carbon fractionation was conducted as described in Rauch and Drewes (2004). In summary, HPI and HPO-A were isolated from secondary treated domestic effluent from the City of Boulder (Colorado) according to Leenheer et al. (2000) by fractionation using chromatographic resin (XAD-8, Rohm and Haas/Philadelphia PA) with a capacity factor of k' = 4. HPI was collected as the non-adsorbable fraction of the sample. HPO-A were recovered as adsorbed carbon from the resin by 0.1 N sodium hydroxide elution. Colloidal organic carbon (6,000 Dalton to 1 μm) was isolated from freshly collected secondary effluent after 1 μm filtration, pre-concentration by vacuum rotary evaporation, and dialysis fractionation (Spectra/Por, Spectrum, 6,000–8,000 Dalton) against milli-Q water (pH 4–5). Dialysis was conducted until total organic carbon (TOC) concentrations in permeate remained below 1 mg/L. Effluent organic matter (EfOM) derived from freshly collected secondary effluent (1 μm pre-filtered) and was used to study the removal of EfOM in comparison to the isolated bulk organic fractions.

Column studies

To assess the bioavailability of the organic carbon fractions, biologically acclimated column systems were employed simulating soil infiltration under saturated aerobic flow conditions. The four columns were 30 cm in length (6.5 cm i. d.; ƒoc = 0.004%; effective grain size: 0.35 mm) as described in Rauch and Drewes (2004) and fed with the four different carbon fractions: organic colloids, HPI, HPO-A, and EfOM, respectively (flow rate of 0.4 mL/min). Feed DOC concentrations in each carbon fraction were adjusted to approximately 5 mg/L. The inorganic background of each feed water sample was adjusted to resemble a conductivity of 1,000–2,000 μS/cm, pH 6.5–7.5, ammonium concentration of approximately 1 mg-N/L, and a phosphate concentration above 0.2 mg/L. After a biological acclimatization period of approximately 3 months, dissolved organic carbon (DOC) was measured one to two times a week at 0, 10 and 30 cm depth for a duration of 2 to 3 months. Average detention time in the 30 cm columns was 19 hours.

Analytical methods

A Sievers 800 Total Organic Carbon Analyzer was used for dissolved organic carbon (DOC) quantification after microfiltration (0.45 μm, Whatman) by the persulfate/ultra violet oxidation method (Standard Method 5310C). Soil biomass was determined as total viable biomass (i.e. viable, not necessarily active bacteria) using phospholipid extraction (PLE) as described in Rauch and Drewes (2005). Soil samples were collected in triplicates from the top soil (0–2 cm, infiltration zone) of the columns.

RESULTS AND DISCUSSION

Biological column studies revealed that each of the three organic carbon fractions showed a unique removal behavior during column transport at steady state operation of the columns (Figure 1) (Rauch and Drewes 2004). All three carbon fractions showed a significantly different overall removal; organic colloids were removed by 68% in average, whereas HPI and HPO-A were only removed by 29 and 11% in average, during 30 cm column transport. The majority HPI and HPO-A carbon (constituting with over 70% the majority of EfOM in Boulder secondary treated effluent) was removed within the first 10 cm of soil infiltration with marginal removal during the subsequent column passage (Figure 1). Organic carbon removal in the first 10 cm of column passage was related to viable soil biomass concentrations as measured during steady state operation of different column experiments in 0–3 cm column depth. Results showed a good positive correlation between organic carbon removal of the different carbon fraction with total viable biomass in the soil infiltration layer (Figure 2). This finding emphasized that biodegradation

was the leading process for organic carbon removal and that BOC concentrations were limiting soil biomass growth during soil infiltration. This conclusion was supported by additional column studies testing the transport behavior of EfOM under abiotic conditions (results not shown). This experiment showed that sorption of EfOM onto the experimental sand was saturated after approximately 8 days with no subsequent sorption of organic carbon.
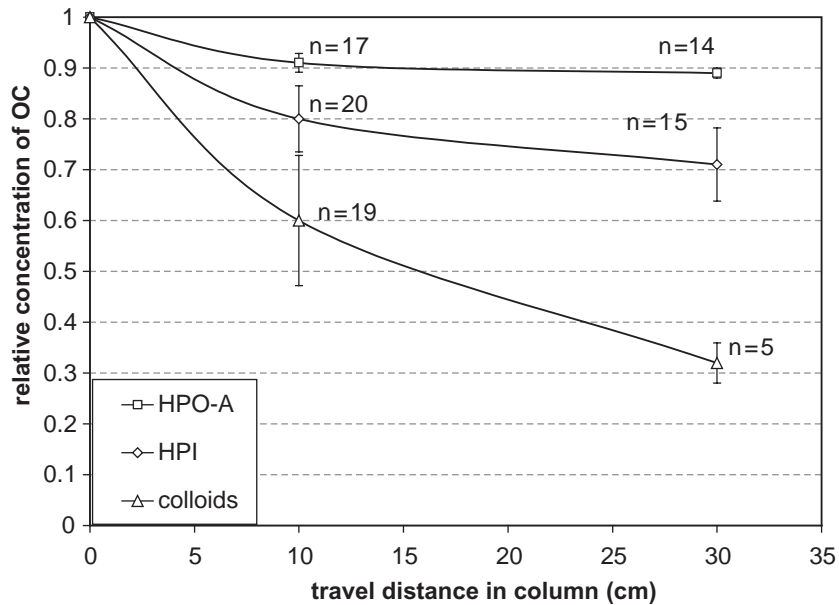
Figure 1. Relative removal of different effluent organic carbon fractions in biological column experiments compared to influent concentrations (error bars = 1 Stdev)
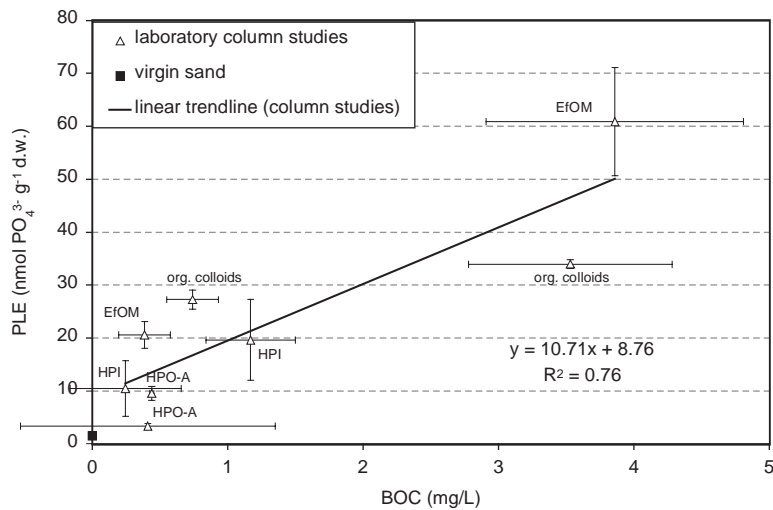
Figure 2. Organic Carbon removal versus soil biomass (0–2 cm) under steady state operation in column studies (virgin sand: experimental sand before to sample application)

The observed relative organic carbon removal of the three bulk fractions at 30 cm column depth was successfully used to model the removal of total EfOM in Boulder secondary treated effluent, as observed in the fourth biological column, by using the following equation [1]:

$$BOC\ (mg/L) = 0.29\ c_{HPI} + 0.11\ c_{HPO} + 0.68\ c_{coll} \qquad [1]$$

where c_{HPI} is the relative organic carbon concentrations of HPI in Boulder secondary treated effluent (37%), c_{HPO} the relative concentration of hydrophobic acids (38%), and c_{coll} is the relative concentration of organic colloids in Boulder secondary treated effluent (16%).

The equation predicted a BOC concentration of 27% which is in very close agreement with average organic carbon removal measured in the column study (24 ± 11%). Equation 1 implies that BOC concentrations in the effluent can be predicted based on the behavior of its individual bulk fractions. This assumption was confirmed by fractionating the effluent of the column fed with EfOM after 30 cm depth. Analytical fractionation resulted in a distribution of 64.4% HPO-A, 34.4% HPI, and 0 ± 8% organic colloids of the sample collected in 30 cm depth. This result was very comparable to the distribution of 62.2% HPO-A, 31.7% HPI, and 6% organic colloids, predicted by equation 1.

In the following we tested whether equation 1 was also applicable to different municipal effluents, which had been characterized in terms of composition (HPO, HPI and organic colloids) and BOC concentrations. Thereby we hypothesized that bulk organic carbon fractions in other municipal effluents receiving similar treatment would contain similar relative portions of easily degradable organic carbon. This assumption is supported by Fox et al. (2001), who found close chemical similarities between HPI and HPO-A fractions isolated from different domestic effluents. The literature review identified 5 water types, for which information on organic carbon composition and BOC concentrations was available (Table 1). Measured BOC data for all effluents or effluent impacted surface waters derived from BDOC batch test studies. Fractionation procedures employed in the cited studies varied in details from the procedure described in this paper (Rauch et al., in preparation). The selected waters differed in terms of absolute organic carbon concentrations (5–12 mg/L as DOC) and bulk organic carbon composition (Table 1). Table 1 compares measured BOC concentrations for all waters as cited from literature and calculated BOC concentrations using the composition of the respective effluents as inputs for equation 1.

Table 1. Comparison of modeled and measured BOC concentrations for different effluents

	Lake Tegel Berlin (mg/L)	SJCE WRP California (mg/L)	Mesa NWWRP Arizona (mg/L)	Tucson SRF Arizona (mg/L)	Whittier NarrowsWRP California (mg/L)	Boulder WWTP Colorado (mg/L)
HPO	4.18	2.77	1.78	4.64	2.39	3.43
HPI	3.73	5.52	3.17	2.44	4.12	3.41 (37.43%)
Colloidal	1.92	*)	neglegible	4.52	*)	1.49 (16.36%)
HPO-N	**)	**)	**)	**)	**)	0.76 (8.34%)
Total OC	9.83 ± 0.67	8.29	4.95 ± 0.68	11.6	6.51	9.11 (± 0.21)
BDOC measured	2.77 ± 0.72	1.94	1.35 ± 1.17	6.3	1.92	2.97 ± 0.34
BDOC modeled	2.83	1.91	1.12	4.26	1.46	2.39
References	Ziegler 2001, Ziegler and Jekel (2001)	Sattler 2004, Fox et al. 2001	Sattler 2004, Fox et al. (2001)	Sattler 2004, Debroux 1998, Fox et al. 2001	Sattler 2004, Fox et al. 2001	Rauch and Drewes (2005), Rauch and Drewes (2004)

*) No data available. Estimated to be neglegible.
**) Included in HPO fraction.

Results show that measured and calculated BOC concentrations are in close agreement, despite significant differences in organic carbon composition in the different effluents. The highest variation between measured and modeled results occurred for Tucson SRF effluent (30%).

CONCLUSIONS

Results of this study give a better understanding for the importance of biological removal for the attenuation of effluent derived organic carbon during soil infiltration under saturated aerobic flow conditions. EfOM from a secondary treated effluent was fractionated into HPI, HPO-A, and colloidal organic carbon. These fractions had a significantly different bioavailability. The majority of easily degradable organic carbon was removed within 30 cm of soil infiltration. An empirical equation was developed to calculate BOC concentrations in Boulder secondary treated effluent based on the relative removal of organic carbon in each fraction. The same equation proved useful for an *a priori* assessment of BOC concentrations in different domestic treated effluents and effluent impacted surface waters. The equation is proposed as an alternative to time consuming laboratory degradation tests and applicable for conventionally treated effluents in a DOC concentration range of approximately 5 and 12 mg/L.

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Fate of wastewater effluent organic matter (EfOM) through soil aquifer treatment (SAT)

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Abstract

This paper employs innovative analytical tools to characterize wastewater effluent organic matter (EfOM) and to track its removal and transformation through soil aquifer treatment (SAT). While the total amount of EfOM is significantly reduced by SAT, there are trends of shorter term versus longer term removals of specific EfOM fractions. The preferential removal of non-humic components (e.g., proteins, polysaccharides) of EfOM occurs over shorter travel times/distances while humic components (i.e., humic substances) are removed over longer travel times/distances, with the removal of both by sustainable biodegradation. Dissolved organic nitrogen (DON), a surrogate for protein-like EfOM, is also effectively removed over shorter term SAT. However, there is some background humic-like natural organic matter (NOM), associated with the drinking water source within the watershed, that persists through SAT.

Keywords

Effluent organic matter, soil aquifer treatment.

INTRODUCTION

Wastewater effluent organic matter (EfOM) consists of natural organic matter (NOM) derived from the drinking water source, dominated by humic substances, plus soluble microbial products (SMPs) derived from biological (secondary) wastewater treatment, reflecting a microbial origin. Soil aquifer treatment (SAT) represents a wastewater reclamation/reuse technology that can renovate wastewater effluent to drinking water levels, and hence can be an important component in an indirect potable reuse system. In application to secondary effluents, SAT has proven very effective in removing total nitrogen (TN) and viruses but is now receiving scrutiny in assessing its ability to remove organics, both bulk organic matter and trace organic compounds. The SAT technology (Figure 1) involves infiltration of secondary effluent through a recharge basin with subsequent extraction from the underlying aquifer through recovery wells, and embodies both treatment, dominant in the vadose (unsaturated) zone, and
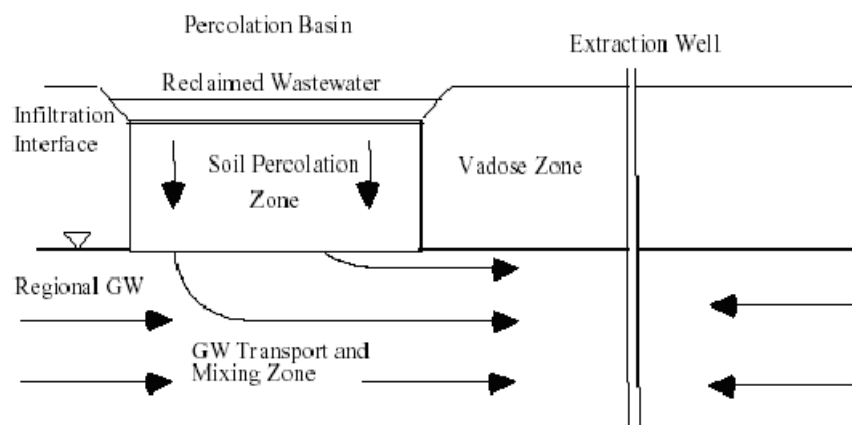
Figure 1. Schematic of Soil Aquifer Treatment (SAT)

storage within the saturated zone (aquifer) (NCSWS, 2001). The emphasis of this paper is removal and on transformation of EfOM and EfOM components during SAT.

From a drinking water perspective, certain EfOM components are problematical in imparting color and serving as a precursor to disinfection by-products (DBPs); other components contribute to membrane fouling. Beyond the traditional measurement of dissolved organic carbon (DOC) as a measure of the amount of EfOM, this paper addresses the *composition* of DOC and the corresponding measurement of dissolved organic nitrogen (DON), of importance given the nitrogenous content of SMPs.

METHODS

To characterize the amounts and compositions of DOC and DON through SAT, samples were collected from three operating SAT facilities situated in the USA: (i) the Northwest Water Reclamation Plant (Mesa WRP) in Mesa, Arizona; (ii) the Sweetwater Underground Storage and Recovery Facility (Tucson WRP) in Tucson, Arizona; and (iii) the Rio Hondo Spreading Grounds (RHSG) in Los Angeles, California. The Mesa WRP site has a dissolved oxygen concentration that remains below 1 mg/L throughout the infiltration zone below the first 1.5 meters, making almost the entire vadose zone of 15 meters thickness anoxic with only the first 1.5 meters aerobic; the plume of reclaimed water extends greater than 2,000 meters down gradient from the recharge site and is about 30 meters thick. A key attribute of the Tucson WRP is its deep vadose zone of 30 meters where high concentrations of DOC and ammonia in the feed water (recharge basin) create a high oxygen demand; as the dissolved oxygen decreases with depth, and the ammonia is nitrified and denitrified, nitrate becomes the electron donor at a typical depth of about 1.5 meters. The RHSG site is unique compared to the other two sites because the entire vadose zone, with a thickness of 3 to 10 meters, remains aerobic.

A sample series was taken from the Mesa WRP and included the effluent fed to the recharge basins, a lysimeter beneath one of the recharge basins, and a series of monitoring wells situated along a transect following the hydraulic gradient of the aquifer (saturated zone). The sample designated as Mesa3eff represents the tertiary wastewater effluent (nitrification-denitrification followed by rapid sand filtration), the P15 sample corresponds to the lysimeter lying beneath the recharge basin at depth of 5 m, the NW- series wells correspond to samples immediately downgradient from the recharge basin, and the -u series wells correspond to samples further downgradient with greater travel distances and longer travel times. Table 1 summarizes pertinent information about these sampling points including (horizontal) travel distance downgradient (m), estimated travel time (days), and percent reclaimed water (based on sulfate levels as a tracer, with sulfate shown to be conservative for the Mesa WRP sys-

Table 1. Sampling location descriptions at Mesa WRP

Sample	Travel distance (m)	Travel time (months)	Reclaimed water (%)
Basin	0	0	100
P15	negligible	Short	100
NW4	388	6–18	100
NW3	655	6–19	86
NW2	885	6–20	99
10u	1,950	12–96	61
26u	1,950	12–96	71
44u	2,700	12–96	39

Table 2. Sampling location descriptions at Tucson WRP

Sample	Travel distance (m)	Travel time (days)	Reclaimed water (%)
RB1	0	0	100
MW5	6	11	100
WR199a	35	35	100

tem). Also taken was a sample of a representative drinking water source within the watershed, the Val Vista drinking water (VVDW) taken from the Val Vista WTP. The samples in the set taken from the Tucson WRP correspond to a monitoring well (MW5) penetrating the vadose (unsaturated) zone and a recovery well (WR 199) tapping into the underlying aquifer (saturated zone) beneath the recharge basin (RB1) fed by a non-nitrified effluent. The drinking water source is a low DOC groundwater. Table 2 summarizes the estimated travel times and percent reclaimed water for the Tucson WRP samples. Finally, the samples taken from the RHSG site include two wastewater effluents feeding into a recharge basin with a downgradient monitoring well. The two effluents are from the San Jose Creek Water Replenishment Plant (SJCWRP) and the Whittier Narrows Water Replenishment Plant (WNWRP), both with tertiary filtration, contributing ~75% and ~25% of the recharge water. Basin 4 represents one of the recharge/infiltration basins and Well 1590 is located less than 30 meters horizontally downgradient from the basin. The corresponding drinking water is derived from the Metropolitan Water District (MWDEff).

The *amount* of EfOM was defined through dissolved organic carbon (DOC) measurements along with UV absorbance at 254 nm (UVA254). The composition of the EfOM in various SAT samples was characterized by: (i) specific UV absorbance (SUVA = UVA_{254}/DOC); (ii) size exclusion chromatography with on-line DOC detection, describing the molecular weight (MW) distribution and classification of organic matter according to polysaccharides (PS), humic substances (HS), and low MW acids (LMA); (iii) fluorescence excitation-emission matrix (EEM), distinguishing humic-like organic matter from protein-like organic matter; and (vi) nitrogen (N) species used to calculate dissolved organic nitrogen (DON) as the difference between total nitrogen (TN) minus the sum of ammonia-nitrogen (NH_3–N) and nitrate-nitrogen (NO_3–N). SUVA represents an index of the aromaticity or humic content of the EfOM. SEC-DOC chromatograms have successfully revealed transformation/removal patterns of PS, HS, and LWA components, with PS components readily removed in the upper vadose zone and HS components more slowly removed through the vadose zone and aquifer; the persistence of some LMA components likely reflects a by-product of HS components biodegradation. EEM spectra, representing a 3-D plot of fluorescence intensity versus excitation and emission wavelengths, have revealed effective removals of protein-like organic matter (corresponding to an EEM peak at lower excitation/emission wavelengths) and partial removals of humic-like organic matter (corresponding to an EEM peak at higher excitation/emission wavelengths). Tracking nitrogen species through SAT serves two purposes: (i) an indication of redox zone transitions (e.g., aerobic to anoxic) that affect biodegradation pathways and (ii) a quantification of DON as a precursor for N-DBPs that exhibit a high cancer potency.

RESULTS AND DISCUSSION

Bulk EfOM parameters; DOC, UVA254, SUVA, and DON; are summarized in Tables 3, 4, and 5 for samples taken from the Mesa WRP, Tucson WRP, and RHSG sites, respectively. At the Mesa WRP (Table 3), SAT is able to reduce the DOC to about 1.5 mg/L, accounting for dilution, which is similar to the drinking-water DOC in the watershed (~2.0 mg/L). The SUVA is less than 2 L/mg-m, consistent with an EfOM signature. The DOC after SAT at the Tucson WRP (Table 4) is about 1 mg/L, only slightly higher than the drinking-water DOC of ~0.5 mg/L. In contrast to the Mesa WRP results, the SUVA values after SAT at the Tucson WRP increase. An increase in SUVA reflect a preferential removal (biodegradation) of non-humic over humic EfOM; this is consistent with the non-nitrified effluent at the Tucson WRP being less well treated than the nitrified/denitrified effluent at the Mesa WRP. The much

lower DOC and DON values at MW5 demonstrate the significant removal achieved through the vadose zone. The DOC at the RHSP (Table 5) is significantly reduced over a short travel distance/time (Well 1590). At the Mesa WRP, the tertiary-treated effluent contained about 2 mg/L of DON; shorter-term SAT reduced the DON to about 1 mg/L while longer-term SAT reduced the DON to a level approaching drinking water (0.2 mg/L). In contrast, at the Tucson WRP, a higher DON level was observed in the non-nitrified effluent, however, the DON was effectively reduced through the extensive vadose zone at this site.

Table 3. Bulk DOC, UVA$_{254}$, SUVA and DON for Mesa WRP samples

Sample	DOC (mg/L)	UVA$_{254}$ (cm^{-1})	SUVA (L/mg-m)	DON (mg/L)
Mesa3eff*	6.10	0.099	1.62	2.1
NW4	1.47	0.024	1.63	1.8
NW3	1.76	0.022	1.25	1.7
NW2	1.52	0.022	1.45	1.1

* Sample of nitrification-denitrification effluent from the Mesa WWTP which feeds the recharge basins at the Mesa WRP.

Table 4. Bulk DOC, UVA$_{254}$, SUVA and DON for Tucson WRP samples

Sample	DOC (mg/L)	UVA$_{254}$ (cm^{-1})	SUVA (L/mg-m)	DON (mg/L)
RB1*	14.1	0.180	1.26	9.4
MW5	4.84	0.120	2.38	BDL
WR199A	0.98	0.02	2.46	BDL
Drinking Water	~0.5	n/a	n/a	n/a

* Recharge basin sample derived from non-nitrified effluent.

Table 5. Bulk DOC, bulk UVA254, and bulk SUVA at RHSG

Sample	DOC (mg/L)	UVA$_{254}$ (cm^{-1})	SUVA (L/mg-m)
SJCWRP	8.2	0.118	1.44
WNWRP	6.0	0.091	1.51
Basin 4	7.7	0.134	1.75
Well 1590	1.8	0.040	2.19
Drinking water	~3.0	n/a	n/a

Two representative sets of SEC-DOC chromatograms are presented in Figures 2 and 3. Results for the Mesa WRP (Figure 2) reveal the almost complete elimination of the PS peak and partial removal of the HS peak, both attributable to (sustainable) biodegradation. A similar trend is seen for the Tucson WRP sample series in Figure 3.
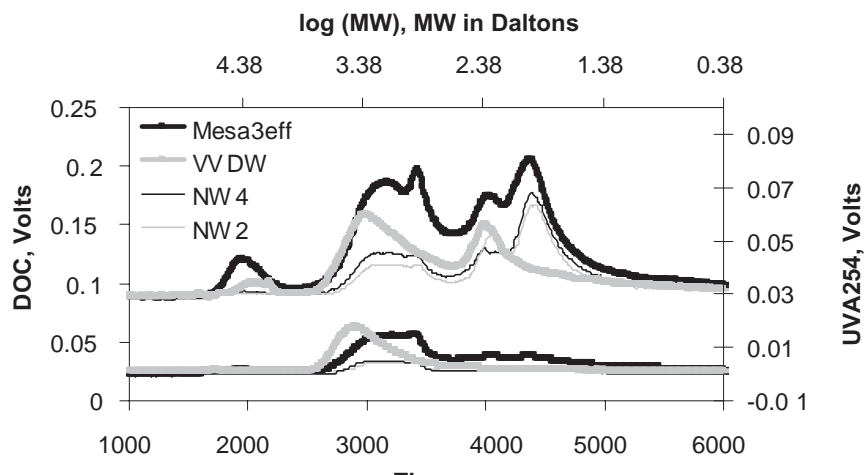
Figure 2. SEC-DOC chromatograms for Mesa WRP samples
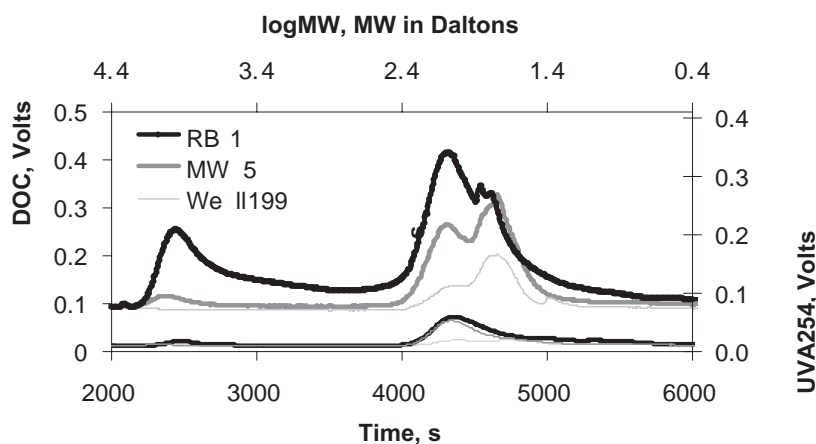
Figure 3. SEC-DOC chromatogram of samples at the Tucson WRP

Two representative sets of EEM spectra appear in Figures 4 through 7. Figure 4 shows EEM spectra for the Mesa WRP samples Mesa3eff (recharge basin), P15 (lysimeter), and NW4 (monitoring well). Alternatively, Figure 5 shows differential EEM spectra where the P15 spectrum is subtracted from the Mesa3eff spectrum (left), revealing shorter-term SAT removals, and the NW4 spectrum is subtracted from the P15 spectrum, revealing longer-term SAT removals. Both humic-like and protein-like EfOM are removed over the shorter term while additional humic-like EfOM is removed over the longer term. More precisely, the significant diminishment of the lower excitation/emission wavelength peak over the shorter term corresponds to protein-like organic matter removed by (sustainable) biodegradation. Figure 5 shows individual EEM spectra for the Tucson WRP samples RB1 (recharge basin), MW5 (shallow monitoring well), and WR199A (deeper monitoring well). Figure 7 portrays corresponding differential EEM where, again, one sees both humic-like and protein-like EfOM being removed in the upper vadose zone while only humic-like EfOM appears to be significantly removed in the lower vadose zone.
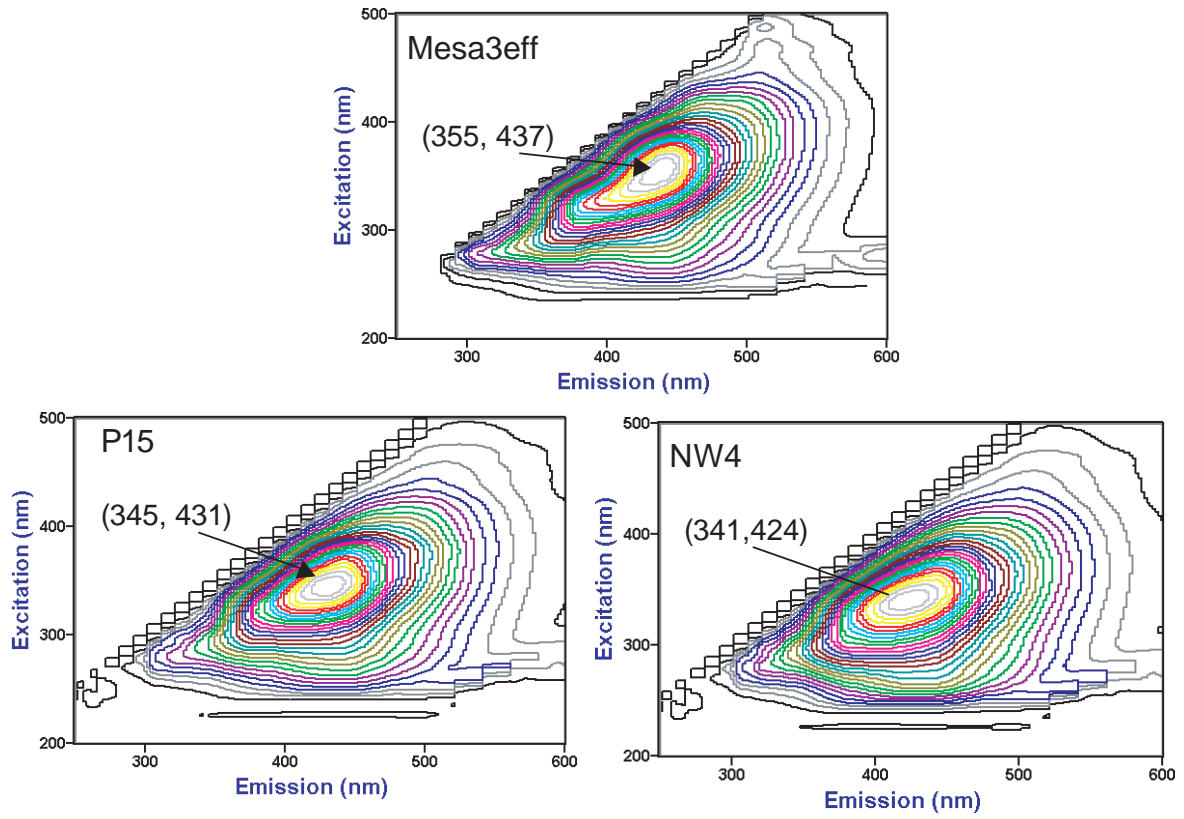
Figure 4. Fluorescence EEM spectra of samples from the Mesa WRP
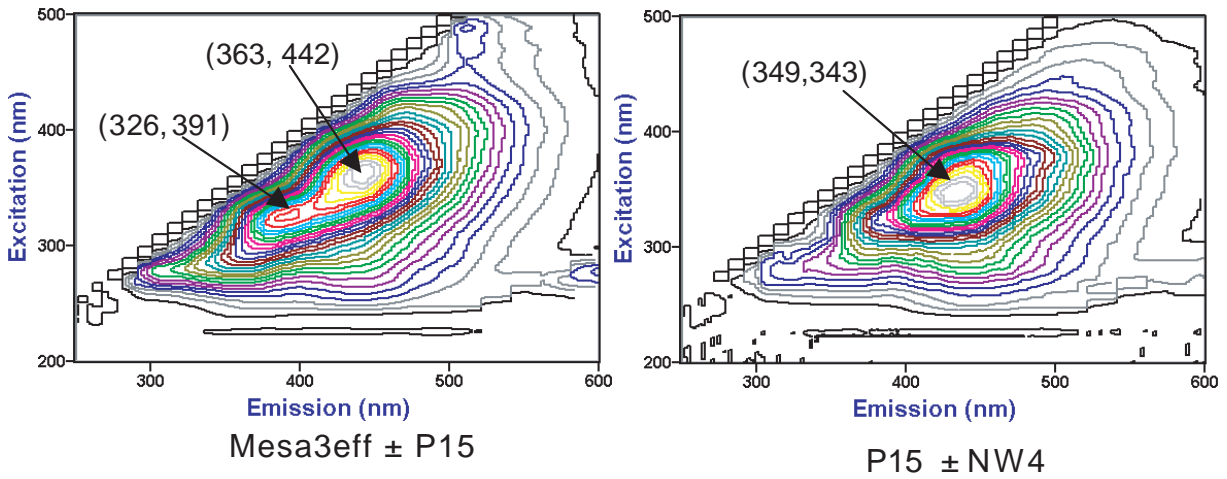
Figure 5. Differential EEM of samples from the Mesa WRP
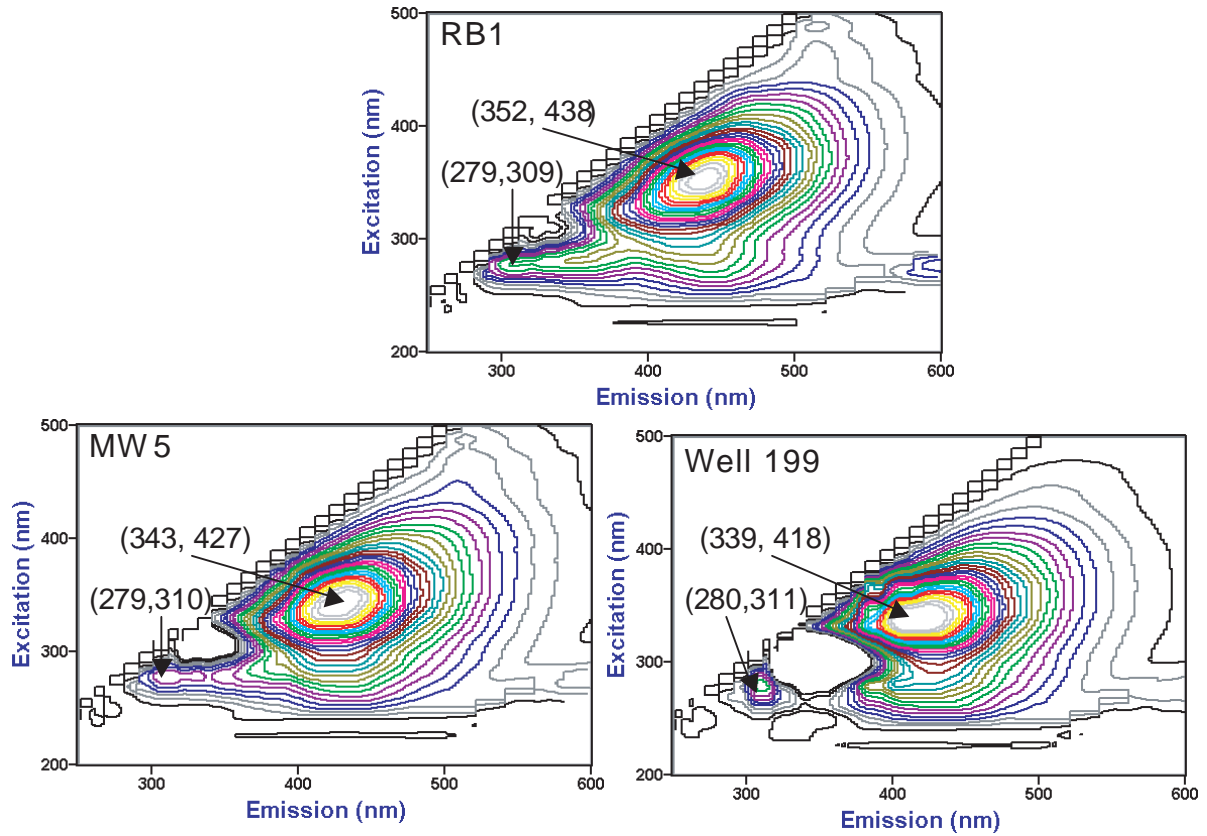
Figure 6. Fluorescence EEM spectra of Tucson WRP samples
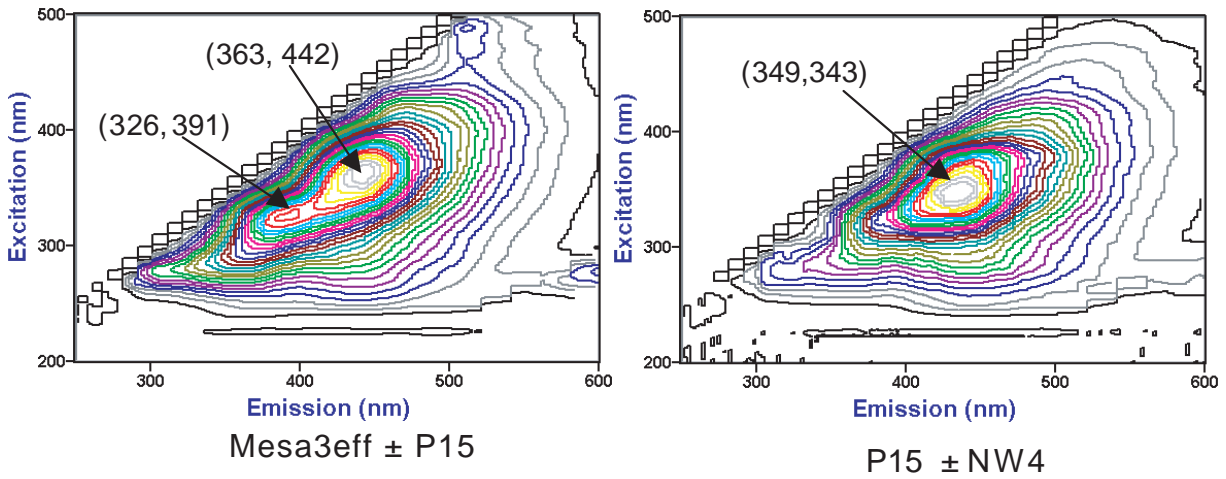
Figure 7. Differential EEM of Samples from the Tucson WRP

CONCLUSION

Overall, significant removals of DOC were observed at all sites, ranging from greater than 50% to almost 75% after accounting for dilution with native groundwater, based on sulfate as a tracer. These DOC removals were accompanied by almost complete elimination of DON. Trends in SUVA, as an index of aromaticity, varied from similar to higher values in samples after SAT, with an increase reflecting a preferential removal of non-aromatic (non-humic) components. Effective removal dominated by biodegradation was observed for polysaccharides and proteins with lesser but significant removals of humic substances under longer-term anoxic conditions; these removal trends were supported by SEC-DOC and EEM results.

Other related work has also shown effective removal of pharmaceutically active (PhACs) and endocrine disrupting (EDCs) compounds by aerobic and/or anoxic biodegradation although some PhACs (e.g., carbamazepine and primidone) have been shown to persist through SAT (Drewes, et al., 2003, Mansell and Drwes, 2004). Thus, SAT represents a sustainable advanced wastewater treatment process that can play an important role in a multi-barrier, indirect potable reuse system.

ACKNOWLEDGEMENT

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Temperature effects on organics removal during river bank filtration

D. Schoenheinz, H. Börnick and E. Worch

Abstract

The objective of the presented investigations was to show the effects of extreme temperature conditions on the organics removal efficiency and transport behaviour during river bank filtration considering both dissolved organic carbon and atrazine as a tracer organic compound. Therefore, temperature controlled column experiments with pumice stone and river bed sediment were carried out contemplating the degradation of dissolved organic carbon (DOC) at 5 °C, 15 °C, and 25 °C. Water from the River Elbe with a mean DOC concentration of 5.6 mg/L was percolated through lab columns with a length of 0.5 m. The pore water velocity was set to 0.14 m/day and 1 m/day. For pumice stone and a pore water velocity of about 1 m/day, the degradation was lower (between 5 % at 5 °C and 15 % at 25 °C) compared to soil sediment (between 10 % at 5 °C and 21 % at 25 °C). However, a reduction of the velocity resulted in a higher removal of DOC for pumice stone but in an increase in DOC concentration for river bed sediment. This observation is explained by taking into account desorption processes of organic material accumulated in the river bed sediment. The investigation into the kinetics of atrazine degradation proved also a temperature dependency. The findings are important for the interpretation of field data and should be considered during design and management of bank filtration and artificial recharge systems.

Keywords

Dissolved organic carbon, atrazine, degradation kinetics, organics removal.

INTRODUCTION

River bank filtration has been used as an effective pre-treatment for drinking water supply in Germany for many decades. Nowadays, more and more countries are getting interested in using this treatment technology instead of abstracting water directly from the river. The application of river bank filtration as a low-cost pre-treatment technology might support the drinking water supply in areas with water scarcity, lack of treatment plants or poor hygienic conditions.

The efficiency of organics removal during aquifer treatment as result of abiotic und biotic processes is of ongoing concern (Braul et al.; 2001; Hambsch, 1992; Ray et al., 2002). Biodegradation and sorption are the main processes during riverbank filtration. The attainable removal efficiency depends on the quality of the infiltrating water and on local conditions as flow path length, flow time, sediment properties, quality of the background groundwater and the climate. Though the removal efficiency for many different compounds has been investigated at many bank filtration sites, there is only a few knowledge regarding the influence of physical boundary conditions as temperature and flow velocity on attenuation processes. However, water temperature depends on climate conditions and underlies seasonal variations. The effect of daily or seasonal temperature changes might be negligible at sites with long flow path lengths due to occurring equilibration of temperature changes. Yet, at sites with short retention times in the first few meters along the infiltration flow path, temperature changes will affect removal processes. Also, the question of transferability of experiences made under temperate climate conditions to warmer or colder climates is unacknowledged.

Field investigations demand high technical equipment, are time and cost intensive and commonly not allowed for simulating shock loads with hazardous chemicals. Thus, laboratory experiments are important tools to gain knowledge about the transport behaviour of single compounds (Worch et al., 2002), especially such of relevance for drinking water quality. Batch and column experiments allow working under defined conditions, e.g. temperature, concentration, reaction time, and column materials. To investigate attenuation processes of dissolved organic carbon and atrazine in river beds at different temperatures, column experiments can be set up using different sediments. To achieve conditions very near to natural conditions, original river bed sediment and river water can be used. Additionally, in order to separate the main processes pumice stone can be exploited. Pumice stone is supposed to be an inert material with high porosity allowing biofilm growth but no sorption onto organic carbon of the material. Consequently, degradation processes occurring within the pumice stone filled columns are attributed to biological degradation and sorption onto the biofilm only.

As far as water of the same origin and consequently of the same hydrochemical composition flows through the columns continuously, equilibrium between liquid phase and solid phase is assumed to be established for both the sediment and the pumice stone. Therefore, biological degradation is the driving removal force. Sorption is of importance as soon as boundary condition changes occur.

The objective of the presented investigations was to show the effects of extreme temperature conditions on the organics removal efficiency and transport behaviour during river bank filtration considering both total concentrations of dissolved organic carbon compounds and atrazine. The influence of changing pore water velocities on DOC removal was also considered. A better understanding of these effects provides the opportunity to transfer European experiences to sites with other climatic conditions. This might support the development of river bank filtration schemes in poorer regions.

METHODS

Experimental set-up

Bench-scale soil column experiments were performed at three different temperatures, 5 °C, 15 °C, and 25 °C.

The stainless steel columns (length 0.5 m, diameter 0.07 m) were filled with different materials and operated according to the experimental set-up either in flow-through regime (Fig. 1a) or circulating flow regime (Fig. 1b). To reduce air trapped in the pores, the water was pumped upwards through the columns. Previous investigations showed that there are no remarkable differences in the elimination efficiency between upward and downward percolation. As a feeding source, water of the same origin as the sediment was used. To operate at the three different temperatures, the columns where built into three thermostatic cabins at 5 °C, 15 °C, and 25 °C. To ensure reproducibility of the results, the experiments were performed at each temperature with two identically prepared columns, being fed by the same source water.

The column systems have been operated one after another with two different materials for about 9 to 12 months each: i) pumice stone that is volcanic rock material and ii) river bed sediment of a bank filtration site at the river Elbe. The characteristics of the materials are summarised in Table 1.

The columns were incorporated for a period of 4 and 8 weeks respectively. During this period, biofilm was established and material flushed out that was resolved due to the disturbed extraction of sediment.

a) flow through regime   b) circulating flow regime
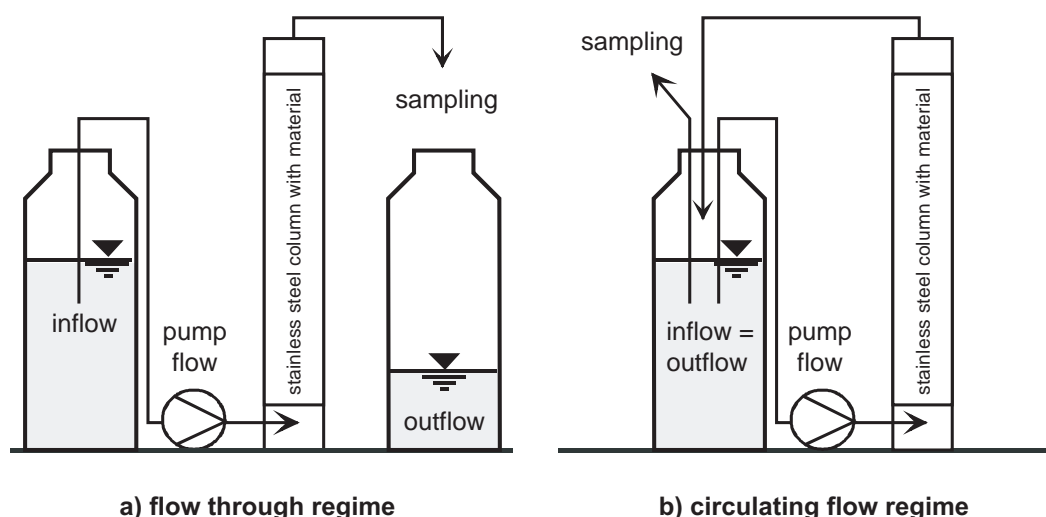
Figure 1. Experimental set-up

Table 1. Characteristics of materials for soil column experiments

material		pumice stone	Elbe river bed sediment
classification		volcanic rock	quartz sand
organic carbon content	f_{OC} (%)	0.024	0.49
hydraulic conductivity	k_f (m/s)	$4.3 * 10^{-2}$	$4.3 * 10^{-4}$
uniformity	d_{60}/d_{10} (–)	1.45	2.3

The flow velocity was changed by controlling the pumping rate. Within a first period the pore water velocity was set to 1 m/day which is equivalent to a contact time of 12 hours in the column. During the second period, a flow velocity of about 0.14 m/day and a contact time of 3.5 days were realised.

- *Flow-through regime*
  Percolation through the columns reflects the soil passage within the first half-meter of river bank infiltration, which is the most active biological zone and experiences the strongest temperature changes. The columns were operated with the same inflow water for cycles of two to three weeks. Water samples of the column inflow and outflow were sampled twice a week and analysed for DOC and ultraviolet absorbance at 254 nm (UVA-254). Occasionally, oxygen, nitrate and sulphate concentrations were also determined.

- *Circulating flow regime*
  Column operation in circulating flow regime enables to record the DOC degradation kinetics. Therefore, the same water percolated repeatedly through the column, the inflow containers served contemporarily as outflow container. By dropping of the column effluent into the container, oxygen insertion js supported. In discrete time steps, samples of a defined volume (100 mL) were taken out of the container. The experiments lasted 1 to 3 months.

Analytics

- *Organics*

Prior to analysis, the samples were filtered through a 0.45 μm cellulose-nitrate filter. For the determination of DOC, 30 mL of each of the samples was analysed with a Carbon Analyzer (model TOC-5000, Shimadzu). The samples were acidified to pH = 2 using hydrochloric acid and sparged by synthetic air for two minutes to strip out carbon dioxide. The DOC concentrations were calculated based on a standard calibration curve. The ultra-violet absorbance at a wavelength of 254nm, UVA-254, was measured with a UV/VIS-spectrometer (model Spekol, Zeiss Jena) using a cell with five-centimeter path length.

Specific ultraviolet light absorbance, SUVA, was calculated as quotient of UVA-254 and DOC. The SUVA is both a measure of the aromaticity of the organic compounds and a control parameter for the DOC data. Due to a higher stability of UVA-254 measurements against analytical and coincidental failures compared to the DOC concentration, outliers of SUVA often indicate outliers of DOC.

- *Atrazine*

To analyse atrazine, 50 mL of water sample were enriched on a conditioned LiChrolutEN-cartridge. After drying in nitrogen gas, the cartridge was flushed with acetonitrile. The extract was concentrated to 0.5 mL under nitrogen gas and filled up to 1 mL with pure water. The final solution was analysed with a HPLC/DAD (DAD L-4500, Merck) using a C-18-column (CC125/4 Nucleosil) and acetonitril/water eluent at isocratic conditions.

Contact time

It is assumed, that biological degradation is dominated by microorganisms settled on the sediment in the column. Consequently, the retention time for the circulating flow regime is calculated by the effective contact time in the column only. This effective contact time depends on the duration of the experiment and the ratio of pore volume and total water volume of the enclosure (Eq. 1). The total water volume changes as function of the water abstraction due to sampling out of the in- and outflow container (Eq. 2).

$$t_c = t \cdot \frac{V_p}{V_t} \tag{1}$$

$$V_t = V_p + V_{in} \tag{2}$$

with

t_c contact time (days),
t experimental time (days),
V_t .total water volume of the enclosure (mL),
V_{in} water volume of the inflow container (mL),
V_p pore volume (mL).

The contact time in the column for the flow-through regime is derived from breakthrough curves measured by chloride as tracer. The contact time is equivalent to that time t_{50}, when 50% of the tracer passed through the column. The pore water velocity, v_a, results from the ratio of column length and the time t_{50},

$$v_a = \frac{l}{t_{50}} \tag{3}$$

with

v_a pore water velocity (m/day),
l column length (m),
t_{50} contact time (days).

RESULTS AND DISCUSSION

The results showed a temperature dependency of both the organics removal efficiency and the degradation rate. Within the considered temperature range of 5 °C to 25 °C, higher temperatures are accompanied by higher removal efficiency and higher degradation rates.

Pumice stone and river water

For the investigations with pumice stone and river water, the averaged results for the removal efficiency at different velocities v_a are summarised in Table 2.

Table 2. Averaged organics removal. Flow-through regime with pumice stone and river water

	n	v_a (m/days)	c_0 (mg/L)	Δc_{DOC} (%)			UVA-254$_0$	Δ UVA-254 (%)		
				5 °C	15 °C	25 °C	m^{-1}	5 °C	15 °C	25 °C
Period 1	21	1.0	5.6*	4.4	11	15	15*	4.2	12	15
Period 2	20	0.14	6.2*	13	20	25	18*	12	21	27

∗: Average value.

During period 1 at a pore water velocity of 1 m/day, an averaged decrease in DOC concentration at 5 °C, 15 °C and 25 °C bei v_a = 1 m/day of about **5%, 10% and 15%**, respectively, was found. The percentage removal of compounds identified by UVA-254 reflects the same behaviour at all three temperatures.

Reducing the pore water velocity to 0.14 m/day in period 2 results in a more efficient organics removal of **13%, 20%, and 25%** at 5 °C, 15 °C, and 25 °C, respectively. This is not surprising but indicates the incomplete organics removal within the simulated transport through the column. The findings prove the importance of the considered contact time for the interpretation of organics removal along the flow path. Longer contact time causes higher concentration decreases. The record of kinetic curves confirms this statement. Therefore, measurements for a pumice stone filled column operated in circulating flow at room temperatures were exploited. In Figure 2, the measured concentration decrease with respect to the realised contact time is compared to the results achieved in the flow through experiments.
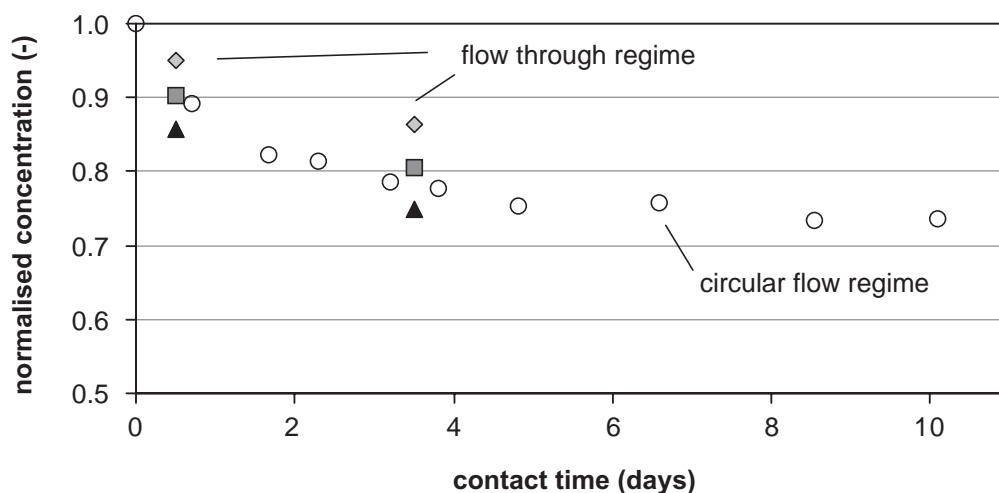
Figure 2. Kinetics of DOC removal in circulating flow (○ room temperature)
in comparison to flow-through regime (◆ 5 °C, ■ 15 °C, ▲ 25 °C) with pumice stone and river water

Though the DOC concentration decrease is not measured compound specific, the degraded organic compounds being responsible for the DOC removal within the observed time period can be combined in a so-called easily degradable fraction c_1. The organic carbon concentration of the compounds that are not degraded within the considered time interval, are summarised in fraction c_2,

$$c(t) = c_1(t) + c_2 \qquad (4)$$

As rough approximation, the recorded decrease can be described by a first order kinetics were the initial concentrations of the degraded organics are combined in c_{01},

$$c(t) = c_{01} \cdot (e^{-\lambda_1 \cdot t} - 1) + c_0 \qquad (5)$$

with

    c DOC concentration (mg/L)

    c_0 DOC concentration at t = 0 (mg/L)

    c_{01} concentration of the easily degradable DOC fraction at t = 0 (mg/L)

    λ_1 degradation rate constant (days^{-1}),

(Schoenheinz et al., 2003).

Applying Eq. (5) to the measurements for river water circulating through pumice stone filled columns (Fig. 2) and solving the equation by nonlinear regression results in the model

$$c(t) = 1.7 \cdot (e^{-0.6 \cdot t} - 1) + 6.5 \qquad (6)$$

for the characterisation of the investigated river water. The model formulates that 1.7 mg/L of the total dissolved organic carbon concentration of 6.5 mg/L are degradable with an average degradation rate constant of 0.6 days^{-1} within a contact time of 10 days.

River sediment and river water

Investigating the behaviour of DOC from the same river but circulating through columns filled with river sediment of the same origin as the river water showed quite a different kinetic behaviour (Fig. 3). While the experiment lasted six weeks, the effective contact time was about 21 days only. The redox conditions measured at the outflow of the column were anoxic.

At all three temperatures, an initial continuous decrease in DOC concentration was followed by a gradual increase in DOC concentration, beginning after an effective contact time of 50 hours (Fig. 3). These findings can be explained by considering desorption processes, taking place at the same time as the degradation occurs.

To describe the measurements, Eq. (3) was extended by a third compound, $c_3(t)$,

$$c(t) = c_1(t) + c_2 + c_3(t) \qquad (7)$$

The fraction $c_3 = c_d$ is supposed to be released from the high organic carbon pool (measured as TOC) of the sediment. The developed model (Schoenheinz, 2004),

$$c(t) = c_{01} \cdot (1 - e^{-\lambda_1 \cdot t}) + c_0 + c_d - c_d \cdot e^{(-k^* \cdot t)} \qquad (8)$$

with

    c_d concentration of the maximum desorbable DOC (mg/L),

    k^* desorption coefficient (days^{-1}),

assumes that the total organic carbon of the sediment is very high compared to the released concentration of DOC.

*Figure 3. DOC variation with contact time during degradation experiments
with sediment and filtrated river water in circulating operation regime*

Thereby, the maximum desorbable DOC concentration is considered to be time independent, c_d is constant within an experimental run. The application of Eq. (8) to measured data enables the determination of c_{01}, λ_1, c_d and k^* by nonlinear regression. In Figure 3, the comparison of the analytical data and the fitted curves is given. Table 3 recapitulates the resulting parameters.

Table 3. Fitting parameters for DOC degradation and desorption at three different temperatures

	Total DOC c_0 (mg/L)	Degradable DOC fraction c_{01} (mg/L)	Degradation coefficient λ_1 (day^{-1})	Desorbable DOC concentration c_d (mg/L)	Desorption coefficient k^* (day^{-1})
5°C		1.45	0.9	1.1	0.04
15°C	5.9	1.7	1.1	1.3	0.08
25°C		2.1	1.3	2.2	0.14

The model to describe the kinetics of both DOC degradation and desorption within the sediment filled columns (Eq. 8) showed good accordance to the observed DOC concentration data. Comparing the results at the three temperatures confirm the temperature dependency of both the rate coefficients and the portion of the DOC fraction degradable under the given conditions. Investigations by Klopp and Koppe (1990) with wastewater at different temperatures affirm these findings: the residual DOC content decreased in biological degradation experiments with increasing temperature.

Analogue to the DOC reduction, the desorption is temperature dependent. For the desorption of DOC from the sediment, desorption rate coefficients of 0.04 day^{-1} at 5 °C and 0.14 day^{-1} at 25 °C were derived from the model. The desorption coefficients were about one order of magnitude lower compared to the degradation coefficients

of 0.9 to 1.3 day^{-1}. This coincides with findings for polychlorinated biphenyls (Coates and Elzerman, 1986). Temperature dependence was also shown for the quantity of desorbable organic carbon. Higher temperature changes the equilibrium between solution and sorbed DOC into the direction of the solution. This agrees with the fact, that sorption is mostly an exothermic process. Consequently, the highest value of potentially desorbable DOC, c_d, was found for 25 °C. The results were confirmed by two more runs. For the degradation and desorption coefficients determined, the activation energy E_a was calculated applying the Arrhenius equation

$$\ln \lambda (or\ k^*) = -\frac{E_a}{R} \cdot (1\,/\,T) + \ln A \qquad (9)$$

to characterise the temperature dependency. With $R = 8{,}31$ J k^{-1} mol^{-1}, an activation energy E_a of 13 and 18 kJ/mol was determined for the dissolved organic carbon degradation (Schoenheinz, 2004), which indicates a low temperature dependency of organics removal within the investigated temperature interval due to the classification given by Atkins (1993). To evaluate a temperature change by 10 K, the factor Q_{10} as ratio of λ at the two temperatures can be used,

$$Q_{10} = \frac{\lambda (\vartheta + 10)}{\lambda (\vartheta)} \qquad (10)$$

The factor for λ at a temperature increase of 10 K was determined to be $Q_{10} = 1.2$. Referring to the desorption coefficients, E_a was calculated in the range of 20 and 43 kJ/mol and indicated a slightly higher temperature dependency.

River sediment and river water spiked with atrazine

Within the demonstrated column run in circulating flow regime, the kinetics of biological atrazine degradation was explored additionally. The river water was spiked with an atrazine concentration of approximately 15 µg/L. During the realised effective contact time of 21 days, the atrazine concentration was reduced to 1.6 µg/L at 25 °C, 3.2 µg/L at 15 °C, and 4.6 µg/L at 5 °C. The measured concentration slopes are given in Figure 4. The observed concentration decrease at all temperatures does not follow the expected first order degradation kinetics. Instead, an initially fast removal with rate constants of 1.1 day^{-1} (5 °C, 15 °C) and 1.2 day-1 (25 °C) occurred, that was
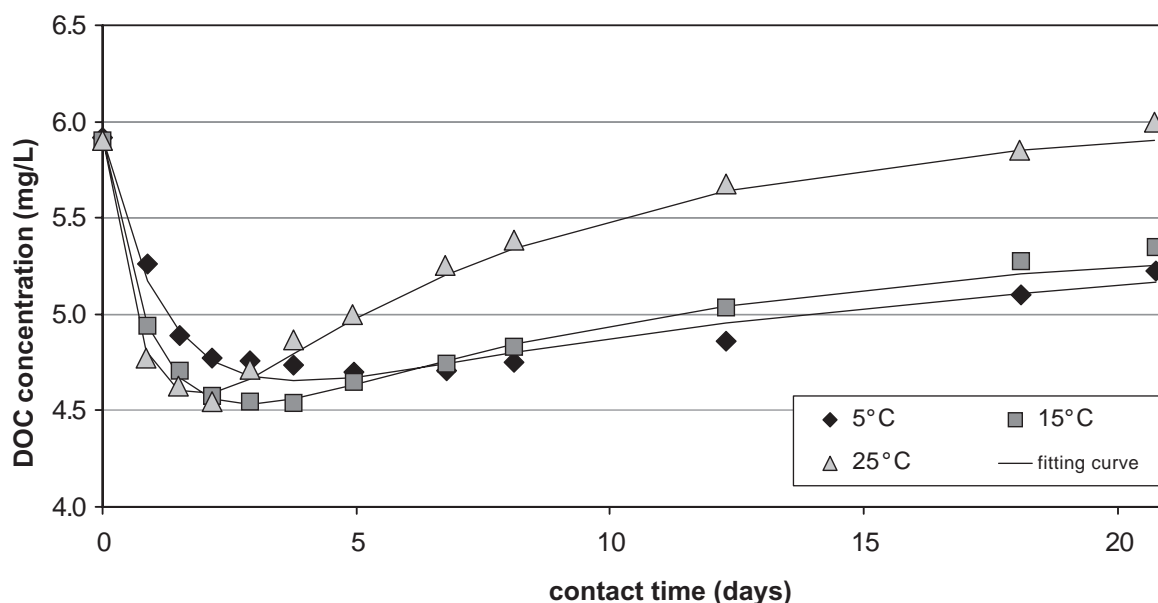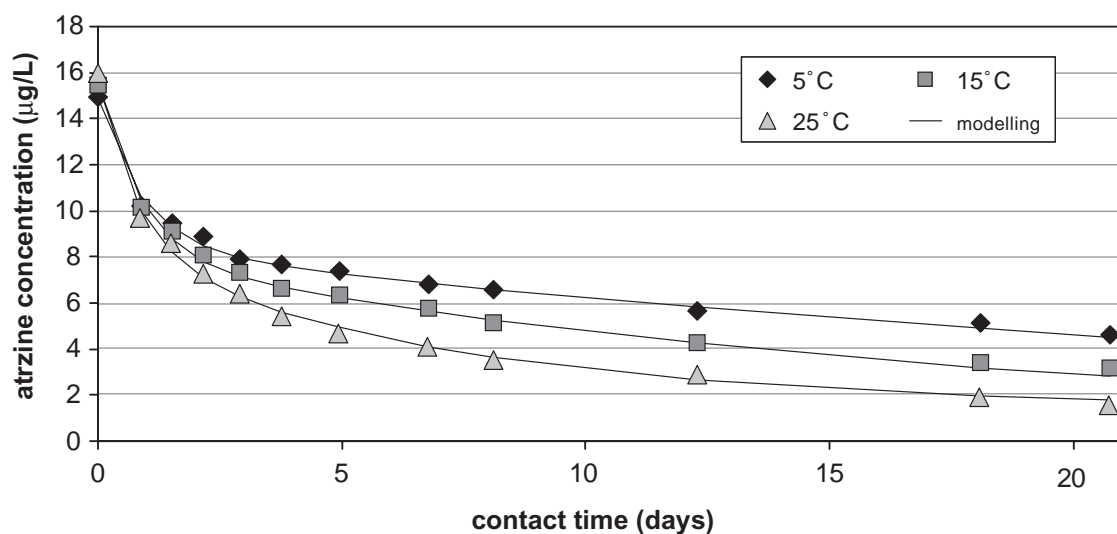

Figure 4. Decrease of atrazine concentration with contact time during degradation experiments with sediment and atrazine spiked river water in circulating operation regime

followed by a slower, obviously more temperature dependent degradation with rates of 0.03 day^{-1} (5 °C) and 0.14 day^{-1} (25 °C, Table 4). The experimentally determined degradation curves could be described formally by a two-component model.

Table 4. Parameters of atrazine degradation. Circulating flow with sediment and river water

	c_0 (mg/L)	c_{01} (mg/L)	λ_1 (d^{-1})	c_{02} (mg/L)	λ_2 (d^{-1})	r^2 %
5 °C	14.9	6.5	1.1	8.4	0.03	99.4
15 °C	15.5	7.6	1.1	7.9	0.05	99.7
25°C	15.5	7.1	1.2	7.0	0.14	99.6

For the slower rate constants, there was a factor Q_{10} = 1.7 for a change from 5 °C to 15 °C and Q_{10} = 3 for an increase between 15 °C and 25 °C. The identification of two rate constants for the atrazine removal can be related to different effects. Most likely, there was a fast initial loading of the sediment by atrazine due to sorption. After equilibrating the atrazine concentration in the solution and the sorbed phase, biological degradation determines the removal. However, another reason might be the necessity of cometabolism. In this case, after the depletion of the easily degradable organic carbon the degradation of atrazine gets slower due to slower degradation of poorer degradable organic compounds. There are further investigations necessary to clarify the observed phenomena.

CONCLUSIONS

Higher average temperatures along filtration paths result in higher degree of DOC degradation, faster DOC and faster atrazine removal. The degradation rates for the dissolved organic carbon removal going along with a temperature increase by 10 °C cannot be simply doubled. For most investigated waters and materials, the increase factor was less than 2, while for atrazine degradation the factor was between 1.7 and 3 depending on the temperature interval.

Because DOC degradation in water depends on temperature, laboratory investigations carried out under room temperature conditions will often result in an overestimation of DOC elimination. The influence is not as important for the evaluation of easily degradable organic compounds that are degraded within in the first centimetres of the infiltration zone. However, for studies into further degradation of the bulk organics, temperature influence cannot be neglected. The equalisation of infiltration water temperature results in Middle Europe in average values of 10 °C which is about the half of average room temperature.

Biodegradation, sorption and accumulation of organics in river bed sediments seem to reach a state of equilibrium, which can be affected significantly by changing flow conditions and concentration changes in the source water. Periods of low flow or stagnation of water in the pore space of the river bed sediment result in an increase in DOC concentration in the infiltrate. This process is assumed as diffusion controlled desorption or resolution of organics from the sediments or biofilms. The investigations into DOC removal at different pore water velocities showed, that longer retention times enable an ongoing organics removal. Longer flow path of the infiltrate would not only be of advantage to equilibrate changes in DOC concentrations, especially in warm climates, but also decrease the organics concentration at the abstraction well depending on landside groundwater quality. Thus, optimisation of flow path length and retention time is the key task for effective management of river bank filtration sites and should also be considered for artificial recharge systems.

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Clogging effects

Characterisation of turbidity and well clogging processes in a double porosity Chalk aquifer during the South London Artificial Recharge Scheme trials

Malcolm Anderson, Rachel Dewhurst, Michael Jones and Keith Baxter

Abstract

Turbidity was a significant issue during the South London Artificial Recharge Scheme trials in the Chalk aquifer. The turbidity of abstracted water exceeded the drinking water standard of 1 NTU for up to 1 week following injection of mains water, typically reducing from maximums of >100 NTU. As turbidity was associated with borehole clogging, a number of approaches were used to investigate the causes of the turbidity and to identify an appropriate turbidity and borehole clogging management strategy. Turbidity was recorded using a meter, and data logger, accurate to 0.01 NTU, whilst step test hydraulic analysis tracked the changes in clogging and fracture opening associated with the turbidity events. Scanning electron microscope (SEM) and chemical analysis identified the mineralogy of the turbidity solids, whilst PHREEQC geochemical modelling was used to identify the processes that were causing the turbidity events. Four separate causes of turbidity were identified: (a) short duration (<4 hours) disequilibria responses in the aquifer; (b) equilibration of injection water, (c) some injection water mixing scenarios and (d) fracture network clearance when boreholes are pumped at higher rates for the first time. Management of the borehole recharge and pumping strategy was found to satisfactorily control turbidity.

Keywords

Artificial recharge, Chalk, double porosity, turbidity, well clogging.

INTRODUCTION

At the inception of the South London Artificial Recharge Scheme (SLARS) investigation programme, it was clear that boreholes were the only feasible artificial recharge solution due to the confined nature of the target aquifer and the lack of available space in the highly urbanised environment of south London. The disadvantage of this approach was the possibility of borehole clogging from suspended solids, contained either in the recharge water, or created by geochemical reactions between the injected water, native groundwater and the aquifer matrix. The assumed solution to this problem was to use the same borehole for both recharge and abstraction, so that any clogging during recharge would be minimised by clearance during subsequent abstraction. Consequently the focus of the testing programme was to validate this assumption by the collection of detailed turbidity, water quality and well performance data in response to recharge and abstraction cycles. Substantial turbidity caused by suspended solids was found in the reabstracted water at the SLARS investigation sites following injection of recharge water. Furthermore, sequential step abstraction tests proved a significant reduction in borehole performance occurred as a result of the first phase of recharge injection. The turbidity issues raised two main concerns; 1) would turbidity cause progressive and long term deterioration in the borehole capacity; and 2) how could the turbidity issue be cost effectively treated or managed in order to meet water quality standards. This paper discusses how the various processes causing the borehole turbidity, aquifer clogging and well performance deterioration were assessed, and how this knowledge was used to successfully select appropriate management and treatment solutions. The discussion focuses on the results from the Streatham test site in south-west London.

METHODS

The testing programme consisted of two testing phases. Phase I consisted of a long duration cycle test split into an initial recharge stage and a subsequent abstraction stage. Phase II contained two short, one medium and one long duration cycle tests, each split into an initial recharge stage and a subsequent abstraction stage. During Phase I the recharge rate was limited by the mains capacity to 7 Ml/d, whilst the abstraction rate was initially 6.5 Ml/d and then was increased to 10 Ml/d during the Phase I constant rate test. Prior to Phase II, the mains capacity was increased so that sustained recharge rates of 14 Ml/d were achieved; abstraction rates were also increased to 12 Ml/d. Step abstraction tests were conducted before and after each stage in order to determine the progressive deterioration or improvement, if any, in borehole performance. The step tests were completed in both abstraction and recharge modes as appropriate, and consisted of 5, one hour steps of progressively increasing rates. These recharge step-up and abstraction step-up tests were analysed using the method of Bierschenk (1963). However in order to assess the rate dependence of the turbidity issues, on one occasion a step-down abstraction test of progressively decreasing rates was carried out.

The turbidity of both the abstracted water and the recharge water prior to injection, were measured continuously using a Hach turbidity meter accurate to 0.01 NTU and recorded on a data logger. Water samples were taken towards the end of each rate step during the step testing and twice a week during constant rate tests. Redox potential, pH, temperature, electrical conductivity and dissolved oxygen were continuously measured using a YSI flow through cell meter, calibrated prior to each test, and recorded on a data logger. These measurements were validated using laboratory analyses of water samples and regular field measurements of these parameters using a hand held instrument. The suspended solids causing turbidity were collected from the turbidity meter filter and were analysed using scanning electron microscopy (SEM) coupled with a chemical analyser.

The potential for mineral precipitation to cause turbidity was assessed using the PHREEQC model. Table 1 records the limited number of possible mixing and equilibration scenarios assessed with the model. PHREEQC calculates the thermodynamically favourable mineral compositions resulting from mixing injected recharge water, native groundwater and equilibrium with the solid minerals of the aquifer matrix, then determines the degree of saturation of the relevant minerals. Whereas it can be assumed that groundwater will always be equilibrated with the aquifer minerals, both unequilibrated and equilibrated recharge water will be present in the aquifer. The constituent concentrations of the injected recharge and native groundwater used for data input into the model were determined from representative water sample analyses obtained from the testing programme. Calcite and chalcedony, chalk and flint respectively, were selected as the aquifer mineralogy.

Table 1. Summary of mineral thermodynamic mixing scenarios

Scenario No.	Recharge Water	Equilibrated Recharge Water	Groundwater	Aquifer Matrix	Result
1	Yes	No	No	Yes	Calcite precipitation
2	Yes	Yes	No	Yes	Silica and calcite precipitation
3	Yes	No	Yes	Yes	Calcite dissolution
4	No	Yes	Yes	Yes	Calcite dissolution

WELL PERFORMANCE TESTING RESULTS

The results of the sequential step tests in the Streatham abstraction borehole (ABH) are shown on Figure 1(a) and show substantial improvement in borehole performance between 6th August 1993 (Line A) immediately following
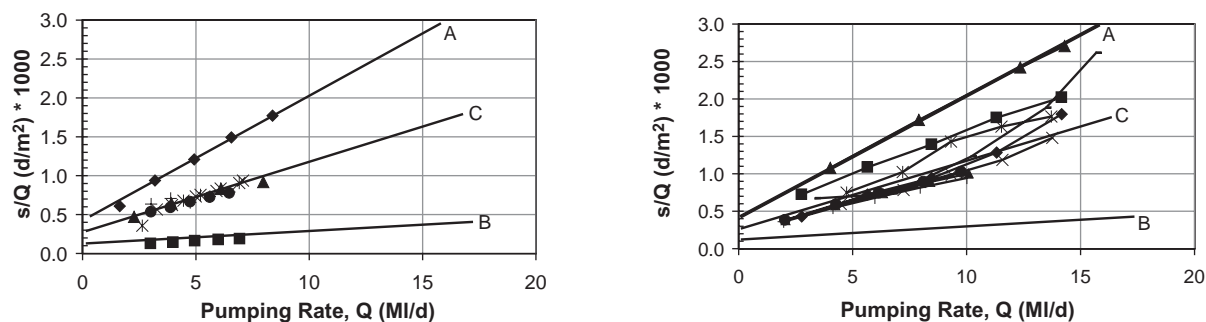
Figure 1. (a) Phase I, and (b) Phase II step test results

borehole construction, and 2nd May 2002 (Line B) immediately prior to the recharge testing programme. The 5 recharge and abstraction step tests undertaken during the remainder of Phase I of the testing programme, including a recharge step test undertaken on 3rd May 2002 all plot along Line C. The 3rd May 2002 recharge step test was the first occasion on which recharge water was injected into the aquifer. As the first step of this step test plots on Line C and not on Line B, or between Line B and Line C, it shows that the reduction in well performance had already occurred by the end of the first hour of this test. Therefore the clogging was effectively instantaneous, whereas well performance thereafter, was apparently constant. However during the Phase II step tests some differences in well performance were recorded, as illustrated in Figure 1(b), where lines A, B and C correspond to the well performance responses on Figure 1(a). Although as in Phase I, the majority of the Phase II step test results fall on line C, two types of tests consistently did not. Firstly, recharge step tests that followed sustained periods of abstraction show deterioration in well performance during steps 3 to 5 only, and are the upward curved responses on Figure 1(b). Secondly, recharge step tests at the end of short period recharge cycle tests of 2 to 10 days duration have significantly reduced well performance; these are the straight lines that plot between Line A and C on Figure 1(b). The first of these tests recorded the worst well performance coincident with Line A, whilst the two subsequent tests recorded well performance that progressively trended towards Line C. In contrast, the results of recharge step tests that followed sustained periods (>10 days) of recharge, plotted just below Line C. As short duration cycle tests were not undertaken in Phase I, it is possible to conclude, in conjunction with the PHREEQC modelling data discussed later, that well performance during recharge deteriorates temporarily over a period of approximately 3 hours to 10 days, but thereafter improves again, approaching the well performance of Line C. Furthermore the degree of deterioration is likely to become less severe as the number of recharge and abstraction cycles is increased, as the aquifer is 'conditioned'.

SEM AND MINERAL THERMODYNAMIC ANALYSIS

The step test results were validated by the PHREEQC thermodynamic modelling, (Table 1). These results show that calcite will precipitate as the recharge water equilibrates in the presence of the Chalk matrix. This occurs as soon as the water enters the aquifer and is likely to have been the cause of the rapid clogging of the aquifer matrix indicated by the step test results shown on Figure (1a). Once the recharge water equilibrates, mixtures of unequilibrated and equilibrated recharge water can also precipitate both calcite and silica. However mixing of either recharge water or equilibrated recharge water with native groundwater results in calcite dissolution. Consequently the model predicts a transition from calcite precipitation to dissolution as recharge progresses, consistent with the step test results on Figure 1. The SEM analysis provided further confirmation of the conceptual model by identifying the suspended solids causing turbidity as principally calcite ($CaCO_3$) and alumino-silicates (clays) with some minor $Fe(OH)_3$. Calcite and clays are the major and minor mineral components of Chalk rock and therefore any dissolution of calcite contained in the aquifer matrix is also likely to release any clays cemented by the calcite. The origin of the $Fe(OH)_3$ in the reabstracted water was not evaluated in any detail as iron minerals are abundant in the overlying and hydraulically connected Thanet Sands, occur independently in the Chalk and were present in low con-

centrations in the injected recharge water. Furthermore sequential water sample analysis showed that iron participates in bacterially assisted redox reactions following recharge. Consequently iron turbidity was not considered in any detail as: (a) iron was a small percentage of the suspended solids causing turbidity; (b) iron was consistently below its PCV in reabstracted water, and (c) the solubility and mobility of iron is affected by the redox reactions.

FRACTURE NETWORK CLEARANCE

It is reasonable to suppose from Figure 1(a) that the improvement in well performance between 6th August 1993 (Line A) and 2nd May 2002 (Line B) was the result of fracture network development in the vicinity of the Streatham ABH during operational abstraction. Progressive clearing out of individual fractures in the network was subsequently well demonstrated during the Phase II constant rate abstraction test conducted between 23rd December 2002 and 6th March 2003, (Figure 2a). It was possible to conclude that the simultaneous occurrence of turbidity spikes and improved well performance must indicate individual fracture network clearance events as the flow meter chart recorder confirmed did not record any changes in abstraction rate. Turbidity data from the August 1993 tests suggest that fracture clearance events occurred frequently over the first 20 days of testing but did not occur thereafter. As a second step test was not carried out at this stage, it is possible the substantial performance improvement (Line B on Figure 1a) had already occurred by the end of the first 20 days of pumping. Notably, the Phase I constant rate test was pumped at the ABH licence rate of 6.5 Ml/d during the period of 12th to 15th June 2002 and no turbidity spikes occurred, (Period A on Figure 2b). However when the pumping rate was increased on 15th June 2002 to 10 Ml/d, a rate significantly higher than the ABH had previously been pumped, turbidity spikes repeatedly and frequently occurred, (Period B on Figure 2b). The only time this process reoccurred, was during the Phase II constant rate test (Figure 2a) when the ABH was pumped at 12 Ml/d, which was the first occasion the ABH was pumped at rates higher than 10 Ml/d. The results show that fracture network clearance in the Streatham borehole is a function of the abstraction rate relative to the previous highest sustained abstraction rate in the ABH and the rate of fracture network clearance attenuates rapidly during constant rate pumping.
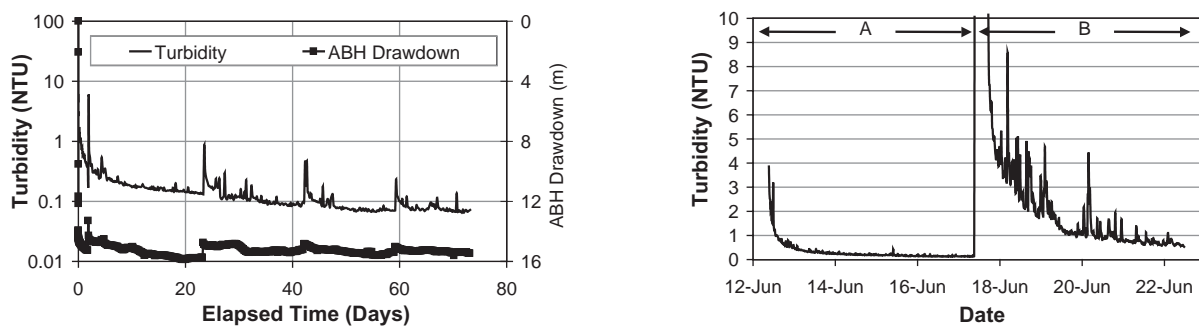
Figure 2. (a) Comparison of turbidity events to drawdown, and (b) Turbidity responses from the first period (6.5 Ml/d) and second period (10 Ml/d) Phase I constant rate abstraction test

TURBIDITY TESTING RESULTS

Elevated turbidity commonly occurs in Chalk abstraction boreholes immediately after pumps are switched on. This response occurs as a result of surging in the rising main (identified by short-lived turbidity spikes of < 5 minutes duration) and temporary disequilibria caused by flow in the fracture network mixing water from different locations of the aquifer matrix. The latter response is well illustrated on Figure 3(a) from the step abstraction test of 2nd May 2002. In this test, which was carried out before any recharge injection was attempted, turbidity peaks at 4.3 NTU before falling back to a baseline level of 0.05 NTU after a period of about 3.5 hours. The results show that providing the test flow rate does not exceed previous highest sustained flow rates, the turbidity in the aquifer is not rate

dependent and is well below the turbidity water quality standard of 1 NTU. This turbidity response is again seen during a post-recharge step abstraction test two days later (Figure 3b), although on this occasion the peak turbidity from the disequilibrium response is much greater, (85 NTU), due to the presence of injected recharge water in the aquifer from the recharge step test on the previous day. However in the second half of the step test following recharge there is also a new response. Although the turbidity attenuates significantly during each step, each time the pumping rate was increased, increasingly greater concentrations of suspended solids were mobilised. The rate dependence of the mobilisation of suspended solids was subsequently proven by the step-down and step-up abstraction tests completed respectively on 10th and 11th June 2002, (Figure 4), following 26 days of recharge during the constant rate recharge test. Turbidity during the step-down test peaked during step 1 and attenuated thereafter. However in the step-up test on the following day, significant peaks in turbidity only occurred during step 5 and to a minor degree during step 4, conclusively showing turbidity is a function of the flow rate following recharge.
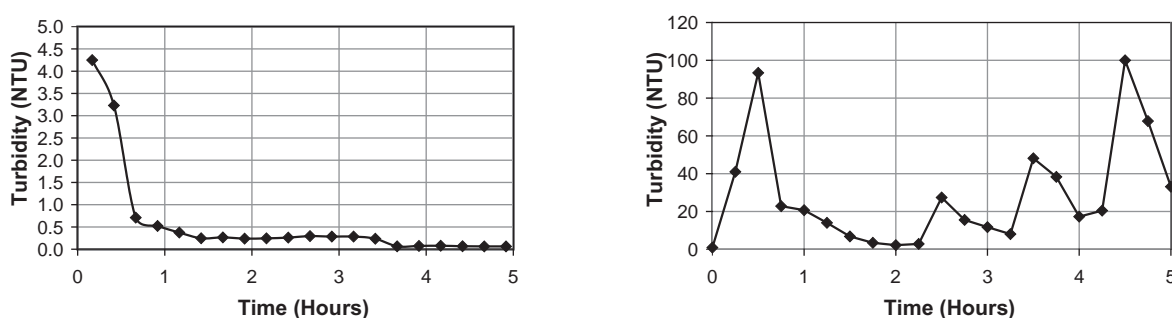
Figure 3. (a) Turbidity during the 2nd May 2002 abstraction step test,
and (b) Turbidity during the reabstraction step test 2 days later
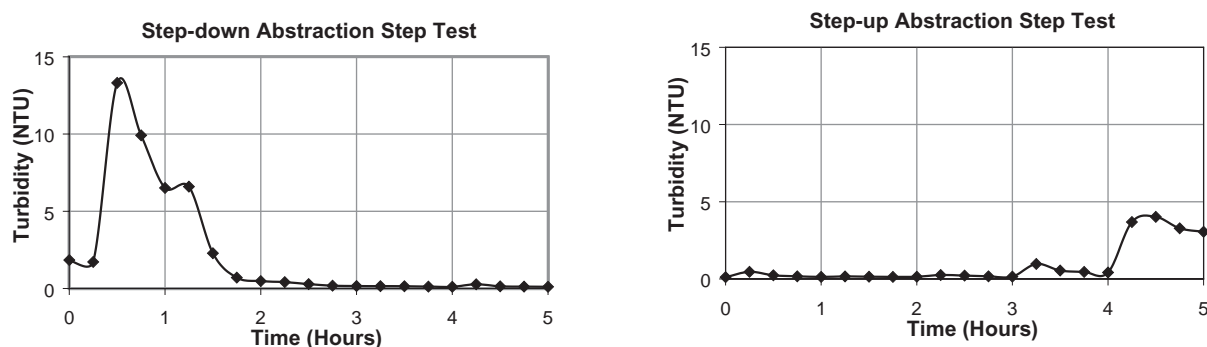
Figure 4: Turbidity responses during the (a) step-down, and (b) step-up abstraction step tests

DISCUSSION

As calcite precipitated in the pores of the aquifer matrix can be expected to clog the matrix and to become immobilised, the calcite turbidity mobilised during reabstraction following recharge is likely to have been precipitated in the fracture network. Similarly, the clay turbidity must be released by calcite dissolution within the fracture network. Bagnold (1973) shows that mobilisation of the precipitates is a function of the cube of velocity and this explains the strong relationship of pumping rate to the occurrence of turbidity. Effectively the data show that providing constant pumping rates are maintained, cleaning by scouring is rapidly achieved and turbidity falls well below the 1 NTU standard. However as the pumping rate is increased, the fracture network is either scoured progressively further afield, or previously immobile precipitates are mobilised in the faster flows in the fracture network. Where the ABH is pumped at a rate that exceeds the previous highest sustained pumping rate, aquifer material in addition to precipitates will be periodically scoured in discreet fracture network clearance events. With

this knowledge it is possible to propose a turbidity management strategy. Pumping to waste at rates the same as, or higher than, the subsequent water supply rate was found to be an effective turbidity clearance strategy following recharge injection operations. Similarly, it was found that turbidity spikes caused by fracture network development could be avoided if the borehole was pumped at a rate below the previous sustained maximum rate.

CONCLUSION

Scanning electron microscope (SEM), mineral thermodynamic modelling, turbidity monitoring and detailed step testing can be used to identify the processes causing turbidity events and fluctuations in well performance. In the SLARS investigation area, four separate causes of turbidity were identified: (a) short duration (< 4 hours) disequilibria responses in the aquifer; (b) equilibration of injection water, (c) some injection water mixing scenarios and (d) fracture network clearance when boreholes are pumped at higher rates for the first time. As a result of identifying each of the processes causing turbidity, it was feasible to identify an appropriate management strategy for turbidity.

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Clogging processes in a bank filtration system in the littoral zone of Lake Tegel (Germany)

Günter Gunkel and Anja Hoffmann

Abstract

Several physical, chemical and biological mechanisms play a role in the clogging of sediment interstices regularly observed in sand filter and infiltration basin systems. Whereas the hyporheic zone has been the focus of many investigations, little is known about the lenitic limnic zone, which is typical in lowland areas with lakes and low flow rivers. One must assume that clogging is regulated by both the build-up and the input of particulate organic matter (POM).

In the present study, we collected samples from the littoral zone of Lake Tegel, Berlin, Germany, to analyze relevant carbon turnover processes. High concentrations of POM were detected in the upper sediment layer, with 3.4% ds down to 20 centimeters depth. A very high biomass of interstitial algae was found in the first 5 cm of sediment (25 μg Chl a per cm^{-3}); this was 1000 times higher than in the lake water. The pore system of the sediment was filled to about 50% with POM, and the algae volume comprised about 25 % of POM. Only low amounts of POM were transported from the lake water downwards into the interstices, and the transport of FPOM (a few centimeters per day) was much lower than the water flow (32–260 cm d^{-1}). The DOC concentrations in lake water (~8 mg L^{-1}) and interstitial water (~6 mg L^{-1}) were determined by the in situ bioactivity of interstitial organisms in addition to DOC input from lake water.

Keywords

Bank filtration, clogging, hyporheic zone, interstices, littoral, meiofauna.

INTRODUCTION

The hyporheic zone is a dynamic ecotone between the aquatic and the groundwater ecosystems. Consequently, the physical, chemical and biological parameters used in limnology and hydrogeology to characterize the two ecosystems tend to display high gradients. The significance of the ecotone is given by the natural and artificial groundwater recharge and the need for adequate groundwater protection.

The hyporheic zone has been the subject of many investigations, most of which have focused on marine sandy beaches, but also on mountain areas with running water containing sandy and stony substrates. This lotic-limnic hyporheic zone must be distinguished from lenitic hyporheic interstices, which are typically found in lowland areas with lakes and low flowing rivers. Compared to lotic systems, fine sand and silt are the dominant substrates in lowland rivers, and infiltration processes are strictly reduced.

Although we have a good understanding of the marine hyporheic interstice (Higgins and Thiel 1988) and of the limnic-lotic hyporheic zone (Pennak 1988, Brunke and Gonser 1997), little is known about the ecological processes that maintain the limnic-lenitic ecotone (Pennak 1988, Hakenkamp et al. 2002). The main mechanisms are the infiltration of water, the mechanical retention of suspended particles, and the mineralization of particulate organic matter (POM) and of dissolved organic carbon (DOC). However, the hyporheic interstice is also a zone of primary production by microalgae, which, like meiofauna, are part of the interstitial food web (Wasmund 1986, Beulker and Gunkel 1996). Thus, biological processes like primary production of biomass, fragmentation of crude POM

(CPOM) to fine POM (FPOM) by consumer organisms, the uptake of DOC by heterotrophic organisms and the excretion of DOC by all interstitial organisms are some factors to be considered. Our investigations at Lake Tegel in Berlin, Germany demonstrated the significance of the bio-coenosis on interstitial clogging. As the numbers of different species (diversity) and organisms (abundance) were remarkably high, we assume that the meioflora and meiofauna play a significant role in the water purification processes related to bank filtration (Beulker and Gunkel 1996).

A better understanding of colmation, the process by which the clogging of sediment interstices occurs, is required since it can severely obstruct the artificial vertical sand filters for groundwater recharge (Rinck-Pfeiffer 2000). The permeability of the hyporheic interstices depends on the hydraulic conductivity of the sediments layers, which is influenced by mechanical, chemical and biological processes related to clogging. Well-known mechanical factors causing clogging are the input of fine sand particles (silt) and the occurrence of gas bubbles (oxygen or methane), while chemical processes are mainly the precipitation of carbonates, iron hydroxides, and sulphur or sulphides. The main biological sources include the excretion of extracellular polymeric substances (EPS), in most cases poly-saccharides or polypeptides (Flemming et al. 1998). Meiofauna presumably influence or regulate the clogging intensity by engaging in activities such as burrowing. However, data on these dynamic interactions, the build-up of EPS and structural destruction by interstitial fauna are scarce. Most of the available information has to do with artificial ground water recharge systems, the inflow conditions of which are significantly different from those of littoral zones.

METHODS

Our investigations were carried in the sandy littoral zone of Lake Tegel, a lake-like extension of the River Havel, in Berlin. Lake Tegel has an area of 2.82 km^2 and a mean depth of 5.5 m; it can be characterized as moderately eutrophic. Along the shoreline of Lake Tegel, 130 wells for water extraction are located close to the surface (30 m) and below the marl (< 62 m). The lake is therefore subject to exfiltration. Sampling equipment was inserted in the littoral zone of the lake (water depth of about 50 cm). Sampling was conducted regularly (infiltration rate and water chemistry every 3 weeks, meioflora and fauna every 6 weeks) for a period of one year.

The infiltration rate was measured in pressureless enclosures 25 cm in diameter; inflow into the enclosures was determined using water-filled bags. DOC was determined using a TOC analyzer (*LiquiTOC*). Sediment cores were extracted by pressing plastic tubes (Ø 5 cm) into the sediment; freeze cores were also obtained following the method of Bretschko and Klemens (1986). The abundance and diversity of interstitial flora and fauna in the samples was assessed after separation by water cascade analysis. The taxonomic determination was reduced to about 90% of the species and biomass, and the species were generally determined to the level of the genus (Beulker and Gunkel 1996). Chlorophyll was assessed by sand extraction after DEV (1996). Carbon and nitrogen levels were determined using a CN analyzer (Carlo Erba Inst.). The structure of the biofilm and the occurrence of EPS was studied by REM-EDX analysis using small sediment cores of 10 x 10 mm, which were spluttered with gold and exposed directly in the REM. POM was measured gravimetriccally as ash-free dry mass after cascade floating separation from sand, silt and meiofauna. Proteins and polysaccharides were identified by light microscopy after staining the sediment samples with Coomassie Brilliant Blue G-250 for proteins and Alcian Blue for polysaccharides.

RESULTS AND DISCUSSION

The Lake Tegel test site is a sandy area (fine sand: 10−30%, medium sand: 55−60%, coarse sand: 5−30%) with some reed stands (*Phragmites australis*) and water lilies (*Nuphar lutea*). Both stands are remnants of an ancient reed zone, which today is protected by a palisade, but is still impacted by swimmers and lake eutrophication. Characteristic parameters of the littoral zone at Lake Tegel are given in Table 1. The infiltration rate was determined to be 3−27 L m^{-2} h^{-1}.

Table 1. Characteristics of the eastern littoral zone of Lake Tegel (near the island of Reiswerder)

	Depth (cm)				
	0 – 5	*5 – 10*	*10 – 15*	*15 – 20*	*20 – 25*
Sediment water content (%)	20.3	20.2	11,5	14.5	17.6
Mean particle diameter (mm)	0.2	0.3	0.5	0.5	0.4
Sorting coefficient (Müller)	1.8	1.7	1.9	1.6	1.5
Nonuniformity	3.8	4.0	3.8	2.4	2.4
Pore volume coefficient	0.40	0.40	0.26	0.31	0.36
Kf (Hazen, 10 °C, m s^{-1})	7×10^{-5}	7×10^{-5}	3×10^{-4}	5×10^{-4}	4×10^{-4}

The littoral sediment is enriched with particulate organic matter (POM), which must be divided in two groups, CPOM (crude POM), a food source for shredder organisms, and FPOM (fine POM), a food source for sediment feeders and filterers. POM can be transported with the water flow into the sediment and is retained in the pore system; this may be one reason for clogging phenomena. The POM concentrations are high (about 1% ds) in the upper layer (down to 20 cm) and decrease to 0.25% at 50 cm sediment depth (Fig. 1). Investigations carried out with monodisperse polymeric resin microparticles and with fluorescein (FITC)-labeled fine organic particles (algae, leaves) indicate that POM is not transported downwards, and that FPOM transport is restricted to the topmost 2 to 3 cm of sediment (Beulker et al. 2005). Water flux, as determined in infiltration chamber measurements (3–27 L m^{-2} h^{-1}), leads to an infiltration velocity of 32–260 cm day^{-1} in the pore system, while the particle transport amounts to a few cm per day. Therefore, water infiltration does not lead to any remarkable extent of passive particle transport. However, significant quantities of CPOM are fragmented to FPOM in the interstitial biocoenosis.

Besides POM input, primary production is also assumed to be another source of organic carbon in the interstices; this theory was confirmed by our microscopic analyses. The sediment particles were densely colonized with algae (mainly diatoms), which are known sources of extracellular polymeric substances (EPS; Fig. 2) and detrital POM. This interstitial flora is a biocoenosis especially adapted to the sediment pore system. That at Lake Tegel was colonized mainly by epipsammic organisms (aufwuchs) attached to sediment grains (abundance: 60–90%). Diatoms were the dominant species; diatoms are able to move within the sediment and can even migrate to micro zones under optimal conditions. The most prevalent diatom species were Amphora pediculus, Cocconeis spp.,Achnanthes clevei, A. minutissima and A. lanceolata. Some interstitial algae like nanoflagellates (Trachelomonas, Euglena) were also found. Pelagic species from the lake water are also transported into the interstices to a small extent (abundance: 0–25%; Beulker and Gunkel 1996).

Figure 1. Vertical particulate organic matter (POM) distribution (> 100 µm) in the bank filtration system within the upper littoral sediment layer of Lake Tegel
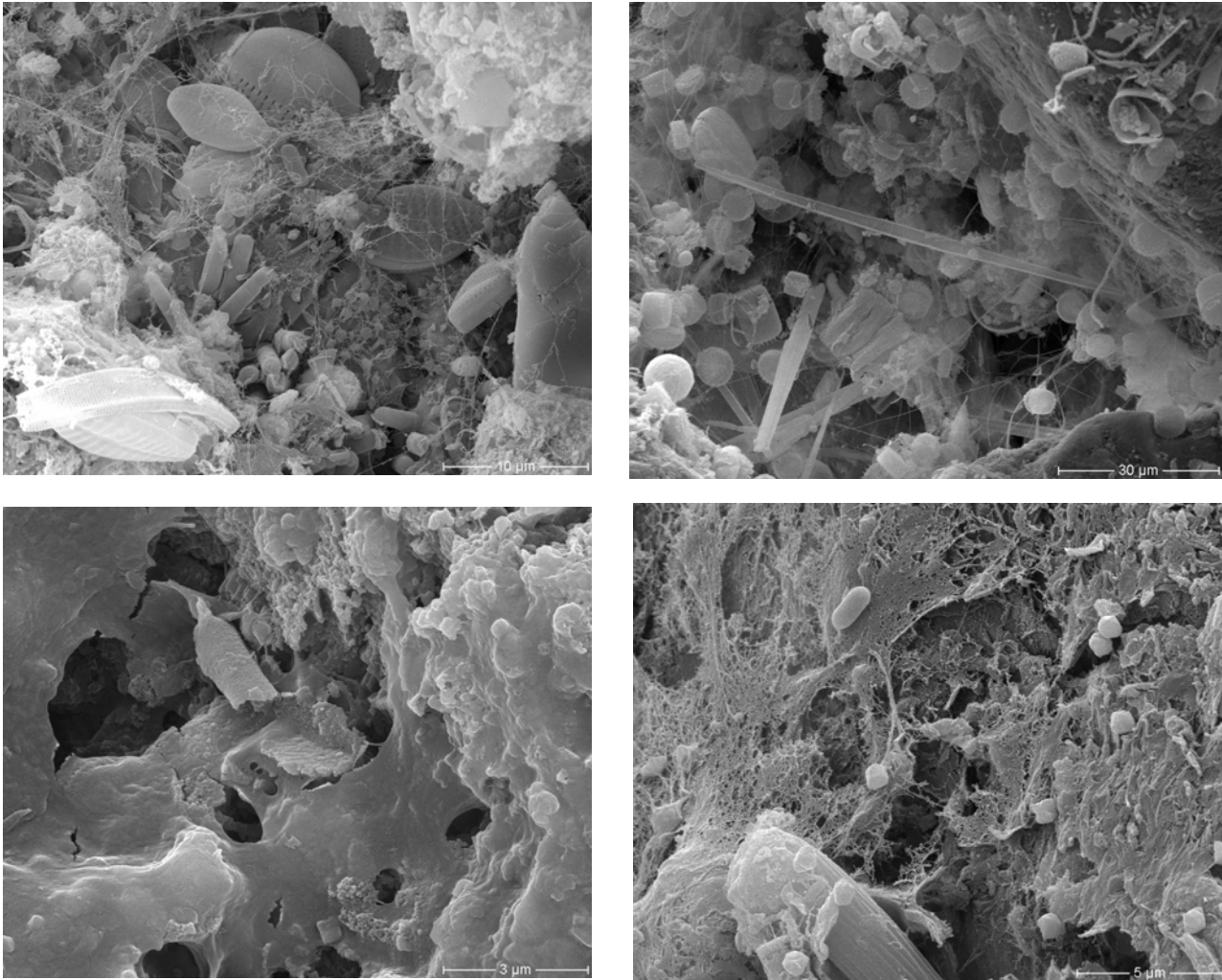
Figure 2. Interstices of the littoral sandy zone of Lake Tegel,
a) 3–4 cm depth, diatoms with fibrillary EPS, 24.8.04; b) 0–1 cm depth, diatoms with EPS 25.02.04;
c) 7–8 cm, flat EPS, 05.02.04; d) 3–4 cm depth, dense EPS web with bacteria and diatom, 17.05.04

The vertical distribution of chlorophyll a (Chl a) confirmed our assumption that algae significantly influence the composition of the upper sand layer (Fig. 3). The Chl a concentration was very high (21 to 28 µg cm^{-1}) in the upper 5 centimeters of sediment and decreased with depth, with only traces of Chl a detected at 9 centimeters depth. A Chl a concentration of about 25 µg cm^3 is extremely high compared with the lake water, which even under eutrophic conditions contained only about 20 µg L^{-1}. The algae biomass in the interstices was about 1,000 times higher than in the overlying water body, and the algae biomass in the upper 5 cm of sediment interstices corresponded to the algae biomass of a 50 m water column. The calculated algae biomass (Margaleff 1983) was about 40 g C m^{-2}; this is certainly high, but similar to figures reported for epipsammic algae in small creeks (Cummins et al. 1966, Ruhrmann 1990).
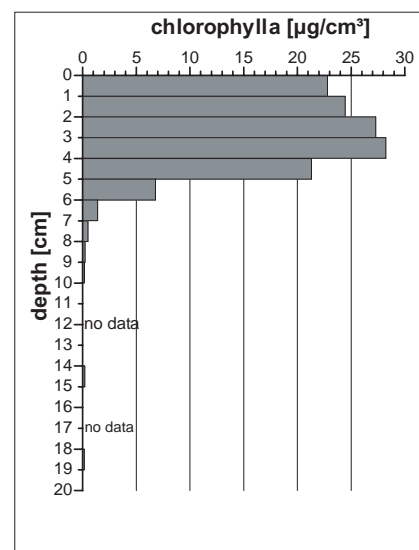
Figure 3. Vertical chlorophyll a concentrations within the upper littoral sediment layer of a bank filtration site at Lake Tegel (mean values from March to August 2004)

The great significance of the interstitial flora was confirmed by the parallel development of a comparably large number of interstitial fauna: a total of 77 taxa were identified, whereby nematodes and annelids (worms) plus crustaceans and copepods (water flies) were the dominant species. With an average 7,000 individuals per dm^{-2} in the upper sediment layer, i.e., at depths of 0 to 10 cm (minimum individuals $/dm^{-2}$: 430; maximum: 9,700; Beulker and Gunkel 1996), this abundance of the interstitial fauna is very high. Comparable abundance and diversity levels can also be found in marine sandy beaches (Higgins and Thiel 1988)

The composition of the sediment in the upper interstitial zone is shown as weight percentages in Fig. 4a and as volume amounts in Fig. 4b, which point out the significance of the POM within the interstices. The average POM volume in the upper sediment zone (0–5 cm) was found to be 19.4%, while the sediment pore volume was calculated to be about 40%. Thus, the sediment pore system is half filled by organic material; 20% of the POM is dry substance and 80% is cell water or bound water.
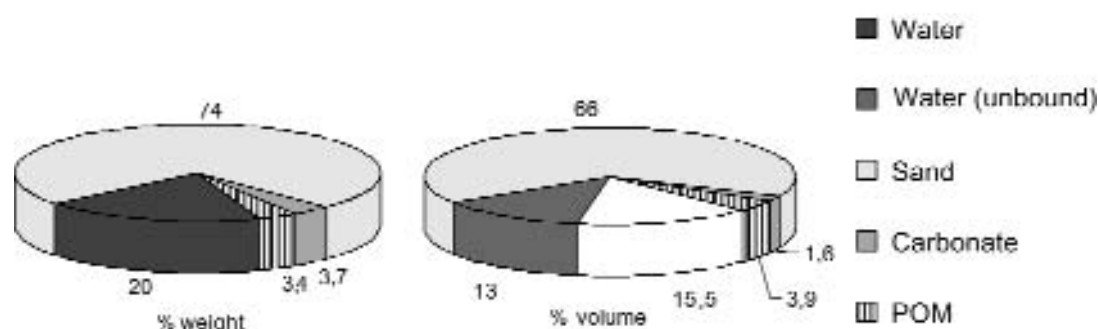
Figure 4. Composition of the upper littoral sand zone (0–5 cm) of Lake Tegel relative to
a) weight (water and dry weight of sand, POM and carbonate) and b) volume.
POM was calculated based on the organic carbon (C org) content under consideration of a biomass of 20%
and a water content of 80% of the algae; leaves are wood debris were excluded

The composition of the POM volume can be specified by correlating chlorophyll a (Chl a) to organic carbon (C org): C org = Chl a x (30 to 100); (Margaleff 1983). Accordingly, the C org concentration of the algae was calculated as 0.8 to 2.5 g/l, while the observed Corg (algae and POM) was 10 g L^{-1}. Living algae therefore comprise about 8 to 25% of the POM volume. This is supported by our findings obtained by correlating Chl a to algae dry substance based on Chl a = 0.5–3% of the dry substance (Margaleff 1983), the analogous algae biomass is 3 to 25 g L^{-1}, corresponding to 7 to 50% of the POM. Considering the wide scattering of the relation between Chl a to carbon content and to biomass, it is notable that about 25% of the POM consisted of living algae.

CONCLUSIONS

Our structural analysis and study of carbon turnover processes at Lake Tegel suggest that interstitial algae play an important role in interstitial clogging. Based on our findings, we must assume that the permeability of the sediment is determined by biological processes. Algae, consumer organisms (shredders, filterers and predators) and bacteria are part of the interstitial food web. Bacteria are maintained in the system by DOC uptake (microbial loop) and POM mineralization. This biocoenosis responsible for clogging of sediment interstices makes up about 50% of the pore system. The biomass of the interstitial algae was used to estimate carbon activity levels (Table 2), assuming a daily reproduction rate of 20%, which is low for limnic systems. These data indicate that the DOC of pore water is mainly determined by algae production, and DOC input from the lake is low. The interstitial biocoenosis is responsible for purification processes of infiltrated water and we must assume that the efficiency is regulated by the stability of the biocoenosis.

Table 2. Parameters of C turnover in littoral sediment interstices of Lake Tegel (means for 2004),
(* = calculated data)

Parameter	Turnover in the interstices
C org total (POC)	5,000 mg C dm^{-2}
C org interstitial flora (algae, 0 – 5 cm depth)	350 – 1,750 mg C dm^{-2}
Primary production of algae (~ 20% d^{-1}) *	70 – 350 mg C dm^{-2} d^{-1}
Infiltration rate to groundwater	0.5 – 6 L dm^{-2}
DOC input by lake water infiltration	3.5 – 42 mg C dm^{-2} d^{-1}
DOC input by interstices algae *	~ 14 – 70 mg C dm^{-2} d^{-1}
DOC output to groundwater	2.5 – 30 mg C dm^{-2} d^{-1}
Δ DOC (input – output)	1 – 12 mg C dm^{-2} d^{-1}
DOC mineralization *	~ 15 – 82 mg C dm^{-2} d^{-1}

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Physicochemical changes in pore water in the sandy littoral zone of Lake Tegel during bank filtration

Anja Hoffmann and Günter Gunkel

Abstract

At Lake Tegel in Berlin, Germany, drinking water is produced by induced bank filtration. Under such increased infiltration conditions, it is very important to understand the natural purification processes in the upper littoral zone (sediment depth of about 0 – 30 cm) in order to maintain a high drinking water quality.

We therefore analyzed the pore water and measured the redox potential at Lake Tegel regularly to detect fluctuations in the concentrations of physicochemical parameters due to seasonal variations in the redox potential.

The redox potential is not only influenced by the biological activity of the interstitial biocoenosis, but also by hydraulic conditions that occasionally produce unsaturated zones leading to an intrusion of gaseous oxygen. The result is an increase in the redox potential, which declines during the summer due to intensive dissimilatory reduction and microbial activity, thus leading to distinctly anaerobic conditions. When this is the case, the oxygen supplied by primary production and bioturbation cannot meet the oxygen demand needed for the mineralization of organic material. Negative redox values (minimal −200 mV) are accompanied by increases in nitrite (max. conc. 150 µg/L) and ammonium levels (max. conc. 0.45 mg/L), while the nitrate concentrations decreased (min. conc. 0.2 mg/L). This indicates that processes such as denitrification and ammonification occur, and that, after depletion of free oxygen reserves, other electron acceptors, such as nitrate and also heavy metal ions (Fe^{3+}), are used.

Keywords

Bank filtration, hyporheic zone, interstices, pore water, redox potential.

INTRODUCTION

In Berlin and other regions, natural bank filtration processes have been successfully exploited for drinking water production for years. In fact, around 70 % of the abstracted groundwater in Berlin originates from induced bank filtration and artificial groundwater recharge (Pekdeger and von Sommer-Jarmerstedt, 1998).

As the water passes through the sediment, distinct self-purification processes such as filtration, adsorption, precipitation, biological oxidation, denitrification and carbon metabolism take place. These mechanisms are well understood due to numerous studies dealing with the functioning of sewage plants and infiltration ponds. However, although increasingly important, the regulatory mechanisms and seasonal changes affecting the self-purification potential as well as the impact of these changes on pore water quality in natural environments influenced by induced bank filtration have not been sufficiently investigated.

A main objective of the current research project was to provide a detailed analysis of the self-purification processes in the upper 30 cm of the hyporheic interstice, i.e. in the sandy littoral zone, of Lake Tegel in northwest Berlin. The self-purification process is an inherent part of the spatial structure and biological activity of the interstice. The biological function is based on the supply of oxygen from primary production, carbon transfer and carbon turnover resulting from meiofaunal activity and the microbial loop, as well as on the regulation of the clogging processes (see also Gunkel and Hoffmann, 2005), and aeration of the sediments by meiofaunal bioturbation, as is well described

elsewhere (Solan and Kennedy, 2002). Degradation of organic material, on the other hand, leads to a depletion of oxygen and the resultant use of other electron acceptors such as nitrate, manganese, iron oxides, hydroxides and sulfate, typically proceeding from the highest energy yield downwards.

Especially during the summer, increased dissimilatory reduction and microbial activity result in anaerobic conditions in an relative extensive layer of sediment. Biological activity in the biocoenosis therefore has a strong impact on the redox situation on which the hydrochemical situation ultimately depends. Because of the induced bank filtration at Lake Tegel, we have to assume that redox conditions in the interstices of the littoral sediments are also influenced to a great extent by temporal and spatial variations in the hydraulic state, which may be either saturated or unsaturated. Similar conditions are assumed for the artificial recharge pond near Lake Tegel (Massmann et al., submitted).

METHODS

Field monitoring was carried out at Lake Tegel. The study site was situated within the catchment area of the Tegel water purification plant (*Berliner Wasserbetriebe*) and the Tegel-West well gallery, which consists of 130 vertical filtrating wells near the islet of Reiswerder. The current trophic state of Lake Tegel is moderately eutrophic (LAWA Directive 1998).

Pore water was recovered using rhizon soil moisture samplers (UMS GmbH Munich). The 5 cm porous polymer tubes (diameter 2.5 x 1.5 mm) were connected via a Luer lock system to nylon tubes and syringes to produce a vacuum. The moisture samplers were permanently inserted in the interstices at 5 cm increments from depths of 1 to 26 cm using a rack-like acrylic glass tube with holes drilled at the appropriate depths. The rack tube was placed in the sediment, and the moisture samplers were inserted horizontally from the inside of the tube into the surrounding and undisturbed sediment. Pore water and lake water samples were extracted every three weeks, then filtered through filters with a defined pore diameter of 0.45 µm and analyzed immediately.

Soluble ions (PO_4^{3-}, NO_3^-, NH_4^+) were assayed by flow injection analysis (FIAstar 5000, Foss Tecator), while metal ion concentrations were determined by atomic absorption spectroscopy (Perkin Elmer 5000) after filtration and acidification (pH ≤ 2) with concentrated nitric acid. Dissolved organic carbon (DOC) was measured using a TOC analyzer (Liqui-TOC). Nitrite analyses were carried out immediately (at lakeside during the summer) using a rapid test kit (Aquaquant, Merck).

Parallel to pore water sampling, a sediment core (length ca. 30 cm) was removed and the redox potential (U_H) was measured immediately by inserting an electrode (WTW, Pt in Ag/AgCl) through a hole in the acrylic glass tube at specified sediment depths (every 2 cm) by successively pushing the sediment upwards.

RESULTS AND DISCUSSION

The highest U_H values (Fig. 1) were observed in the winter in all sediment layers studied. At this time algae biomass values in the interstice were very high (Gunkel and Hoffmann, 2005), while microbial activity levels probably were relatively low. These conditions, in conjunction with a high primary production potential and cold water temperatures, promote a high redox potential. From February to April 2004, peak values of > 400 mV were measured from sediment depths of 8 cm downwards, indicating oxic conditions. In the same period, the upper horizon also showed a high redox potential but was rather anoxic; this means that the free oxygen produced by algae primary production was also rapidly depleted by respiration and chemical oxidation. Besides primary production, this zone also demonstrated the most intensive heterotrophic metabolism (highest meiofaunal abundance). Decomposition

of leaf material is another factor, that leads to increased oxygen consumption. From April 2004 on, the redox potential decreased rapidly from the uppermost sediment layer down.

This resulted in anaerobic conditions over the entire sediment horizon studied in June, the time when infiltration and pumping rates were highest. In that month, the high temperatures, which also reduce oxygen dissolution in lake water, led to increased heterotrophic activity and mineralization. In addition, primary production in the sediments was probably reduced due to a worsening of the light climate, while enhanced primary production in the water column led to increasing amounts of dead algae entering the sediment.

Here, the bulk of organic material (dead algae) also caused high mineralization rates. Moreover, we have to assume that aerobic microniches with free oxygen exist, but were not detected by the redox electrode. The fauna activity also has great effects on the total O_2 consumption rates and thus on mineralization. Wenzhöfer and Glud (2004) found that light-dependent changes in fauna activity result in strongly elevated O_2 uptake rates (5.3 mmol/m² h) at the onset of darkness. On the other hand, they found that the volume of oxic sediment around burrow structures grossly exceeded that of the primary sediment surface. During the summer (Fig. 1), the U_H values increased continuously from depths of 20 cm upwards, indicating the intrusion of gaseous oxygen due to a change in hydraulic state, while values in the top 10 cm remained minimal. The reason for the intrusion of gaseous oxygen into deeper sediment layers and the resultant rise in the redox potential was probably a drop in the groundwater level caused by large groundwater delivery rates, whereas clogging limits the amount of infiltrating water. This is similar to the observations of Massmann et al. (submitted), who investigated the artificial groundwater pond near Lake Tegel. From October on, the U_H values indicated anoxic to oxic conditions except in small microzones that remained anaerobic.
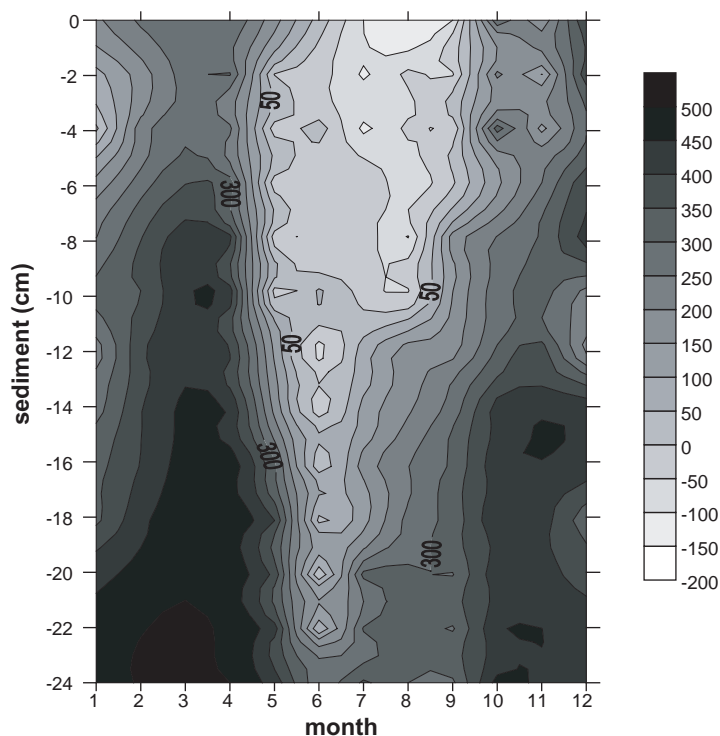
Figure 1. Changes in U_H values (mV) in the upper 24 cm of the sandy littoral sediment during 2004

As shown in Figures 2–4, the concentrations of distinct soluble nitrogen compounds in pore water were similar to those in infiltrating water most of the year. Under the anaerobic conditions occurring in the summer, the levels of nitrate in the interstices decreased. First, nitrite concentrations increased to a maximum (150 µg/L), which is indicative of denitrification, while nitrate concentrations in both pore water and infiltrating water remained moderate (roughly 1 mg/L). Compared to infiltrating water, nitrate concentrations in pore water declined sharply, from 0.9 mg/L to 0.2 mg/L, at depths of 6 to 26 cm (maximum difference: 0.7 mg/L) in August and September 2004. At the same time, ammonium concentrations in pore water increased to maximal 0.4 mg/L, indicating ammonification. The infiltrating nitrate concentrations presumably were not sufficient to compensate the low redox potential. Apart from the low redox potential, the high temperatures also promoted denitrification and ammonification. As these are fast processes, the intermediate product, nitrite, was not always detected in high concentrations.

After oxygen depletion, other electron acceptors are utilized sequentially, in the order of the highest energy yield downwards. Microbially catalyzed reduction of nitrate starts below E_H values of +421 mV, followed by the
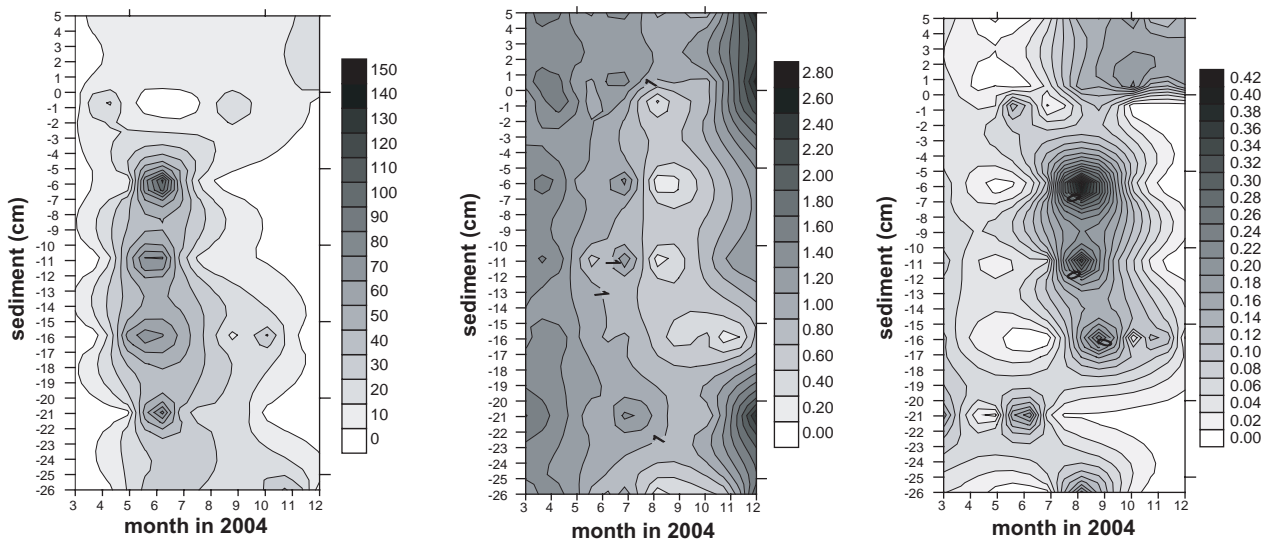
*Figures 2–4. Concentrations of nitrogen compounds in infiltrating and pore water
in the upper 26 cm of the sandy littoral zone from March to December 2004.
Figure 2. NO_2–N (μg/L). Figure 3. NO_3–N (mg/L). Figure 4. NH^4–N (mg/L)*

reduction of manganese below E_H values of +396 mV (Schachtschabel et al., 1982). If the redox potential continues to decrease, reduced iron appears and, finally, desulfurification starts at E_H values of –100 mV and lower, depending on which complex reactions of organic compounds occur (Matthess, 1990).

An increase in the rather low soluble iron concentrations in the interstices of Lake Tegel was only observed at 6 cm and deeper between June and August (Fig. 5). The changes in sulfate concentrations in pore water were not indicative of desulfurification, but scanning electron microscopy revealed pyrites at different sediment depths. The sulfate concentrations ranged between 114 mg/L and 136 mg/L in both pore water and free water. From March to October, soluble reactive phosphorous (SRP) concentrations in the interstices were relatively low, mostly in the range of 10 to 40 μg/L (Fig. 6) and, except in June, were also very similar to those in infiltrating water. The highest SRP concentrations were observed from October to December in the overlying water column and in the relatively deep sediment below 20 cm. The high SRP concentrations in the free water can be attributed to desorption, as was also suggested by the very low concentrations in the uppermost sediment layer. Desorption of SRP is mainly controlled by temperature and mineralization as well as by the subsequent reduction of iron, the most important ligand. Since desorption processes were normally observed at the hypolimnic sediment water interface under anaerobic conditions, distribution of SRP from deeper water layers through mixing is also a possibility. Total phosphorus concentrations in sediment were rather moderate (ca. 70 mg/g dw).

Figure 7 shows the changes in pH over time. Two distinct zones of high pH values, corresponding to high primary production, were apparent: one in the free water during the vegetation period, and a second in the upper 6 cm sediment layer in October and during the winter. At all sampling dates, the pH values decreased with sediment depth; this suggests that metabolism is exclusively heterotrophic at depths below 10 cm.

The electric conductivity ranged between 705 μS/cm and 820 μS/cm, but no depth gradients were detected.

No significant changes in pore water concentrations of dissolved organic carbon (DOC) were observed, but differences (mostly < 1 mg/L) between DOC levels in the infiltrating water (range: 6.3–7.7 mg/L) and the upper interstice were detected. In the whole sediment horizon, DOC concentrations ranged between 5.9 and 7.0 mg/L. Our previous results about the biocoenosis (Gunkel and Hoffmann 2005) led us to the assumption that very short cycles of carbon release and uptake, i.e. microbial loops, were responsible for this observation.
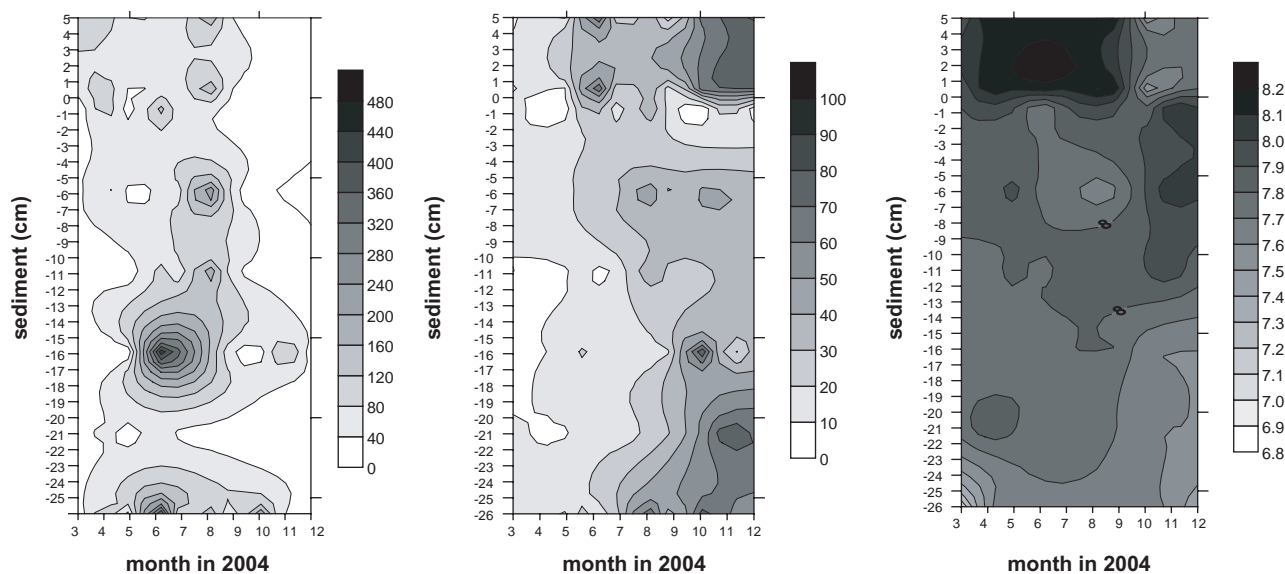
*Figures 5–7. Concentrations of soluble Fe (µg/L); SRP (µg/L) and pH in infiltrating water and pore water
in the upper 26 cm of the sandy littoral zone from March to December 2004*

CONCLUSIONS

During the summer, enhanced respiration and mineralization rates combined with presumably lower primary production rates led to a huge oxygen demand that was not compensated by functional biological activity. Therefore, biological oxidation of water compounds as a part of the self-purification process does not seem to be guaranteed at that time. The low redox potential regulates chemical reactions that determine the pore water concentrations of soluble ions. The resultant use of other electron acceptors in lieu of free oxygen led to denitrification and ammonification, resulting in a beneficial reduction of nitrate concentrations. A hydraulic shift, that is, a change from saturated to unsaturated conditions, led to reoxidation of the sediments by gaseous oxygen. Later, the redox situation seemed to stabilize due to a more balanced relationship between primary production (high algae biomass and depth expansion, good light climate) and heterotrophic metabolism. Carbon metabolism, which is related to algae and meiofauna abundance, was highest in the upper 5 cm of sediment.

These findings lead to the assumption, that the efficiency of bank filtration exhibit a high seasonal variability. During the summer the quality of bank filtrate will be diminished due to the extensive anaerobic conditions in the hyporheic zone, that also affect the stability of mineral surface coatings, such as ferric and manganese oxyhydroxydes. Consequently the sorption or coprecipitation of trace metals and others as well as the filtration of microbial pathogens is no more ensured.

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ACKNOWLEDGEMENTS

Experiments to determine clogging and redevelopment effects of ASR-wells at laboratory scale

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Abstract

Aquifer Storage and Recovery (ASR) can be a useful tool for water management in countries with distinct rainy and dry seasons. ASR can be used to store water in the rainy season and to use it in the dry season, but also to improve the groundwater quality of saline aquifers. Surface water, which contains soil particles, nutrients, micro organisms and other pollutants is infiltrated through ASR-wells direct into the aquifer. Besides quality problems of the stored water the infiltration of polluted water leads to clogging of the well screen and the aquifer next to the ASR-well. Laboratory experiments were carried out at soil columns using natural storm water run-off to find out the effects of single clogging processes and the effect of back-washing. By back-washing suspended particles can be withdrawn from the aquifer matrix. For that reason run-off water was infiltrated into the soil column in several cycles of infiltration and back-pumping. The results of the experiments show that the biological clogging process has the largest influence on the reduction of the aquifer conductivity when using untreated surface water for infiltration. The deposition of particles had a relative small influence on the changes in conductivity and the deposition was removable by back-washing periods.

Keywords

Clogging, redevelopment, laboratory experiments, ASR-well.

INTRODUCTION

A major problem of direct infiltration of surface water into the groundwater aquifer is the clogging of the well screen and the aquifer nearby (Pyne, 1995). In general the surface water, which is available for infiltration, contains fine soil particles, inorganic and organic material and micro organism. The substances are introduced through the ASR-well into the aquifer and block the filter layer and the aquifer. The pore volume is reduced and with that the infiltration capacity of the ASR-well.

The physical processes of the blockage are accumulation of organic and inorganic suspended matters and adsorption of silt and clay particles. In addition to this, chemical reactions (e.g. oxidation of iron and manganese, precipitation of lime etc.) lead to clogging of well screens. The accumulation and growth of algae, bacteria and micro organisms in the filter material belong to the biological processes of clogging.

Bouwer (2002) suggested cleaning up the infiltration water to 'drinking water quality', so that blockage processes can be excluded. Dillon and Pavelic (1996) and Dillon (2002) found out during their measurements in Australia, that particle loads in the infiltrated water of not more than 150 mg/l did not give considerable blockages. In some countries (Netherlands, Great Britain, USA) it is not allowed to infiltrate water with opacity degrees of more than 2 to 5 NTU. Also the guidelines for artificial groundwater recharge EWRI/ASCE (2001) say that surface water is to disinfect and be cleaned as far as possible.

This show, that for the prevention of clogging and to secure the durability of the infiltration wells, the quality of the infiltrated water is evident. One other possibility to improve the life time of ASR wells is to reverse the flow for a

period in order to pump some particles out of the aquifer. The effectiveness of this method depends on the kind of clogging. Especially with biological clogging it may be difficult to redevelop the aquifer. Dillon et al. (1999) refer to the risks of the biological growth. But the examinations of the biological processes are quantitatively still insufficient. It's not clear, how great is the influence of the biological processes in comparison to the physical and chemical influences.

In this study laboratory experiments were carried out at 1-D soil columns using natural storm water run-off to find out the magnitude of influence of the individual processes on the changes of the permeability of a porous medium. The possibility of raising the conductivity was checked by back-washing with clean water. By back-washing suspended particles can be withdrawn from the aquifer matrix.

LABORATORY EXPERIMENTS

The general experimental set up is shown in fig. 1. Water with suspended solids is flowing through a 1-dimensional column with a length of 63 cm and a diameter of 11 cm. The column was filled with sand with a grain size of 1–2 mm (k_f = 2.2 10 3 m/s; n = 0.38). The column has 6 sectors, the 1st three sectors have a length of 8 cm each and the last three 13 cm length each. Samples can be extracted from the soil column at 12 points and 6 observation pipes show the hydraulic head. During the experiment the soil column was darken to reproduce the conditions of an aquifer.
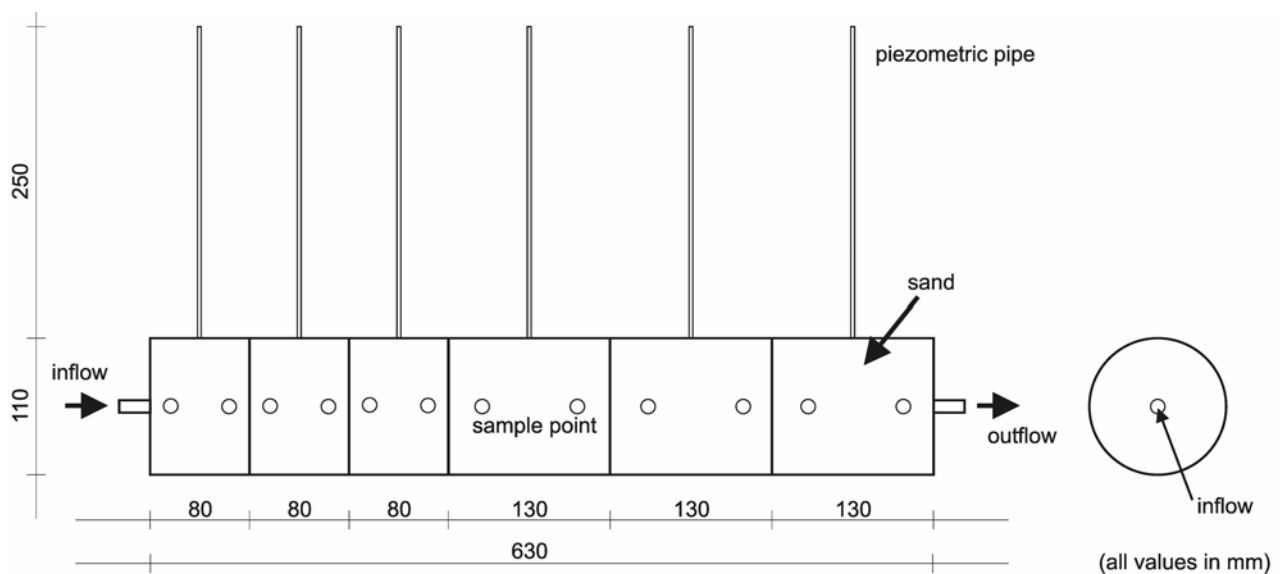
Figure 1. Experimental set-up

The general opinion is that the major causes for clogging are physical deposition, air entrapment and biological effects. To quantify these effects and to determine redevelopment effects, the following three laboratory experiments were carried out.

1. Experiment (1) with air-free tape-water with a suspended solid concentration of 1 g/l and a 12% NaOCl (Sodium Hypochlorite) concentration of 1 ml/l.
2. Experiment (2) with non-air free tape-water with a suspended solid concentration of 1 g/l and a 12% NaOCl concentration of 1 ml/l.
3. Experiment (3) with none-air free tape-water with a suspended solid concentration of 1 g/l without NaOCl.

The 1st experiment concerns only physical deposition. All other effects were excluded by choosing pure sand (no

geochemical reactions), air-free water (no air entrapment) and by adding NaOCl no biological growth. In the 2nd experiment air entrapment was allowed and in the 3rd experiment also biological growing of micro organisms.

During the experiments the hydraulic head at inflow side was constant. The initial inflow rate was fixed to 125 ml/min. The concentration of particles in the injected water was also kept constant. The grain size distribution of the infiltrated particles was measured with an automatic laser optical instrument (GALAI CIS-100). The largest particle was 63 μm.

All three laboratory experiments had the same course as follows:

- infiltration during 72 h from the inflow side of the column, followed by simulated pumping of 10 h. This means that clean water was injected at the outflow side for 10 h;
- infiltration was continued for 24 h from the inflow side of the column, followed by simulated pumping of 6 h;
- infiltration was continued for 18 h from the inflow side of the column.

The infiltration water was stored first for several hours in a vessel to reach a constant temperature of 18.5 to 21.5°C. Outflow rate, temperature, pH-value and oxygen content of the inflow and of the outflow water were measured every hour by taking water samples. Every two to four hours water samples were taken at the sampling points (Fig. 1) with a syringe. The particle content and distribution of the particle size in these samples were measured by drying at 105°C and by analysing the samples with an automatic laser optical instrument (GALAI CIS-100) wich is able to determine particles sizes between 0.1 and 600 μm. This detection was done every two to four hours. During the experiments the hydraulic head was taken from the piezometric pipes every hour.

During the experiments for investigating the biological clogging, filtrated COD, organic dry matter and filtratable matter were taken. The filtrated COD was determined by COD test LCK 414 (Dr. Lange) which allows a detection of the filtrated COD in the range from 5 to 60 mg oxygen. The effluent water was tested every 12 h and also all segments at the end of the experiment. The organic dry mater was taken together with the filtrated COD. The samples for the organic dry matter had to be filtered by a membrane (0.45 μm) so that the filtratable matter was also detected.

RESULTS AND DISCUSSION

Fig. 2 show the dependence of the relative conductivity k_r on the deposited particles in the soil pores represented by the flow-effective pore volume n_f for all three experiments.

The flow-effective pore volume nf (related to the total pore volume) is defined as:

$$n_f = 1 - \frac{V_P}{V_H}$$

where: V_P : deposited particle volume [cm^3], V_H : total pore volume [cm^3]

This means with an increasing deposition of particles the water content decreases. The deposited particle volume V_P was calculated from the difference between inflow and outflow particle mass divided by the bulk density of the particles (ρ_P = 1.46 g/cm^3).

$$V_P = \frac{m_P}{\rho_P} = \frac{Q_{in}\,(c_{in} - c_{out})}{\rho_P}$$

where: m_P : particle mass [g], c_{in} : inflow concentration of the particles [g/cm^3], c_{out} : outflow concentration of the particles [g/cm^3]

The relative conductivity k_r was determined from the actual inflow rate Q_{in} divided by the initial rate ($Q_{in,t=0}$) at the beginning of the experiment.

$$k_f = \frac{Q_{in}}{Q_{in,t=0}}$$

The largest reduction of the relative conductivity was found in experiment (3), (15 % of the initial value), while the conductivity rate in the 1st experiment decreases lowest (58 % of the initial value). During the first infiltration (72 hours) in the 1st experiment the conductivity decreases linear. The second and 3rd experiment shows a faster reduction during the first time and the conductivity decreases slower than in the 1st experiment (Fig. 2).
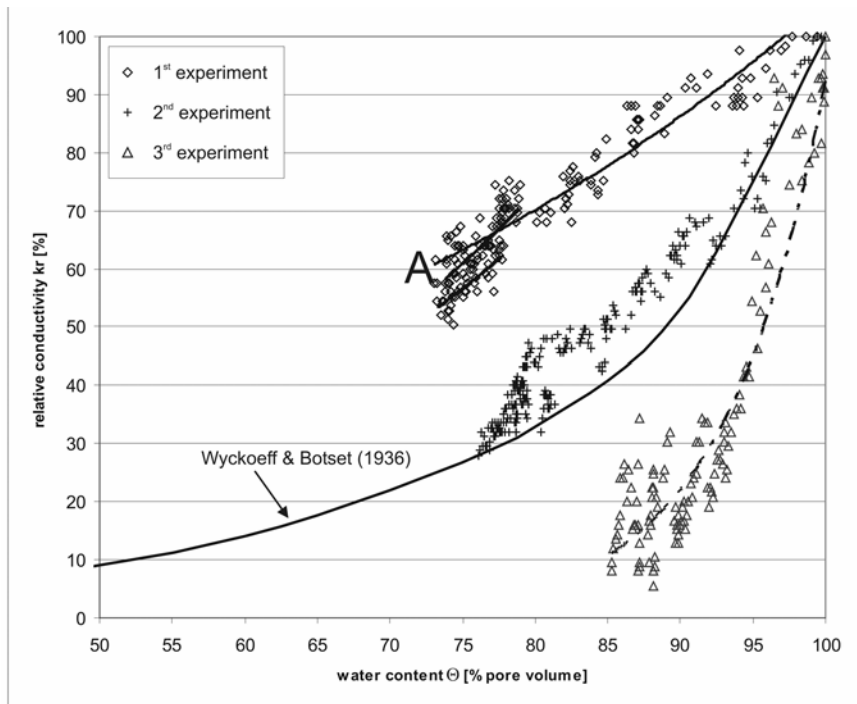
Figure 2. Relative conductivity in relation to the flow-effective pore volume

Between the infiltrations, there is a reverse flow of fresh water for flushing out the suspended particles with a hydraulic gradient, which is about double of the hydraulic gradient while infiltrating. Fig. 3 shows that the flux for the 1st experiment rises within 15 to 30 minutes in a speedy manner. After this flush the flux is more or less constant with a slight tendency of declining, which is caused by consolidation of the sediments inside the aquifer material. Similar the particle concentration, which was found at the outflow during the first 15 to 30 minutes was very high and reduced too less than 1 g/l after that time. The second
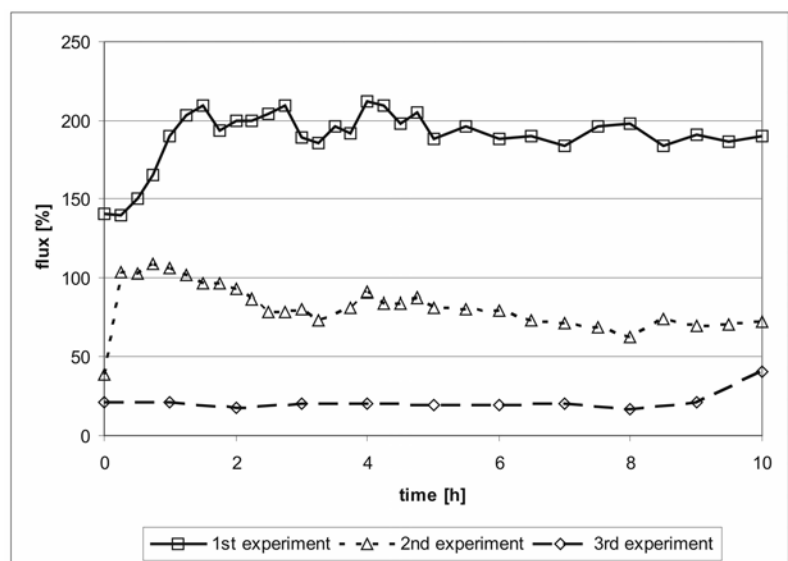
Figure 3. Relative flux during 1st back flushing lume

reverse flow period was giving similar results but the particle mass which was carried out of the soil column was smaller.

The characteristic of the second and third infiltration is similar to the first infiltration. The flux decreases with time. Also the pumping interval between the second and third infiltration is similar to the first pumping interval.

In the 2nd experiment (water with air and suspended solids) the same characteristic were measured than in experiment 1. The difference between the two experiments is a faster reduction of the flux. The tendency of declining is non-linear and trying to reach a final value (Fig. 2) (40% of initial infiltration rate).

The reduction of flux is faster because bubbles were leaving the water, while the air was flowing through the soil column. The main effect is that the water is getting warmer, so that the saturation level of the air, which is dissolved in the water, is reducing.

The positive effect of discharge after the infiltration was also measured in the 2nd experiment, but it is lower than in the 1st experiment. The mass of particles, which is moving out is smaller, but the time interval (first 15 to 30 minutes) is the same. The longer the discharge is going on, the lower is the flux. At some points this flux is rising than again and falling afterwards. This happens due to the clean water (with air) which was infiltrating the (recharge) outflow side. Bubbles were built and sometimes bubbles were moving out of the system through the piezometers.

The 3rd experiment (water with air, suspended solids and allowed biological growth) also shows similar results. The flux was reduced within short time to a low level (Fig. 2). This was also not changed by the discharge. In the first 15 to 30 minutes no particles flow out of the system. After several hours a lot of particles flow out at once and after that nothing anymore (Fig. 3). The time when they were flowing out was not the same in the first and in the second discharge. The flux was rising at that event and was constant afterwards.

A dissolved oxygen balance shows that inside the column oxygen is used. The difference between input of dissolved oxygen and the output rises linearly with time so that it can be seen that inside the system biological growth occurs. Since dissolved oxygen can be found at the output of the column, aerobic microbes are also growing. The biofilm has not been investigated directly.

The result of the growth is that the very small particles which can be found in the outflow in the 1st and 2nd experiment are not there in the 3rd one. This means the particle are also deposited in the column and the transportation way for the particle is shorter.

Since the flux in the 2nd and in the 3rd experiment was lower than in the 1st experiment, the mass of particles are smaller. Fig. 4 shows the mass in the soil column versus time. The drawdown during the curves shows the reverse flow periods. It is visible that in the first reverse period the reduction of the particle mass is larger than in the second phase. The second point which is visible is that physical clogging can be reduced by reverse flow but if biological clogging also occurs, reverse flow with a doubled gradient than during infiltration has nearly no effect.

The laboratory experiments showed that biological clogging is the most significant process for the reduction of the aquifer conductivity. The biological clogging was found on the first 20 cm of the soil column and back-washing was hardly unable to withdraw particles or other materials. The entrapment of gas bubbles has also a great influence. But it is reversible and back-flushing gives an improvement on raising the conductivity. Clogging by infiltrated particles has the lowest effect. The deposition of particles takes place mainly in the dead-end-pores and does not affect the active pore volume very much. Particles which were deposited in the active pore volume were washed off the soil column by back-washing or were redeposited into the dead-end-pores.

Figure 4. Particle mass in the soil column

CONCLUSIONS

The laboratory experiments showed that biological clogging is the most significant process for the reduction of the aquifer conductivity. The biological clogging was found on the first 20 cm of the soil column and back-washing was hardly unable to withdraw particles or other materials. The entrapment of gas bubbles has also a great influence. But it is reversible and back-flushing gives an improvement on raising the conductivity. Clogging by infiltrated particles has the lowest effect. The deposition of particles takes place mainly in the dead-end-pores and does not affect the active pore volume very much. Particles which were deposited in the active pore volume were washed off the soil column by back-washing or were redeposited into the dead-end-pores.

Laboratory experiments showed that biological processes have a very large impact on clogging when using storm water run-off for direct infiltration by ASR-wells. Biological processes complicate the improvement of aquifer conductivity by back-washing. Physical deposition has a minor effect and particles can be withdrawn from the aquifer. A probable reason is that physical deposition of particles takes place more in the dead-end-pores than in the active pores.

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Changes of water quality and bacterial community structure during artificial groundwater recharge with humic lake water

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Abstract

This research aims at evaluating changes of water quality, i.e. content of natural organic matter, and microbial community structure during full-scale artificial recharge of groundwater with humic lake water for drinking water supply. Three recharge sites studied were located in public water works, referred here as sites A, B and C, in Finland. On site A groundwater recharge is performed by basin infiltration (since 1976) and sprinkling infiltration (since 1995), on site B by sprinkling infiltration (since the end of 2000) and on site C by basin infiltration (since 1979). Samples were taken in January 2005, when the raw water lakes were ice covered. Reductions of total organic carbon at sites A, B and C were 91, 81 and 70%, respectively. Larger molecular fractions were removed more efficiently than the smaller ones. Total cell counts in raw waters indicated 1.6 million cells/ml at sites A and B and about 0.7 million cells/ml at site C. Recharge process at sites A, B and C resulted in removal percentages in bacterial numbers by 94, 94 and 75, respectively. Denaturing gradient gel electrophoresis (DGGE) of the amplified 16S rRNA gene illustrated diverse microbial community structures throughout the process co-occurring with the decrease in dissolved organic carbon content and total amount of cells in water.

Keywords

Artificial groundwater recharge; bacterial community analysis; basin infiltration; DGGE; humic water; sprinkling infiltration.

INTRODUCTION

In Finland, 65% of the population uses groundwater as drinking water source (Korkka-Niemi, 2001). However, due to the unequal distribution and size of groundwater sources, the number of water works, which apply artificial groundwater recharge, has been increasing in recent years. In Finland, groundwater areas are often located in glacial formations, i.e. eskers, which are favourable sites for artificial groundwater recharge by basin- and sprinkling infiltration. Sprinkling infiltration comprises a pipe network through which lake or river water is distributed on top of natural forest soil. The location of the pipe network is regularly changed to minimise potential adverse environmental effects that can result from the process (Helmisaari et al., 1999). Sprinkling infiltration is a relatively new method and currently favoured technology of aquifer recharge at new Finnish sites.

Crucial issue of groundwater recharge is the introduction of potentially reactive organic matter into an aquifer. The natural organic matter (NOM) is partly retained in the aquifer by physical and chemical processes and partially degraded by microorganisms. Changes in the amount and nature of organic matter in the aquifer may result in significant alterations in removal processes. (Aiken, 2002) Microbial populations adapt to new conditions either physiologically or by changes in the community composition (Miettinen et al., 1996; Eiler et al., 2003). In general, NOM is not easily biodegradable because of the many types of chemical bonds within the organic macromolecular structure, which require diverse groups of microorganisms for biodegradation.

From microbial point of view, subsurface habitats can be described as dark, relatively constant in temperature, relatively large in mineral surface area and generally oligotrophic (Ghiorse, 1997). The major parameters, which govern bacterial abundances in subsurface are the following: dissolved organic carbon (DOC), nitrogen, phosphorus, sulphur, moisture, pH, electron acceptors, grazing by predators and immigration of microorganisms from other habitats. Topsoil is generally a favourable environment for the enrichment of microorganisms but the decrease in the main microbial carbon source DOC with increasing depth leads to the decline in numbers of microorganisms. (Madsen and Ghiorse, 1993; Steffen et al., 2002).

Generalized features of the indigenous subsurface microbiota can not be given because of the many microhabitats that favour different populations. Bacteria with simple life cycles are by far the most widely distributed organisms in groundwater habitats (Madsen and Ghiorse, 1993). Autochthonous bacteria refer to bacteria likely to utilize refractory humic substances. Low but constant activity is typical for these bacteria. In subsurface most of the bacteria live attached on soil particles (Seppänen, 1997) as biofilms, thereby receiving nutrients from over flowing water. The communities of attached and unattached bacteria appear to be different, though they probably represent overlapping components of a dynamic community. (Madsen and Ghiorse, 1993).

Little is still known about the microorganisms that decompose and recycle humic substances in aquifers. This study aims at characterising bacterial community diversity and dynamics and removal of NOM during the recharge process, the key aspects affecting artificial groundwater recharge.

METHODS

Characteristics of the study sites and groundwater sampling

The studied recharge sites were unconfined aquifers located in public water works, referred here as sites A, B and C, in Finland. Selected parameters are listed on Table 1. Samples were collected in January 2005 when raw water lakes were ice covered. Three parallel set of samples were taken from each site: raw water (for recharge), monitoring well water (located at about the midway of recharge) and extracted groundwater. Prior to sampling, stagnant water was removed from the monitoring wells by pumping. Samples were taken from 1 m below groundwater table. Samples for microbial analyses were stored at 5 °C in the dark until analyses and samples for chemical analyses were acidified on site with HCl to a pH below 2.

Table 1. Characteristics of groundwater recharge sites.

	Method	*Start of recharge*	*Recharge [m³/d]*	*Flow distance [m]*	*Retention time [d]*
A	Basin	1976	20,000	1,000−1,300	90
	Sprinkling	1995			
B	Sprinkling	2000	15,000	200−500	15−30
C	Basin	1979	10,000	500−700	30−60

Analysis

Absorbance, DOC/TOC, HP-SEC and total cell count. Samples were acidified and filtrated through 0.45 µm cartridge filters (Polyethersulfone Membrane, Whatman) for absorbance and DOC determination, and analysis by high performance liquid chromatography (HPLC) (Hewlett Packard 1100 series) with a size exclusion chromatography column (SEC) (TSK-GEL G3000W). Filters were rinsed for 10 times to remove any removable carbon prior to

usage. Absorbance and DOC/TOC measurements were performed at Tampere Water Works Laboratory (Shimadzu pharmaspec UV-1700 254 nm, Shimadzu TOC-5000A, respectively). Characterisation of the molecular weight distribution of NOM was performed by HP-SEC with detection at 254 nm. Chromatograms obtained from HP-SEC analysis were drawn as peak height versus retention time. Retention times were modified to the closest whole and half numbers. Total cell count in water samples was performed within 24 h of sample collection by filtrating 2–6 ml of water through 0.2 μm membrane filters (Millipore) and staining the cells with DAPI (4',6-diamidino-2-phenylindole) nucleic acid staining solution (Molecular Probes). Cells were counted 20 times from duplicate samples from the filter surfaces using epifluorescense microscope (ZEISS Axioscop 2).

DNA extraction, PCR and DGGE. Sample filtration for DNA extraction was performed within 24 h of sample collection at 300 ml to 2,000 ml of water. Filters were stored at -20 ᵒC until DNA extraction by physical and chemical lyses of cells. Parallel water samples were filtrated using two different filters: 0.2 μm pore size hydrophilic polycarbonate filter (Cyclopore Track Etched Membrane, Whatman) and sterile 0.2 μm pore size glass fibre filter membrane (Schleicher and Schuel). Filters were inserted to tubes and the following extraction was performed according to the kit protocol (Ultraclean Soil DNA kit, Mobio Laboratories Inc.). The crude DNA sample was used as a template for polymerase chain reaction (PCR). A pair of Eubacterial primers GC-BacV3f (5'-CGC CCG CCG CGC GCG GCG GGC GGG GCG GGG GCA CGG GGG GCC TAC GGG AGG CAG CAG-3', including GC clump for DGGE) and reverse 907r (5'-CCG TCA ATT CMT TTG AGT TT-3') were used to amplify a fragment of 16S rRNA gene having a length of about 570 bp. PCR master mix contained 5 μl of 10 x buffer (1 x is 10 mM Tris-HCl, pH 8.8 at 25ᵒC, 1.5 mM $MgCl_2$, 50 mM KCl and 0.1% Triton X-100), 0.5 μM of each primer, 100 μM of each deoxynucleotide triphosphate, 1.5 U of DyNAzyme II DNA polymerase, 0.02 mg of bovine serum albumin, and sterile DNase RNase free water to give a finale volume of 49 μl. 48 μl of master mix was used with 2 μl of template in the PCR. The PCR was performed to amplify the DNA in a thermal cycler (Peltier Thermal Cycler PTC-200, MJ Research) with the following program: initial denaturation at 95 ᵒC for 5 min, 30 cycles of denaturation at 94 ᵒC for 30 s, annealing at 50 ᵒC for 1 min, and extension at 72 ᵒC for 2 min and a final extension at 72 ᵒC for 10 min. Amount of product was verified in 1% (w/v) agarose gel electrophoresis using Ethidium bromide stain.

Denaturing gradient gel electrophoresis (DGGE) was performed with a universal mutation detection system (BIO-RAD). Samples were loaded into 8% (w/v) polyacrylamide gel (bisacrylamide gel stock solution) in 1 x TAE (40 mM Tris, 20 mM acetic acid, 1 mM EDTA, pH 8.3) with denaturing gradients ranging from 30-65% (100% denaturant contains 7 M urea and 40% (v/v) Formamide). Electrophoresis was run at 60 ᵒC with 100 V for 16 hours. After the run, the gel was stained using SYBR Gold solution and photographed on a UV transilluminator with Kodak DC290 Digital camera.

RESULTS AND DISCUSSION

Organic matter removal

The changes of molecular weight distribution of NOM and efficiency of its removal during artificial groundwater recharge were studied. TOC and absorbance results were as shown in Figure 1 (DOC not shown). At site A, initial DOC/TOC content of raw water was the highest and at site C the lowest. Recharge resulted in the most efficient removal of organic carbon at site A (DOC/TOC 1.12/1.12 mg/l) and poorest at site C (1.59/1.75 mg/l) however, differences were small. The DOC content comprised on average 93% of the TOC in all samples. Absorbance measurement resulted in similar reductions as those of the DOC/TOC. In other words, results indicated efficient removal of total and dissolved organic carbon during recharge process. Helmisaari et al. (2003) found that retention time had a greater role in TOC removal than recharge distance but similar results were not obtained in this study. However, at site A where both the recharge distance and the retention time were the greatest, the most efficient purification occurred.
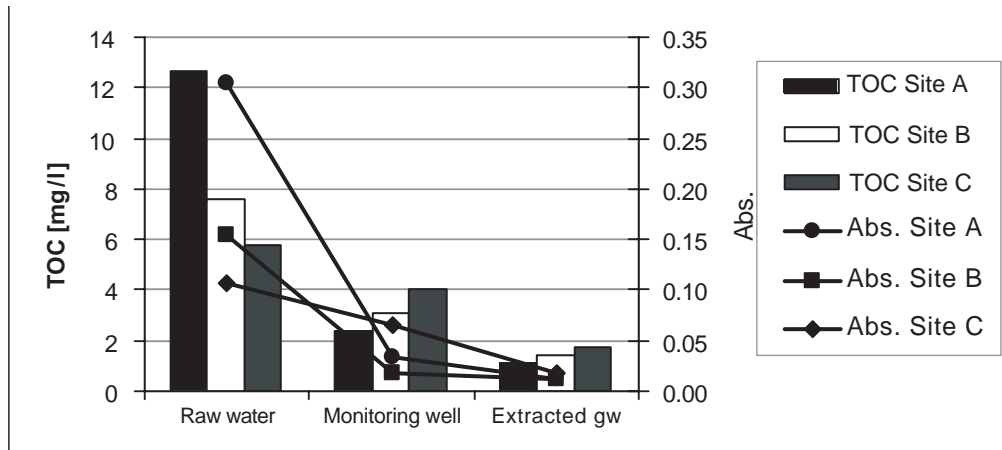
Figure 1. Changes within TOC and absorbance during groundwater recharge at three sites

The molecular weight distribution of organic matter changed significantly during the recharge at all sites (Figure 2 except for site C). Total height of the peaks as well as the peak area (not shown) became smaller indicating a decrease in total amount of organic matter in water. Largest molecules appeared predominantly within 7.5–8.5 min and smallest between 10 and 11 min of HP-SEC retention time at all sites.
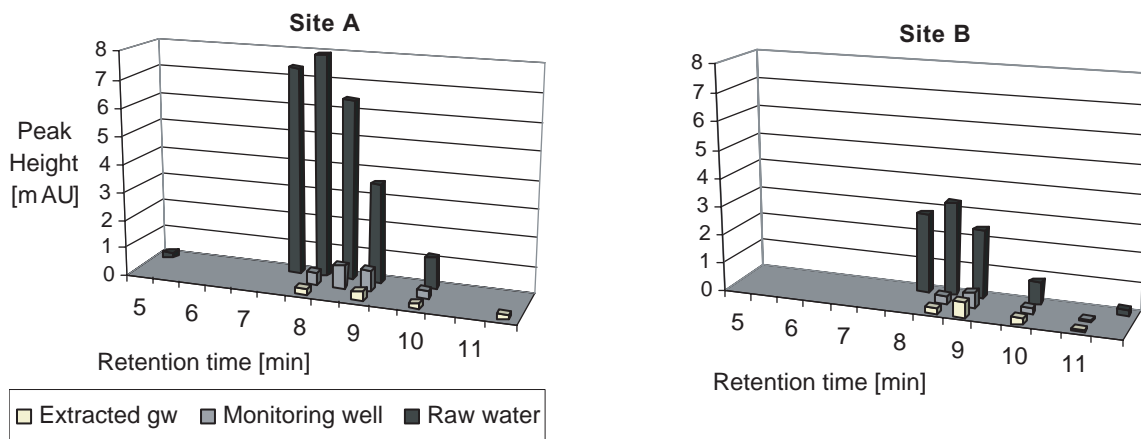
Figure 2. HP-SEC analysis illustrating molecular weight distribution of NOM during recharge at sites A and B

The sum of peak heights in raw waters were as follows: 26.6 at site A, 9.5 at site B and 7.0 at site C. At site A, the largest molecular fractions (at 5 and 7.5 min) were removed completely. Concurrently, smaller molecules were formed. Results were similar during the purification process at both sites B and C. Larger molecular fractions were removed more efficiently than the smaller ones, which was similar to the results obtained by Helmisaari et al. (2003) in a study made also in Finland.

Microbial communities

Changes of total cell counts and bacterial community diversity and dynamics in water during artificial groundwater recharge were studied. Total cell counts in raw waters were approximately 1.6 million/ml at sites A and B and about 0.7 million/ml at site C (Figure 3). However, recharge process resulted in smallest cell counts at sites A and B and greatest at site C. In extracted groundwater there were less than 200 000 cells/ml at all sites.

Diverse bacterial community structure occurred in all steps of recharge process and similar patterns were seen at all sites as seen in DGGE analysis (Figure 4). Slight changes within the community structure, however, did occur
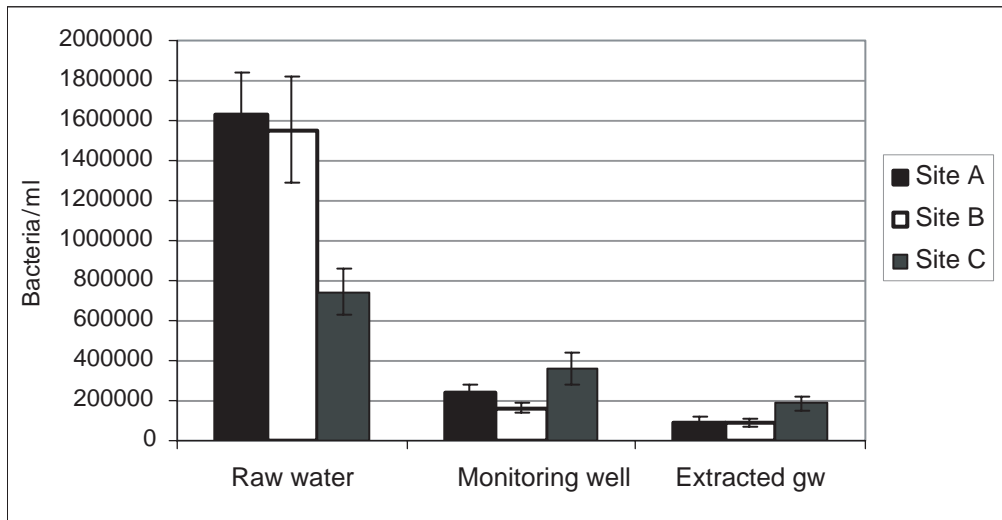
Figure 3. Total amount of cells in water samples. Error bars indicate standard deviation
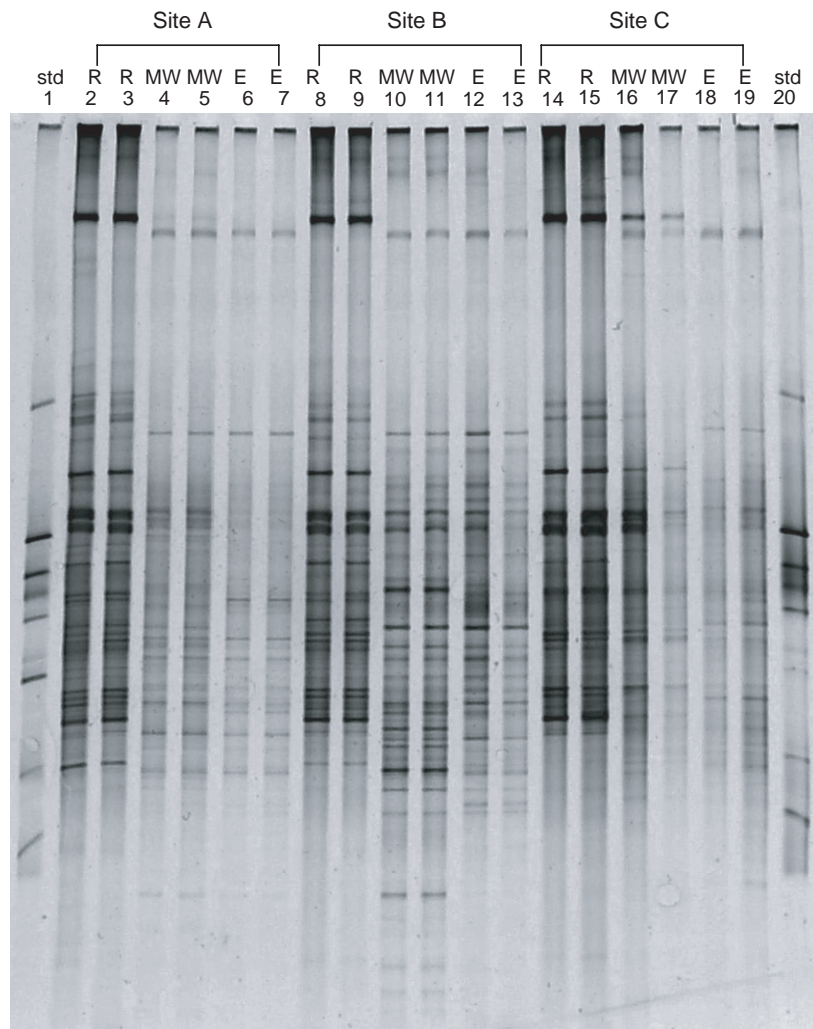
*Figure 4. DGGE band patterns of amplified DNA fragments coding for Eubacterial 16S rRNA gene
for sites A, B and C. For each site, three parallel set of samples are presented: raw water (R),
monitoring well water (MW) (midpoint of filtration process) and extracted groundwater (E).
Site A monitoring well water and extracted groundwater and site B monitoring well water
were parallel only from the point of DGGE due to the failure in DNA extraction of the true parallel samples.
Lanes std represent standard reference patterns.*

during the process: some bands disappeared and some new ones appeared. Also, band intensity decreased during the process. Band patterns of different raw waters were almost the same at all sites. Figure 5 shows a dendrogram as an example for site B. Similarity of band patterns was 70% between monitoring well and extracted groundwater samples and less than 50% between those two and raw water. In case of site A monitoring well water and extracted groundwater and site B monitoring well water, samples were parallel only from the point of DGGE due to the failure in DNA extraction of the true parallel samples.
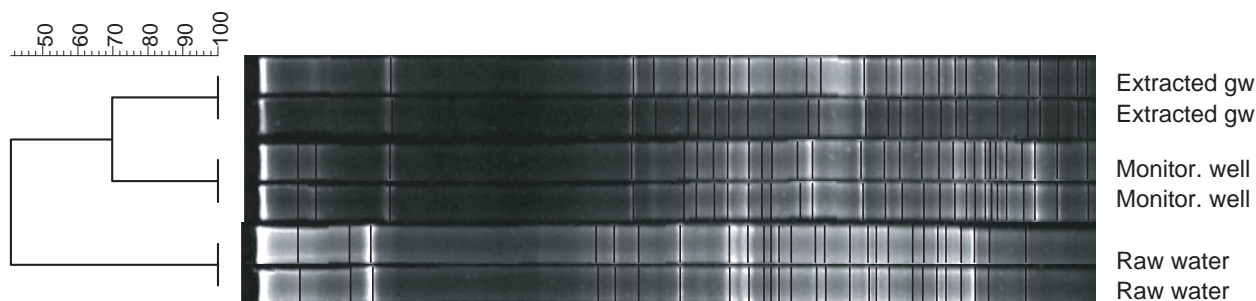
Figure 5. Dendrogram of Eubacterial band patterns at site B (GelCompar II software)

CONCLUSIONS

The results demonstrated high removal of NOM during the recharge process at all three Finnish sites. This was seen in DOC/TOC, absorbance and NOM molecular size distribution (HP-SEC) measurements, which agreed well with each others. Reductions in TOC were 91, 81 and 70% for sites A, B and C, respectively. HP-SEC analysis indicated changes in NOM molecular size distribution during the recharge process: larger molecular fractions were removed more efficiently than the smaller ones. Initial carbon content of raw water did not affect the removal efficiency; however recharge distances and retention times varied between the sites.

Total amount of bacterial cells in raw waters was about 1.6 million/ml at sites A and B and about 0.7 million/ml at site C. Recharge process at sites A, B and C resulted in removal percentages of bacterial numbers by 94, 94 and 75, respectively. Despite the decrease in total amount of bacteria and amount of organic matter in water, bacterial community remained diverse during the recharge process. However, some bands on the DGGE gel disappeared and some new ones appeared during the recharge. Even though sites A and C have been operating since the end of 1970s and site B since only the end of 2000, no significant differences occurred in organic matter removal efficiency, the quantity of bacteria or the diversity of the bacterial community during the recharge processes.

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Laboratory column study on the effect of ponding depth on infiltration rate during SAT

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Abstract

A sound understanding of soil clogging processes is fundamental in the design and operation of a proposed soil aquifer treatment (SAT) trial at Alice Springs in central Australia. A multi-factorial column experiment was undertaken to assess the effect of soil type, level of effluent pre-treatment, light and ponding depth. Only results pertaining to the effects of ponding depths of 10, 30 and 50 cm for two soil types fed with high quality reclaimed water are reported here. For both the sand and loam columns there is a ponding depth above which flow rate and hydraulic conductivity are reduced rather than enhanced. This is attributed to greater penetration or compaction of clogging agents within the soil, rather than filter cake compression in the low turbidity reclaimed water tested. The sand clogged more substantially than the loams, leading to a reduction in the hydraulic conductivity contrast between the two soil types from a factor of >200 initially, to 15–20 after the four cycles. Ponding depths of 30 cm or less are recommended for proposed SAT operations at Alice Springs.

Keywords

Clogging; ponding depth; reclaimed water; SAT.

INTRODUCTION

The success of a proposed soil aquifer treatment (SAT) trial with reclaimed water in the central Australian town of Alice Springs relies on defining appropriate design and operational parameters which will ensure that adequate rates of infiltration are maintained within recharge basins in the long-term (Knapton et al., 2004). Understanding the factors that cause infiltration rates to decline due to soil clogging is essential to achieving this goal. A laboratory column study was devised to systematically test a range of SAT management variables. The variables included soil type, source water quality, ponding depth and environmental factors such as solar radiation and temperature. Presentation of all the results of the study is not possible here. Instead, this paper focuses on the role of the ponding depth of water within the basin. A report by Pavelic et al., (in prep.) describes the results of the entire study.

Ponding depth is an important factor for SAT management since it affects rates of infiltration. Typically, SAT basins have been operated under a range of ponding depths, ranging from less than 10 cm to greater than 50 cm. The literature on the effects of waterlevel on infiltration rates is sparse and the findings often conflict. Some studies have suggested a positive correlation between ponding depth and infiltration rate (Houston et al., 1999); others a negative correlation (Rice, 1974; Bouwer and Rice, 1989). An increase in ponding depth is expected to increase infiltration rates due to the enhanced hydraulic gradient across the clogging layer on the basin floor and the increase in wetted surface area of the basin. However, this increases the residence time of the water in the basin and hence the opportunity for regrowth of algae and time needed to allow the basin to dry-out. Also, filter cake compression is accelerated by higher gradients and this may have a significant effect on infiltration.

METHOD

The column study was conducted to compare the infiltration rates of two different soil types with four qualities of water, in a controlled temperature environment in the absence of light. The water types included three effluents that had undergone varying degrees of pretreatment and a potable groundwater. Only the dissolved air floated reclaimed water from the Bolivar wastewater treatment plant (DAF) is of relevance to this paper. For that reclaimed water, the effect of three different ponding depths was tested at a constant temperature of 20°C, although during the last cycle the temperature rose up to 26°C due to a mechanical breakdown in the air conditioning system. For that same water type, columns of standard and extended lengths were tested in a nearby glasshouse where the temperature fluctuated and the soil surface and source water were exposed to solar radiation. Duplicate columns, each with a diameter of 2 cm and length of 10 cm, were run in parallel over an eight week period (October to December 2003) for four cycles of wetting and drying that were each of 7 days duration. The results for sand and loam soils with DAF water that compare ponding depths of 10, 30 and 50 cm are reported here.

One of the soils collected from the investigation site near Alice Springs is a coarse textured palaeochannel deposit, which for convenience, will be referred to as the 'sand'. The other is a finer textured sandy soil that we shall call the 'loam'. Physical properties are summarised in Table 1. The turbidity of the feed water varied from 0.3–1.1 NTU.

Measurements of infiltration rates and basic measures of water quality before and after soil passage were made routinely with detailed chemical analyses conducted in the last wetting cycle. At the end of the experiment the soils within the columns were sectioned and microbial analyses performed on some columns.

Table 1. Physical properties of the two soils types tested

	Property					
	K *(m/day)*	*Porosity* *(%)*	*Bulk density* *(g/cm3)*	*% Sand*	*% Silt*	*% Clay*
Sand	23	54	1.6	97	2	1
Loam	0.1	44	1.35	70	17	13

RESULTS AND DISCUSSION

Total cumulative flux

Figure 1 presents a plot of the cumulative flux of DAF water through the two replicate sand and two replicate loam columns, as a function of the cumulative number of days of wetting during the four cycles. For the sand columns, a 3-fold increase in ponding depth relative to a depth of 10 cm led to a 90% increase in the hydraulic loading after the four cycles (314 cf. 168 m). In the case of the loams, the corresponding increase was lower at 50% (15 cf. 10 m). A 5-fold increase in ponding depth initially led to highest infiltration rates for both soil types. The enhanced flux for 50 cm compared with 30 cm was maintained until the end of the third cycle for sands and midway through the fourth cycle for loams. Whilst the average behaviour of the 30 and 50 cm loam columns closely resembled each other in the fourth cycle, there was an appreciable increase in the deviation between two duplicates for 50 cm compared to smaller ponding depths. Thus, soils tended to clog less predictably at higher ponding depths. It is possible that higher seepage forces would provide greater opportunities for water to flush clogging agents from preferential flow paths than lower seepage forces, thereby increasing degree of micro-scale heterogeneity within the soil and increased variance between the duplicate columns. At lower ponding depths the flow field within the column was more uniform.
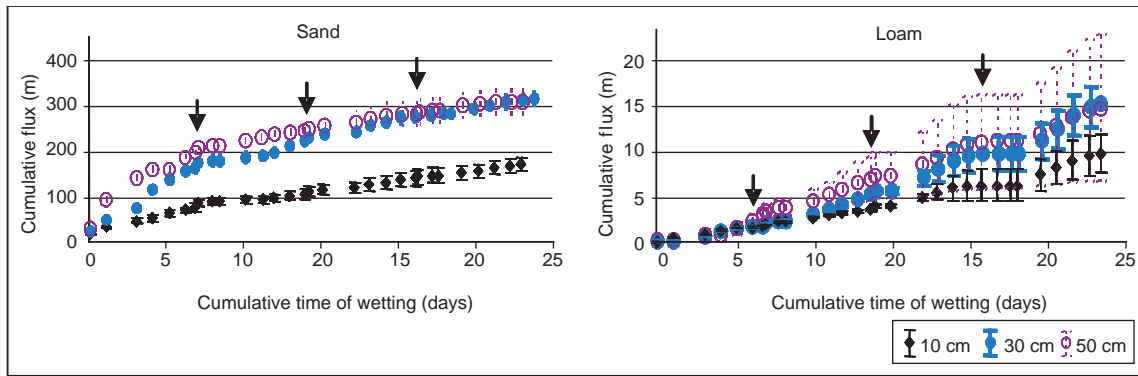
Figure 1. Total cumulative flux of DAF-fed sand columns (left) and loam columns (right) at ponding depths of 10, 30 and 50 cm. The averages are plotted as the symbols and the horizontal bars give the value for each duplicate. Periods when drying occurred are indicated by the vertical arrows. Note the 20-fold difference in the vertical scale between the two plots.

Relative hydraulic conductivity (K/K$_{sat}$)

Although the infiltration behaviour of the sand and loam soils were quite similar, the 15–20 fold difference in flux between the sands and the loams at the same hydraulic gradients should be noted in Figure 1. This compares to the >200-fold difference in the initial hydraulic conductivity of the two soils (Table 1). Figure 2 presents the results for the ratio of the hydraulic conductivity to the initial hydraulic conductivity (K/K$_{sat}$) as calculated from Darcy's Law. This clearly shows that the sand clogged more than the loam, and explains why the contrast in hydraulic conductivity between the two soils diminished over time. For the sands, clogging occurred marginally faster at the highest ponding depth. Flow rates were maintained at levels comparable to 30 cm, and higher than 10 cm only as a result of the higher imposed hydraulic gradient across the soil. Minimal restoration in K/K$_{sat}$ occurred as a result of the drying periods. For the loams, the strong initial declines in the first cycle were followed by steady increases in K/K$_{sat}$ during subsequent cycles in all but one of the 50 cm columns, where the response was more like that of the sands. K/K$_{sat}$ often declined within each wetting cycle but the restoration following each drying period and often during wetting overcompensated for the reduction, leading to significant net improvements in hydraulic conductivity. In two cases (for a 10 cm and a 30 cm column) the final K/K$_{sat}$ exceeded the initial conditions. Whilst there is no clear explanation for the increase over time, this phenomenon was not systematically observed for the loam columns with the three other water qualities tested, nor for (even) the tests conducted with the DAF water in the glasshouse.
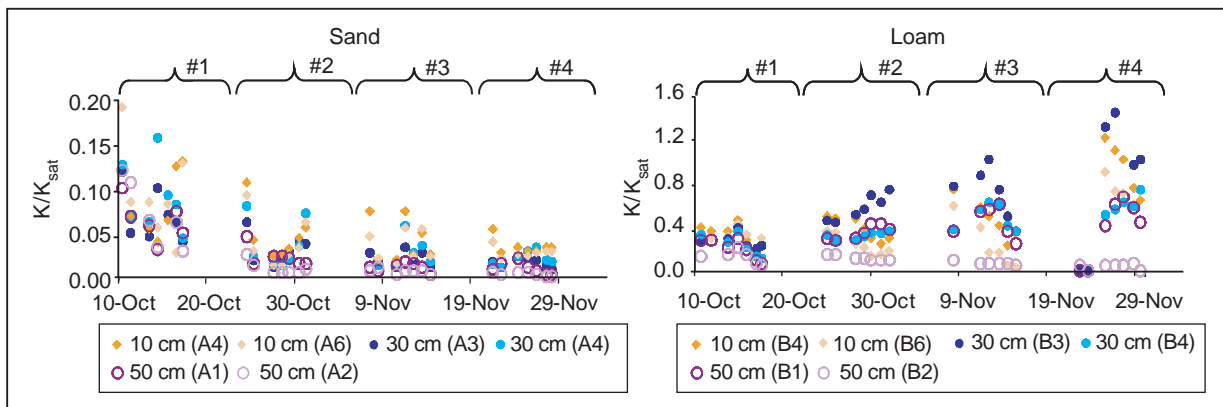
Figure 2. Relative hydraulic conductivity (K/Ksat) of DAF-fed sand and loam columns for ponding depths of 10, 30 and 50 cm. Note the different y-axis scale for sands and loams.

One of the more striking observations in the hydraulic conductivity data are the low K/K_{sat} values, even from the initial cycle. This occurred even though each soil type was packed to an identical bulk density (Table 1) and the initial K's using chlorinated potable water prior to commencing the experiment varied by less than 10% for sand columns and 20% for loam columns. Note that air bubbles entrained within the soil during the initial wetting-up phase were eliminated from the column prior to K testing. Where some influence was observed due to bubbles during the initial testing, this resulted in an increase in K over time rather than a decrease as the pore throats became cleared by flushing with water. The mechanism for the rapid initial drop in hydraulic conductivity within the first day of the experiment is currently being investigated in a follow-up study and preliminary results indicate that redistribution of soil particles within the column is a plausible explanation.

Clogging with soil depth

The observed soil clogging arises from the combined effect of physical and microbial processes. Due to the low particulate content of the DAF feed water, a clogging 'layer' did not develop on the surface of the sand or loam. Rather, the clogging occurred within the soil profile, since some of the organic matter introduced into the column in the influent was retained within the soil. Table 2 shows that, if suspended solids are proportional to turbidity that was measured, then between 29% and 64% of the particulate matter was captured within the soil in the latter stages of the experiment, with the loam being a more effective filter than the sand due to the smaller pore throats. The trapped particles are primarily of organic origin and provide a source of substrate for microbial growth that supplements the nutrients within the dissolved phase.

Table 2. Turbidity changes during soil passage through sand and loam columns with 10 cm ponding depth during the middle of the fourth wetting cycle

	Turbidity (NTU)		
	Inflow	*Outflow*	*Avg % change*
Sand	0.7	0.4−0.6	29
Loam	0.7	0.2−0.3	64

The most direct evidence on where the clogging occurs within the soil profile comes from a sand and loam column that were fitted with manometers at three depths, thereby allowing determination of K/K_{sat} over four intervals: 0−1, 1−3, 3−8 and 8−10 cm. Figure 3 shows that K/K_{sat} values were generally low for all of the intervals, with negligible difference between each interval observed.
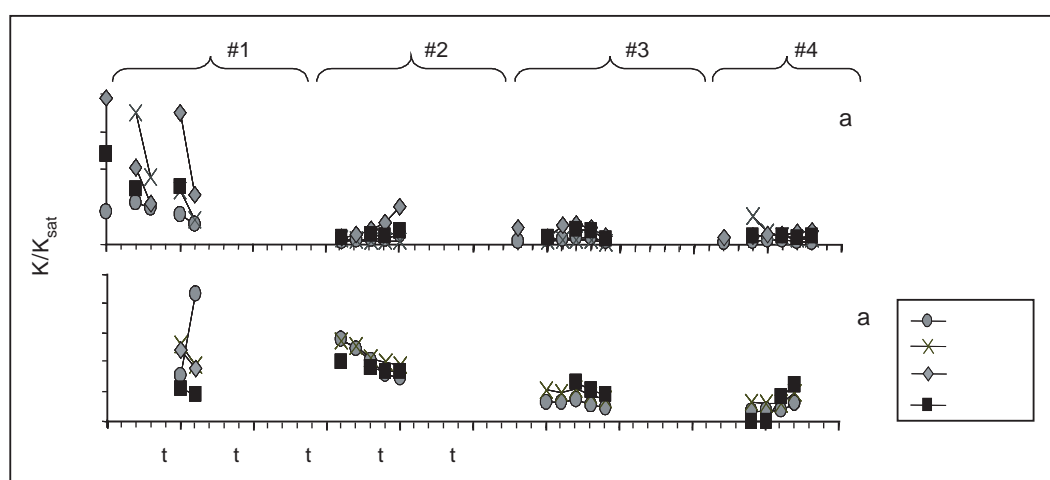
Figure 3. K/K_{sat} data of four soil intervals for a sand and a loam column with a 50 cm ponding depth. Due to re-equilibration caused by resaturation at the beginning of each wetting cycle some initial readings were omitted.

Figure 4 presents the microbial biomass index at the end of the experiment. Biomass values vary by no more than a factor of two along the column length. Polysaccharide values (not presented) also indicate a similar trend. The consistency of clogging with depth supports the K/K_{sat} data presented in Figure 3.

The enhancement of clogging rates at higher ponding depths may be due to either or both the compaction of the clogging layer and/or greater penetration depth of particulates into the pore space of the soil matrix. Rice, (1974) recommended that a higher ponding depth should be maintained since the faster rate of clogging would be overcompensated by the high initial infiltration rates and thus result in a higher net hydraulic loading. This was observed to be the case from 10 to 30 cm, but not from 30 to 50 cm. The reversibility of clogging due to the restorative effects of desiccation also appears to be negatively affected by increasing ponding depth.
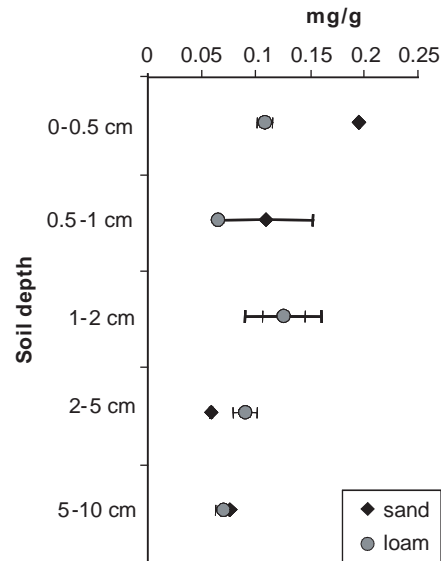
Figure 4. Changes in biomass index with soil depth at the end of the fourth cycle for sand and loam columns with a 10 cm ponding depth

CONCLUSIONS

The depth of ponding above the soil surface has a considerable impact on the rates of infiltration for both of the soil types tested. Of the three values of ponding depth tested (10, 30 and 50 cm), higher ponding depths were found to increase infiltration rates in the short-term but did not translate to higher rates over the longer-term. Fluxes at smaller ponding depths appeared to converge towards larger depths over time and the benefits of increased ponding depth was minimal. The differences in hydraulic loading between the two soil types was 15–20 fold, which narrowed over time since the sand clogged more rapidly and to a higher degree than the loam, presumably due to the higher mass flux of nutrients and particulates entering the soil. For a 30 cm ponding depth the average final K/K_{sat} for the sand was 0.015 and for loam was a significantly higher 1.0. Clogging agents were largely distributed over the entire 10 cm length of the columns, although there was a slight tendency for greater clogging in the shallowest interval of the soil.

Ponding depth plays a role due to its effect on the resistance, depth and persistence of clogging 'layers' within the soil. This may be because clogging is enhanced by greater downward movement of clogging agents into, or compaction of agents on the soil matrix, as opposed to compression of a surficial filter cake. Based on these results, ponding depths of 30 cm or less are recommended for proposed SAT operations at Alice Springs.

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Effect of grain size on biological clogging in porous media leached with synthetic nutrient solutions

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Abstract

Aquifer storage and recovery (ASR) provides beneficial storage and reuse options for reclaimed water. Injection of water containing nutrients into an aquifer creates a suitable environment for microbial growth and this can lead to clogging of the well screen, an increase in the hydraulic head and a loss of injection efficiency. For the long term sustainability of direct injection of reclaimed waters it is important to understand the factors controlling the biological clogging. Laboratory column experiments were used to investigate how the grain size of porous media affects the rate and degree of biological clogging. Three grain size distribution of silicate sand were investigated; fine (0.053–0.212mm), medium (0.125–0.355mm) and coarse (0.250–0.710mm). Over the experimental test period of 35days the hydraulic head increase due to biological clogging within the fine-grained sand was 2.8 times greater than the medium grained sand and 20 times greater than the coarse grained sand. This work has clearly shown that the grain size of the receiving aquifer subjected to ASR using nutrient rich waters has a significant impact on the amount of biological clogging and subsequent hydraulic head increase.

INTRODUCTION

Aquifer storage and recovery (ASR) of reclaimed waters is emerging as a potential storage and reuse option for reclaimed waters (Dillon at al. 2005). It has been well documented that sub-surface storage of reclaimed waters can lead to several different types of aquifer clogging. These clogging processes include, the filtration of suspended solids, precipitation/dissolution of minerals, entrained air binding and biological activity (Rinck-Pfeiffer at al. 2000, Vanderzalm at al. 2002). The potential for microbial activity and subsequent clogging around the well screen is enhanced when the injected water contains substantial nutrient concentrations and can lead to significant build-up in the injection pressure resulting in a loss of injection efficiency.

In order to be able to better manage aquifer clogging upon injection, it is important to understand the factors controlling the clogging processes. The work discussed in this paper was conducted to understand how the grain size of a porous media affects the rate and degree of biological clogging upon injection of nutrient rich waters. Previous studies investigating the relationship of grain size and permeability reductions due to biological clogging have observed that finer grained porous media were subject to more rapid decreases in permeability where concentrations of nutrients leached through the columns were considerably higher than those observed in tertiary treated waste waters (Cunningham at al. 1991, Vandevivere and Baveye 1992a). Cunningham at al. (1991) observed a reduction in hydraulic conductivity to 1–5% of the initial values for silicate sands and glass beads with a mean grain size of 0.12 mm, 0.54 mm, and 0.7 mm through biological clogging. Their porous media reactors were packed between 1 and 3 particles thick. A single strain bacteria *Pseudomonas aeruginosa* was used to induce clogging and the reactors were fed with a constant supply of 25 mg/L glucose substrate under constant hydraulic head resulting in a decrease in flow rate with time. Under these conditions the majority of the permeability reductions were observed within the first 5 days of leaching. The work presented here is part of a larger project investigating how variations in the nutrient composition of waters intended for ASR, and grain size of the porous media are affected by biological clogging. Laboratory columns packed with pure silicate sands, of three different grain sizes, were leached with a synthetic nutrient solution with similar nutrient characteristics to tertiary treated effluent injected into a confined aquifer at the Bolivar ASR site in South Australia.

METHODS

Experimental infrastructure

Laboratory column studies were used to investigate how the grain size of porous media affects the rate and extent of hydraulic head build-up over a 35 day test period. The columns were made from clear plexiglass (16 cm in length and 2.5 cm internal diameter) (after Rinck-Pfeiffer, 2000). Stainless steel mesh was placed inside the end caps to hold the porous media in place. The columns were wrapped in aluminium foil to prevent the growth of photo-trophic micro-organisms.

Porous media

Non-reactive silicate sand was used throughout all experiments in order to avoid any potential geochemical reactions between the solid and liquid phase. The grain size of the three sands included: a coarse grained sand (0.250–0.710 mm), a medium grained sand (0.125–0.355 mm) and a fine sand (0.053–0.212 mm). Prior to packing the columns the sands were acid washed in a 10% solution of nitric acid (HNO_3) for 24hours to remove potential carbonates from the sand and then rinsed thoroughly in ultra-pure (U.P.) water.

Source solutions

A synthetically composed nutrient solution was used throughout the experiments to allow for controlled manipulation of the nutrient characteristics. The chemical characteristics of the synthetic solution mimicked the bulk chemical characteristics of tertiary treated Dissolved Air Flotation/Filtration (DAF/F) effluent used for direct injection at the Bolivar ASR site in South Australia. Table 1 outlines the physical and chemical properties of the synthetic nutrient solution.

Table 1. Summary of the physical and chemical characteristics of the synthetic nutrient solution used throughout the column experiments

Physical and chemical properties of synthetic nutrient solution	Value (mg/L unless otherwise stated)
Total Organic Carbon (TOC)	2
Total Nitrogen (TN)	8
Total Phosphorus (TP)	1.6
Ca^{2+}	22.9
K^+	35.2
Na^+	157
Mg^{2+}	20
Fe^{2+}	0.1
Cu^{2+}	0.004
Cl^-	280.7
SO_4^{2-}	79.2
HCO_3^-	80.3
pH	7.5
E.C. (mS/cm)	1.61
Total Bacterial Count (cells/ml)	10,000–60,000

Cellobiose ($C_{12}H_{22}O_{11}$) was used as a carbon source, potassium nitrate (KNO_3) was selected as the nitrogen sources and potassium-di-hydrogen orthophosphate (KH_2PO_4) and di-sodium hydrogen orthophosphate ($Na_2HPO_4.12H_2O$) were used as phosphorus sources. The synthetic nutrient solution was seeded with a fresh

sample of unchlorinated DAF/F effluent to introduce an effluent microbial community to the synthetic solution (10 ml DAF/F per 1 L of synthetic nutrient solution).

A total organic carbon (TOC) concentration of 2mg-C/L was used for all the synthetic nutrient solutions. This concentration of carbon was based on the average observed removal of TOC from the DAF/F effluent under field and laboratory conditions (Vanderzalm 2004).

Experimental procedure and analytical techniques

Duplicate columns were packed at 1cm increments with the fine, medium and coarse grained acid-washed sand. All of the columns were inoculated for two weeks with a fresh unfiltered sample of groundwater in order to establish a native population of groundwater micro-organisms in the clean silicate sand. The columns were then fed with synthetic reclaimed water in an upward flow arrangement at a constant flow rate of 2 L/day (*ie.* residence time in columns was 22 minutes). The development of biological clogging and subsequent build up of hydraulic head was measured along the length of the columns using a combination of pressure transducers (WIKA, Model S-10, Germany) and manometers. Pressure transducers were used at the inlet of the columns where the greatest hydraulic head was expected and manometers were positioned along the columns at 3 cm, 8 cm, 13 cm and the outlet.

At the end of the experimental period the columns were dismantled and sampled at 0.5 cm intervals for the first 3cm of the column length and then at 1cm intervals for the remaining length of the columns. The sand from each interval was then analysed for polysaccharide concentrations using a phenol-sulphuric method (Dubois at al. 1956) and total viable biomass concentration (Haack at al. 1994).

RESULTS AND DISCUSSION

Hydraulic head response

Hydraulic head increase at the inlet of the columns was observed in all the sands during the 35day experimental period (Figure 1). The rate at which clogging occurs and the magnitude of hydraulic head increase varies quite significantly for the different grain size porous media. The grain size distribution of a porous media plays a crucial role in the change in hydraulic properties upon commencement of biological clogging. Clogging of the fine-grained porous media resulted in a final hydraulic head build up 2.8 times greater than that observed in medium grained sand and 20 times greater than the hydraulic head build-up in the coarse- grained sand. Observed clogging is associated with biological activity as the influent was initially free of suspended solids, however while biological clogging was investigated there is a potential for physical clogging due to the experimental design. The nature of the material causing the physical clogging is believed to be organic as the synthetic solution was composed of mineral salts and organic nutrients below saturation and the only suspended component added to the solutions was the DAF/F seeding solution which did contain bacterial and algal cells. The supply tank from which the columns were fed contained a nutrient rich solution seeded with a sample of DAF/F water. This would have provided suitable conditions for growth and death of cells within the supply tank and subsequent settling out of the dead cells. Refilling of the supply tanks may disturb sediments at the bottom of the tanks which may subsequently be sucked into the supply tube and filtered out at the inlet of the columns resulting in a sudden increase in hydraulic head. However, further experimental investigation is required to validate these findings. The supply tanks were refilled approximately every 5 days. The cyclic sudden increases in hydraulic head coincide with filling events. It is evident that the fine grained porous media was most sensitive to this due to the narrower pore apertures becoming blocked by the suspended load. Monitoring of the nutrient concentrations showed a TOC removal of 0.2 mg-C/L over a five day period within the supply tanks suggesting that there was some microbial activity potentially leading to the suspended solids in the influent.
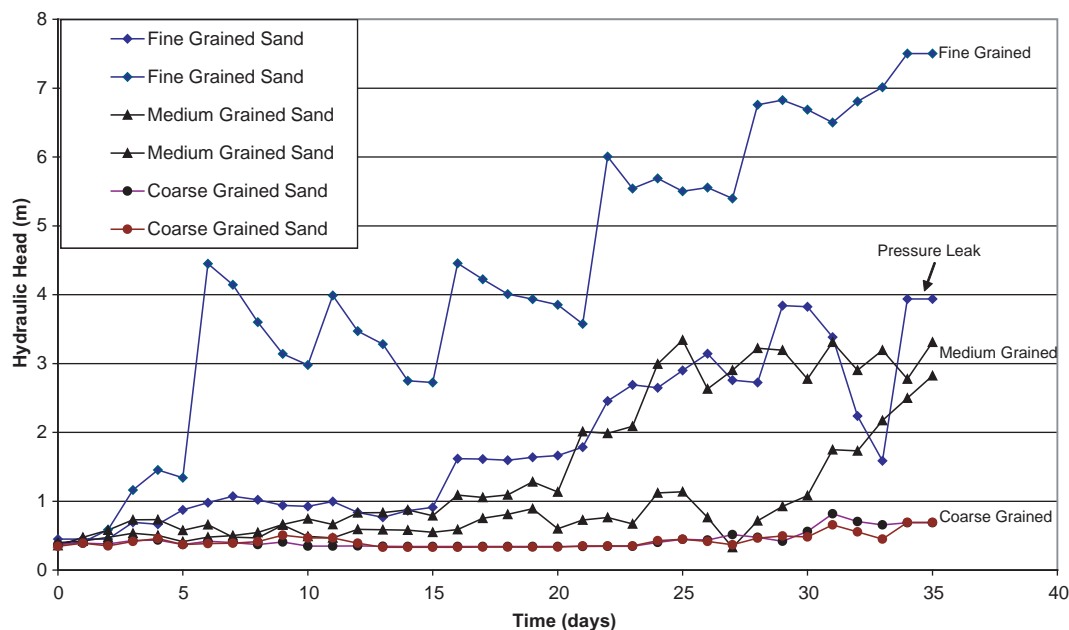
*Figure 1. Hydraulic head build-up at the inlet of duplicate columns
packed with fine, medium and coarse grained sands*

The initial clean, un-clogged saturated hydraulic conductivities at the inlet interval (0–3 cm) of the columns were 2.6–3.8 m/d for the fine grained columns, 8.6–18.7 m/d for the medium grained columns and 9.6–10.7 m/d for the coarse grained columns. For each of the grain sizes the saturated hydraulic conductivity decreased over the experimental test period. The final saturated hydraulic conductivities for all the columns were between 1 and 5% of the initial saturated hydraulic conductivity. The final Ks for the fine-grained columns was 0.02 m/d, and was 0.05 m/d for the medium-grained columns and between 0.34–0.56 m/d for the coarse- grained columns.

Biomass and polysaccharide production

At the end of the experimental period the columns were dismantled and the porous media sampled at 0.5 cm interval for the first 3 cm of the column and 1cm intervals for the remaining length of the column. All of the intervals were sampled for polysaccharide concentrations while a few selected intervals were sampled for biomass concentrations. Figure 2 shows the average polysaccharide concentrations along the length of duplicate sets of columns packed with fine, medium and coarse-grained porous media. The greatest concentrations of polysaccharide were found within the first 2–3 cm of the column inlets. This pattern of accumulation of microbial products at the inlet of columns has been observed in most studies investigating biological clogging of porous media (Allison 1947, Vandevivere and Baveye 1992b, Rinck-Pfeiffer 2000).

The highest polysaccharide concentrations of 4–5.2 mg/g were measured over the inlet interval (0–0.5 cm) in the fine grained porous media, followed by 2.3–3.9 mg/g at the inlet of the medium grained sand and the lowest polysaccharide concentrations of 1.8–2 mg/g was measured at the inlet of the coarse grained sand. Rapid decrease down to minimal concentrations was observed in all columns regardless of the grain size.

Biomass concentrations at the inlet interval of the fine, medium and coarse grained columns show similar patterns. Table 2 summaries the biomass and polysaccharide concentrations over the inlet interval for the three different porous media.
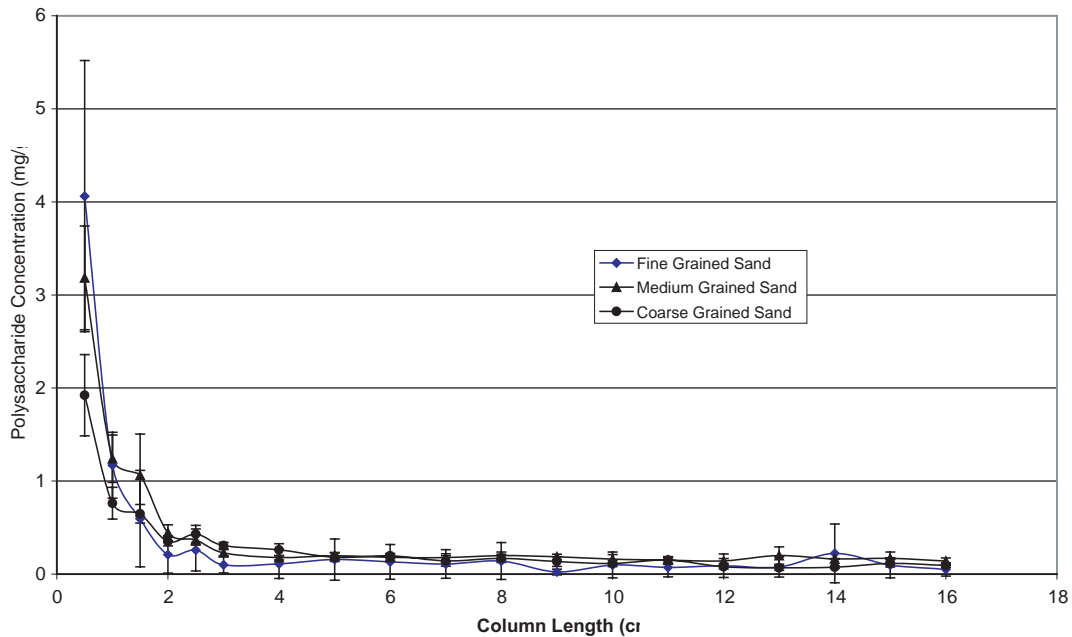
*Figure 2. Polysaccharide concentrations along the length of the columns packed
with fine, medium and coarse grained sand. Data represent means for duplicate columns*

Table 2. Summary of the biological and hydraulic parameters measured at the inlet (0–0.5 cm) of the columns

Porous media	Biomass conc. (mg/g)	Polysaccharide conc. (mg/g)	Final hydraulic conductivity (m/day)
Fine Grained	10 – 17	4 – 5.2	0.002
Medium Grained	17*	2.3 – 3.9	0.005
Coarse Grained	7 – 10	1.8–2	0.34 – 0.56

Note: * Analytical problem resulting in no duplicate data.

CONCLUSIONS

Laboratory column experiments were carried out with three different grain sizes of pure silicate sand to investigate the effects of injecting nutrient rich solutions on the hydraulic properties of the porous media. The key observations for the given experimental conditions were the following:

1. An increase in hydraulic head at the point of injection and subsequent reduction in hydraulic conductivity, due to biological clogging, was sensitive to the grain size of the porous media. More rapid clogging occurred in porous media with finer grains and narrower pore apertures.
2. The final saturated hydraulic conductivities for all the columns were between 1–5% of the initial un-clogged saturated hydraulic conductivity.
3. A decrease in the grain size of porous media resulted in greater concentration of polysaccharides and biomass at the inlet of the column.

It is evident that the injection of nutrient rich waters resulted in the development of biological clogging at the point of injection. The growth and development of microbial communities and their extra cellular products will eventually lead to the reduction of pore apertures and subsequent decrease in hydraulic conductivity. At a field scale this implies that greater injection pressures would be required to maintain constant injection rates.

ACKNOWLEDGEMENTS

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Case studies

Region issues and artificial recharge case studies

Groundwater resource management on the urban environment

Eyilachew Yitayew

INTRODUCTION

Background

Water is a fundamental resource for socio-economic development and is essential for maintaining healthy environment and ecosystems. Consequently, there is a raising demand for fresh water resources as a result of increasing population and advancement of technology, which it is becoming increasingly difficult to satisfy in the current context of growing pollution world-wide. This is a matter requiring urgent attention, since water is a scarce and such an important resource that it needs detailed scientific research all over the world in order to sustain and protect the water resource from pollution and for its wise utilization.

Natural fresh water is a finite resource, essential for agriculture, industry and human/domestic existence. Without fresh water of adequate quantity and quality, sustainable development will be impossible and life is in danger. But human intervention in the natural system has a significant effect on the quality of natural water. Human activities like discharge of untreated toxic chemical and industrial waste into streams, unplanned urban development, lack of sewerage system, over pumping of aquifers, contamination of water bodies with substances that promote algal growth (possibly leading to eutrophication) and global circulation (heating) are some of the prevailing causes of water quality degradation.

Dire-Dawa is the second largest urbanized centre in Ethiopia, next to Addis Ababa, along the Addis Ababa-Djibuti railway, with the population of more than 270,000. It has an enormous development potential. Industries mainly comprise food-processing plants, textile and cement factory. The main source of water for domestic supply is groundwater (Sabiyan well-field) within the urbanized part of the town. In addition, there is water supply from the springs and hand dug wells. Although easier to exploit, it has to be treated before it meets the WHO standards for domestic supply. Hence, the water supply from the Dire Dawa and its vicinity is not completely safe due to the presence of different sources of pollutants that increase the risk of chemical pollution of the water resource.

Objectives

- Determine water quality of the aquifers and impacts of the present and future exploitation;
- To identify the different sources of pollutants affecting the quality of groundwater;
- Give recommendation on the strategy of groundwater resources development and protection.

Methodology

The methods applied are:

- Data collection and review of previous studies i.e geological, hydrogeological , urbanization;
- Water points inventory, sampling & field measurements (location, EC, PH, EH, temperature);
- Water quality survey, hydrochemical, isotope analysis and pollution source identification by applying different softwares.

URBANIZATION AND GROUNDWATER POLLUTION IN DIRE DAWA TOWN

General situation

The geomorphologic, geologic and hydrogeologic characteristics of the basin facilitate the effect of pollution. The town consists of mainly unconsolidated permeable alluvium deposit that can easily be penetrated by disposing wastewater. Percolation of wastewater is also facilitated by gently dipping or almost flat nature of the topography of the town. Ground water table is encountered at shallow depth in the northern part of the town that is as low as ten meters below ground level. Such distance is covered with in a short time interval by descending pollutants.

Degradation of the town's groundwater quality may be caused by urban agriculture in the south (valley) – western and eastern sides of the basin, or by point sources such as septic tanks, pit latrines, garbage disposal sites and cemeteries. Line sources such as poor quality water; wastewater drainages from factories and seepage from polluted streams affect the hydrogeologic system in a considerably high magnitude which tend to move laterally in the direction sources of the pollution.

The present contamination may be at its early stage because the movement of both groundwater and pollutants is so slow that it takes many years to be in contact. After the contact is made, it is difficult to cleanup and rehabilitate from the aquifer since the degree of contamination will show a growing/pluming trend.

Spatial conditions and urbanization

Dire Dawa is the second largest and one of the fast growing towns in Ethiopia. It has recorded a dramatic growth since its foundation. The first Master plan of Dire Dawa was prepared in 1967 that has now become obsolete. The landuse master plan that dates back to late 1967 and 1994 (NUPI) indicates that the total planned area was 2,928 and 3,241 hectares respectively. Presently (2004), it is extended to 8,386 hectares. The existing and future land use for Dire Dawa town is shown on the landuse map.

The landuse of the town is dominantly mixed, especially residential areas with commercial activities. This is true notably in the central part of the town where almost all buildings along the streets are used for commercial activities and their backyards or internal courtyards are used for dwelling purpose. Residential areas cover around 680 ha (10.38%), squatter settlement is estimated 980 ha (12%) and all about consists of 50.14% of the total built-up area.

The sanitation situation in Ethiopia is very low; the development is limited and has not been a major concern. Most of the population in rural and urban areas do not have access to safe and reliable sanitation facilities. As a result, above 75% of the health problems in Ethiopia are due to contagious diseases attributed to unsafe and inadequate water supply, and unhygienic waste management, particularly human excreta.

In Dire Dawa town, there is no municipal sewerage system at present. The sanitary system and practice in the town is very poor unlined traditional pit latrines are the most common technology in use in the town. At present the town doesn't have any system for the safe disposal of wastewater. Each household is in charge of disposing of its own waste. It is clear that the existing facilities do not cover the needs of the town in terms of sanitary state of affairs.

Sanitation in Dire Dawa town at the moment is the responsibility of both the Water Supply and Sewerage and the Health Offices, even though they have insufficient means at their disposal to adequately execute their role. In the town there are three trucks and 84 transfer containers for refuse collection and two vacuum trucks for emptying the filled toilets. Sullage is domestic wastewater used for body washing, laundries and cleaning of cooking utensils.

Sullage water represents a significant proportion of water consumption in the town and the volume produced is greatly dependent on the volume of domestic water used. In Dire Dawa town, washing is mostly carried out at the entrance of households and the resulting wastewater is disposed on the ground or into drainage ditches outside the compound.

Table 1. Solid waste disposal situations in Dire Dawa and Rural as per 1998 CSA

Status	Vehicle container	Dug out	Thrown away	Others	Total
Rural	3.1%	1.1%	93%	2.8%	100%
Urban	46.55%	11%	37.4%	5.1%	100%

Source: 1998 CSA WMS.

Solid waste disposal sites are not selected according to hydrogeological priorities. The main solid waste disposal area is the sandy dry stream Channel of Dechatu River that divides the town into two almost equal parts. Solid waste stack is clearly seen in the dry river channel starting from the upper part of the town (Addis Ketema) to the lower part (Kebelle 22).

Dire Dawa textile, meat factory and soft drinks factory discharge untreated wastewater into the drainage. All of them drain northward unprotected and open to additional surface contaminations. Due to high permeable nature of the geological formations, there is a sharp drop in the amount of wastewater into the system from the initial points.

It is noted that dysentery and malaria are the second and the third causes of death in the region which are caused by ingestion of contaminated water. Even the first cause of death TB- the thermometer of the living standard is highly connected with water, sanitation and environment.

Sources of pollution

Due to the rapid growth and urbanization in Dire Dawa, there is an increase in the size of population, number of commercial establishment and number of industries. Consequently, the amount of waste generated has also increased since there is no good integrated waste management system (no recycling, sewage network and landfill sites) in the town. The geology and aquifer system is easily exposed for contamination. The major pollution sources are related to anthropogenic activities that are domestic, industry and agriculture. On the other hand, natural hardness is a major problem of the region since it is a sedimentary terrain and the major aquifer is limestone with Ca-Mg bicarbonate of water type is dominated.

Industrial Activities: Industry is the second important economic activity in urban area. There are six major industries and more than 100 small scale manufacturing industries in the town of Dire Dawa. These are Dire Dawa textile, Dire Dawa food complex, ELFORA meat processing, East Africa bottling (soft drink), and Dire Dawa Cement.

Major wastes and by products of the factory are carbon dioxide, carbon monoxide, dust and some time sulfur dioxide. Carbon monoxide is produced when there is incomplete combustion of raw material. The main raw material is lime ($CaCO_3$). This lime when it partially combusted gives cement, CO_2, and CO. Both CO_2 and CO liberate to the air, which ultimately contribute to the green house effect in the atmosphere. Other nuisance waste product is sulfur dioxide that liberates from the furnace when there is less air or less ventilation. This gas has been causing bad smell for the near by residents.

The Dire Dawa textile factory is the main source of contaminant in the urban area. The chemicals used in this factory pollute the ground and surface water. The factory has no waste treatment plant and it directly releases all sort of wastes into the near by stream. Most of the time, PH of the waste is more than 12.

In the town, there are about 98 medium and small-scale factories. All medium and large-scale industries do not treat their effluent or liquid waste. They merely discharge open field, near by the Dry River or sandy stream channel, which is a threat for the ground water source. The cumulative and long-term effect of these pollutants in the environment, particularly in the surface and ground water resource potential of the town would be significant.

Municipal Wastes. There is no waste treatment and proper waste disposal system in Dire Dawa town. Domestic as well as industrial wastes have been discharged directly into the open ditches and sandy streams. From the nature of its topography and soil, the ground water resource is very venerable to pollution. Degradation of ground water quality is exacerbating by point source contamination such as septic tanks, pit latrines and industrial effluents.

In Dire Dawa town, there is no central municipal sewerage system at present. The sanitary system and practice in the town is very poor; unlined traditional pit latrines are the most common technology in use in the town. At present the town doesn't have any system for the safe disposal of wastewater. Each household is in charge of disposing its own waste. It is clear that the existing facilities do not cover the needs of the town in terms of sanitary state of affairs.

Table 2. Urban toilet facilities of Dire Dawa by housing unit, as to 1994 census

Towns	All Housing units	Type of toilet facilities					
		Has no Toilet	Flushed Toilet private	Flushed Toilet shared	Pit Private	Pit Shared	Not stated
Dire Dawa	34,680	21.6%	4.7%	2.4%	29.7%	39.6%	1.9%
MelkaJebdu	1,702	61.4%	–	0.5%	31.1%	5.4%	1.4%
DDAC	36,382	8,531	1,662	851	10,831	13,811	696

According to CSA (1994), about 68% of the household in Dire Dawa use pit latrines. This situation is increasing the risk of the ground water contamination by human wastes. These pollution problems are clearly observed from the water quality analysis in the past 30 years. Water quality test from different sites of the town (bore holes) shows that nitrate concentration of water samples from Sabiyan and inner part of the town highly exceeds the WHO and Ethiopian Drinking Water Quality Guideline Values.

Agricultural wastes. In the town and its vicinity, there are numerous urban and rural agricultural activities such as Tony farm, chat farm, Amdael diary farm, Hafecat diary, and other small-scale cattle breeding and horticulture producers in the town. Generally, the agricultural inputs and by-products are the major constitutes of wastes and have a chance to contact the groundwater system.

Animal wastes are classified as solid and liquid. Such animal waste may become the source of groundwater pollution. The groundwater quality is directly affected by microorganisms in organic fertilizers, particularly when raw animal feaces are applied without being subjected to thermal and anaerobic stabilization prior to application (Castany, Groba, Tomija, 1986; and references therein).

Miscellaneous activities. In Dire Dawa area, in addition to the above-mentioned potential sources there are also possible pollution sources like markets, cemeteries, fuel stations, garages and etc. Urban activities produce numerous sources of contamination to the environment.

The other possible source of water pollution, which can be inferred, is cemeteries. Most of the Churches in Dire Dawa have graveyards /cemeteries/ away from their compounds. There are two main cemeteries associated with the religious consideration such as Muslim and Christian. But both cemetery sites are in the inner part of the town and near to Dechatu river.

Sources of groundwater contamination

The chemical, biological and physical properties of natural water are variable and depend on the composition of precipitation, geology, climate, biological activities, geochemical processes, the contact surface and contact time between water and rocks and human factors. As a result of disposal of liquid, gaseous and solid wastes, chemical substances and pathogens can pollute water sources; and their potability may be impaired by troublesome odor, test and color. Also substances that can be used in agriculture such a fertilizers, insecticides, and herbicides may pollute water sources. Their harmful effect on water quality can be caused by inappropriate application or the chemical nature of the substance used.

Generally domestic as well as industrial wastes affect water quality in urban centers. In Ethiopia the degree of pollution is generally not large except the surface waters of Addis Ababa and groundwater of Dire Dawa town. In general, the influence of human activity in the wastewater in the urban centers is indicated by the increase of nitrate and chloride concentration in the water bodies.

Groundwater moves slowly and responds slowly to quality changes. The movement of pollution is determined by the movement of water and therefore depends on physical factors. Pollution tends to attenuate as it moves through soil and groundwater systems due to physical dilution and dispersion, and a combination of chemical and biological actions. Of these precipitation-solution, ion exchange and adsorption, and biological degradation (oxidation-reduction) are most significant to groundwater quality. The importance of the various physical, chemical or biological process depends on the type and source of pollution, the nature of the soil and /or aquifer material, the hydrology and the well field.

The causes, types and extent of groundwater pollution range from wide spread agricultural sources to the primary domestic pollution sources of solid and liquid waste disposal. Individual septic tanks cause local and up to regional contamination problems of nitrate buildup if there concentration is very high. The land disposal of municipal sewage is a potential nitrate source and landfills are sources of metals and variety of organic and inorganic compounds.

Most potential groundwater contaminants are released at or slightly beneath the land surface. Here the wastes are subjected to the processes of leaching and percolation that may lead to their introduction into the ground water system. As they move through the unsaturated zone above the groundwater table there is the tendency to attenuate; a process that sometimes eliminates potential contamination sources as serious problems, because contamination simply does not reach the groundwater in sufficient strength.

The movement of a solute through the unsaturated zone, or zone of aeration, to water table is primarily vertically downward from the surface, and then horizontal displacement has undergone within the saturated zone. In unsaturated zone hydraulic and mass transport properties influence the degree of pollutant movement (FAO-Rome, 1979).

Contaminants are solutes reaching aquifer systems as a result of human activities. Pollution occurs when contaminant concentrations reach objectionable levels. Man's interference with natural flow patterns and his introduction of chemical and biological material into the ground usually results in undesirable groundwater quality changes. Contaminant sources include: municipal sewer leakage, liquid waste disposal, solid waste disposal, urban run off, lawn fertilizer application, industrial liquid waste disposal, tank and pipeline leakage, mining activities, chemical spills etc.

Point Source Contamination: Dire Dawa has no sewer system, well-studied waste disposal sites and no systematic disposal methods. For the past nine decades, pit latrines have remained the main human excreta disposal facilities. There are more than fifteen thousand pit latrines and more than one thousand septic tanks throughout the town. In

the other part of the town, where the population density is higher, the latrines are closely spaced and this poses cumulative effect on the hydrogeologic environment. With total population of more than 270,000 and poor feeding style, the annual tonnage of human excreta dumped is considerably high. Such exponential increase will have a direct and higher effect on the hydrogeologic system since it is only in the past few decades that the town has well developed and populated. The effect of population in polluting the hydrogeologic system is still in its early stage.

Human excreta contain large amount of water, 20% organic matter, 2.5% urine, nitrogen, phosphoric acid, sulpher and other in organic compounds (Ehlers and steer, 1976). As can be understood from this fact, nitrogen is the main component of human excreta. Of the organic matter contained in average domestic sewage, about 40% is made up of nitrogenous substances, 50% of carbohydrates and 10% of fats (Fair and Geyer, 1971). Such high influx of nitrogenous substances will result in nitrate pollution of groundwater.

Line Source Contamination: Industrial and domestic waste water drain in to the sandy stream channels and perco- late with in few hundreds of meters distance from their source. Dire Dawa textiles factory, meat factory and soft drinks factory discharge untreated wastewater through drainages and into the hydrogeologic environment, unpro- tected and open that makes additional surface contaminations. On the other way, they create favorable conditions for the decomposition of organic matter dumped as a solid waste in the sandy water channels.

Nitrate and Extent of pollution: The laboratory analyses conducted on the water samples taken from different local- ities at different times indicate that the level of groundwater pollution is increasing at an alarming rate. For instance, according to the hydrochemical analysis conducted by an Israeli geologist in 1959,the maximum concen- tration of NO_3 at the center of the town was 45 mg/l. After twenty-two years (1982) Ketema Tadesse has reported a NO_3 concentration of 320 mg/l within the town. On the other hand, while preparing this research the water sam- pled from Dire Dawa food complex bore hole (FBH) at August 2003 and analyzed on October 2003 by EIGS-labora- tory shows the result is still high to 266mg/l (2003).

Nitrate may also play a role in the production of nitrosomines in the stomach, which are known as carcinogens. This was considered as a possible reason for a higher death rate from gastric cancer in a group of people that had high nitrate level in their drinking water (Bower and references their in, 1978). Point contamination also has high- er contribution to the higher concentration of sulphate, sodium and chlorine in Dire Dawa ground water (refer nitrate map).

Presently, the factors that are assumed to cause pollution (declining of the rainfall, increment of household wastes, absence of proper waste management, etc.) are being aggravated. Hence the level of pollution is exceeding the maxi- mum allowable limits set by World Health Organization (WHO) and the National Standard (ENWQG).

High nitrate, chloride, and sodium concentration is observed in the wells found within Dire Dawa town. The water quality of alluvial aquifer is highly contaminated by human interferences, especially the alluvial water in the Dire Dawa town and dug wells near the community. The Total dissolved solids of polluted water is from 1,000 mg/l to more than 3,000 mg/l. The spatial coverage and trend of TDS, EC, hardness, major cations and anions of the basin in general and the Dire Dawa town in particular is explained on the maps

From the land use map of the town and in relation with the hydrochemical evolution, the concentration of EC, TDS, chloride, nitrate and sulphate are mainly depend on the groundwater flow direction and anthropogenic activi- ties. For further understanding, please deal with the figures attached here with.

Aquifers vulnerability

There are two aquifers in Dire Dawa area that is the alluvial and the upper sandstone aquifer. The main aquifer that
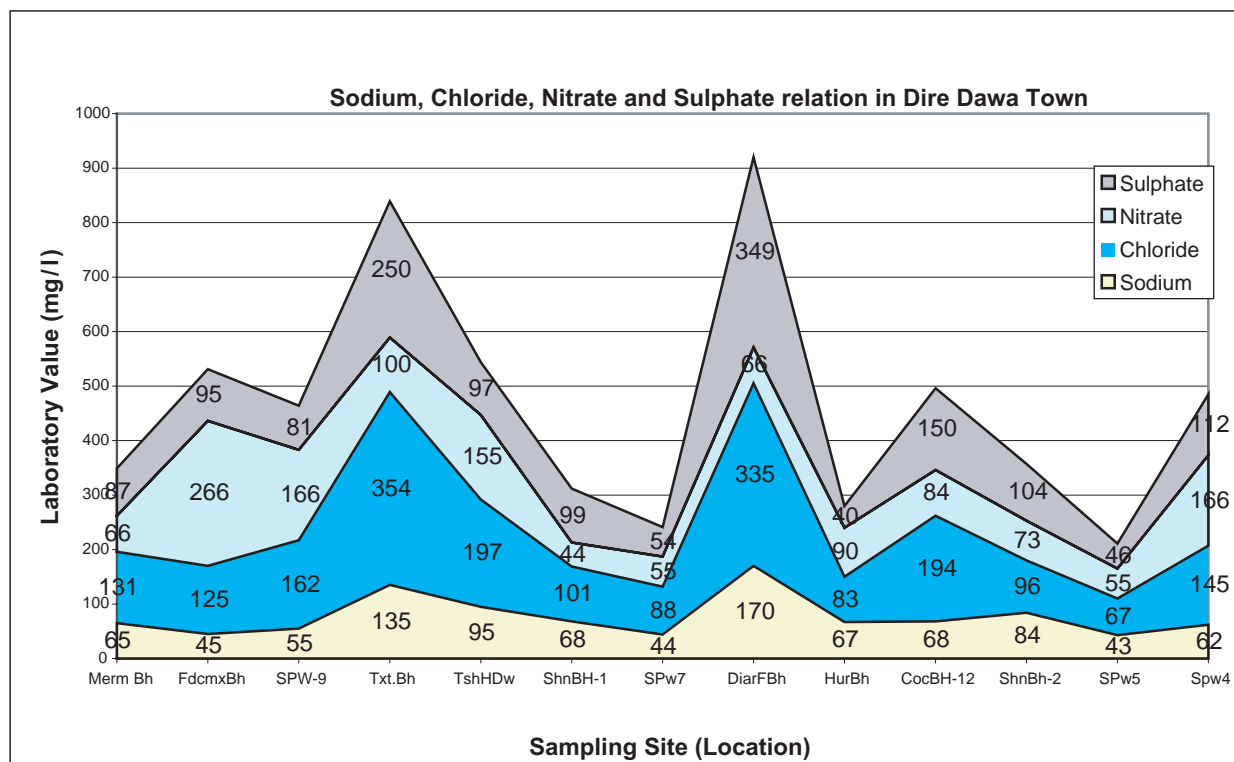
Figure 1. Sodium, chloride, nitrate and sulphate relations within Dire Dawa groundwater

is exploited for the Dire Dawa town water supply is the upper sandstone. Upper sandstone is semi-confined overlaid by the alluvial aquifer. The groundwater elevation map show that the groundwater level is deep along Dechatu river and at the same time the overlying alluvial and the upper sandstones are highly permeable and the groundwater flow concentrates along the river. In general, the upper sandstone aquifer at Dire Dawa town is vulnerable to pollution due to the high to moderate permeability of the alluvial sediments overlying the main aquifer.

From nitrates, chlorides and sulphate concentration distribution in the aquifer of the Dire Dawa basin the following observations are made:

- High concentration is directly related to high population density and industrial areas
- The high concentration plume is flowing along the groundwater flow direction
- At Sabiyain well field a localized plume of nitrates, chlorides and sulphate are flowing to the well field. At present, some of the wells in the well field are located within the zone of 50–166 mg/l of nitrates.

Impacts of pollution on the environment

One of the major difficulties with groundwater contamination is that it occurs underground with a complex system. The pollution sources are not easily observed nor are their effect seen until damage has occurred. The tangible effects of groundwater contamination usually come to light after the incident causing the contamination has occurred. This long lag time is a major problem.

All the scientific evidence that has been considered so far lead to the conclusion that continued disposal of wastes to the subsurface will produce progressive and largely irreversible pollution of the groundwater. Although physical, chemical and biological processes have been identified which may attenuate pollutants, these processes have finite limits or are operative only over a limited range of phsico-chemical conditions. Persistent pollutants will therefore

accumulate in the groundwater until a dynamic balance is reached between the rate of input of the pollutants to the groundwater flow system and the rate of discharge of the pollutants from the system.

In order to evaluate the extent of pollution in the study area the number and distribution of possible pollutant sources, hydrogeolgical characteristics of rocks and soils and results of water analysis have been given due emphasis. Based on these factors the study area divided in to three different areas as shown in the map (Fig. 1). The sources of pollutants are closely associated with landuse pattern and to some extent to population density in the area. Moreover, the extent of pollution also varies between urban and suburban part of the town.

Remedial measures and aquifer cleanup

The general characteristics of contaminants from, common point and non-point sources are seepage pits and trenches, percolation ponds, lagoons, waste disposal facilities, streambeds, landfills, deep disposal wells, injection wells, surface spreading and irrigation areas, and farming areas.

The largest component of municipal land disposal of solid wastes is paper, but substantial food wastes, yard wastes, glass, menials, plastics, rubber, and liquid wastes are also included. Landfill leachiest can contain high levels of BOD, COD, iron, manganese, chloride, nitrate, hardness, heavy metals, stable organics, and trace elements. Gases such as methane, carbon dioxide, ammonia, and hydrogen sulfide are by-products of municipal landfills. Many municipal waste disposal sites receive industrial process residuals and pollution control sludge. Radioactive, toxic, and hazardous wastes have been disposed in some municipal landfills and applied an integrated waste management system.

Municipal waste water may reach to aquifers by leakage from collecting sewers, leakage from the industries during processing, land disposal of the treatment plant effluent, disposal to surface waters which recharged aquifers, and land disposal of sludge. Sewer leakage can introduce high concentrations of BOD, COD, nitrate, organic chemicals, bacteria, and heavy metals in to groundwater. Potential contaminants form sludge includes nutrients, heavy metals, and pathogenic organisms. Potential contaminants from industrial waste disposal sites cover the full range of inorganic and organic chemicals including phenols, acids, heavy metals, and cyanide.

These remedial measures are time-consuming (months to years) and very expensive (tens of thousands to millions of dollars). It often takes longer to decontaminate an aquifer than it took to contaminate the aquifer.

Remedial measures remove or isolate point sources and/or pump and treat contaminated ground water (JRB Associates, 1982). Remedial measures include: changing the surface drainage so that runoff does not cross the source, using source subsurface drains and ditches, constructing low-permeability caps above the source, installing a low-permeability vertical barrier (slurry wall, grout curtain, or sheet piling) around the source, lowering the water table where it is in contact with the source, chemical or biological in situ treatment of the source plume, modifying nearby production well discharge patterns, changing water table hydraulic gradients though the installation of injection wells, artificial recharge and extracting contaminated ground water via production wells.

 Removal of contaminated water through extraction wells with aquifer advection and dispersion mechanisms but without aquifer sorption mechanisms requires that a volume of groundwater about twice the volume of the contaminant plume be removed from the aquifer. Biological activity is another method to identify the nitrate after investigating the issue in detail.

CONCLUSION AND RECOMMENDATIONS

Chemical, physical and biological processes in addition to the geological formation and man-made factors influence the hydrochemical variation of groundwater both spatial and temporal in water quality directly and indirectly. Due to the fact that different nuclei have seen on the hydrchemical variation maps within the basin area. Mostly, the concentration EC, TDS and chloride are increasing along the groundwater flow direction. But total hardness and bicarbonate are directly related with the PH conditions of the groundwater and geological formation.

The poor sanitation condition together with the lack of proper waste disposal mechanisms attributed to sever effect of pollution of both surface and ground water resources of the area. The most sever effect of pollution was observed in shallow wells, which is the reflection of all anthropogenic impact on water bodies of the area.

The existing industries should be enforced or advised to treat their wastes and good waste management system should be established in the Dire Dawa town. When industrial establishments are considered budget for the treatment plant should be included (considered) in the main cost of the plant. Develop and implement effective program of solid waste disposal and sanitation system. More solid waste containers with the required truck should be allocated as one alternative for solid waste collection and disposal. To minimize the impact on public health and environment by treating wastes down to an acceptable standard and dispose it.

Waste generated from industries, agricultural activities, households, market centers, institutions, garages, fuel stations and the health centers are the main sources of pollutants that may affect the quality of water in the area. In general, the causes, types and extent of waste pollution range from the wide spread agricultural use of fertilizers to a single incident of an industrial chemical spill.

Since there is a danger of potential pollutant risk of the water bodies of the area, planners and policy makers and other decision making bodies in the region should put clear environmental policy. Rules and regulation should be formulated to control utilization of groundwater, effluent standards and properly follow the implementation of the policy; otherwise future generation will not be out of these problems.

Recommendations

Based on the present investigations the following recommendations are made.

- Artificial recharge should be done to maximize the groundwater potential and Proper groundwater management should be under taken and boreholes must be drilled out side the town to the north direction.
- Establish environmental standards related to chemical management and develop concept of integrated waste management strategy (source reduction, sorting, recycling, incineration and sanitary landfill) and construct sewerage system.
- Remedial measures should in practices i.e. change the surface drainage so that runoff does not cross the source.
- Aquifer cleanup/removal of contaminated water through extraction wells with aquifer advection and dispersion mechanisms, but aquifer sorption mechanisms requires that a volume of groundwater about twice the volume of the contaminant plume be removed from the aquifer. Biological activity is another method to identify the nitrate and to dismantle from the contaminated zone.

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Identification of groundwater recharge conditions in crystalline basement rock aquifers of the southwestern Nigeria

Segun Adelana, Peter Olasehinde and Peter Vrbka

Abstract

Groundwater recharge conditions in the crystalline Basement of southwestern Nigeria are a crucial factor for understanding the groundwater flow regime. It is therefore the overall aim of this research to get adequate information on groundwater isotopic and chemical characteristics with possible origin dynamics and infiltrating conditions. Results of the present study show groundwater isotopic signature range from −19.2 to −11.5 ‰ for $\delta^2 H$ (with mean of −15.7 ‰) and −3.6 to −2.2 ‰ for $\delta^{18}O$ (mean: −3.0 ‰) as against rainwater ($\delta^2 H$: 0.8 ‰, $\delta^{18}O$: −1.0 ‰). A plot of $\delta^2 H$ against $\delta^{18}O$ show that the isotopic composition of groundwater cluster closely around a regression line defined by the equation: $\delta^2 H = 8.6\, \delta^{18}O + 10$. This is in agreement with the meteoric line defined for the northern hemisphere and almost falls perfectly along the Mean Meteoric Water Line except for the different gradient. Most of the water samples are having deuterium excess less than 10. This was interpreted as indicating similar recharge conditions for the groundwater of this area and that mainly from precipitation. The low d-excess in the groundwater of the study area is also indicative of enrichment by evaporation.

Keywords

Southwestern Nigeria, groundwater, recharge, environmental isotopes, basement aquifer.

INTRODUCTION

The present study area is situated in the southern end of Kwara State in southwestern Nigeria comprising of fast growing business and educational centres in Nigeria. The towns included are Offa, Omu-Aran and Osi-Opin (Fig. 1). These towns have been rapidly expanding during the last 15 years as a result of rapid industrialization and expansion of Local Government Authorities. The climate of the study area is characterized by dry and wet seasons. The dry season is short and usually last 3–10 weeks while the wet season starts at about March and extends till October during which the rivers are at their peak. The mean annual rainfall is generally between 1,200 and 1,400 mm while temperature ranges from 25°C to about 35.0°C. Thus the area is a transition between the tropical savannah in the north and rainforest in the south. The nearest available long-term record of precipitation is at the Ilorin International Airport (Fig. 2). The wind pattern over southwestern Nigeria is dominated by the southerly Monsoon, moisture-laden air masses from the Atlantic leading to the development of clouds and, eventually, rainfall for most of the year. During the short dry season the Harmattan coming from the north blow dusty and cold wind across the area.

In the years ahead, southwestern Nigeria is most likely to face a two-fold problem. On the one hand, due to population growth the demand for drinking water supply and irrigation will increase. While on the other hand, due to changing climatic pattern, rainfall will become erratic hence sustainability of water supply for irrigation and domestic use will pose greater challenge under such unpredictable climate. Therefore, sustainable management programmes, alongside with other conservative measures, will not be out of place in order to meet the requirements for adequate water supply. It is therefore the overall aim of this research to get adequate information on groundwater isotopic and chemical characteristics with possible origin dynamics and infiltrating conditions.
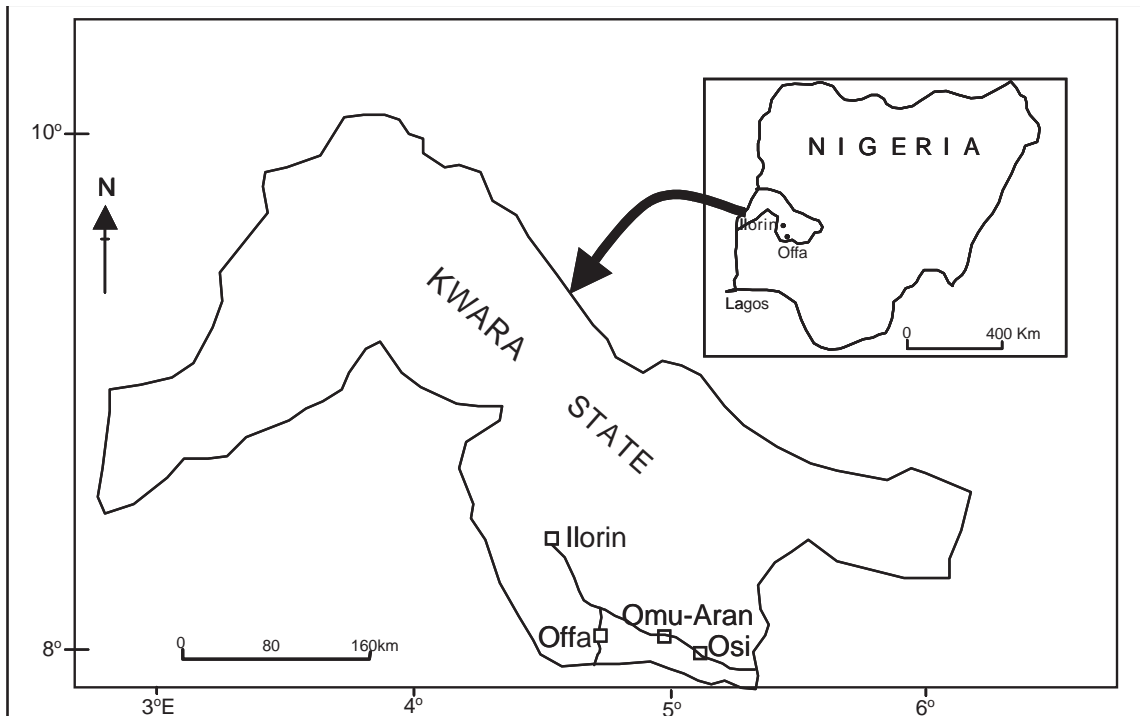
Figure 1. Map of Kwara State in SW Nigeria showing the location of Offa and Ilorin
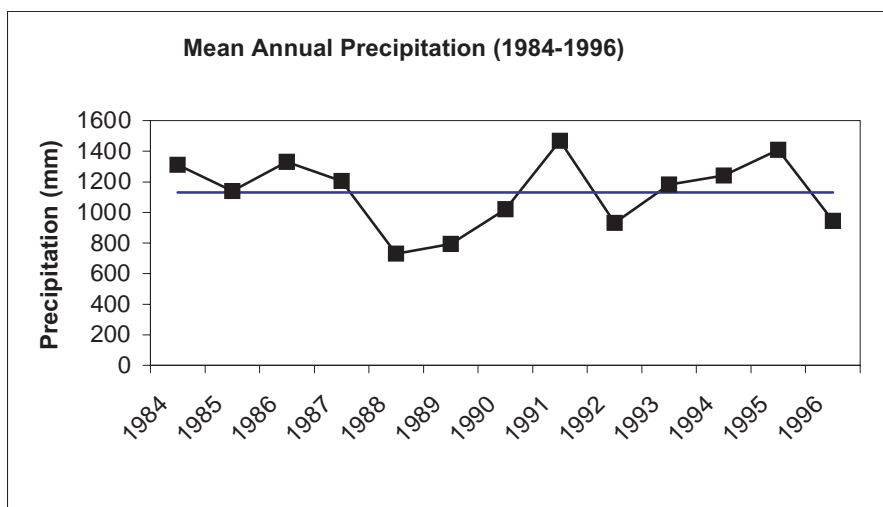
Figure 2. Mean annual precipitation in Ilorin

METHODS

Sampling and analyses of the groundwater took place at different times and were analyzed in different laboratories. All field measurements were made directly at the well site. Physical parameters determined in the field include: Temperature, pH, electrical conductivity (EC), dissolved oxygen, which were all measured using portable (battery-operated) meters and alkalinity determined by titration. As a reference to groundwater of the study area actual precipitation have been sampled from two different stations for chemical as well as isotope analyses. Samples for chemistry were collected in order to attempt a definition of water groups, according to major ion concentrations, and together with field measurements, to determine the geographical distribution of extracted water quality. Chemical analyses were carried out in the following laboratories: Geochemie labor, Geologie institut, Technische

Universitat Darmstadt (Germany) and Chemical laboratory of the Kwara State office of the United Nations Children Education Fund (UNICEF)-assisted Water Projects, Ilorin (Nigeria).

Analyses of the stable isotopes were jointly performed at the Isotope Geochemistry Laboratory, Monash University, Australia and Centre for Environmental Research (UFZ), Leipzig-Halle, Germany. Stable isotopes ratios are expressed as delta in per mil (i.e. δ‰) relative to VSMOW (Vienna Standard Mean Ocean Water). The isotope precision of measurement based on VSMOW is ±0.15‰ for ^{18}O and ±1‰ for ^{2}H.

GEOLOGY AND STRUCTURES

The rocks covering the study area in southwestern Nigeria consist mainly of the Precambrian Basement (Fig. 3). Radiometric dating indicates that the Basement rocks of southwestern Nigeria is polycyclic and has responded to several tectonic events with differing intensities from Archean to late Proterozoic (Pan African). Offa town is underlain by the hard rock suite consisting mainly of biotite gneiss with lenses and bands of biotite schists and pegmatite intrusions. Pegmatites in this area, within the Ilorin map sheet (Sheet 50) area, occur as veins, dykes and irregular bodies that are ubiquitous in all other units. Extensive outcrops of pegmatites which occur as irregular bodies have been mapped 14 Km NE of Offa (Oluyide et al., 1998). Prominent structural trends (faulting and folding) have been observed in southwestern Nigeria (Oyawoye, 1972; Cooray, 1974; Rahaman, 1989). Further information from the
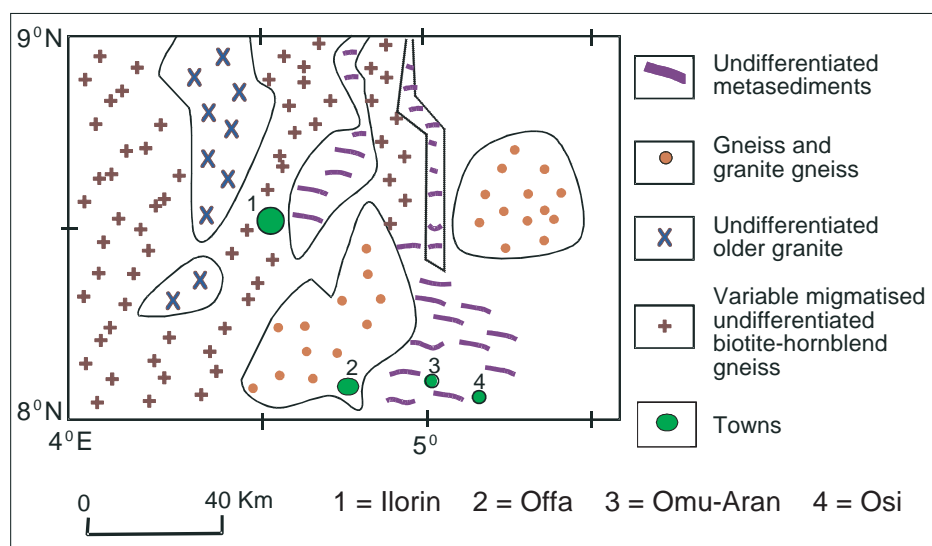
Figure 3. Geological map of the southern end of Kwara State in southwestern Nigeria

available borehole lithological logs (Adelana et al., 2003) revealed that weathering is fairly deep and that the rocks have been jointed and fractured severely. These joints and fractures have, to a large extent, controlled the flow direction of the rivers in the area (Olasehinde, 1979).

GROUNDWATER SYSTEM

The potentiometric map shows a fairly radial flow away from the City centre to a nearby Reservoir and Agun river and subsequently towards River Oyun. The potentiometric map resulting from each year's field survey was very similar, only with slight differences. Therefore only one representative potentiometric map is presented here (Fig. 4). The aquifer has a fairly heterogenous geological structure, being covered by approximately 3–7 m of unsaturated soil composed of a humus layer and a sandy loam overlaying a lateritic and clay layer. The highly permeable

weathered/fractured zone varies locally and is sometimes composed of quartzofeldspathic and amphibolitic materials, especially near Offa Dam. The groundwater in the western part flows in an easterly direction until it meets the
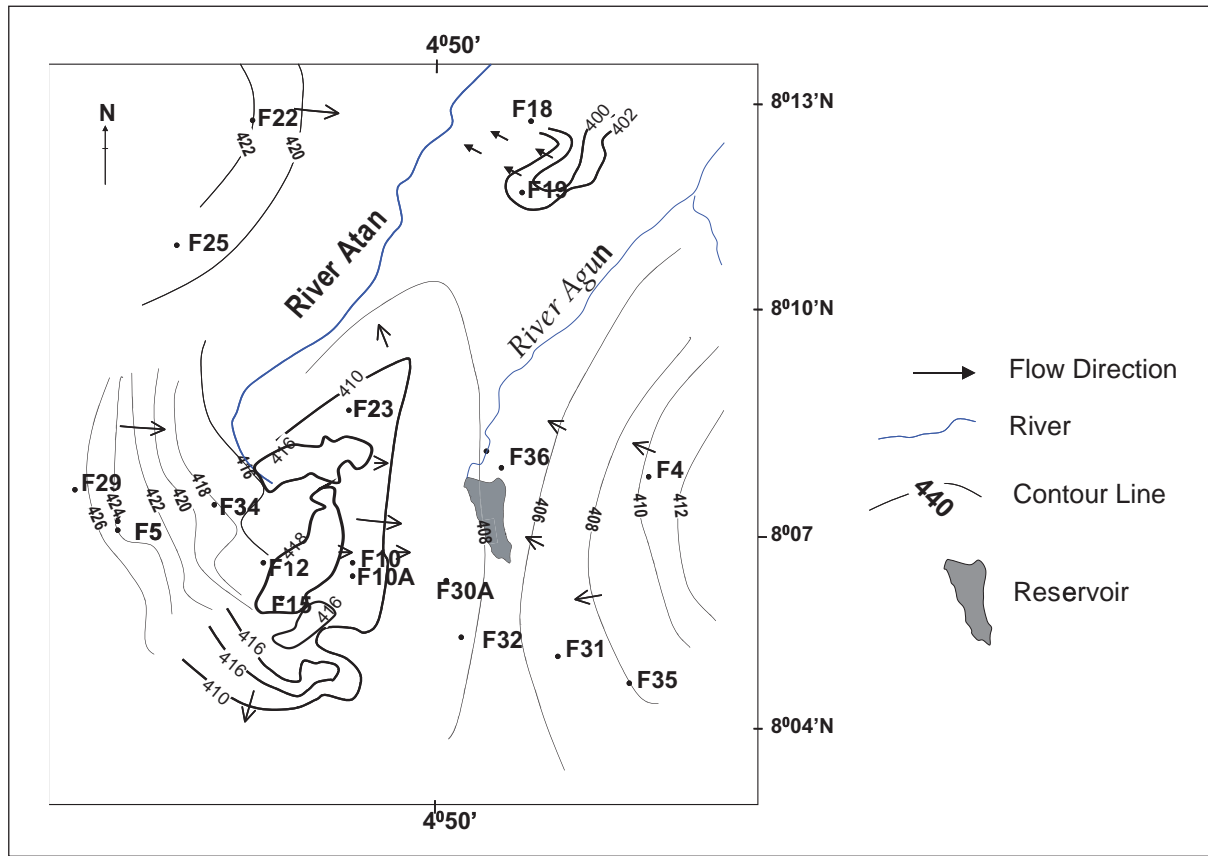
Figure 4. Potentiometric and groundwater flow map in Offa

groundwater recharged in the east and flowing northwestwards towards Agun and Atan rivers. These rivers are tributaries that receive recharge from rainfall, water from surface runoff and contributions from groundwater before flowing to Oyun River; a river that has substansive volume of water throughout the year and has therefore been dammed (at the outskirt of the town) for water supply for both domestic and industrial use in Offa municipal and its environs.

INTERPRETATION OF ISOTOPIC COMPOSITION

Precipitation

In southwestern Nigeria, the average weighted isotopic composition of precipitation in the two stations within the close vicinity of the study area is given in Table 1. The long-term weighted mean of precipitation in Kano, the longest available record in Nigeria (1961–1973), is included for comparison. The isotopic composition

Table 1. Isotopic composition of rain water

Location	$\delta^{18}O$	$\delta^{2}H$	Elevation (m a.m.s.l.)	Average rainfall (mm/a)	EC (µS/cm)	Cl (meq/L)
Offa	−1.00	0.8	366	1,300	34	0.11
Ilorin	3.20	−8.8	305	1,127	117	0.12
Kano	−3.62	−20.17	481	745		

of precipitation clearly indicates the effects of altitude, amount and evaporation. The stable isotope composition of precipitation in Ilorin (45 Km NW of Offa) is more enriched and shows much deviation from that of Offa sampled around the same period and from that of Kano (at a higher altitude and latitude). The only reason is the single heavy rainfall event that produced the sample as compared to the lighter showers sampled in Offa. The samples were collected in early April when sporadic but irregular showers characterize the onset of the rainy season in Ilorin.

Shallow groundwater

The groundwater in the area occurs under unconfined conditions. Most of the dug wells extend into the lithomarge zone. Some of the wells have encountered hard rocks. The depth of wells (hand-dug) ranges from 3 to 12 m. Depth to water level rarely exceeds 24 m; while the mean dry season water level in this region is 12.2 m. Analysis from hand dug well taken from different locations in Offa show static water level between 1.2 and 9.2 during rainy season and average depth of about 4 m – 5 m, which is not lower than the laterite level. The isotopic composition of groundwater from the shallow wells is given in Table 2 and shows a similar history to that of surface water in the area. The majority of the water samples are having deuterium excess less than 10. This was interpreted as indicating similar recharge conditions for the shallow groundwater of this area and that mainly from precipitation. The low deuterium excess in the groundwater of the study area is also indication of enrichment by evaporation.

Table 2. Shallow wells (hand-dug) in Offa
*Surface water from reservoir

Well No.	Well depth (m)	EC (μS/cm)	^{18}O (‰)	^{2}H (‰)	d* (‰)	Cl (meq/L)
FF12	2.50	275.0	–2.16	–11.5	5.78	0.90
FF15	6.00	617.0	–2.38	–13.2	5.84	2.12
FF18	7.25	997.0	–2.67	–11.8	9.56	1.54
FF19	5.10	174.0	–2.72	–15.0	6.76	0.35
FF23	7.65	522.0	–2.70	–15.2	6.40	1.14
FF30A	2.65	854.0	–2.61	–15.3	5.58	2.35
FF32	8.00	125.0	–3.15	–15.0	10.20	0.25
*FF36	–	879.0	3.54	–1.2	–29.52	2.16

Deep wells of the study area

The isotopic composition of groundwater flowing at depth from the east and west towards the valley and the city center is illustrated on the analyses given in Table 3. Some of these wells are tapping from the fractured basement aquifer but their water originates from rainwater infiltration. This is reflected in the isotopic content of the groundwater.

Table 3. Deep wells in Offa

Well No.	Well depth (m)	EC (μS/cm)	^{18}O (‰)	^{2}H (‰)	d* (‰)	Cl (meq/L)
FF4	28.50	218.0	–3.55	–19.2	9.20	0.08
FF5	27.00	193.0	–3.46	–19.1	8.58	0.25
FF10	23.00	928.0	–2.61	–14.6	6.28	1.52
FF10A	38.00	1013.0	–2.30	–15.2	3.20	2.47
FF22	27.00	155.0	–3.24	–18.3	7.62	0.21
FF25	32.60	181.0	–3.48	–15.8	12.04	0.32
FF29	28.00	84.0	–3.13	–16.1	8.94	0.22
FF34	68.20	360.0	–3.60	–17.7	11.10	0.37
FF35	38.00	264.0	–3.44	–18.8	8.72	0.09

The isotopic composition of the wells resembles that of rainfall only that they are slightly more depleted in both stable isotopes of water molecules due to travel times. The isotopic composition does not show evidence of upward leakage through the secondary permeability as claimed by earlier geophysical work. This is evidence from the consistent depletion of water samples from these wells and agrees with the local geology of the area that fractures occur at depth but in most cases not linked to the surface. The chloride concentrations are lower than those of the shallow aquifer water with the exception of FF10 and FF10A. These two wells are production boreholes (located about than 5 metres apart within a private residence) that are reported as heavily polluted with nitrate i.e. NO_3 > 200 mg/L (Adelana, 2004).

SUMMARY OF DISCUSSION

The $\delta\,^2H$–$\delta\,^{18}O$ relationships of all the analyzed samples (Fig. 5) show that the isotopic composition of groundwater cluster closely around a regression line defined by the equation: $\delta\,^2H = 8.6\,\delta\,^{18}O + 10$. This is in agreement with the meteoric line of Dansgaard (1964) for the northern hemisphere and almost falls perfectly along the Mean Meteoric Water Line except for the different gradient. Most of the water samples are having deuterium excess (d-excess) less than 10. This was interpreted as indicating similar recharge conditions for the groundwater of this area and that mainly from precipitation. The low d-excess in the groundwater of the study area is also an indication

*Figure 5. A plot of $\delta\,^2H$ against $\delta\,^{18}O$
for all the analyzed samples in Offa area*

of enrichment by evaporation. Similar isotopic composition was displayed by the groundwater from Omu-Aran/Osi area, except that they are slightly more depleted in $\delta\,^2H$ (Adelana and Olasehinde, 2004).

Water that is ponded at the Reservoir (FF36) has been subjected to some degree of evaporation and is therefore more enriched in $\delta\,^2H$ and $\delta\,^{18}O$. Although the isotopic composition of shallow groundwater close to the Reservoir is less enriched in heavy isotopes it is difficult to established leakage from this standing water because of the prevailing groundwater flow direction indicated by the potentiometric map. There are, however, indications from the isotopic composition of precipitation, shallow and deep groundwater samples that the rivers and reservoir captures the recent recharge from the surrounding shallow aquifer flowing locally from the west and east and from direct infiltration. The salinity and chloride contents of these waters are higher (EC up to 997 µS/cm and Cl up to 2.35 meq/L), which reflects high evaporation rates. There are no indications of groundwater leakage from the deep fractured aquifer upwards indicating fractures are not linked to the surface or covered by the thick weathered zone.

Isotopic and chemical data indicate that groundwater has major contribution from rain in the study area as against base-flow recharge that seems to be predominant in other regions. It must be pointed out, however, that the $\delta\,^2H$ and $\delta\,^{18}O$ depleted waters of the study area do not seem to follow the isotope depletion range defined by Dray et al. (1983) for old as well as recent groundwater in the southern Sahara. The values of Dray suggest that groundwater was recharged during a more humid climatic condition in the past.

CONCLUSIONS

The knowledge of the groundwater infiltration conditions is crucial to quantifying recharge. The alignment of the groundwater samples of the study area with the Global Meteoric Water Line (GMWL) underlines their meteoric

origin. Isotopic data indicate 'modern' precipitation as recharging the area. The δ ^{2}H and δ ^{18}O do not exhibit a wide range but vary from −19.2‰ to 0.8‰ and from −3.6‰ to 3.5 ‰ respectively and cluster around the GMWL.

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Domestic-scale ASR with rainwater at Kingswood, South Australia

Karen Barry and Peter Dillon

Abstract

Rainwater derived from roof surfaces can offer sufficient quantities of good quality water if it can be successfully harvested, particularly in semi-arid areas where alternative sources are scarce. A demonstration project in its second year is currently underway to investigate the operational performance of domestic-scale aquifer storage and recovery (ASR) with rainwater. A shallow alluvial aquifer in the southeast Adelaide metropolitan area was targeted after an earlier regional assessment had suggested this area has potential for ASR. The mean annual rainfall here is approximately 550 mm with most of this falling in winter between May and September. Annual evaporation (A class pan) is about 2,000 mm, most of which occurs in the warm, dry summer months creating a significant irrigation demand.

Two wells, located 5 m apart, were installed to a depth of 24 m. Their ambient salinity of 2,500 mg/L TDS is considered too brackish for irrigation of the clayey soils. The run-off from a 250 m^2 roof area of a single dwelling has been plumbed to the ASR well under gravity feed via a 4 m^3 storage tank and 100 mm filter. Flow rate, volume, piezometric head, electrical conductivity and temperature are monitored continuously in the injection well and the samples of the influent and recovered water analysed periodically. After the first full 12 months of operation, with 563 mm rainfall, 142 KL had been injected with no observable clogging. Although the salinity of the groundwater has been reduced, it is not yet sufficiently fresh to be useful for irrigation supplies due to mixing losses and the effect of regional groundwater flow. Productive use of the groundwater may be possible once greater volumes have been injected, however the supply is limited by available roof area and rainfall variability. The main challenge is therefore to meet operational performance criteria under the current constraints.

Keywords

ASR; rainwater harvesting; salinity; water quality.

INTRODUCTION

In semi-arid areas, rainwater derived from roof surfaces may offer useful quantities of good quality water if it can be successfully harvested and stored. This may have value in developing countries where alternative sources of stormwater are often contaminated. Successful ASR allows water to be stored below ground for recovery at a later date. Following a research project on stormwater ASR at municipal scale in a Tertiary limestone aquifer (Dillon et al., 1997), that led to guidelines for this practice (Dillon and Pavelic, 1996), ASR has been taken up widely in Adelaide where there are now 22 sites recycling 2 million m^3/year of stormwater (Hodgkin, 2004). This is a small but growing practice, which is contributing to the security of Adelaide's water supply (Dillon et al., 2004). An evaluation of a shallow Quaternary system of alluvial aquifers of the Adelaide Plains (Pavelic et al., 1992), revealed that much of the metropolitan area has potential for storage of water on a smaller scale in brackish aquifers. For

example, by recharging domestic roof runoff for use in garden watering in summer to further reduce demand on a stressed water supply system. This is one of the activities being explored in support of improved stormwater management as part of plans by the Patawalonga Catchment Water Management Board (2002) and conforms with the South Australian Government (2004) strategy 'Water Proofing Adelaide'. However, it was recognised that distributed management by householders of many small ASR schemes also has potential for creating problems, through inadequate knowledge on how to operate and maintain collection systems and ASR wells and the potential for groundwater levels to rise to problematic levels and for pollution of groundwater (eg. see Dillon et al., 2005). It was also not clear whether recovered water quality would meet the water quality requirements for its intended uses and whether, if water was recovered from nearby wells, the pathogen attenuation that aquifers provide could be relied on for producing supplies to substitute for some in-house uses. Hence the objectives of this study are, (i) to measure operational performance of domestic scale ASR, determine departures from predicted performance, and identify solutions to any problems that arise, so that in future appropriate advice can be given when licensing such installations, (ii) to provide a rainwater ASR comparison for an intended operational treated stormwater runoff ASR site in shallow alluvium nearby, and (iii) to provide a site to determine whether recovered water quality from a nearby well meets all drinking water criteria, after the plume of freshwater reaches that well.

SITE DESCRIPTION AND OPERATION

In June 2003 a Domestic Scale Rainwater ASR Demonstration site was established in the rear garden of a residential dwelling in Kingswood, South Australia. The suburb of Kingswood is located in the base of the foothills 6 km south east of the central business district of Adelaide. In 2002 two wells (100 and 125 mm diameter) located 5 m apart, were installed to a depth of 24 m in a Quaternary alluvial aquifer. The profile is largely made up of clay containing several layers of sand and gravel to 21 m depth and underlain by more clay. Results from 'pumping-tests' showed their well yields to be 1.1 L/s (100 mm, northern) and 0.3 L/s (125 mm, southern) with an ambient groundwater salinity of 2,500 mg/L TDS, and depth to ambient groundwater level of 12 m. From these results it was decided to install the injection line in the 100 mm (northern) well with the higher hydraulic conductivity and hence greater capacity to receive and recover injected water. The run-off from a single residential dwelling of 250 m^2 was then plumbed to this ASR well under gravity feed via a 4 m^3 tank (with 3 m^3 active storage) and 100 mm filter. Two low-head flow meters were installed to monitor inflows and outflows to and from the ASR well. Groundwater levels have been monitored using a combination of pressure transducers and capacitance probes, with a backup of regular manual measurements. In July 2003 a YSI© water quality sonde was installed in the ASR well to continuously monitor the electrical conductivity and temperature. Sampling for basic water quality parameters for each well was done in December 2002, prior to the start of any injection to the well. Periodic samples of the roof run-off have also been collected before, during and after injection events. Since the start of injection there have been on average 0.6 KL recovered per month to purge the well of suspended solids and to assess the quality of the recovered water.

During the first year of operation (July 03 to June 04) the injection line was installed in the 100 mm diameter well. On the 22 July 2004 following a second set of pumping tests the injection line was moved to the lower yielding well (125 mm) with the aim of increasing the volume of fresh water that could be recovered for irrigation.

RESULTS AND DISCUSSION

This report presents data from the first 18 months of the study, July 2003 – December 2004. Table 1 shows the monthly volumes of injection and recovery in each well. Figure 1 compares the household water consumption of mains water with net volume recharged as an indicator of the relative potential contribution of harvested water to household water demand. Over the 18 month period a total volume of 235 KL was injected and 18 KL was recovered, leaving a net increase in storage of 217 KL.

Table 1. Monthly water volumes

	Injection Volume (KL)	Recovered Volume (KL)	Rainfall (mm)	Mains (KL)
July	11.56	1.12	42.50	20.39
August	22.38	0.46	85.30	22.29
September	15.03	0.58	59.40	20.29
October	12.87	0.29	62.40	19.51
November	1.66	0.57	7.40	29.62
December	7.31	0	35.20	38.50
January	3.24	0	15.00	40.77
February	1.25	0	5.10	54.32
March	7.80	0.66	29.30	33.77
April	2.86	0.58	18.70	23.26
May	21.94	0.58	74.60	24.39
June	34.60	0.58	128.20	26.67
July*	23.51	9.20	101.60	33.29
August	25.18	1.48	90.00	21.13
September	18.89	0.86	65.60	18.93
October	1.06	0.63	16.20	22.76
November	19.18	0.60	74.70	22.76
December	4.04	0.60	17.40	36.57
Total Volumes July 03 -Dec 04	234.3	18.8	928.6	509.2
Volumes 100 mm well	154.4	12.0	606.0	376.0
Volumes 125 mm well	79.9	6.8	322.6	133.2

* Conducted 'pump-tests' on both wells 21/22 July 04 and moved pump from 100 mm to 125 mm well.

Recovery consisted mostly of intermittently purging the well to remove any accumulated particulate matter. This water was discharged into a nearby sump and soaked into the ground as additional recharge to shallower alluvium, along with a small amount of runoff from paved areas adjacent the house. The salinity of recovered water in this first year of operation was too high to use the banked water for garden irrigation in summer. In the first year the overall ratio of injected water to mains water usage was 40%. It was observed that during the wet winter months injection rates approximated mains usage rates, however in the drier summer months, when water demand peaks, there is little runoff, showing the value of having substantial storage to meet summer demand (Figure 1).
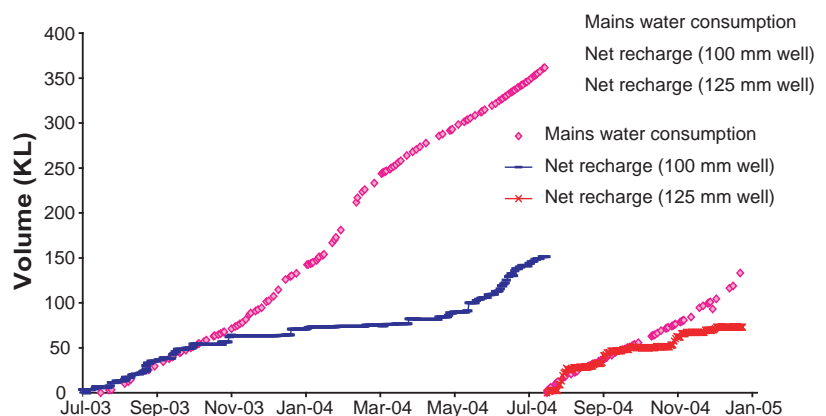
Figure 1. Net volume of recharge and mains water consumption

Daily standing water levels have been monitored since July 2002 using a combination of loggers and manual measurements (Figure 2). During injection periods it has been observed that the head only rose by about 1 metre
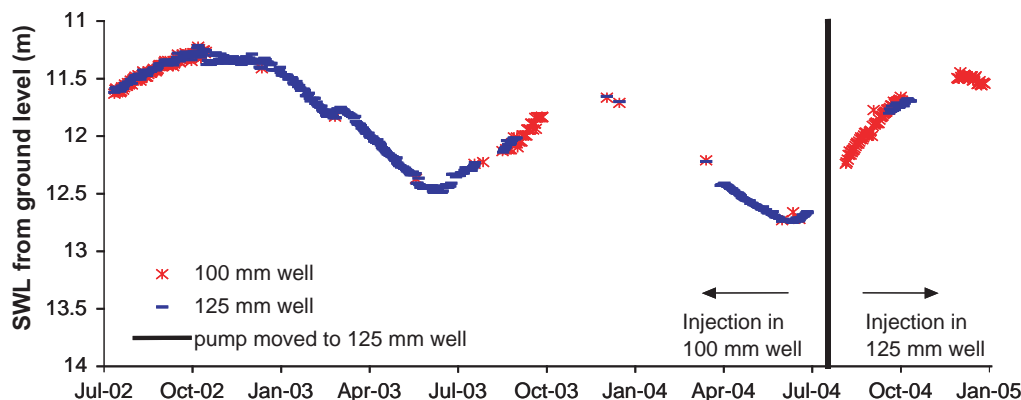
Figure 2. Monitored daily standing water levels from July 2002 to January 2005

(with injection rates typically at 0.3 L/s). Drawdown during recovery events has been logged with the pressure transducers during pumping tests. Purging events in the 100 mm well, were run at up to 1 L/s for 10 minutes, with the well never running dry. Pumping at the same rate in the lower yielding well (125 mm) drawdown rapidly reached the level of the pump at 18 m. Recovery events in the 2nd year of operation are now run at a lower average rate of 0.3 L/s for 35 minutes, including intermittent short periods of pumping at 1 L/s. The annual groundwater level fluctuation is approximately 1 m, with the peak in December and the lowest levels in June, approximately a 3–4 month lag on seasonal rainfall fluctuations. In spite of rainfall being near average, surprisingly there is a downward trend in groundwater levels that eclipses the effect of the increased recharge from the ASR system. Both wells record the same level when there is no recharge or recovery.

Electrical conductivity (EC) and temperature have been continuously logged in the ASR well for 18 months, using a YSI water quality monitoring sonde (Figure 3). During injection events in the first year of operation, the ambient EC (>4000 uS/cm) was diluted with fresh rainwater (<100 uS/cm). With regular events in winter the EC in the well was maintained below 1,000 uS/cm. However once injection events become less frequent, as in summer and during pumping, the EC of water in the ASR well increased rapidly towards ambient conditions. Water recovered from the well also showed a similar pattern, with salinity increasing during recovery. Insufficient freshwater has accumulated around the well in this first year to create a buffer against the saline ambient groundwater. The temperature of ambient groundwater is 18°C, and during winter rainwater can be less than 10°C and in summer up to 25°C. There is sufficient recharge in winter to depress the average temperature of ASR well water by 5°C. Temperature returns to equilibrium more quickly than salinity due to the thermal mass of aquifer material. For the second year of operation
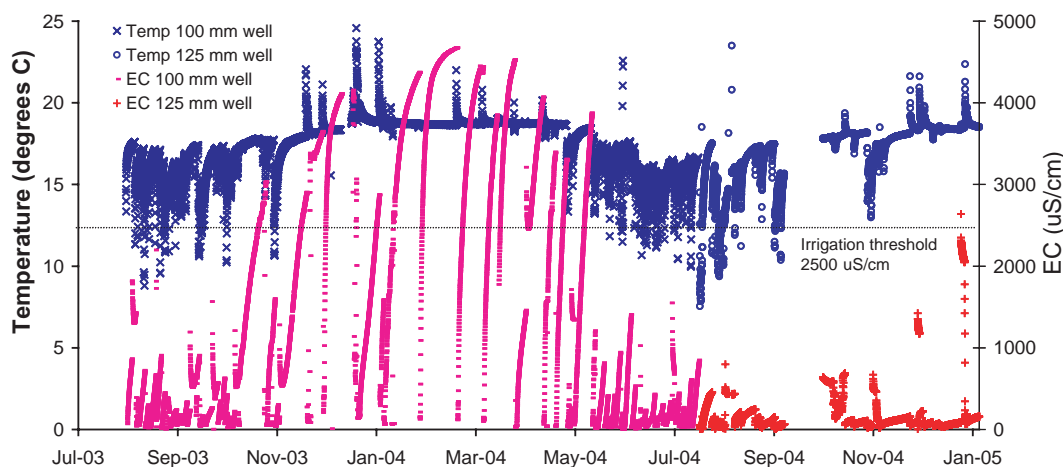
Figure 3. Electrical conductivity and temperature measurements in the ASR well

the injection line was installed in the 125 mm well, though recovery rates were slower, the plume of fresh water around the well has still risen to the irrigation threshold of 2,500 uS/cm (Figure 3) during purging events following periods of low injection.

Conductivity and temperature profiles have been carried out periodically through the trial using a YSI profiling sonde. In the first year of operation, the electrical conductivity and temperature profiles were measured 3 times in the initial observation well (125 mm) to detect any possible breakthrough and once in the ASR well (100 mm) when the pump was temporarily removed. The profiles show distinct stratification within the 24 m depth of each well with the fresher water overlying saltier water. This may explain why the salinity of recovered water increases so quickly. It is possible that the low salinity at the watertable in the observation well in September 2003 was due to injectant spreading laterally above the saline ambient water, however a pre-existing salinity gradient due to the presence of the nearby stormwater drainage sump cannot be ruled out. Since injection has occurred in the 125 mm well two sets of profiles have been carried out in each well, one near the end of winter and one mid summer. Here it is observed, that the water remains fresh (<500 uS/cm) above a depth of 20 m and below this salinity increase rapidly, reaching ambient values (>4,000 uS/cm) at the base of the well (24 m). It is possible that by limiting the ASR well's depth to 20 m, an improvement in recovery efficiencies could be observed. This will be assessed by performing down hole flow meter profiling, to establish the hydraulic conductivity profile of each well and allow development of strategies to enhance and maintain recovery efficiencies.

Detailed water quality analyses in groundwater have not extended beyond the initial sampling in December 2002 from each bore before the start of injection and grab samples collected from the rainwater tank since injection commenced on 29 June 2003. Table 2 assembles these data. The ambient groundwater is saline with low organic carbon and phosphorus but has significant nitrate. The rainwater is fresh with minimal nutrient levels but contained detectable zinc levels, from a galvanized steel roof. The leaf trap sample was taken in May 2003 from a roof gutter where the down-pipe leaf guard had become choked with decaying leaves and water had been sitting in the gutter for some time. This sample contained coloured water with a musty odour and is taken to be a worst case for the potential for nutrient loading in the rainwater available for injection. A siphonic pipe system results in accumulation of detritus in the pipes so that rainwater discharging to the tank is sometimes depleted in oxygen.

CONCLUSIONS

Operation and maintenance inputs required so far have been at a level beyond that which could reasonably be expected of normal householders. This is not unexpected for a first trial, and demonstrates the value of running trials to iron out problems before such methods are permitted, advocated or promoted.

No clogging has been observed in the well, with the current regime of water quality management and with pumping out approximately 0.6 KL each month during months with moderate recharge. Initially quite turbid water is recovered on each cycle and discharged onto the garden or into the sump. This suggests that purging has real value in helping to prevent clogging. Salinity of recovered water has increased quickly beyond values acceptable for irrigation supplies in the first year of operation. The decision in July 2004 to assess the use of the 125 mm well, with its lower hydraulic conductivity as the ASR well, has to date resulted in fresher water on recovery and has not been restricted by the volume injected from rainfall events. The EC profile data suggests that density stratification could contribute to the low recovery efficiency.

The trial is intended to run for three years to enable the objectives to be met. Results to date suggest some changes are required in order to achieve a viable ASR system. In the next 12 months it is proposed to obtain additional information and operational experience to define requirements for a viable system.

Table 2. Water quality data

Parameters (mg/L)	100 mm Well 12/12/02	125 mm Well 12/12/02	Leaf trap 19/05/03	RainwaterTank			
				21/05/03	17/08/0	1/10/03	12/11/04
EC (uS/cm)	4,005	4,150		42.3	21.5	28.2	23
pH-field	6.64	6.63		7.82			5.8
Temp-field (°C)	19.6	19.8		16.7			
DO-field	2.17	2.31		4.32			
Eh-field (mV SHE)	365	372		609			
Turbidity (NTU)				2.12	0.75	1.05	1.2
Total Organic Carbon	2.0	2.6	694	<1.0	2.2	0.8	3.0
Calcium	151	149	57	1.1	0.7	0.4	0.4
Magnesium	173	165	12.4	0.5	0.4	0.4	0.3
Potassium	12.1	12.4	13.1	<1	<1	<1	<0.1
Sodium	503	530	14	2.9	1.8	2.3	2.8
Bicarbonate	675	717	288	7	7	6	
Chloride	990	1,000	20	7	9	8	4.2
Sulphate as S	140	140	33.1	<1.5	<1.5	<1.5	0.1[a]
Ammonia as N	0.027	0.022	30.1	0.116	0.347	0.053	
Phosphorus - Total as P	<0.025	<0.025	39.7	0.018	0.026	0.047	<0.1
Kjeldahl Nitrogen as N	<0.25	<0.25	275	0.17	0.39	0.2	
Nitrate + Nitrite as N	5.82	5.87	0.006	0.202	0.389	0.075	
Total Nitrogen							0.8
Total Arsenic	<0.001	<0.001	0.066	<0.001		<0.001	
Total Aluminium							<0.1
Boron							<0.1
Total Copper			0.162	<0.001		0.002	<0.02
Total Iron	<0.03	<0.03	148	<0.03	<0.03	<.03	<0.1
Total Manganese							<0.05
Total Lead			3.361	<0.0005		0.0006	
Total Strontium	<0.003	<0.003					<0.1
Total Zinc			16.99	0.138	0.248	0.124	0.08
E.coli (/100 mL)	0	0		2,400		7	
Coliforms (/mL)	680	520		>2,400		>2,400	
Faecal Coliforms (/100mL)						7	
Colony Count (20°C) Aerobic (/mL)				>10,000		>10,000	

a. As S only.

ACKNOWLEDGEMENTS

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Improvements in wastewater quality from soil and aquifer passage using infiltration galleries: case study in Western Australia

Elise B. Bekele, Simon Toze, Jeannette Rümmler,
Jonathan Hanna, Palenque Blair and Nick Turner

Abstract

There is a growing interest in using reclaimed water from managed aquifer recharge (MAR) to reduce pressure on potable water supplies. In Perth, Western Australia, covered infiltration galleries are currently being trialled to compare the performance of a MAR system for enhancing the percolation of treated effluent into different aquifers, one karstic limestone and the other unconfined sand. The galleries are located at a maximum depth of 1 m at the Floreat site in sand and at the Mandurah site in limestone: one trench contains 10 mm-diameter gravel, while the other contains the Atlantis Leach System®. The research aim is to quantify the attenuation of microbial pathogens, nutrients and chemicals of concern in the subsurface. Water quality changes will be investigated at the Floreat site within a forced-gradient setting created by pumping continuously from an extraction bore. The water table is about 9.5 m below the base of the galleries and several m below the transition from calcareous sands of the Spearwood Dune System to an aeolianite (the Tamala Limestone). Several research activities, including infiltration tests, hydraulic property estimation from sediment analyses and MODFLOW-MODPATH simulations, have led to insights into the design criteria for managing residence times in the aquifer to enable water quality improvements to occur.

Keywords

Artificial recharge, covered infiltration galleries, managed aquifer recharge.

INTRODUCTION

Background

There is a major impetus to develop water reuse in the Perth region of Western Australia to reduce pressure on potable water supplies for different purposes (e.g. green-space irrigation, horticulture and industrial usage). Perth is semi-arid with hot dry summers and rainfall occurring mainly during the winter, and there has been a sustained decrease in winter rainfall in the southwest of Western Australia in recent decades (IOCI, 2002). The declines in rainfall and streamflows have prompted support for the efficient use of water and research and development projects to develop reclaimed water as a resource. In the Perth where land values are at a premium, pilot-scale, covered infiltration galleries are being tested in hydrogeological settings that are representative of anticipated full-scale projects. The sites are located within sedimentary deposits of the Swan Coastal Plain, which consist mainly of dune sands, carbonate cemented sands or limestone, and podsolized soils with varying amounts of iron and organic matter (McArthur and Bettenay, 1974, Figure 1a).

Infiltration galleries at the Halls Head Wastewater Treatment Plant in Mandurah and at an experimental field at the CSIRO-CELS lab in Floreat are among the first research sites for managed aquifer recharge (MAR) in Perth. The

Halls Head galleries were recently commissioned; thus, the focus of this paper is on research at the Floreat site. Treated effluent enters a subsurface concrete chamber, which connects to slotted PVC pipes in each gallery or infiltration trench (Figure 1b). The galleries are 0.5 to 1 m below ground and approximately 2 m and 9.5 m above the water table at the Mandurah and Floreat sites, respectively. A forced-gradient setting will be established by pumping continuously from a recovery bore at the Floreat site. In one of the galleries, the PVC pipe is surrounded by 10 mm graded and washed gravel, whereas in the second gallery, the Atlantis Leach System® is used, consisting of a modular series of lightweight polypropylene tanks to compare different designs. At both sites, secondary treated wastewater will receive further treatment from an Amiad Filtration System®, prior to recharging the aquifer via the infiltration galleries. The Amiad® system consists of a 200 micron stainless steel filter and a dual bed (anthracite and fine sand) filter.
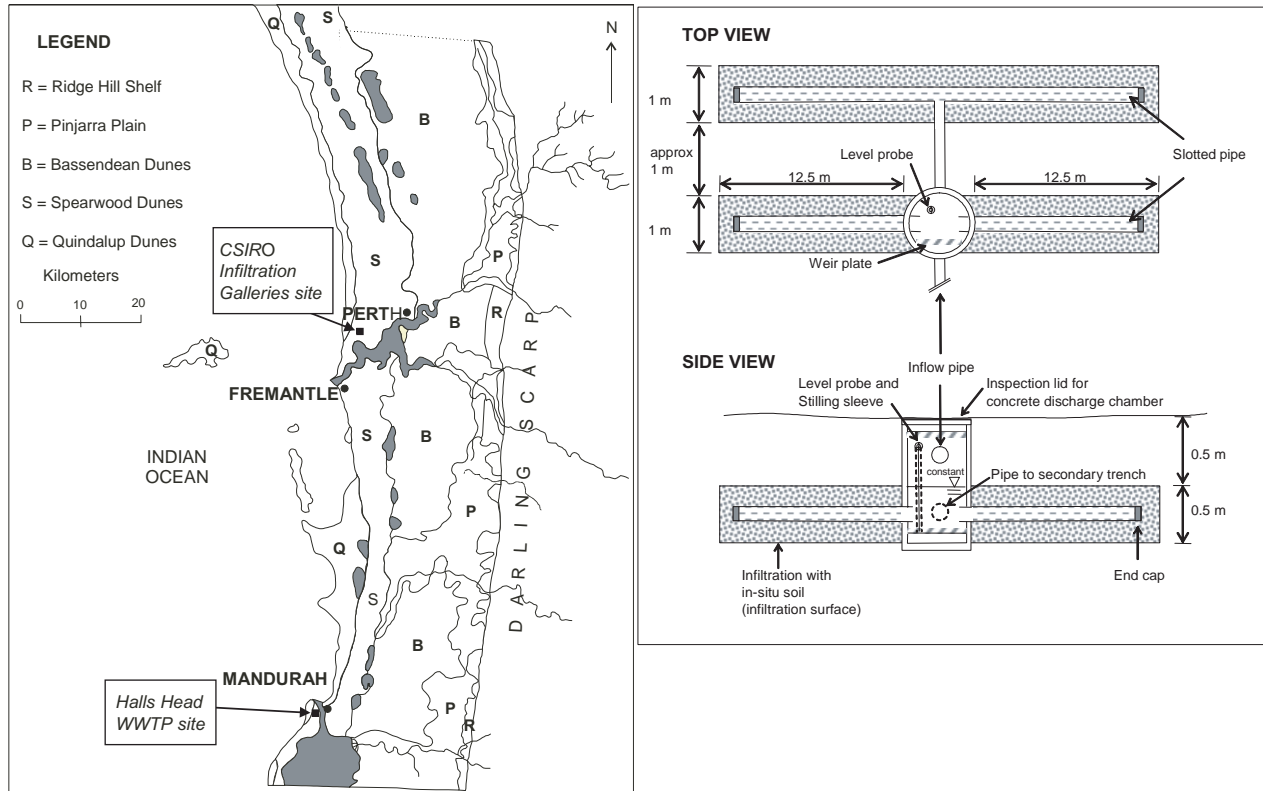
Figure 1. a) Location map showing the geomorphology and soils on the Swan Coastal Plain (after McArthur and Bettenay, 1974) and field sites, b) design features of the covered infiltration galleries

Water quality changes from soil-aquifer treatment

There is a broad range of criteria for evaluating the quality of reclaimed water. The most immediate risks to human health and the environment are pathogenic microorganisms, nutrients, organic compounds, and adverse effects from low-level, chronic exposure to compounds such as disinfection by-products, endocrine-disruptors, and other manufactured chemicals, such as pesticide, insecticides, industrial waste products, and pharmaceuticals. Field and lab research on water quality improvements from artificial recharge reveal the potential to remove viruses and parasitic protozoa (Schijven et al., 2003) due to adsorption processes and the activity of indigenous groundwater microorganisms (Gordon and Toze, 2003). Other studies have examined the fate of organic matter (Rauch and Drewes, 2004), selected pharmaceuticals (Kreuzinger et al., 2004) and endocrine disruptors (Mansell et al., 2004) during groundwater recharge.

Research goals and scope

Guidelines for wastewater reuse in Western Australia are currently being developed to clarify the water quality criteria and treatment levels for different intended uses. One of the objectives of this study is to assist in developing the required monitoring and aquifer assessments for recharge projects. A major part of this is to quantify the attenuation of pathogenic microorganisms, nutrients and specific chemicals of concern, particularly for non-potable reuse of reclaimed water. A second objective is to monitor gallery performance and flow rates for the two types of gallery designs.

Attenuation of microbial pathogens during MAR may occur due to microbial processes and adsorption to aquifer materials in both the unsaturated zone and the aquifer. One of the research aims is to quantify microbial pathogen and trace organic attenuation during transport through the unsaturated zone and through the aquifer by microbial processes and adsorption to aquifer materials. Lysimeters will be used to collect samples for detecting water quality changes near the infiltration galleries, while a series of piezometers between the galleries and an extraction bore will be used to monitor water quality changes in the aquifer once the infiltration galleries are fully operational. Tracer experiments will be conducted using bacteriophage as surrogates for human enteric viruses and a conservative tracer (e.g. bromide) to study the fate and transport of pathogens in the aquifer. The extent of pathogen removal due to the activity of indigenous microorganisms versus adsorption will be investigated using flow-through chambers installed in selected piezometers. The chambers will allow contact between selected pathogenic microbes (mainly enteric viruses) and groundwater microorganisms, but prevent contact and adsorption with aquifer material, thus enabling a comparison of removal processes in-situ.

At the Floreat site, preliminary work has involved characterising infiltration rates, hydraulic properties of the Spearwood Sand and upper part of the Tamala Limestone, and the groundwater chemistry. The next stage will be to extend the characterisation of hydraulic properties to the base of the aquifer. Flow metering and step-drawdown aquifer testing are being considered to investigate variability in aquifer properties. Additional knowledge of the aquifer is needed to constrain a MODFLOW model, which has been developed to aid decision-making regarding the placement of monitoring bores, screened intervals, and the pumping rate for the extraction bore.

METHODS

Infiltration testing

Infiltration rates in the Spearwood Sand using treated effluent were measured using a double-ring infiltrometer (concentric cylinders). The basic design of the infiltrometer is similar to the apparatus described by Reynolds et al. (2002), with a probe and recorder added to measure water levels at small time intervals in the internal cylinder. The procedure involved ponding a constant depth of water (up to 35 cm) inside the measuring cylinder and measuring the rate at which water infiltrated into the sand, while a constant supply of water was applied to the surrounding buffer cylinder to reduce lateral capillary forces on the infiltrating water. An average infiltration rate was calculated by repeating of the experiment several times where the water was lowered by about 1 cm and the final rate of complete infiltration of the ponded water. Effluent discharge to the infiltrometer was metered to obtain another estimate of the infiltration rate.

Preliminary hydrogeology characterisation

Three exploratory boreholes were made at the field site by auger drilling with samples collected from the aquifer by wireline coring. Sediment core samples were analysed to determine hydraulic properties, while groundwater samples were analysed to acquire background data on the water quality in the aquifer. Bore logs were obtained to a

maximum depth of 12.75 m below the ground surface. The water table is at 10.5 m depth. The thickness of the Spearwood Sand is variable across the Swan Coastal Plain with increasing carbonate cement and the Tamala Limestone with depth. There was variability detected in the thickness of Spearwood Sand at the three boreholes that were drilled at distances of 19, 24 and 27 m apart. The base of the superficial aquifer where it overlies shale and silt-stone of the Kings Park Formation is about 30 m below the ground surface. Sediment core samples were analysed for bulk density, volumetric water content, total porosity, and grain-size distribution as described in Rümmler et al. (2005). Saturated hydraulic conductivity of core samples was calculated using the grain size data and the Beyer and Hazen formulas (Rümmler et al., 2005).

MODFLOW-MODPATH simulations to aid MAR design

One of the principal concerns with operating a managed aquifer recharge scheme is ensuring adequate residence time within the aquifer for water quality improvements to occur. It is also necessary to avoid pumping excessively from the recovery bore to avoid large drawdown of the water table and production of an inordinate amount of water. Groundwater simulations were developed using MODFLOW linked to a particle tracking code (MODPATH) for predicting travel times in the aquifer based on advection.

The initial aim of modelling was to determine the criteria need to achieve a minimum aquifer residence time of 70 days. 70 days is the initial estimate of residence time necessary for the removal of viral pathogens. The main controls on the residence time are the hydraulic conductivity of the aquifer, and the pumping rate and screen length in the recovery bore. It is anticipated that the infiltration galleries will receive approximately 25 KL/day of treated effluent. At the Floreat site, the separation distance between the infiltration galleries and the recovery bore is 50 m. This distance was decided based on land availability and sensitivity modelling with a range of hydraulic conduc-tivity values. The model was also used to investigate any interference from an irrigation bore nearby. Another purpose of the model will be to assist with locating screened intervals in the monitoring wells to intersect possible preferred flow paths; however, hydraulic properties at greater depths below the water table are being obtained to properly constrain groundwater simulations.

RESULTS AND DISCUSSION

The average infiltration rate obtained from seven repeats with the infiltrometer was 2.71 L/min, which compares favourably to the rate obtained based on complete infiltration of the water in the infiltration meter (1.58 L/min) and the total volume of effluent infiltrated over the course of the experiments (2.12 L/min). There was only slight clog-ging detected by a small increase in water levels, which was controlled by altering the flow rate. Despite the slow rate of water level rise, the water level dropped rapidly when inflow ceased or reduced significantly, indicating that the infiltration capacity is fairly high in the Spearwood Sand.

The bore logs and sediment analyses performed on cores from the three exploratory boreholes reveal consistency within approximately the upper 740 cm of the unsaturated zone. This upper section is predominantly medium-grained quartz sand with an increasing amount of fine-grained, limey sand from the surface to about 600–750 cm depth (Figure 2). There were difficulties encountered while coring through the limestone between about 785 and 1,125 cm, which prevented sampling entirely from this section. Depth profiles of bulk density, porosity and water content show greater variability below about 740 cm, particularly in relation to increasing amounts of cemented sand or limestone. The bulk density of the sands is generally between 1.4 and 1.8 g/cm^3 within the upper 750 cm, while porosity is between 0.3 and 0.47. Estimates of saturated hydraulic conductivity decline steadily from about 20 m/day near the surface to 5 m/day at a depth of 825 cm, but estimates for deeper cemented sands that were sparsely sampled showed greater variability (3–35 m/day).

Groundwater flow and transport simulations for the MAR site in Floreat are ongoing, pending further knowledge of variability in hydraulic properties of the Tamala limestone. Assuming the hydraulic conductivity is 40 m/day, the pumping rate must be at least 150 m^3/day in the recovery bore with a 3 m screen to capture the plume of infiltrated water and the predicted aquifer residence time is about 110 days. A longer well screen in the pumping bore (6 m) will require a higher pumping rate (160 m^3/day) to achieve similar results. If the aquifer hydraulic conductivity is higher (~100 m/day), then the screen length in the pumping bore below the water table should be longer (25 m) to capture the plume of infiltrated water and the minimum required pumping rate should be increased (400 m^3/day) to achieve the minimum residence time of 70 days. The maximum drawdown near the pumping well generated by these scenarios is less than about 0.5 m.
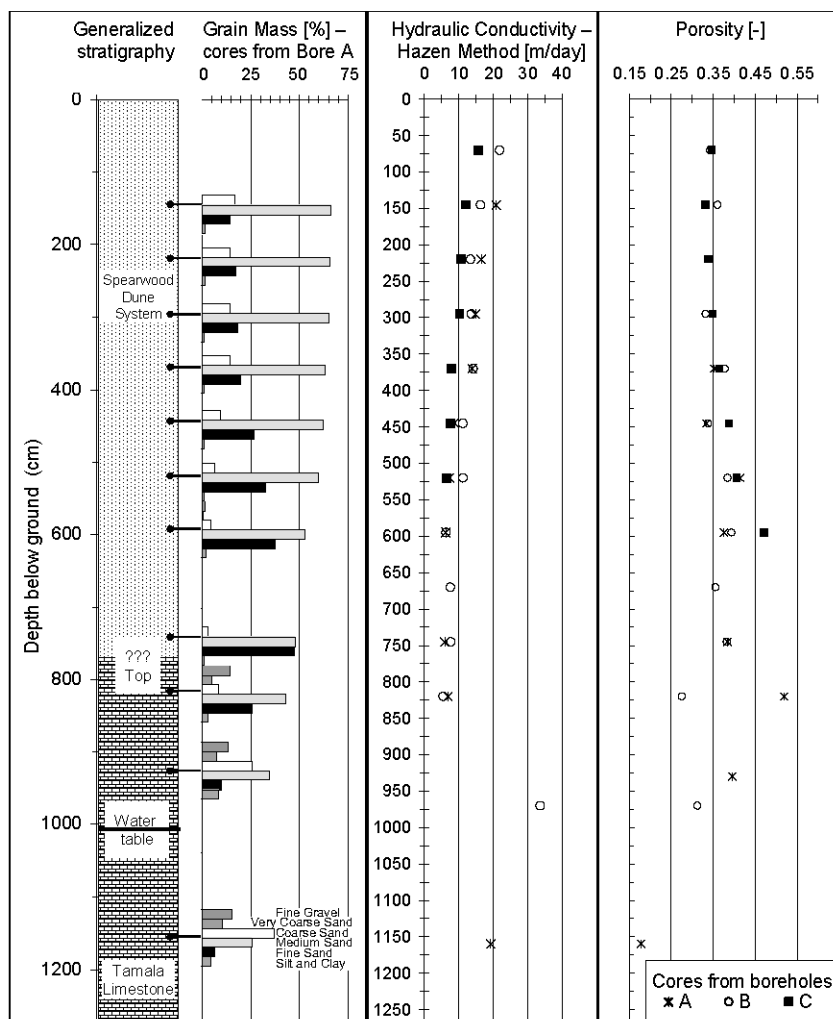
Figure 2. General stratigraphy for the Floreat site showing the results from grain-size analysis for borehole A cores, and hydraulic conductivity and porosity from boreholes A, B and C

CONCLUSIONS

This case study provides an overview of current research to underpin the use of covered infiltration galleries and MAR in Western Australia. At the Floreat site, infiltration tests and sediment analyses reveal that the there is fairly high infiltration capacity in the unsaturated zone, which is mainly medium-grained sand with increasing fines to a depth of 740 cm and an accompanying decrease in hydraulic conductivity from about 20 to 5 m/day. Below about 740 cm, there is a greater proportion of cemented sand grading into the Tamala limestone and greater variability in aquifer properties. Additional work is needed to characterise hydraulic properties at greater depth in the aquifer

where the plume of infiltrated water is likely to migrate. Future work will include installation of a bore through the full ~30 m from the ground surface to the base of the aquifer, a step-drawdown pump test and possibly flow metering to investigate vertical variability in hydraulic conductivity, and tracer tests. This work is part of an ongoing effort to understand the hydrogeological setting and to determine aquifer residence times required for microbial pathogen attenuation.

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Underground infiltration system for treatment of stormwater from metal roofs - Investigation on laboratory rigs

P. Göbel, C. Dierkes, W. Benze, W.G. Coldewey

Abstract

A new pollution control pit was developed with a hydrodynamic separator and a multi-stage filter. The pit is connected to a concrete infiltration pipe with a partly sealed base. The underground infiltration system acts as a pre-treatment device containing a specially designed filter. The treatment process is based on sedimentation, filtration, adsorption and chemical precipitation. Sediments are trapped in a special chamber within the pit and can be removed easily. Other pollutants are captured in the filter upstream of the sediment separation chamber. Filters can be easily replaced. Filters have been adapted to treat polluted stormwater loads from metal roofs. Investigation on laboratory rigs indicates that more than 65% of heavy metals can be removed. Soils and groundwater are effectively protected from heavy metal contamination.

Keywords

Heavy metals; metal roofs; stormwater infiltration; stormwater treatment.

INTRODUCTION

Source control by on-site retention and infiltration of stormwater is used in many countries as a sustainable alternative to classical drainage methods. Infiltration reduces the impact of sewer systems on receiving waters, allows installation of storm-sewers with smaller diameters and helps return the urban water cycle to its pre-urbanized state (Göbel et al. 2004). In recent years, however, it has become apparent that sediments and pollutants from drained surfaces cause clogging and endanger soil and groundwater during long-term operation of devices (Zimmermann et al. 2005). German water authorities recommend the use of infiltration devices such as swales or swale-trench systems. Infiltration by underground facilities such as pipes, trenches or sinks without pre-treatment of runoff is generally not permitted, especially for runoff from metal roofs, traffic areas and industrial sites. The decreasing size of private allotments and restrictions on land use often prevents the use of space-consuming swale infiltrations. Small decentralised stormwater treatment facilities can be an effective solution to this problem, provided they have the capacity to remove important pollutants. Dierkes et al. (2005) describe the development of a new pollution control pit, which acts as a pre-treatment device containing a specially designed filter. The pit is connected to a concrete infiltration pipe with a partly sealed base.

The aim of the study reported in this paper was to develop an optimal multistage filter and to identify the best materials to use in the filter for removing heavy metals in different types of runoff. The filter needs to be optimized to achieve both high pollution retention capacity and a satisfactory flow-rate. Solid and dissolved heavy metals need to be removed from runoff by filtration, adsorption (ion exchange) and chemical precipitation. The study investigated the mobility of heavy metals and the impact on soil and groundwater.

The research was undertaken using runoff from metal roofs. While roofs differ in material, age, slope, exposure and location, a reasonable classification for drainage purposes can only be developed based on material (Bannermann 1994). Most roofs in Germany consist of tiles and concrete. Metal roofs of copper and zinc are found in city centres (Gromaire et al. 2001). Runoff from these roofs cannot be infiltrated in some states of Germany. Besides material,
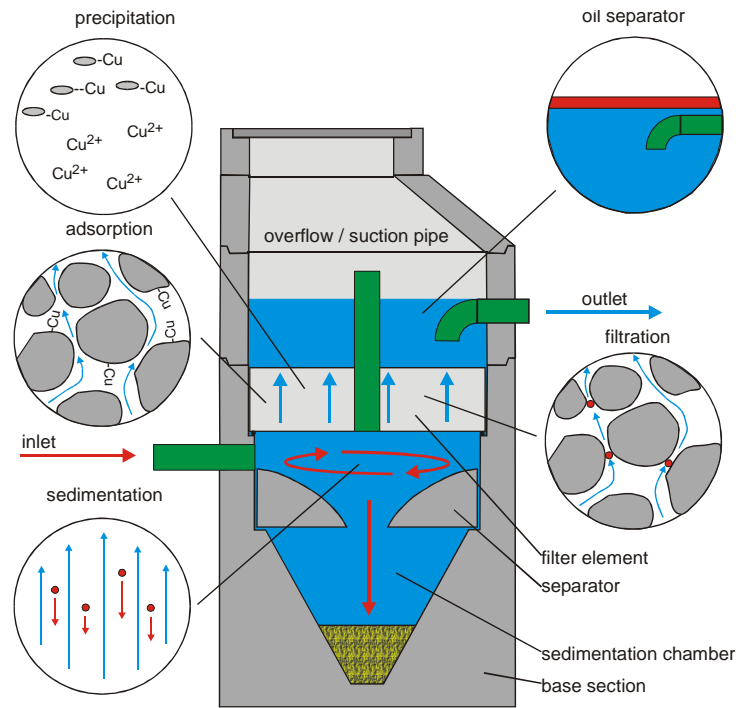
Figure 1. Filter pit for the stormwater treatment system

location is an important factor. Roofs in industrial and comparable commercial zones show higher pollutant concentrations than in residential areas. A search of the literature (Zimmermann et al. 2005) identifies minimum, maximum and mean zinc and copper concentrations from metal roofs (Table 1). By comparison, the mean concentrations of runoff from traffic surfaces are lower both for zinc (< 585 µg/l) and copper (< 86 µg/l).

Table 1. Minimum, maximum and mean of heavy metal concentrations of certain metal roof runoff (modified after Zimmermann et al., 2004)

Roof runoff	Zn (µg/l)			Cu (µg/l)		
	Min.	Max.	Mean	Min.	Max.	Mean
Fibr. cement, pantiles, concrete tiles & tar felt with zinc gutters	280	4,880	1,851	34	2,733	153
Zinc sheeting	1,731	43,674	6,000	11	950	153
Copper sheeting	24	877	370	2,200	3,797	2,600

METHODS

The filters tested consisted of two layers of specially coated porous material separated by a filling layer. The porous material was made of quartz gravel (grain size: 2 mm to 5 mm) bonded with concrete (Concr.) or epoxy resin (Epoxide). Iron hydroxides (IH), precipitated and dried iron sludge (FE), recycled concrete (RC), expanded clay (EC) and zeolithes (Z) were used as filling. Components of the test filters were of varying thickness.

The filters were assessed in laboratory rigs. The discharge area of the test filter in the laboratory rig was equivalent to 1/71 of the discharge area of a normal size filter. Generally, treatment systems are designed with a maximum of 500 m^2 connected area to one DN1000 pit. The test filters were charged with artificial water spiked with pollutants. De-ionized water was charged with sulphuric acid to a pH value between 5.8 and 6.0, which corresponds to the pH value of rainwater in Germany. Dissolved heavy metals salts at a 5–10 times concentration (Table 1) were added and the water mixed by loop pumping. The rain yield factor used of 11.8 l/(s×ha) corresponded to a rainfall intensity of 12.9 mm/d ie between steady rain and a shower. This required a flow rate of 30 l/h (90 l/d) in the laboratory rig at a hydraulic pressure of 1.5 m (Figure 2). Four different tests were carried out. Each test ended when heavy metals appeared in the outflow. Concentrations of metals in the inflow and the outflow were measured. After evaluation of each test, the composition of the filters in the subsequent test was optimised.
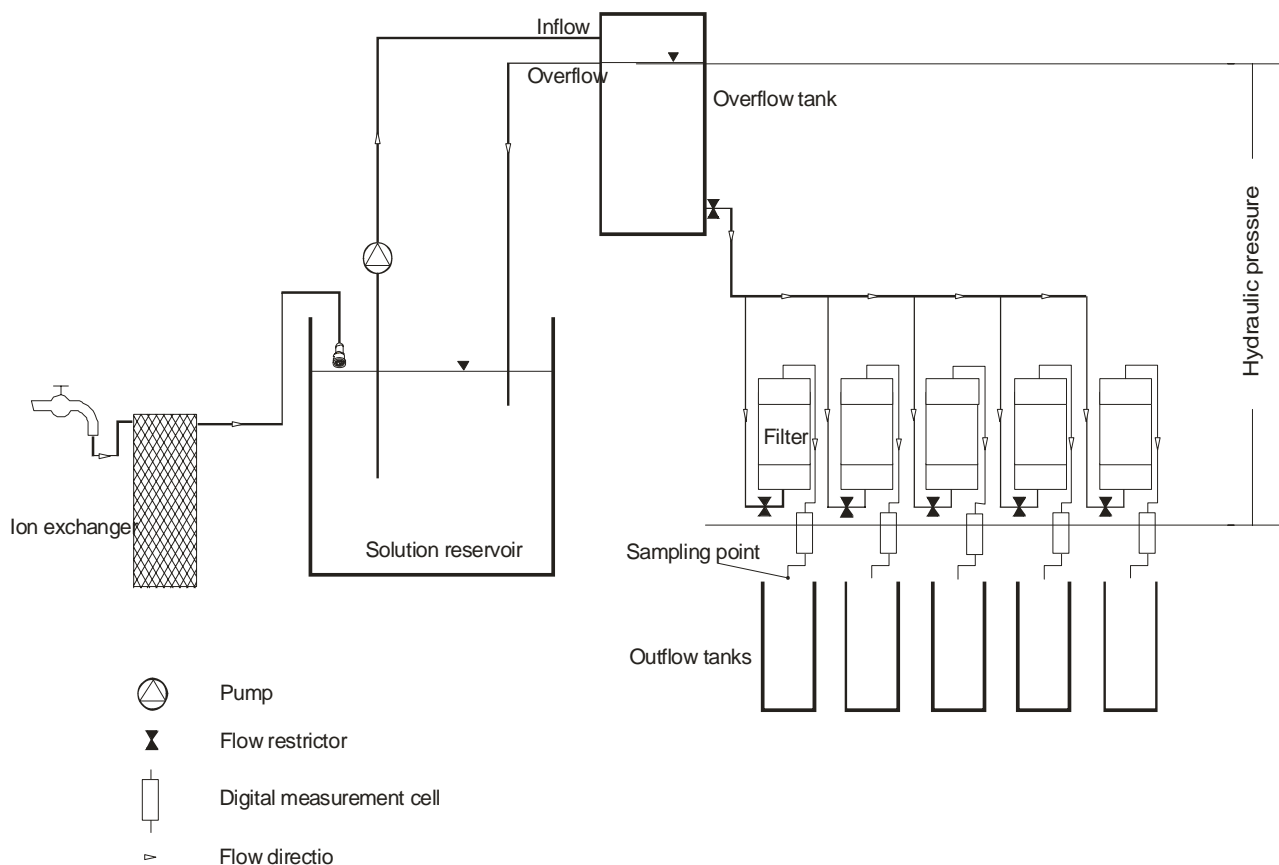
Figure 2. Schematic configuration of the laboratory rig

During one test (Figure 2) five filters were investigated in parallel. Only dissolved copper and zinc were simulated, since particulate metals would have been removed in the pre-filtering sedimentation chamber (Figure 1). Table 2 shows the composition of 20 different filters tested during the study. All tests simulated worst case scenarios. Therefore the result of the laboratory investigation only allows relative estimation within one test.

Table 2. Composition of test filters

Test		Unit	1	2	3	4
Inflow	Solution		Copper Sheeting	Copper Sheeting	Zinc Sheeting	Zinc Sheeting
	Concentration		x 5	x 5	x 10	x 5
	pH value	(1)	5.8	5.8	6.0	6.0
Filter	Discharge area ratio		1/71	1/71	1/71	1/71
	Thickness	(cm)	24 (8+8+8)	24 (8+8+8)	24 (8+8+8)	40 (8+24+8)
Duration		(d)	8	8	2	8
Composition	Filter		Concr. + IH+RC 1:1	Epoxide + IH+RC 1:1	Epoxide + IH+RC 1:1	Concr. + RC
	Filter		Epoxide + IH	Epoxide + IH+RC 10:1	Epoxide + IH+RC 10:1	Concr. + RC+IH 3:1
	Filter		Concr. + IH	Epoxide + IH+RC 50:1	Epoxide + IH+RC 50:1	Concr. + EC+IH
	Filter		Concr. + FE	Epoxide + IH	Epoxide + IH	Concr. + Z
	Filter		Concr. without Filling	Epoxide + FE	Epoxide + FE	Concr. + FE

RESULTS AND DISCUSSION

Generally, retention and flow rates are inter-related; a higher flow rate means a lower retention rate. Figure 3 shows the results for runoff from a copper roof in test 1. The amount of copper removed is cumulative over an 8 day test period. The test simulates an 8 month period in areas with an annual rainfall of 800 mm. The retention capacity of copper decreased over the duration of the test. The epoxy resin based filter showed the highest retention rate of nearly 99% at the beginning of the test and also the highest retention decrease of about 34%. The concrete based filter filled with iron-hydroxide gives the most stable retention rate of about 65%.
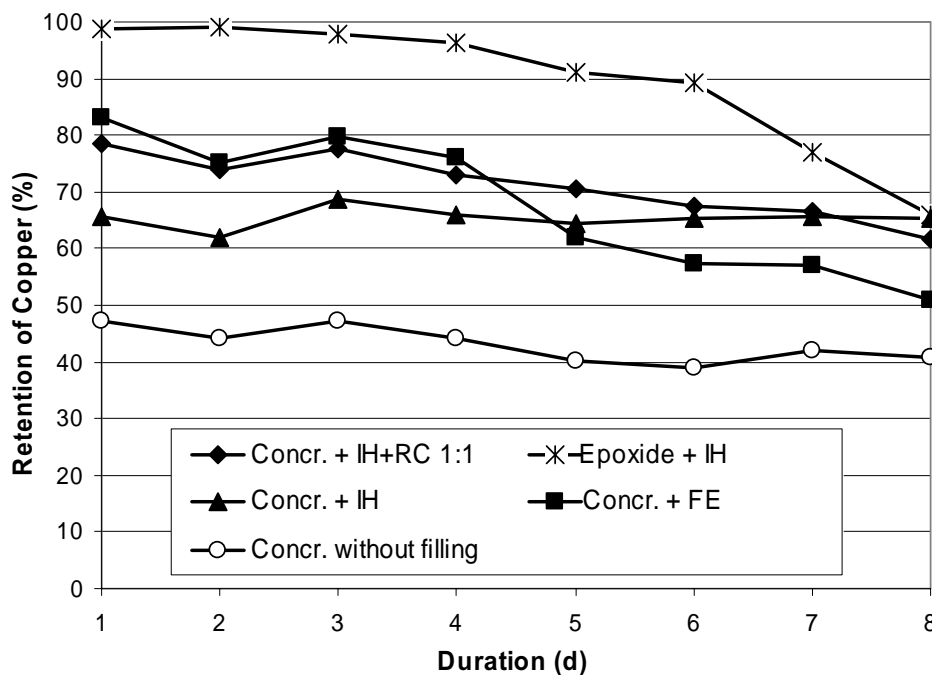
Figure 3. Removal efficiency for copper in test 1

Table 3 gives separate readings for dissolved and precipitated heavy metals. The practical implications of the readings are that a high amount of dissolved metals in the outflow from the filter pit will require further treatment by, for example, concrete infiltration pipes. Precipitated heavy metals can be filtered out by the trench of washed sand surrounding the infiltration pipe. Epoxy resin based filters generally show higher ratios of dissolved metals than concrete based filters. Retention in the latter filters is attributable to adsorption processes in the filling material. The concrete buffers the pH level of the stormwater, which otherwise is typically acidic. Heavy metals are removed by chemical precipitation as stormwater flows through the porous filter.

Table 3. Test results

Test		Filter: Porous Material	Filter: Filling	Flow amount	Percentage of Inflow		
					Total Retention of Filter	Outflow Dissolved	Outflow Precipitated
				(l)	(%)	(%)	(%)
Copper Sheeting	1	Concr.	IH+RC 1:1	642	61.8	0.2	38.0
		Epoxide	IH	581	66.0	24.2	9.8
		Concr.	IH	647	65.3	0.3	34.4
		Concr.	FE	640	50.8	0.2	49.0
		Concr.	without	664	40.7	1.6	57.7
	2	Epoxide	IH+RC 1:1	728	27.0	47.9	25.1
		Epoxide	IH+RC 10:1	728	36.8	58.1	5.1
		Epoxide	IH+RC 50:1	771	42.9	55.9	1.2
		Epoxide	IH	800	38.1	61.4	0.5
		Epoxide	FE	653	46.7	35.6	17.7
Zinc Sheeting	3	Epoxide	IH+RC 1:1	179	50.7	43.9	5.4
		Epoxide	IH+RC 10:1	176	33.8	65.2	1.0
		Epoxide	IH+RC 50:1	161	32.8	67.6	-0.4
		Epoxide	IH	160	25.8	73.3	0.9
		Epoxide	FE	157	26.1	68.5	5.4
	4	Concr.	RC	751	39.7	4.9	55.4
		Concr.	RC+IH 3:1	778	54.1	5.4	40.5
		Concr.	EC+IH	720	69.8	14.9	15.3
		Concr.	Z	722	39.7	7.5	52.8
		Concr.	FE	226	99.6	0.1	0.3

Best results for runoff from copper sheeting were achieved by the epoxy resin based filter filled with iron hydroxide. Adsorption is the main process occurring in this case. A light addition of carbonates such as recycled concrete or iron sludge decreased the percentage of dissolved copper in the outflow of the filter pit because of a higher level of chemical precipitation. Concrete based filters allow chemical precipitation as well as adsorption processes to take place. In the laboratory, while concrete based filters showed lower retention capacity initially, efficiency did not decrease over the test period as in the case of filters with an iron hydroxide filling (Figure 3). Concrete based filters can be expected to have a longer operational life. Since the amount of dissolved copper in the outflow from these filters is less than 1%, the remaining copper particles can be removed easily in the trench of washed sand surrounding the system.

Zinc retention is due to chemical precipitation rather than adsorption processes (Table 3). Best results for zinc sheeting were obtained from concrete based filters filled with a blend of iron hydroxide and expanded clay.

Concrete based filters filled with a blend of recycled concrete and iron hydroxide showed a lower amount of dissolved zinc in the outflow. The high retention capacity (about 99%) of concrete based filters filled with iron sludge was due to the lower flow rate. It is apparent therefore that retention capacity is much higher with lower flow rates. The zeolithe filling material showed low retention capacity under test conditions. An increase in thickness of filters (test 4) had positive effects on retention capacity.

CONCLUSION

Concrete based filters can be expected to have a long operational life. Based on the findings of this study, the retention capacity of filters is at least 65%. However, filters are only one component of the new pollution control system (Dierkes et al. 2005). After several months of operation, the retention capacity of the filter decreases and higher amounts of copper or zinc are able to pass through as precipitated hydroxides and carbonates. As heavy metal retention works by means of adsorption and chemical precipitation, dissolved pollutants can be removed by the filter pit and other upstream components of the system e.g. concrete infiltration pipes. The precipitated heavy metals can be filtered out in the trench of washed sand surrounding the infiltration pipe. The use of different kinds of filters allows the system to treat runoff from a large range of roof surfaces. Further investigations under near natural conditions (Dierkes et al. 2005) show that concrete filters should be replaced every 5 and 10 years. Concrete filters are more cost effective than the use of other adsorbing materials such as zeolithes or activated carbon. Heavy metals deposited on the sealed base of concrete infiltration pipes can be removed by back flushing. The filter in the pollution control pit traps even the smallest particles, preventing them from reaching and clogging the infiltration pipe. This extends the life of the underground infiltration system. Soils and groundwater are effectively protected from heavy metal contamination.

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The 'careos' from Alpujarra (Granada, Spain), an historical example of previous to XII century artificial recharge system applicable to the XXI century – Characterization and inventory

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Abstract

The *careos* system canals constitute one of the first devices for artificial recharge of aquifers of the Iberian Peninsula. Operative from the Muslim period, they present a construction system and distribution still operative at the present what constitutes an example to keep in mind on groundwater management. This paper present a bibliographical recompilation of their origin, construction and operation. We had upgraded the inventory on the field and finally present practical recommendations in order to the design and management of systems of superficial artificial recharge of aquifer. This paper arise from the pH Thesis of the first author.

Keywords

Careos canals, artificial recharge of aquifers, Alpujarra, Sierra Nevada, Granada.

INTRODUCTION

The *careos* canals, traditionally dug in soils or rock, are used like a technique for artificial recharge of aquifers in the Alpujarra area, mainly by mean of meltwaters coming from Sierra Nevada range. Although their origin goes back, at least a year 1139 (Espinar, 1988), its most extensive employment was reached in the last centuries of the Muslim time, XIII to XV centuries, when it was developed an intricate canalisation system for the maximum use of the water (Díaz Marta, 1989).

In the year 2000 the Autonomous Organism of National Parks, National Park of Sierra Nevada, carried out an inventory and reconstruction of *careos* canals, work executed by the Company of Agrarian Transformation (TRAGSA) of Granada. The performances of repair of the inventoried canals are in the address that marks the National Park, like one of their key objectives of management in the administration of the patrimony of canals (Cano-Manuel and Grupo Tragsa, 2000; González Ayestarán, 2000).

Their recovery and maintenance has demonstrated that they help to maintain a vegetation of great interest, they serve for support to a particular fauna and collaborate from a very important way to regulate the hydrological cycle of the region, conditioning a future line of unavoidable performances to be envisaged by the agents of the National Park.

OBJECTIVES

This article, offers an inventory of the net of the *careos* canals of the Alpujarra. The constructive and administration approaches of the *careos* give ideas to the management of current systems and devices of artificial recharge of aquifer, carried out at the present time in similar scenarios. The study area, it has been limited to the canals located in the south hillside of Sierra Nevada National Park. The following works have been carried out:

- Inventory of the current situation of the net of *careos* canals.
- Inventory of Communities Users that use this system.
- Study of the geology to know the permeability of the materials and the effectiveness of the *careos*.
- Determination of the type of materials and the used techniques, for their reconstruction and maintenance.

INVENTORY

According to the inventory of the year 2000 (Cano-Manuel and Group Tragsa, 2000), the *careos* canals are more frequent and more important in East side of Sierra Nevada, area where the smallest altitudes in the mountain range are located, and therefore, smaller precipitation are registered and consequently, the necessity of regulation of the water is bigger. According to the function that they carry out, we find two types of canals:

- *Careo* canals: They facilitate the infiltration of the water. For it, the water of the rivers and streams, is diverted by these canals during the winter and the spring, to the flat areas where it finally infiltrate (Figure 1). Each canal has its area of recharge called *simas* or *cimas*.
- Irrigation canals: They transport the water, generally from the streams to terraces lands (Navarro Pérez, 1983). In these canals the infiltration also has great importance.

From the Muslim period up to the present the *'acequiero'* remains yet like the person in charge of the *careos* administration and the only authorised to control the floodgates (Al- Mudayna, 1991 and Vidal, 1995).

They have been classified and defined a total of 23 *careos* canals from 127 inventoried (Table 1). Their position is located in figure 3.

The *careos* canals considered more important for their size and preservation degree are:

- Canals of Mecina-Bombarón. It has a system of canals very well developed. The canal of Mecina is the biggest canal of Sierra Nevada. It has 20 *simas* where the water is distributed (Cara, 1989; Ben Sbih and Pulido-Bosch, 1996).
- Canals of Trévelez. Good conservation state (Delaigue, 1995) and with scarce presence of 'new materials' employees in maintenance works.
- Canals of Bérchules. It has big irrigation canals and two *careos*.
- Canals of Valor. The Users Community negotiates three *careos* canals: canals of the Vadillos, La Loma y El Monte. These *careos* are used for the urban supply.

One overlapping GIS between the *careo* nets and the geologic map has been carried out in order to study their effectiveness, also adding other geographical coverings. The study reveals that there is a general tendency to build *careos* from fluvial beds to: 1) limestone outcrops, with simas and ponors; 2) metamorphic areas with weathering zone superficial zone, where the regolit acquires aquifer potentiality (Castillo et al, 1996) and 3) quaternary detrital aquifer of diverse typology (piedmont, etc.), under which carbonated or metamorphic formations of low permeability underlie (in general phyllites, schists and calc-schists).

Figures 1 a) and b).
Examples of careo´s morphology.
Lanjarón and Mecina sites

Table 1. Inventory of 'careos'

Name	Diverted river	Community Users	Observations
Del Espino	Chico de Bérchules	YES	Well conserved
Bérchules	Trevélez	YES	Almost abandoned
Mecina	Grande de Bérchules	YES	Well conserved Urban water supply
De la Mogea	Nechite	YES	Well conserved
Del Horcajo	Mecina	YES	Well conserved
Yegen	Mecina	YES	Sealed by concrete Irrigation and *careo*
De los Vadillos	Valor	YES	Well conserved
Del Monte	Valor	YES	Well conserved. Irrigation and *careo*
De la Loma	Valor	YES	Well conserved. Earth
De la Fuente del Espino	Nechite	YES	Well conserved
Del Boy	Laroles	YES	Well conserved Concrete and earth
Nueva de Bayarcal	Bayarcal	YES	Very well conserved
De las Hoyas	Andarax	YES	Well conserved Small *careo* made by earth
Del Pecho	Andarax	YES	Well conserved. Small *careo* made by earth
Del Maguillo	Río del Pueblo	YES	Well conserved. Close the line, there is a pathway
Del Prado Llano	Río del Pueblo	YES	Well conserved
Del Prado Largo	Río del Pueblo	YES	Well conserved
De *careo* de Beires	Andarax y Ohanes	NO	Fair conserved. Stability problems
Del Garbanzal	Ohanes	NO	Fair conserved
Del Canal	Ohanes	NO	Well conserved. Partly it takes advantage of an old channel of hydropower station.
De Tices	Ohanes	YES	Whit careless
Del Corazón	Alhorí	YES	Abandoned recently.
Del Jaral	Alhorí	YES	Well conserved

MAINTENANCE AND RECOVERY

In general, the *careos* require an important maintenance. They are usually affected by diverse problems: hillside movements, fall of solids that cut or break the conduits, rill erosion, bad lands, etc. The most important recovery tasks that they carrying out are the following ones:

- Excavation of the bottom of the channel and rebuilt by buried stones (Ayuso et al., 1986, Rodríguez de Velasco, 1993 and Medina, 1996), (Figure 2 a) and protection of the borders of the channel with flagstones (Figure 2 b).
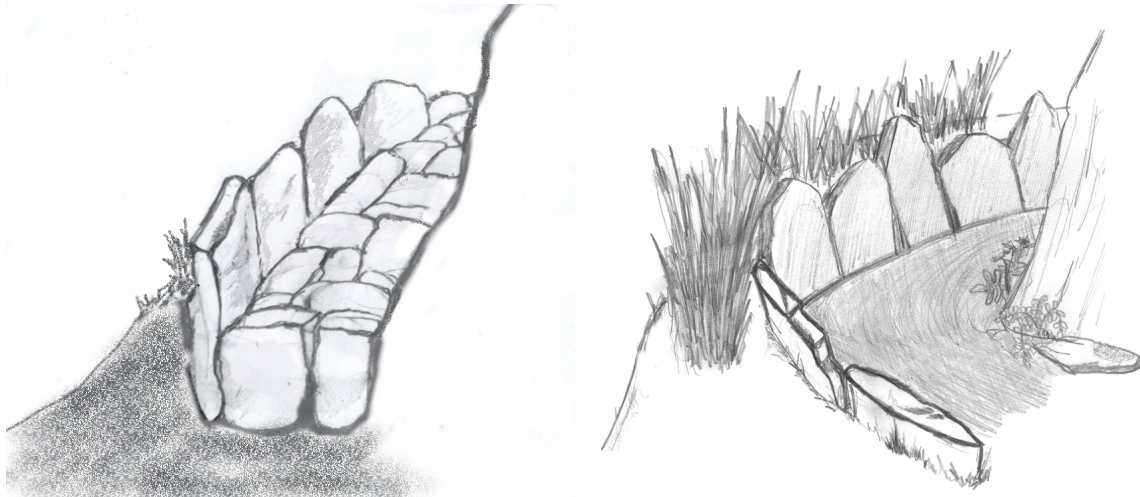
Figures 2 a) and b).
Drawings of hydraulic masonry (for courtesy of Cano Manuel and Group Tragsa of Granada).
a) Structure of the channel protected by stones buried in tracts of great slope and subject to a strong erosion,
b) Flagstones protecting the external border of the channel.

- Enlarge of the channel in specific points and amplification of the longitude of the canal increasing this way the infiltration surface and construction of earth channel, by means of the opening of ditches, in order to improve the artificial recharge of the aquifer for direct infiltration in device type grave.
- Homogenisation of the slope to avoid damming in not wanted areas and construction of masonry aqueduct in the points where the canals cross ravines in those that lose great quantity of water.

CONCLUSIONS

- The *careos* canals, constitute a specific system of artificial recharge of aquifer. Of the 127 inventoried canals, 23 are *careos*. The most important are those of Trevélez, and Bérchules, Mecina-Bombarón and Valor. These arise from the XII-XIII centuries.
- They have a joint administration with a person in charge 'the acequiero' who is responsible for the floodgates. In general, the Communities of Users, main protectors of the system of canals, have scarce resources for their maintenance, so a part of the conservation expense is supported by the National Park.
- In the study area, the best favourable geologic materials for the artificial recharge by mean of *careos* are the limestone and formations permeable detrital formations in crop areas. The topography is a strong condition of the design and layout of the canals.
- It would be important to preserve and to maintain these systems of *careos*, given their high historical and environmental value, appealing if it is necessary to the externalities of the expenses in concept of environmental costs.
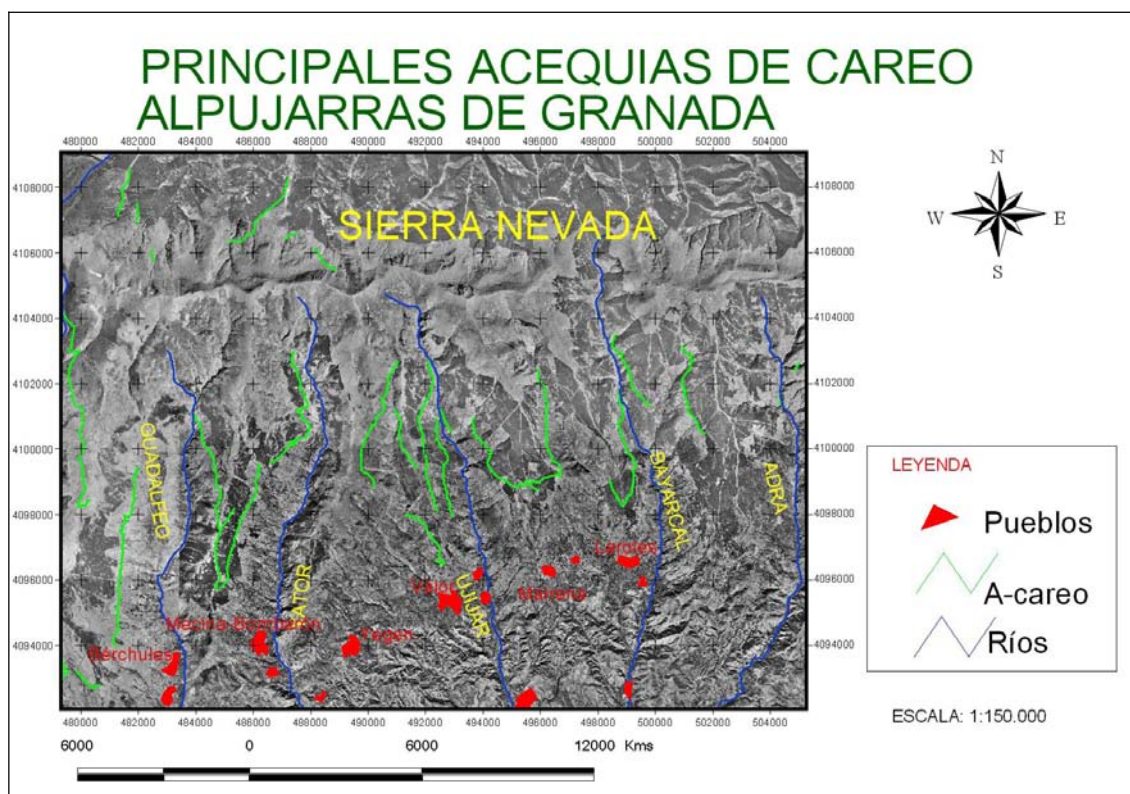
Figure 3. Inventory of 'careos' map

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Pumping influence on particle transport properties of a chalk karst aquifer exploited for drinking water supply

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Abstract

This study aims to know the drainage conditions of a chalk karst aquifer exploited for drinking water supply and to determine if pumpings modified the particles transport properties. The study site located in the Seine river edge corresponds to a chalk karst aquifer of the western Paris Basin. It consists of i) a sinkhole, where the surface water is introduced, ii) an overflow spring which drains naturally chalk aquifer, iii) a well for drinking water supply.

We associate the grain size distribution with the groundwater piezometric level and the Seine river tidal range. To identify the drainage conditions and the pumping impact, we added qualitative variables concerning the Seine river dynamics, and the pumping period. Then we treated the whole by a statistical method allowing the combined use of the qualitative and quantitative variables.

The results show that this chalk karst aquifer is naturally and mainly drained by the overflow spring and by the Seine river. Pumping periods carries out an artificial drainage of this aquifer which allows to increase the grain size distribution and the concentration of suspended matter released at the well.

Keywords

Karst hydrology, transport properties, human impact, multivariate analysis.

INTRODUCTION

The hydrological functioning of aquifers may be complex in particular for the karst aquifers. This complexity is due to the significant number of parameters which can affect the functioning of aquifers (boundary conditions, base level variation, pluviometry, human impact, interactions with a river or sea ...). The transport of particles in karstic media is a complex process implying deposition and release phenomena (Massei, 2001; Massei et al., 2003). The observed turbidity would have two potential origins: (i) the direct transfer of particles from the inlet to the outlet of the karstic system (allochthonous origin) and (ii) the resuspension of previously deposited sediments (sub-autochthonous origin). The particle size distributions (PSD) of suspended particulate matter may constitute a tracer of particulate transport which is related to flow conditions within hydrological systems.

The aim of this study is to identify the hydrodynamic variables which control the transport properties of particles within a karst aquifer of western Paris basin. The PSD of water samples swallowed at a sinkhole have been compared with the PSD released at a spring and a well. This comparison allow to identify the particles transport properties within this karst aquifer. Then, the granulometric characteristics of water samples at the sinkhole, spring and well, have been compared with the hydrodynamic variables of karst system (recharge and depletion of ground water, Seine river dynamics, human impact). The variables studied (Table 1) are numerous (49) and mixed (quantitatives and qualitatives). Under these conditions, it is difficult to identify the parameters which influence the karst

aquifer functioning. Thus, a multivariate analysis (Hill and Smith Analysis, Hill and smith, 1976), which allows the joint analysis of quantitative and qualitative variables, was used to identify i) the hydrodynamic parameters controlling the transport properties of suspended matter, ii) the human impact on particle transport properties.

METHODS

Study site

The karst system studied is located in the Pays de Caux (Haute-Normandie, France) on the right bank of the Seine river, about 40 km away from the Seine estuary (Figure 1). This kind of system is typical of the karst of the lower Seine valley. The Norville system has been widely studied and its boundaries became quite well known (Massei,
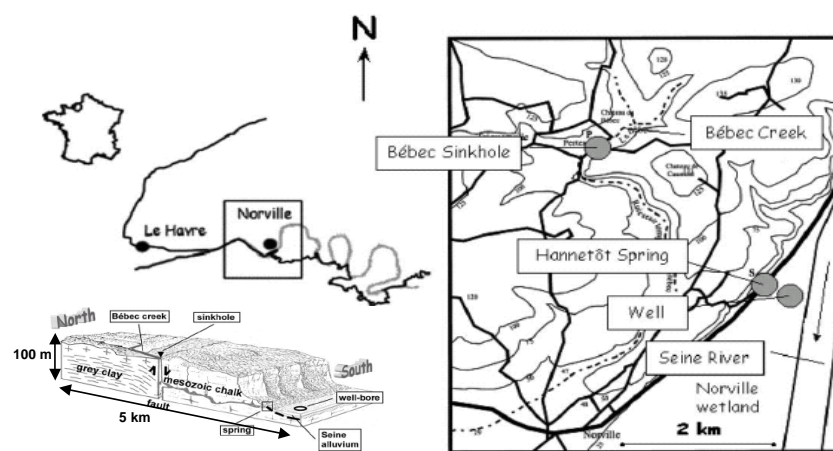
Figure 1. Study site location

2001; Massei et al., 2002a, b; Dussart-Baptista, 2003; Massei et al., 2003). The system is composed of a sinkhole, a spring, and a well for drinking water supply.

Water head in a tank controls the automatic release of pumping in well. During the day (8 AM to 10 PM), pumping starts automatically when the water level in the tank reaches 2,50 m. In the night (10 PM to 8 AM), pumping occurs when water level in the tank reaches 1,40 m. The demand of the population during the day generates a multiplication of the pumping sequences which last 1 hour (4 to 6 times of 8 AM at 10 PM). In the night, only one pumping sequence occurs which lasts 5 hours.

Materials and methods

ISCO 6700 automatic sampler were installed at the three points of the karst system (sinkhole, spring and well). Water samples were collected in response to two rain events (February 2002 and October 2002) and one dry event (March 2003). February 2002 was an intense rain event with a mean intensity equal to 1.24 mm.h^{-1} during 4 days. Five water samples were taken during this period. The mean intensity of October 2002 rain event was 0.22 mm.h^{-1} during 4 days (Dussart-Baptista, 2003). Nine water samples were taken during this period. In March 2003 (four water samples were taken) we realized a continuous pumping at the well during 14 hours to understand the influence of a long pumping on particle transport properties. The particle size distributions of water samples (PSD) were determined with a Coulter Multisizer particle counter, using a 100 µm aperture. Artificial tracer test were carried out to know hydraulics connections of the sinkhole-spring and sinkhole-well systems (Massei, 2001; Dussart-Baptista, 2003). Knowing the transit times, the granulometric characteristics of each water sample intro-

duced at the sinkhole may be compared with those released at spring and well. The PSD were gathered according to grain size (clays < 2 µm; 2 < very fine silts < 4 µm; 4 < fine silts < 8 µm; 8 < medium silts < 16 µm; 16 < coarse silts < 31 µm; 31 < very coarse silts < 63 µm). From each grain size class (GSC), two indices (A and T) have been computed to identify the transport properties within the karst system:

$$A_{gsc} = W_{gsc}/Sp_{gsc}$$
$$T_{gsc} = Si_{gsc} - (Sp_{gsc} + W_{gsc})$$

with S_i=sinkhole, S_p=spring, W=well

These equations have been computed separately for each grain size class (GSC).

For each GSC, the A index indicates the proportion of the particles released at the well compared to the particles released at the spring. It allows to identify the human impact on the PSD at the well during the pumping periods. For each GSC, the T index indicates the proportion of the particles released at the spring and well compared to the particles introduced at the sinkhole. It allows to identify the transport properties within the karst system:

- For each GSC: if $T_{gsc} < 0$, the PSD released at the spring and well are more important than PSD swallowed at the sinkhole. It puts in evidence the resuspension of intrakarstic sediments, noted by R (in Figure 2, for example R.c indicates the resuspension of clays and R.vcs indicates the resuspension of very coarse silts).
- For each GSC: if $T_{gsc} > 0$, the PSD released at the spring and well are less important than PSD swallowed at the sinkhole. It puts in evidence the deposition of suspended matter, noted by D (in Figure 2, for example D. c indicates the deposition of clays and D.vcs indicates the deposition of very coarse silts).

These granulometric variables have been compared with the hydrodynamic variables of karst system. As the variables (Table 1) are numerous (49) and mixed (quantitatives and qualitatives), the Hill and Smith Analysis (HSA, Hill and Smith, 1976), which allows the joint analysis of quantitative and qualitative variables, was used to identify the hydrodynamic parameters controlling the transport properties of suspended matter and the human impact on particle transport properties.

Table 1. Identification of the variables

	Granulometric variables	Hydrodynamic variables
Quantitative variables	Particle size distribution (PSD) according to the grain size class (gsc) at the sinkhole (Si.gsc), spring (Sp.gsc) and well (W.gsc)	Piezometric level of karst aquifer (piezometry) Water height at the Seine river (tidal range)
Qualitative variables	PSDwell > PSDspring : A.gsc Resuspension (R.gsc) or deposition (D.gsc)	Groundwater dynamics (recharge or depletion) Seine river dynamics (flood or ebb and falling or rising) Human impact (pumping or no pumping and daylight cycle or night cycle or continuous pumping)

Multivariate analysis

According to Hill and Smith (1976), the intuitive notion of the correlation between a qualitative variable and a quantitative variable can be formalised using the analysis of variance. That is to say, the correlation between the two variables may be interpreted as the correlation which would be registered if the quantitative variable is approximated as closely as possible by an additive effects model based on the observed character states of the qualitative variable (Hill and Smith, 1976). Therefore, the Hill and Smith analysis (HSA, Hill and Smith, 1976) is capable of 'recognizing' when two character states are essentially the same (i.e. when they do not differ in their relations to the

other characters), and that it automatically makes allowance for this fact. In this way it is possible to define the correlation between a qualitative and a quantitative variable.

The HSA proceeds in three steps i) the quantitative variables are processed by principal component analysis, ii) the qualitative variables are processed by correspondence analysis, iii) the HSA studies the interrelationship between the quantitative and qualitative variables. The results of HSA are interpreted in two steps i) the variable space interpretation, ii) the individual space interpretation. Variables and/or individuals that are plotted close to each other on the factorial plane are related or correlated to each other, whereas those variables and/or individuals that are plotted separately are not related or correlated with other variables and/or individuals. Consequently, in the factorial plane of individuals, if two individuals plot close to each other, then these two individuals are strongly correlated. In the factorial plane of variables, it is the same interpretation between two quantitative variables, between two qualitative variables and between quantitative and qualitative variables. Qualitative and quantitative variables have been treated by HSA with the ADE4 package (Thioulouse et al., 1997) of the R software (Ihaka and Gentleman, 1996; The R Development Core Team, 2003).

RESULTS AND DISCUSSION

Variable space

The HSA is a multivariate analysis which allows to identify the correlations between quantitative and qualitative variables. In the variable space (Figure 2) to identify the hydrodynamic variables which influence the granulometric characteristics of water at the spring and well, we seek the groups of quantitative and qualitative variables which are plotted i) close one to the other, i.e. which are correlated, or ii) opposed, i.e. which are anti-correlated. The first factorial plane (F1F2) expresses 58.9% of the total variance, i.e. of all the variations of variables. F1 is related to PSD at the sinkhole (Si.gsc), spring (Sp.gsc) and well (W.gsc), the piezometric level (piezometry and recharge), then the flood and the Seine river tidal range in the negative part (F1), opposed to ebb, and depletion of chalk aquifer in the positive part (F1+). F1 can be interpreted as the PSD of particles released according to the natural hydraulic gradient defined by the piezometric level and the water height at the Seine river. The negative part (F1-) expresses a strong hydrodynamic conditions and the positive part (F1+) expresses a weak hydrodynamic conditions.

PSD=Particle Size Distribution
gsc=grain size class (c=clay, vfs=very fine silt, fs=fine silt, ms=medium silt, cs=coarse silt, vcs=very coarse silt)
hydrodynamic variables of the karst aquifer, the Seine river and the human impact
PSD according to the grain size class at the sinkhole (Si), spring (Sp) and well (W)
Transport indices according to the grain size class

Figure 2. Variable space of HSA (For more clearness, only the arrows of the hydrodynamic variables have been preserved, the positions of the other variables are represented by points)

F2 is related to the resuspension of intrakarstic sediments whatever the grain size class (R.gsc) and strong released at the well (A.gsc), then rising at the Seine river, pumping, continuous pumping and night cycle at the well in the positive part (F2+), opposed to deposition of suspended matter (D.gsc), daylight cycle and no pumping in the negative part (F2-). F2 can be interpreted as a human impact on particle transport properties at the well. The negative part

(F2-) expresses a weak hydrodynamic conditions and the positive part (F2+) expresses a strong hydrodynamic conditions.

Whatever the grain size class, the particle recovery is more or less the same at the sinkhole and spring. The PSD of suspended matter recovered at the spring strongly depends on PSD of suspended matter swallowed at the sinkhole, i.e. of the rain event intensity. At the well, whatever the grain size class, the deposition of suspended matter (D.gsc) are located in the negative part of F2 (F2-), which defines weak hydrodynamic conditions (no pumping, daylight cycle, falling tidal range condition). On the contrary, the A.gsc and R.gsc indices, which indicate an important resuspension of intrakarstic sediments at the well, are located in the positive part of F2 (strong hydrodynamic conditions). Thus, after rain event intensity which determine the granulometric characteristics (number and size) of particles swallowed at the sinkhole, the granulometric characteristics of water samples are determined by the hydraulic gradient (between piezometric level in karst aquifer and water height in the Seine river) at the spring then by this hydraulic gradient and pumping periods at the well.

Individual space

In the variable and individual spaces, principal components (F1 and F2) are the same. The position of the samples on the individual space depends on the values of their variables. Therefore, we can interpret the position of the individuals on F1 and F2 as strong correlations or anti-correlations with the variables which define the principal components. For example, individuals close to the negative part of F1 in the individual space (Figure 3) are strongly determined by variables close to the negative part of F1 in the variable space (Figure 2), etc.

The Figure 3 shows that the water samples are well separated according to the sample period, i.e. according to the rain event intensity. The samples of February (individuals number 1 to 5) are localised in the negative part of F1 (F1-). The February 2002 rain event was an intense rain event. Thus, a huge amount of particles swallowed at the sinkhole was released at the spring and well. The fourth sample has been taken in daylight during the ebb under falling tidal range condition. There were no pumping at the well and the Seine river height was low. This sample shows an important deposition (strong values for D.gsc) and a weak particle recovery at the well (weak values for A.gsc). On the contrary, the fifth sample shows an important resuspension (strong values for R.gsc). It has been sampled during pumping in night cycle under the same conditions at the Seine river.
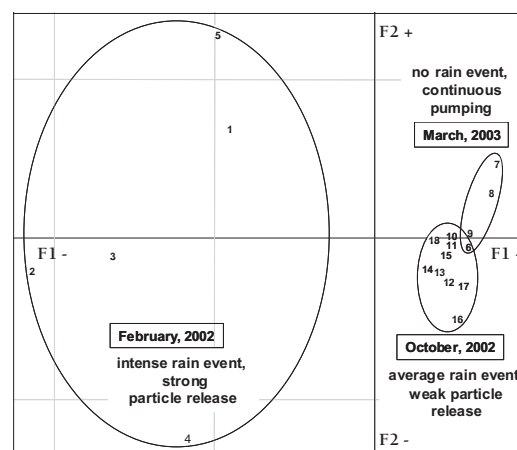
Figure 3. Individual space of HSA

The October 2002 rain event was a mean event. The samples (individuals number 10–18) are located in the positive part of F1 (F1+), because the particles recovered at the spring and well are less important than in February. The sixteenth sample has been sampled during no pumping period. It shows an important deposition and a weak particle recovery at the well. On the contrary, the tenth sample shows an important resuspension and a strong particle recovery at the well. It has been sampled during a pumping period.

In March 2003, no rain event occured during the continuous pumping. Thus, not a huge amount of particles swallowed at the sinkhole were observed. March samples (individuals number 6–9) are well defined by A.gsc which indicate the most important proportion of particle released at the well during the continuous pumping experiment. All samples have been sampled during pumping periods except for the ninth sample.

In the individual space, the PSD of water samples are separated on F1 according to the rain event intensity and the hydraulic gradient and on F2 according to the pumping periods.

CONCLUSION

HSA puts in evidence the qualitative and quantitative variables which have an impact on transport properties of suspended matter within a karst system. It allowed to dissociate the natural functioning and the human impact. The natural functioning is controlled by i) the pluviometry which defined the characteristics of the PSD swallowed at the sinkhole, ii) the hydraulic gradient which is defined by the piezometric level and the Seine river tidal range. Pumping periods produced an artificial hydraulic gradient which is added to the natural gradient. This impact is all the more significant as pumping times are long.

The number and size increases of particles, by resuspension or direct transfer, are responsible of well water turbidity and an arrest of drinking water supply. From these results, obtained by this analysis, we can define a management of aquifer to protect the karst water resources. As the Seine river tidal range is defined by tide coefficient, we can determine the periods when the hydraulic gradient will be strong; pumping will have to be very limited during these periods as well. The pumping periods must be defined according to the tide coefficient variations to prevent the artificial gradient to be added and amplify an already strong natural hydraulic gradient.

The HSA offers the interesting prospects for hydrogeologists. It allows to identify the parameters which influence the global functioning of the aquifers from a large data set of quantitative and qualitative variables. This study illustrates the results which the hydrogeologists could obtain for the management of aquifer recharge:

- identification of the variables which influence the aquifer recharge by statistical treatment of large data sets acquired during experiments,
- identification of the hydraulic properties of aquifer during the recharge,
- definition of the optimal conditions for the management of aquifer recharge.

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Basin artificial recharge and groundwater mound formation: A case of the Rokugo alluvial aquifer, northern Japan

Noboru Hida and Yukiko Ohizumi

Abstract

The Rokugo alluvial fan lies in northern Japan. The unconfined aquifer of the fan consists mainly of gravel and sand. Four artificial recharge basins were constructed on the central part of the fan. This time, we discuss the effects of artificial recharge on the formation of groundwater mounds under the area of basins No. 1 and No. 2. Results obtained are based on the recharge operations which have done three times, first: from September 8 to November 10, 1998 (63 days), second: from November 15, 1998 to April 5, 1999 (141 days) and third: from April 19 to April 29, 1999 (10 days). As conclusion, groundwater mounds were formed under the bottom of basins No. 1 and No. 2. The enhancing of the groundwater cycle in the aquifer has resulted in sustainability for the groundwater environment especially in the distal fan.

Keywords

Artificial recharge, aquifer, groundwater mound, environment, sustainability.

INTRODUCTION

The Rokugo alluvial fan lies around 39"25'N and 140"34'E in northern Japan. The distance between the proximal fan at 90 meters above sea level and the distal fan at 45 meters is about four kilometers. The center of Rokugo town, which is situated on the distal fan, numbers 6,000 inhabitants. They privately bore wells, pump groundwater, and supply the water for their own domestic uses. In addition, there are over 70 water springs including large and small ones in the area of the distal fan. The springs are associated with the regional water environment as well as multi-purpose water uses.

An artificial recharge for the aquifer, using four basins was operating during the non-irrigation period in order to enhance unconfined groundwater. The basins, Nos. 1–4, were constructed on the central part of the Rokugo alluvial fan. Souse water to the basins is withdrawn from irrigation canals in which water flows through the year.

In this paper, we discuss the effects of artificial recharge on the formation of groundwater mounds under the area of basins No. 1 and No. 2. Results obtained are based on the artificial recharge operations which have done three times, first: from September 8 to November 10, 1998 (63 days), second: from November 15, 1998 to April 5, 1999 (141 days) and third: from April 19 to April 29, 1999 (10 days). We previously pointed out the outcome from the basin No.4 experience (Hida et al., 1999).

HYDROLOGICAL ENVIRONMENT

The unconfined aquifer of the Rorugo alluvial fan consists mainly of gravel, of which hydraulic conductivity is of 10^0–10^{-2} cm/sec and specific yield is over 20%. The depth of the aquifer is over 100 meters around the center of the fan. Annual mean precipitation is 1,653 mm and annual mean potential evapotranspiration is estimated at 660 mm.

Maximum snow depth appears during a period from mid-January to mid-February. It averages 130 cm in the distal fan and 150 cm in the proximal fan.

The annual groundwater level changes regularly. The level is high during the period of paddy field irrigation from May to August, and low during the non-irrigation period. As for land use, the paddy field accounts for 70 per cent of the total surface of the fan.

ARTIFICIAL RECHARGE BASINS AND OBSERVATION EQUIPMENT

Site and structure of the four basins

Table 1 shows the site and structure of the basins, Nos. 1–4, such as the Bottom area of the basin, Depth from the ground surface at the basin, Latit., Longt., Elevation, and Construction year.

Table 1. Site and structure of the basins

	Area	Depth	Site	Elv.	Const. yr.
No.1	1,192	1.0 m^3	9°25"05'N, 140°33"35'E	58.0 m	1991
No.2	2,120	3.4 m^3	9°25"00'N, 140°34"05'E	68.0 m	1992, 1994
No.3	212	2.9 m^3	9°25"01'N, 140°33"44'E	61.0 m	1998
No.4	1,045	3.0 m^3	9°25"27'N, 140°34"05'E	62.0 m	1998

Observation equipment

Fig. 1 shows the site of basin No. 1 and three observation wells, Nos. 1–3, and Fig. 2 shows the site of basin No. 2 and three observation wells, Nos. 4–6 and one piezometer.
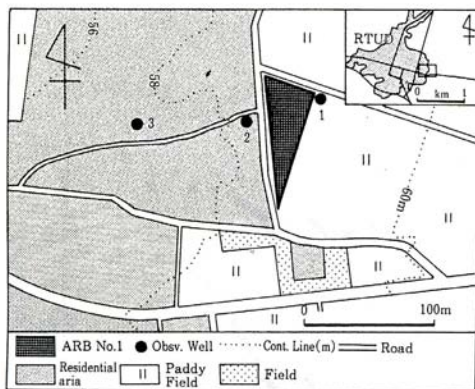
Figure 1. Artificial recharge basin No. 1, observation wells 1–3 and surround
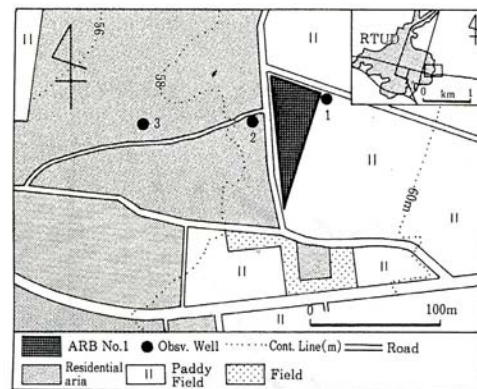
Figure 2. Artificial recharge basin No. 2, observation wells 4–6 and surround

RESULTS AND DISCUSSIONS

Amount of supply water and infiltration rate: Basins No. 1 and No. 2

The irrigation canal laid out beside paddy fields supplies water sources to the basins. The water received no pre-treatments. Table 3 shows the amount of water supplied in l/sec and Table 4 shows the infiltration rate in cm/h.

Table 3. Amount of water supplied in l/sec

	First time	Second time	Third time
Basin No. 1	(40)	(40)	40
Basin No. 2	(96.6)	(96.6)	96.6

Table 4. Infiltration rate in cm/h

	First time	Second time	Third time
Basin No. 1	(12.0)	(12.0)	12.0
Basin No. 2	(16.4)	(16.4)	16.4

Note: The amount was calculated from the measurement of supplied water made on April 4th, 1999. The other infiltration rates are estimated to be equal to or greater than the the third time.

Growth and decay of groundwater mound

Case of Basin No. 1

Growth of groundwater mound (Figs. 3 and 4)

In the duration of 10^4 minutes after the beginning of the water supply to Basin No.1, we recorded growing water table at observation wells 1 to 3. Below is the record of water table changes observed at well 2 for the three respective experiments (Fig. 3).

In the first experiment, water table reached the highest value, 51.85 meters as of 2,920 minutes (ca. 49 hours) in process. The rise in the water table was 1.16 meters during this time. The water table averaged a five centimeter per hour rise until 180 minutes after supplying water, and 10 centimeters rise per hour after that. This high amount is associated with the rise of water level in Basin No.1. In the second experiment, the water table averaged 10 centimeter rise per hour after 95 minutes from the beginning of water supply to Basin No.1, and reached the highest value, 51.48 meters as of 6,920 minutes (ca. 115hours) into the process. The water table rose 1.47 meters. In the third experiment, the water table did not rise as much as in the first and second experiments because the initial water table was high and the quantity of water supplied to Basin No.1 was relatively small due to unstable flow of water in the irrigation canal. The water table rose 0.46 meters. In each case, groundwater mound was formed under Basin No. 1 as shown in Fig. 4 which depicts a longitudinal section along observation wells 1, 2 and 3.

Decay of groundwater mound (Figs. 5 and 6)

In the duration of about 104 minutes after the stopping water supply to Basin No. 1, we recorded a decaying water table at observation wells 1 to 3. Following is the record of water table changes observed at well 2 for the three experiments (Fig. 5). In the first experiment, the water table almost showed no change until 370 minutes (ca. 6 hours) after stopping water supply. After that, it sank at a rate of 2 to 4 centimeters per hour, which was recorded after water level in the Basin No.1 reached zero. The decrease in the water table was 1.07 meters over the period of ca. 4 days. In the second experiment, the water table rose 4 centimeters even after stopping water supply to the Basin No.1. It will be seen that water from snow melt was infiltrating the basin. As of 1,340 minutes (ca. 22 hours) after stopping water supply, water table sank down to 50.55 meters.

In the first experiment, the water table almost showed no change until 370 minutes (ca. The declining amount of water table was 1.58 meters in the ca. 22 hours. In the third experiment, water depth inside Basin No.1 was 91 centimeters right before stopping water supply to the basin, and reached almost zero centimeters as of 960 minutes. The decrease of the water table was small during the time when the water in Basin No.1 was present, and showed 4 centimeters per hour and then 1–2 centimeters per hour after the water in the basin disappeared. As of 10,155 minutes (ca. 7 days) after stopping water supply, the water table sank to 49.75 meters. The decrease in the water table was 1.39 meters over the ca. 7 days. In each case, groundwater mound decayed under Basin No. 1 as shown in Fig. 6, which is drawn in a longitudinal section along observation wells 1, 2 and 3.

Figure 3. **Rise of the groundwater levels after supplying water to artificial recharge bsin (ARB) No.1**

Experiment 1: Sept. 8~27, 1998
Experiment 2: Vov. 15~30, 1998
Experiment 3: April 19~28, 1999
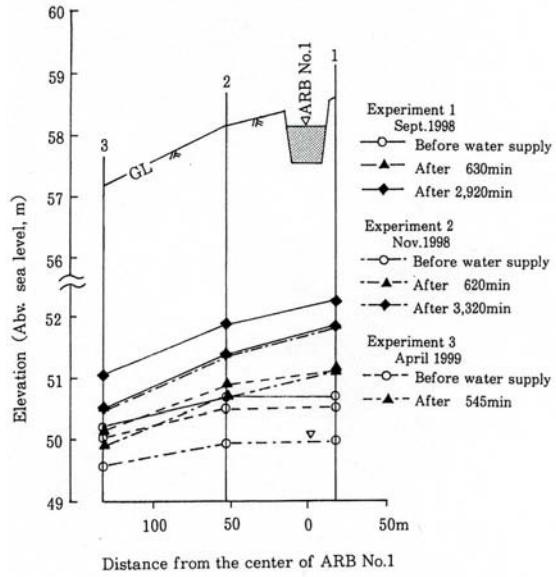
Figure 4. **Changes of longitudinal profile of the water table between observation well 1 and observation well 3: after supplying water to artificial recharge basin No.1**

1~3: Observation well
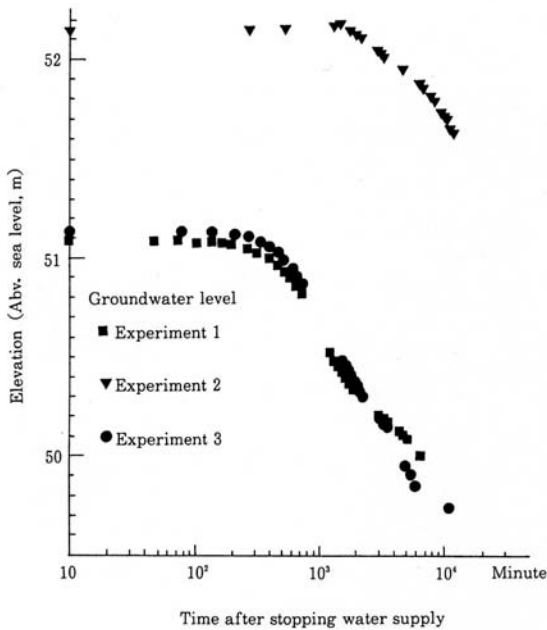
Figure 5. **Decline of the groundwater levels after stopping water supply to artificial recharge basin (ARB) No.1**

Experiment 1: Nov. 10~14, 1998
Experiment 2: April 5~12, 1999
Experiment 3: April 29~May 6, 1999
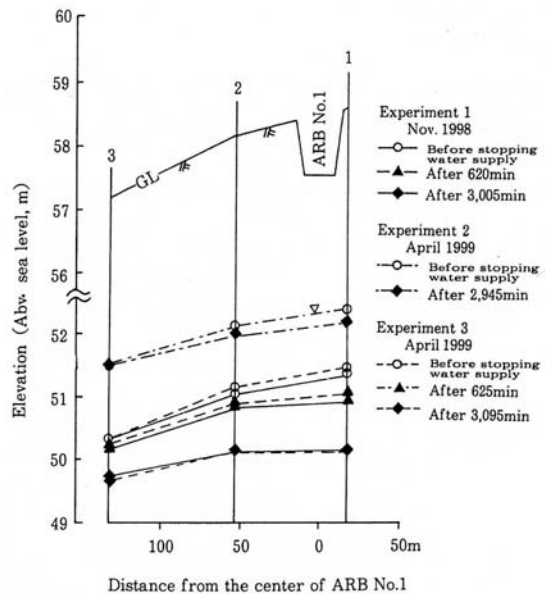
Figure 6. **Changes of longitudinal profile of the water table between observation well 1 and observation well 3: after stopping water supply to artificial recharge basin No.1**

1~3: Observation well

Case of Basin No. 2

Growth of groundwater mound (Figs.7 and 8)

About 10^4 minutes after beginning water supply to Basin No. 2, we recorded a growing water table at observation wells 4 to 6 and the piezometer. Initial water table under Basin No .6922, measured at observation well 5, was about 57.5 meters in each of the three experiment. The water table began to rise after supplying water to the basin, and reached the highest value, 60.39 meters as of 43 hours in the first experiment, 60.66 meters as of 56 hours in the second case test and 60.66 meters as of 172 hours in the third experiment, The ithe third experiment, in respective order (Fig.7). The increase in the water table was 3.03 meters for the first, 2.87 meters for the second and 3.23 meters for the third. In each experiment, mound was formed under Basin No.2 as shown in Fig. 8 which is drawn in a longitudinal section along observation wells 4, 5, 6 and the piezometer.
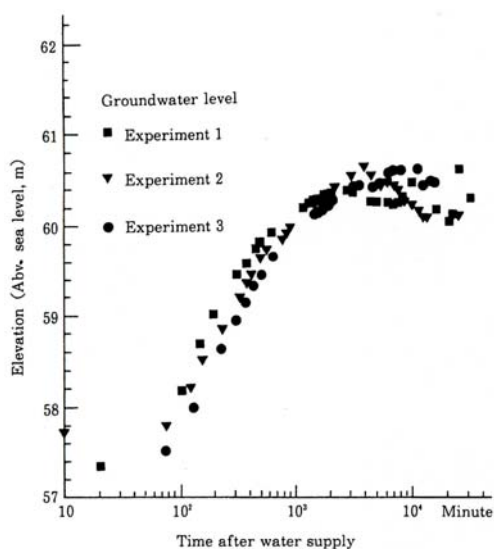
Figure 7. **Rise of the groundwater levels after supplying water to artificial recharge basin (ARB) No.2**

Experiment 1: Sept. 8~27, 1998
Experiment 2: Vov. 15~30, 1998
Experiment 3: April 19~28, 1999
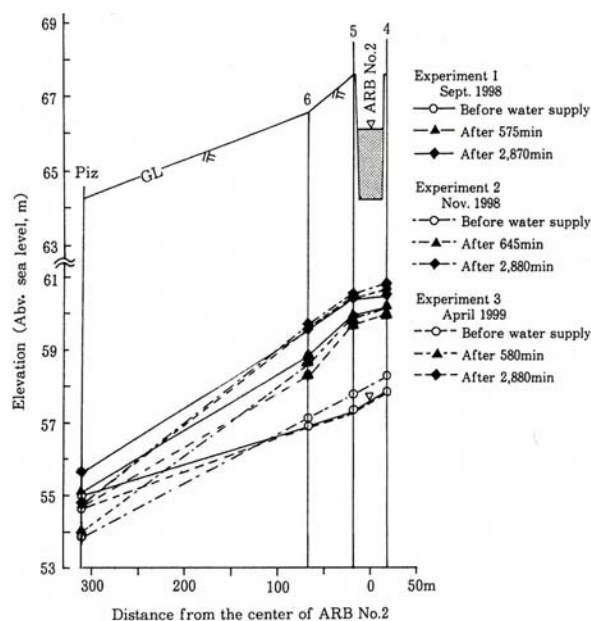
Figure 8. **Changes of longitudinal profile of the water table between observation well 4 and piezometer: after supplying water to artificial recharge basin No.2**

4~6: Observation well, Piz: Piezometer (N-20)

Decay of groundwater mound (Figs. 9 and 10)

About 10^4 minutes after the end of water supply to Basin No.2, we recorded decaying water table at observa-tion wells 4 to 6 and the piezometer. Below is the record of water table changes observed at observation well 5 for the three respective experiments (Fig. 9). In the first experiment, the decrease in water table was 2.49 meters as of 2,820 minutes (47 hours) and 3.71 meters as of 5,460 minutes (91 hours) in the third experiment, after stopping water supply. On the other hand, the decrease in the second case, 0.65 meters as of 2,785 minutes (ca. 46 hours) after stopping water supply, was not as much as in the first and second cases. This would be due to melting snow, whose period coincides with the end of March and beginning of April, and result in a bigger infiltration rate from the surface. In each case, groundwater mound decayed under Basin No. 2 as shown in Fig. 10, which is drawn in a longitudinal section along observation wells 4, 5, 6 and partly including the piezometer.
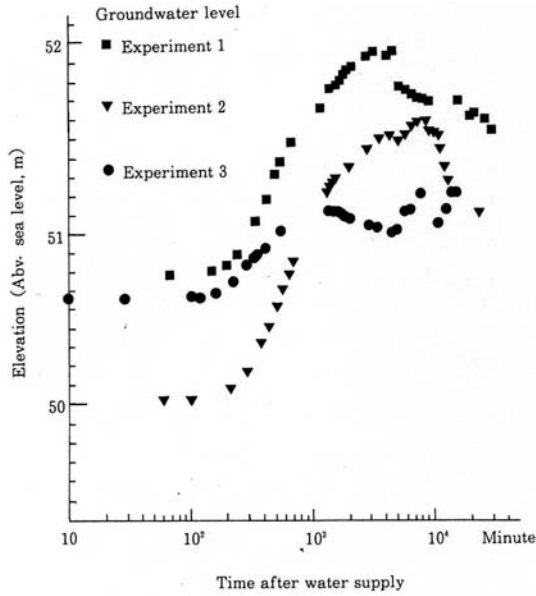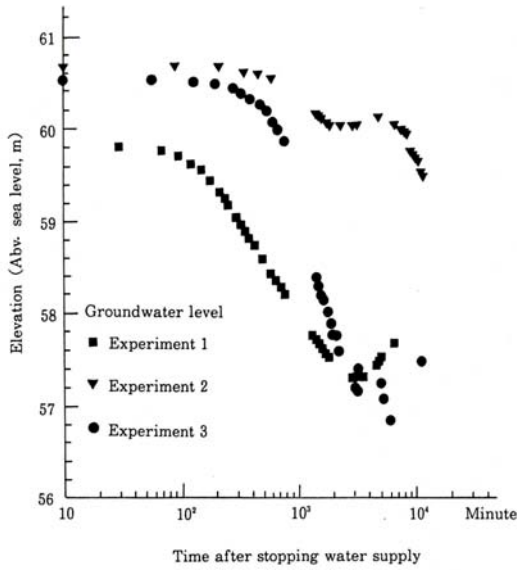
Figure 9. Decline of the groundwater levels after stopping water supply to artificial recharge basin (ARB) No.2

Experiment 1: Nov. 10~14, 1998
Experiment 2: April 5~12, 1999
Experiment 3: April 29~May 6, 1999
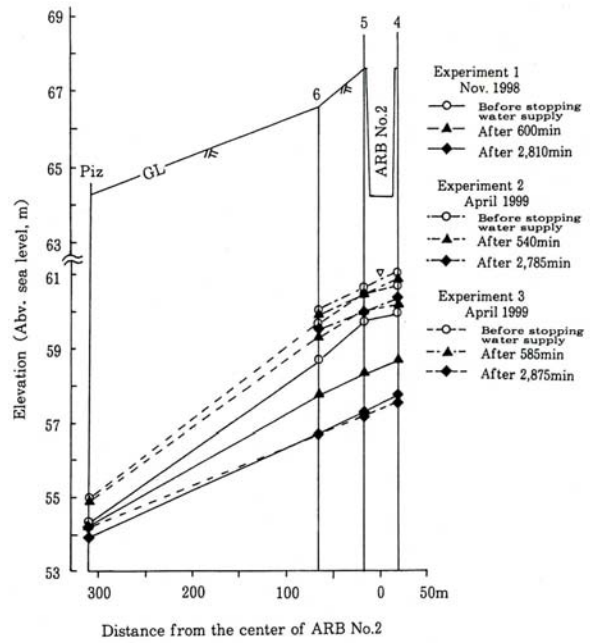
Figure 10. Changes of longitudinal profile of the water table between observation well 4 and piezometer: after stopping water supply to artificial recharge basin No.2

4~6: Observation well, Piz: Piezometer (N-20)

CONCLUSION

Groundwater mounds were formed under the bottom of the artificial recharge basins that were constructed on the center aria of the Rokugo alluvial fan. The enhancing of the groundwater cycle in the aquifer has resulted in sustainability for the groundwater environment especially in the distal fan.

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Investigations of alternative filter materials for slow sand filtration

Ulrike Hütter, Dominik Mueller-Töwe and Frank Remmler

Abstract

The aim of this study was to investigate the use of recycled crushed glass and coconut fibre as alternative low cost filter materials for their application using slow sand filtration (SSF). Pilot plant testing in laboratory and the field was used to compare the performance of recycled crushed glass and coconut fibres to a typical sand filter medium. The investigations evaluated the performance of both materials under defined conditions concerning temperature (5–10°C, 20°C, 30°C) and raw water composition (high concentrations of DOC and ammonium). Concerning the reduction of DOC, recycled crushed glass and sand, showed rather similar results whereas the coconut fibres were characterised by an organic emission increasing with higher temperatures. The degradation of ammonium was similar for all tested filter materials. The results of the investigations showed that the replacement of sand for the application using slow sand filtration, applying low cost local available filter materials, in general is possible. The alternative filter materials recycled crushed glass and coconut fibre have specific advantages and disadvantages compared to a standard filer sand used in Central Europe. These aspects have to be considered for their use for drinking water production under local conditions.

Keywords

Ammonium, coconut fibre, DOC, recycled crushed glass, slow sand filtration, temperature.

INTRODUCTION

For more than 100 years slow sand filtration (SSF) has been recognized in Europe as a natural and cheap method of water treatment, and in it's many years of use it has become a tested and scientifically well researched procedure. Beside the low waste production and the low energy and maintenance requirements, the ability to customize the procedure to individual requirements and conditions is advantageous.

The low cost use of suitable filter sands with reproducible characteristics for SSF is usual in Germany but not naturally practicable in other countries. The aim of this study was to investigate alternative low-cost filter materials in laboratory scale experiments as well as in larger scale experimental facilities for their application using slow filtration. The conducted experiments simulated climatic conditions and raw water compositions deviating from Central European circumstances to check whether slow filtration can be adapted to different conditions in other countries. A quality-orientated guideline for the adaptation of water treatment plants with slow filtration to local conditions different from Central Europe (for example, concerning climate, raw water composition, alternative low cost filter materials) is not yet in existence.

This paper reports on investigations made in the subproject 'Limitations of slow sand filtration, possibilities of technical modifications and adaptation to local conditions', which belongs to the central research project 'Optimisation and expansion of the application of slow sand filters'.

MATERIALS AND METHODS

After preliminary examinations the alternative filter materials 'recycled crushed glass' and 'coconut fibre' were chosen for further experiments. Both materials are readily available in many parts of the world. Water filtration using coconut fibres in combination with burnt rice husks is used e.g. in Southeast Asia and Pakistan (Frankel, 1981; Mughal, 2000). Recycled crushed glass was already investigated in different studies for its performance in a potable water treatment application (CWC, 1995, 1998; Evans et al., 2002).

In laboratory scale experiments the effects of temperature and filter material on the cleaning efficiency were investigated (Figure 1). The laboratory filter columns were constructed of glass tubing of 200 mm internal diameter and 1,000 mm length and designed to simulate slow filtration conditions (continuous operation with constant supernatant, filtration rate 2.2 m/d). The different filter materials (coconut fibre, recycled crushed glass, standard filter sand = reference material) were placed over a bed of gravel, which supported the media. A characterisation of the applied filter materials is given in Table 1. The experiments were carried out in an air-conditioned room at three different temperatures (5–10°C, 20°C and 30°C) with a filter run time of ca 6.5 weeks in each case. Before each test the filter materials were renewed. The filter influent (surface water: river Ruhr) was heated up respectively cooled down, aerated and in addition spiked with DOC (yeast extract) and ammonium (ammonium sulfate).

*Figure 1. Filter columns
in an air-conditioned room*

Table 1. Characteristics of the applied filter materials

	Sand medium	Recycled crushed glass medium*	Coconut fibre medium**
Media identification	River sand	Brown and green glass	Shredded coconut fibre
Particle size distribution [mm]	0.2–2	0.2–5	–
Effective size [mm]	0.28	0.55	–
Unconformity coefficient [–]	2.3	5.2	–
Depth installed in pilot plant column [mm]	500	500	500
Support gravel layer [mm]			
(particle size/layer depth)	3.15–5.6/80	3.15–5.6/80	3.15–5.6/80

* Preliminary treatment: washing and disinfection (2 h at 180°C)

** Fibre: average length ca 2 cm, average diameter ca 0.09 mm

The subsequent experiments conducted in a pilot scale field test plant (basin surface area: 4 m^2) focused on the influence of operating conditions (filtration rate, continuous or intermittent operation) on the filter performance (Figure 2). Controllable test conditions like influent composition (= spiked surface water), filter design and filter materials were comparable to the column experiments. For the simulation of climatic conditions deviating from Central Europe the surface water (= influent) was heated up to ca 30°C. During the operation time (in all ca 15 weeks) four different tests were conducted without a change of filter material in between:

Stage 1: continuous operation, filtration rate 2.2 m/d
Stage 2: intermittent operation, filtration rate 2.2 m/d
Stage 3: intermittent operation, filtration rate 3.8 m/d
Stage 4: continuous operation, filtration rate 1.0 m/d

Samples of influent and effluent of the filter columns and the pilot scale field filter basins were collected and analysed for temperature, conductivity, oxygen, oxygen reduction potential, turbidity, SAK_{254}, pH, metals (iron, manganese, copper, nickel, zinc), DOC, TOC, nitrate, nitrite, ammonium and sulfate. Sampling procedures and determination of chemical parameters were accomplished according to German standard methods (Drinking Water Ordinance, 2001). Microbiological parameters were analysed in another subproject of the central research project.
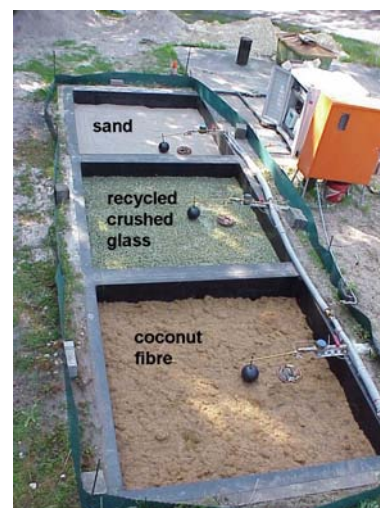
*Figure 1. Filter columns
in an air-conditioned room*

RESULTS AND DISCUSSION

The results of the experiments showed differences in the purification efficiency of the investigated filter materials depending on temperature and parameter. In Table 2 the mean concentrations of DOC and ammonium in the influent of the test plants are listed. With increasing temperatures the DOC and ammonium concentrations were reduced to avoid anaerobic filter conditions. Figure 3 illustrates the relative DOC and ammonium concentrations in the columns effluents (Coconut fibres were not investigated at the temperature 5- 10°C).

Table 2. Mean concentrations of DOC and ammonium in the influent of the test plants

Test plant and temperature	Influent (= spiked surface water)	
	DOC [mg/L]	*Ammonium [mg/L]*
Column tests		
5–10°C	7 mg/l	5.2 mg/l
20°C	mg/l	2.9 mg/l
30°C	5 mg/l	1.9 mg/l
Pilot scale field tests		
30°C	5 mg/l	1.4 mg/l

Concerning the reduction of DOC a comparable development of the relative concentrations was detected for the different filter materials (Figure 3, left part). The range of the C/C_0 – ratio grew with increasing temperatures. Recycled crushed glass and sand showed rather similar results whereas the coconut fibres were characterised by an organic emission (predominantly humates) increasing with higher temperatures. This effect became even more intense in den start-up stages of the tests. The main part of this organic emission of the coconut fibres consisted of humic substances.

The degradation of ammonium was similar for all tested filter materials (Figure 3, right part). The results proved the strong dependence of this process on the temperature. An optimum for nitrification is given at a temperature between 28 and 36°C. Below 12°C this process slows down and below 8°C it nearly stops. The pH value of the effluents ranged from 7 to 8 which is also an optimum for nitrification. The development of the concentrations of nitrate and nitrite on the other hand was different for the tested filter materials. A significant accumulation of nitrite

was determined for the coconut fibres only shortly and lower compared to the other materials. In the effluents of the recycled crushed glass and the sand the concentrations of nitrites were about two times higher as in the effluent of the coconut fibres. This effect was also determined at the pilot scale test plant. In contrast to the recycled crushed glass and the sand the coconut fibre filter showed a much lower accumulation of nitrate.
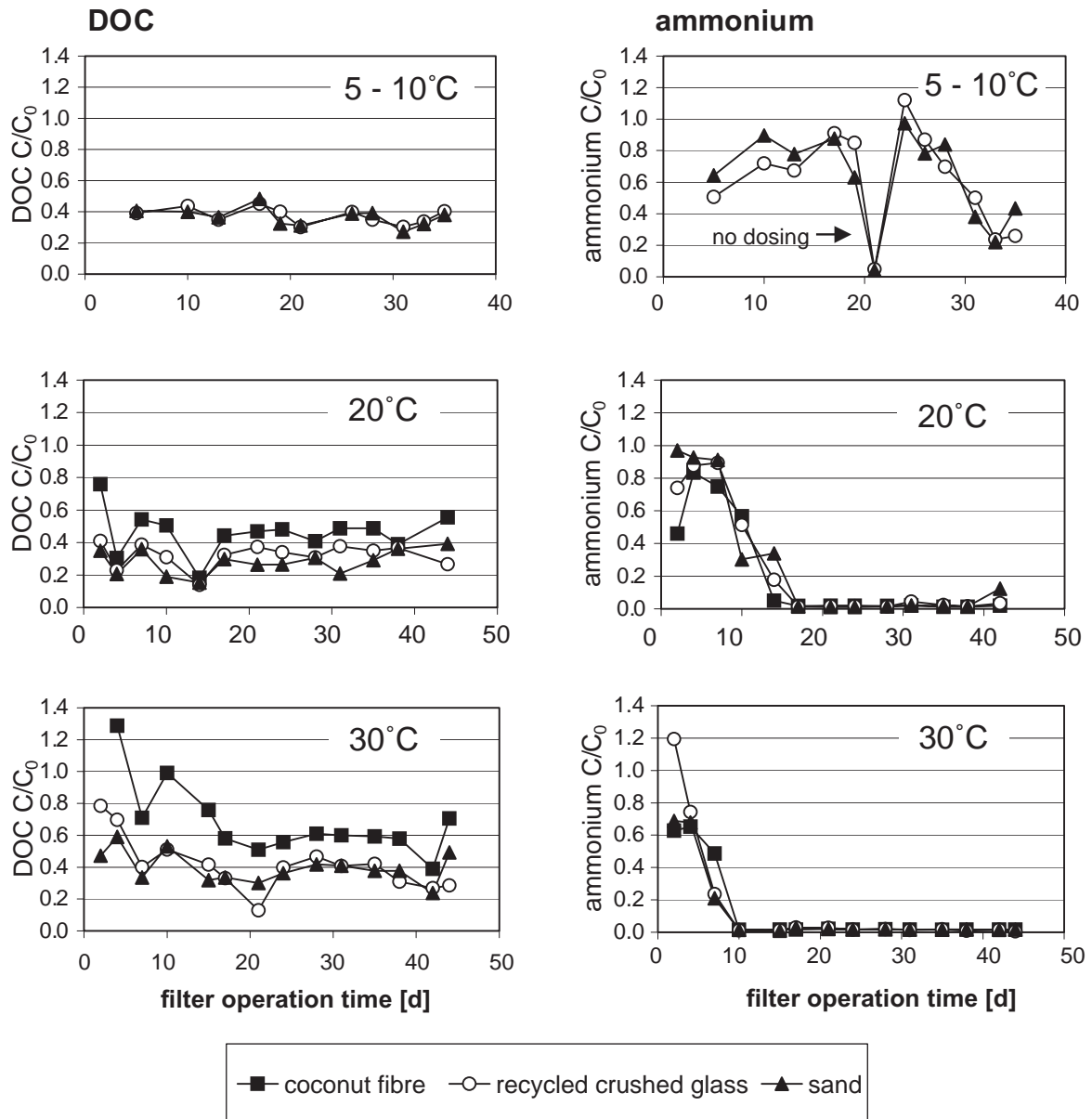
Figure 3. Relative concentrations of DOC and ammonium in the effluent of the filter columns

Coconut fibres and sand showed a comparable reduction efficiency concerning the removal of iron and zinc. For the recycled crushed glass a distinct lower elimination for zinc in general and for iron at the temperature 5–10°C was detected. The results of the investigations proved in all a slightly better reduction efficiency with increasing temperatures.

Especially in the start-up stages the filtrate of the coconut fibre was characterized by a pH value below 7. This organic filter material also showed at higher temperatures a higher oxygen reduction during the filtration process compared to the other filter materials.

The sand showed the highest headloss during all tests because of the particle size distribution (Table 1). The sand filter and the coconut fibre filter (high algae vegetation) had to be cleaned during the pilot scale field tests whereas the recycled crushed glass in all cases showed the longest filter run times. Evans et al. (2002) investigated a glass medium, that was similar to that of the tested typical sand medium (concerning effective size and unconformity coefficient). In that study the glass medium had the benefit of taking 10–15% longer than the sand to reach particle breakthrough, and it also appeared to accumulate headloss in most runs at a slightly lower rate than the sand.

The pilot scale field investigations proved that intermittent filter operation (stage 2 and stage 3) causes a high mobilisation of substances during restart stages. This effect was much more intense at the higher filtration rate in stage 3.

In Table 3 the achieved mean DOC and turbidity reduction in the different filter materials under the defined conditions at the test plants are given. The filtrates showed no great differences in turbidity; the recycled crushed glass and the sand filter had a slightly lower turbidity than the coconut fibre filter. All filtered water turbidity levels were within typical target levels for drinking water treatment processes, except in start-up stages or shortly in restart stages during intermittent operation. The results summarised in Table 3 prove for the recycled crushed glass and the sand that nearly similar reductions for DOC and turbidity were detected. The coconut fibre filter showed a little lower reduction efficiencies.

Table 3. Mean DOC and turbidity reduction of the filter materials

Filter material	Temperature	mean DOC reduction [%]	mean turbidity reduction [%]
Sand	5 - 10°C	60	60
	20°C	70	80
	30°C	60/45 – 55*	70/90 – 95*
Recycled crushed glass	5 - 10°C	60	65
	20°C	70	70
	30°C	60/40 – 45*	65/90*
Coconut fibre	20°C	55	65
	30°C	25/30 – 40*	25/85*

* Pilot scale field test plant.

CONCLUSIONS

The results of the investigations showed that the replacement of sand for the application using slow sand filtration, applying low cost local available filter materials, in general is possible. The alternative filter materials recycled crushed glass and coconut fibre have specific advantages and disadvantages compared to a standard filer sand used in Central Europe. These aspects have to be considered for their use for drinking water production under local conditions.

The results of these investigations completed with literature data will allow an assessment to be made for the application of the alternative filter materials in slow filtration under defined conditions (temperature, raw water quality). On the basis of this work a guideline for international water treatment companies and engineers will be prepared which contains the results of the joint project and presents important notes for the transfer of practical expertise outside of Germany.

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ACKNOWLEDGEMENTS

Groundwater recharge through cavity wells in saline groundwater regions

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Abstract

The north-western Indian State of Haryana is a part of Indo- Gangetic alluvial plains. About two third of the geographical area is currently underlain with saline groundwater and the situation is deteriorating further due to disproportionate pumping vis- a- vis groundwater recharge. In most of the marginally saline groundwater regions, low discharge shallow cavity wells are used for irrigation, which are inexpensive abstraction structures without a strainer. Deep tubewells are not feasible due to increasing groundwater salinity with depth while many shallow tubewells are abandoned due to upconing of salts from the deeper layers. Under these conditions, it is imperative not to disturb the saline water but to selectively skim fresh water accumulated over the native saline groundwater and by enhancing groundwater recharge.

The paper discusses the features of a combined skimming cum recharging system for marginally saline groundwater regions of Haryana. The system, consisting of two cavity tubewells installed at 7 m and 40 m depth in different quality zones and a recharging filter, permits separate or combined pumping from or recharging of one or both cavities. Salient results on performance evaluation of the system through pumping and recharging tests, geo- resistivity surveys and periodic observations of groundwater levels and quality are presented in the paper. It is reported that recharge rates through injection in cavity wells were low at about one quarter of the pumping rates under shallow groundwater conditions.

Keywords

Cavity wells; groundwater recharge; groundwater salinity; groundwater skimming; Indo-Gangetic plain; upconing.

INTRODUCTION

Excessive pumping of fresh groundwater to meet agricultural, municipal and industrial demands is causing critical groundwater salinization in several inland- irrigated regions of India. Such a situation exists in the alluvial plains of the north- western Indian states of Haryana, Punjab, Rajasthan and Uttar Pradesh. In Haryana, about two third geographical area is underlain with saline groundwater and the situation is deteriorating further due to disproportionate pumping vis- a- vis groundwater recharge. Deep tubewells are not feasible due to increasing groundwater salinity with depth while many shallow tubewells are getting abandoned due to *upconing* of salts from the deeper layers. Under these conditions, it is imperative not to disturb the saline water but to selectively skim fresh water accumulated due to recharge from rainfall, irrigation and/or canal seepage over the native saline groundwater through specially designed tubewells and by enhancing groundwater recharge. The basic concept of all skimming structures is to modify the flow lines in such a way to maximize horizontal contribution of aquifer zones of acceptable quality to pumped water (Sufi et al., 1998). Different variants of skimming structures tested in alluvial unconfined plains are multi-well point systems installed close to canals/distributaries in Punjab (Shakya, 2002) and in the Indus plains of Pakistan (Sufi et al., 1998; Mazhar Saeed et al., 2003), radial collectors wells in coastal sandy regions in India (Raghu Babu et al., 2004) and scavenger wells which involve simultaneous abstraction of fresh and saline waters through two wells having screens in different quality zones, for controlling the rise of interface. The

scavenger wells have been tested in the lower Indus basin of Pakistan (Sufi et al., 1998) and have shown their potential in skimming of fresh water despite apparent problem of disposal of sizable quantities of saline water.

CAVITY WELLS

Haryana has two distinct topographical and hydro- geological settings: high water yielding fresh groundwater areas where rice- wheat is the dominant cropping sequence and the saline groundwater regions where aquifers of relatively poor transmission characteristics occur. The number of private shallow tubewells has increased twenty fold to about 0.6 million over the last four decades. The sustainability of state's groundwater resources is threatened due to rise of water levels in the saline areas and alarming decline in the fresh water areas forcing farmers to regularly deepen the tubewells or resorting to submersible pumps with significantly higher energy costs.

In most marginally saline groundwater regions of Haryana, low discharge (5–10 litre per sec) shallow cavity wells are used for irrigation (Boumans et al., 1988). These inexpensive structures do not require a strainer or a gravel pack and are constructed by drilling a hole until a sandy layer is encountered below a layer of stiff clay. After retracting the casing pipe into the clay layer, sand is pumped out through a centrifugal pump until a stable cavity is developed below the clay layer and clear water is obtained. The life of cavity tubewell varies from a few months to almost a decade or more depending upon the quality and thickness of clay layer and grain size of the sandy zone. The farmers operate the well for several hours per day to skim the least saline water and stopping when salinity of pumped water increases. Despite low transmissivity (< 500 m^2/day) of these aquifers, there is considerable scope and need to enhance groundwater recharge and improve groundwater quality by injection through cavity wells (Taneja and Khepar, 1996). This paper presents the features and field evaluation of a combined groundwater skimming cum recharging system consisting of two cavities installed in different quality depth zones. The system is similar in features to a scavenger type skimming structure discussed above but consists of cavity wells instead of strainer tubewells. Keeping in view the short life of cavity wells in the study area, groundwater recharge component was added to the skimming structure to improve sustained supply of reasonably good quality pumped water.

GROUNDWATER SKIMMING CUM RECHARGING SYSTEM

The test site was selected at village Jagsi- Sarfabad (block: Saffidon) in Jind district of Haryana after detailed geo- hydro- chemical characterization of the area, interaction with farmers and field tests. The field investigations included test borings up to 30 m depth and collection of soil and water samples of different zones. Particle size distribution, electrical conductivity (EC), pH, ionic composition of water and soil samples and resistivity profiles were determined to characterize the lithology and depth- wise groundwater quality. Kamra et al. (2004) presented the soil texture, chemical characteristics of groundwater and resistivity profiles of three test bore locations at Jagsi- Sarfabad. Based on the horizon- wise texture, thickness of clay layers and chemical characteristics of soil and water samples, scavenger well type skimming structure consisting of two cavities at different depths was initially considered. However, the groundwater at the chosen site for the skimming cum recharging structure had marginal salinity (EC < 4 mS/cm) upto 17 m depth during June 2001; its quality deteriorated between 17−30 m depth in terms of salinity (EC: 5- 6 mS/cm) but more seriously with respect to residual sodium carbonate (RSC). Considering such adverse conditions, groundwater recharge component was incorporated into the original skimming scheme.

A skimming cum recharging system was constructed at a downstream location prone to runoff flooding during rainy season. The system (Fig. 1) consists of two cavity tubewells installed at 7 mand 40 m in the respective fresh and saline groundwater zones. The two cavities can be operated separately or together to obtain water of different qualities. A recharge chamber of 30 m^3 (6 m x 2.5 m x 2 m) capacity and containing a graded filter of fine sand, coarse sand, gravel and boulders was constructed close by to facilitate recharging of one or both cavities with

filtered surface runoff during rainy season or excess canal water. The objective was to increase the availability of good water in upper cavity or improve the quality of lower cavity through dilution for possible use either directly or after treatment with gypsum. During installation, the deep cavity was finally successful at 40 m where water with little RSC problem was available. Temporal changes in quality of shallow and deep cavities during 2003–2004 are summarized in Table 1. The quality of pumped water during simultaneous operation of two cavities can be controlled through gate valves.
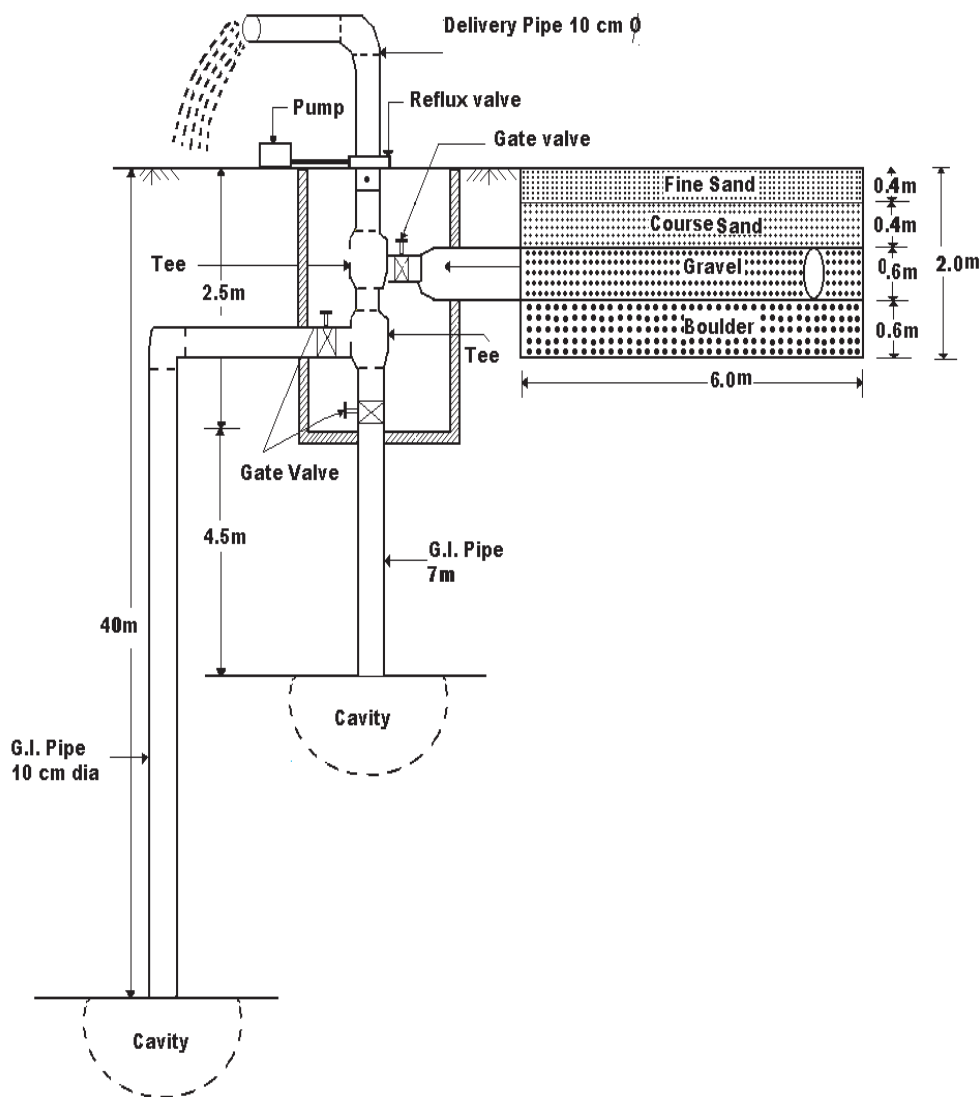
Figure 1. Groundwater skimming cum recharging system at Jagsi-Sarfabad (Haryana)

Table 1. Groundwater quality of shallow and deep cavity tubewells

	EC (mS/cm)			RSC			PH		
Cavity	Oct. 2003	Feb. 2004	Oct. 2004	Oct. 2003	Feb. 2004	Oct. 2004	Oct. 2003	Feb. 2004	Oct. 2004
Shallow	2.0	1.8	1.4	4.0	2.6	2.1	8.1	7.9	8.0
Deep	7.8	4.2	3.8	0.2	0.5	0.7	8.4	8.4	8.5

Hydraulic evaluation of the system

The hydraulic impact of the system was evaluated through pumping and recharging tests, observations on groundwater levels and quality, changes in pumped water quality and periodic resistivity surveys. For shallow cavity at 7 m depth, the clay layer of about 30 cm thickness is available at about 6.5 m depth. Kamra et al. (2004) describe the feature of a network of 32 observation wells installed at radial distances of 5, 15, 30, 60 and 120 m, in 16 pairs at 5m (above clay layer) and 8 m (below clay layer) depth. During these tests, water table depth was monitored only in observation wells but not in well pipe; water quality was monitored both in observation wells as well as of pumped water.

Figure 2 presents part results on the variation in water table depth and associated changes in groundwater salinity (EC) during pumping cum recovery test conducted during July 2003 and recharge studies performed during November 2004 on the shallow cavity. These and similar results are being analyzed to estimate aquifer parameters including resistance of clay layer, area of influence of cavity wells and for application of hydro-salinity models. Analytical approaches of Taneja and Khepar (1996) and Sharma et al. (1987) were applied for pumping phase (Fig. 2a) data in cavity wells to estimate mean hydraulic conductivity of shallow aquifer as 14 m/ day within a range of 10.6 to 17.5 m/day (NATP, 2004). The corresponding average value of specific storage – coefficient was 1.5×10^{-4} m^{-1}; its range was quite wide indicating the limitation of analytical approaches. Small values of EC in observation well at radial distance (r) of 15 m vis-a-vis other observation wells (Fig. 2b) are probably allied with the orientation of the shape of the developed cavity in relation to the direction of groundwater flow.

Due to manual operation of all gadgets and large distance between the work and study site, recharge studies during actual rain events could not be conducted. A series of recharge studies were, however, conducted by transporting pumped water with flexible PVC pipes either from a nearby tubewell (within the radius of influence of cavity to study interference phenomenon) or from a distant tubewell. During different tests, the recharge was first initiated through the bed of the recharge chamber and later through injection in one or both cavities. The results of one such test for shallow cavity (Fig. 2 c,d) indicate restriction of the impact of recharge on water table regime and quality to within a small radius of 5 m. The injected water, brought from a distant tubewell, had a mean EC of 2.5 mS/cm during the course of the test, almost half of the initial groundwater salinity at the recharge site (Fig. 2d). It is pertinent to explain here that most of the natural groundwater recharge in the area occurs during the rainy (July–October, *South West Moonsoon*) season. This recharge replenishes the shallow groundwater in terms of quantity as well as improvement in quality as illustrated by depth and EC of groundwater during and post rainy seasons of 2003 and 2004 (Fig. 2a,b,c,d). Relatively higher groundwater salinity during November 2004 (Fig. 2d) is reflective of *upconing* of saline water from deeper zones due to excessive pumping by farmers to meet water requirements during post rainy season. Considering small volumes of injected water during about two hour duration of this and other tests, the results should be treated as indicative only.

The estimated recharge rates of different studies have been summarized in Table 2. It is seen that the recharge rates of 2 to 3 litre per sec through shallow cavity were low at about one quarter of the pumping rates while of deep cavity were very low at less than one litre per sec. It seems some sort of pressurized injection may be needed to enhance well injection rates under high groundwater conditions. Specific studies on clogging and efficiency of different recharge filters are needed, particularly in relation to the role of fine sand.

Table 2. Recharge rates through injection in shallow and deep cavities

S	.	Recharge rates (litre per sec) through		
No.	Water table depth (m)	Recharge filter bed	Shallow cavity	Deep cavity
1.	2.5	0.5	2.2	0.2
2.	4.0	0.7	3.0	0.9
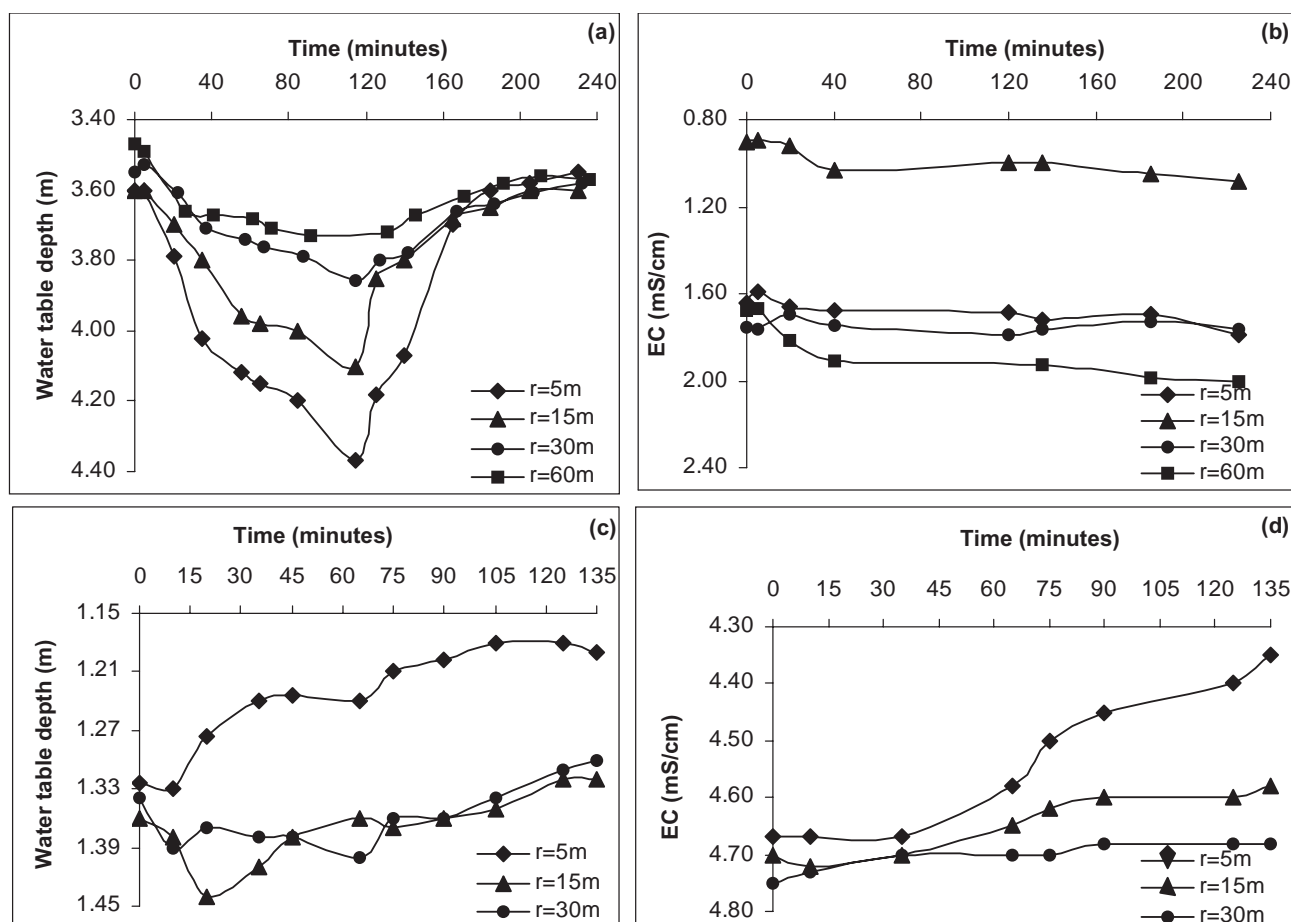
Fig. 2. Changes in water table depth and electrical conductivity (EC)
at different radial distances during pumping and recovery test (a, b) and recharge studies (c, d)
[Pumping stopped after 120 minutes in pumping test (July 2003); gate valve of shallow cavity opened
after 50 minutes in recharge study (November 2004)]

The improvement in groundwater quality in the area was evaluated through periodic resistivity surveys and quanti-fying the area under different apparent resistivity zones. It is reported (NATP, 2004) that there has been a gradual increase in the aquifer zone representing good to marginal groundwater from 31 to 54 % over a period of two and a half years due to combined effect of the natural and imposed recharge interventions. The farmers in study area have started directing excess runoff, whenever available, to cavity tubewells through the reflex valves without filtration, though filtration is crucial. The total cost of the system is Rs. 50,000/(US $ 1,100) including Rs. 30,000/(US $ 660) for the pumping unit and Rs. 20,000/(US $ 440) for the recharging components.

CONCLUSIONS

Indo-Gangetic alluvial state of Haryana in India has distinct fresh groundwater regions with declining water table trends and saline groundwater zones with continuously rising water tables. A number of activities and projects on groundwater recharge have been executed by governmental agencies and NGOs to arrest and sustain the ground-water decline in falling water table areas. The number of such projects in saline/ shallow groundwater regions is almost negligible. In this paper, features of a skimming cum recharging system have been presented for a mar-ginally saline shallow groundwater area in Haryana. The system permits separate or combined pumping from or recharging of two cavity tubewells installed at different groundwater quality depths. Due to shallow groundwater, the recharge rates are low but it is recommended to undertake pilot studies on pressurized injection considering the

vast extent of brackish/saline groundwater in India. Pressurized well injection may also find application in disposal of treated waste/sewage waters. Incorporation of properly designed inexpensive filters in the existing or abandoned cavity or strainer tubewells belonging to farmers with small land holding can significantly enhance groundwater recharge in these areas.

ACKNOWLEDGEMENT

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Variability and scale factors in artificial recharge from the Deccan basalts of India: a case study from Kolwan valley of Pune district in Maharashtra, India

Himanshu Kulkarni, Uma Badarayani and Vinit Phadnis

Abstract

The inhomogeneous nature of the Deccan basalt in central-west India offers a great challenge in understanding the hydrogeology of these rocks. The disposition of basalt lava units, their weathering and fracture patterns and their setting within a watershed or a river basin hold many answers to understanding groundwater resources, especially artificial recharge, in these rocks. Scale and variability are important factors in artificial recharge studies in the Deccan basalts. The variability in aquifer conditions is clearly evident only on the scale of a single recharge structure, or at the scale of a single aquifer. A typical microwatershed includes more than one aquifer and hence, the natural and artificial recharge mechanisms are far more complicated to be reflected in a simple watershed scale water budget. River basin budgets have other intricacies that are significantly different from the artificial recharge structure or watershed scale budgets. Only integrated studies of water resources on these three scales help understand complexities in conditions controlling artificial recharge in Deccan basalt areas and more so in highlighting limitations in regional studies.

Keywords

Deccan basalt, hydrogeology, microwatershed, scale, variability, water budget.

INTRODUCTION

The Deccan Volcanic Province in central-west India is a unique hydrogeological setting representing a major 'hard rock' groundwater province. The Deccan basalts cover a an area in excess of 5,00,000 km2. Their thickness is considerable, with estimates for the Saurashtra region being 0.36 to 1.3 km (Kaila et al, 1981). A large portion of the Deccan basalts outcrops in the state of Maharashtra, although it also extends into the adjoining states of Gujarat, Madhya Pradesh, Karnataka and Andhra Pradesh (Figure 1). The type of basalt, its degree of weathering, and the nature and intensity of fracturing control groundwater accumulation and movement in the Deccan basalt
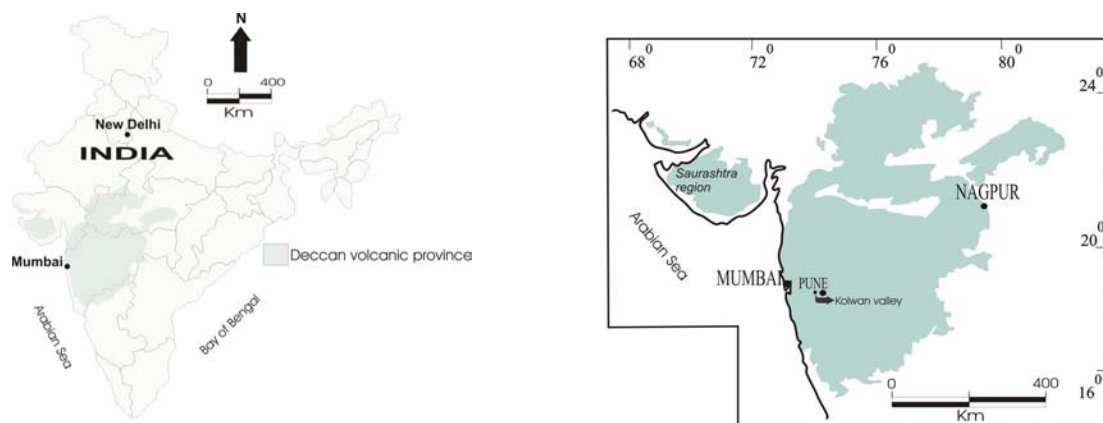
Figure 1. Deccan volcanic province of India

(Deolankar, 1980; Kulkarni et al., 2000). Hydrogeologically, the Deccan basalts are quite inhomogeneous in nature with highly variable conditions controlling groundwater occurrence and movement. This is evident from the fact that degree of weathering and intensity of fracturing are often quite different even in adjacent wells.

Water budget studies in the Deccan volcanic province are increasingly gaining importance on the back of large-scale watershed development activities, especially in regions underlain by hard-rocks. The hydrogeological nuances of the basalt rocks make water budget calculations or water audit estimates a challenging task. The appropriateness of computing water budgets in this region depends significantly on factors like scale of study, variability, relationship between meteorological, hydrological and hydrogeological units, and the process of estimating various parameters of the water budget. The work by Sutcliffe (2004) was especially useful in this regard. This paper presents water budgets on three different scales, especially highlighting estimates of natural and artificial recharge and how they relate to other components of the water budgets on different scales of study. The results presented in this paper are part of a larger study of the Kolwan valley, along with three other sites in India, called 'Augmenting Groundwater Resources by Artificial Recharge' (AGRAR), funded by DFID-UK and led by the British Geological Survey, UK.

METHODS

Monitoring of water resources, including different types of primary and secondary data collection, formed the basis of the AGRAR project, and therefore also for this study.

Setting

Water resources development in the Walki river basin (commonly and henceforth referred to as Kolwan valley) from Mulshi taluka of Pune district in the state of Maharashtra was strategised on three scales. These three scales are compatible with the scales on which artificial recharge measures are undertaken in India and can be described as:

- Several microwatersheds constitute a river basin that may cover areas of the order of 100 km^2 and constitute the more 'regional' unit.
- A microwatershed is a small catchment of less than 20 km^2 area, forming a part of a larger river basin (1 above) considered as the primary unit for watershed development, wherein several individual structures like small check dams are constructed.
- Masonry and earthen check dams constructed for harvesting runoff and spreading it over the surface, in anticipation of recharge to groundwater during the dry season forms part of watershed development programmes. One or more such check dams are located within a microwatershed (2 above).

The Kolwan valley forms a perfect setting to study various aspects of artificial recharge on the three scales mentioned above. It is entirely constituted of Deccan basalt lavas that erupted some 65 million years ago. The lava 'flows' vary in thickness from a few meters up to 10s or even 100s of meters. Each lava flow can further be divided into sub-units. In general, the Deccan basalts can be grouped into two categories, 'simple' or 'compound' depending on the viscosity of the primary lava (Deshmukh, 1988; Kale and Kulkarni, 1992). Basalt lavas in Kolwan valley are mainly of the compound type. The compound basalt units observed in Kolwan valley are similar in their physical characters to those reported from other areas in the Deccan volcanic province (Bondre et al., 2000).

Figure 2 is a geological map of Kolwan valley showing the disposition of horizontal lavas in the Walki river basin. The distinction between different basalt units is based on research conducted over the last twenty years on the hydrogeological characterisation of Deccan basalts, summarised in Kulkarni et al., 2000. Further, regional linear features in the form of fracture zones and dykes often transect a sequence of basalt lavas. These lineaments in the Kolwan valley, were identified, using remote sensing data and field checks (Kulkarni et al., 2003).
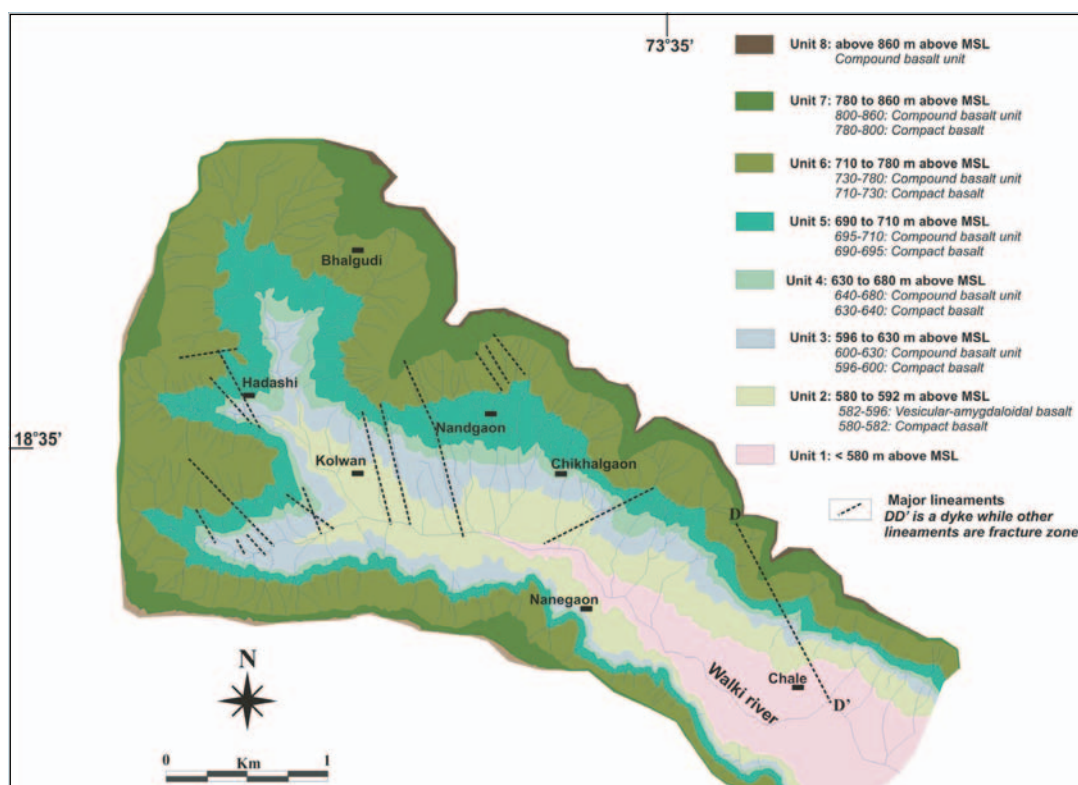
Figure 2. Geology of Kolwan valley

A microwatershed in Kolwan valley, is therefore constituted of 6 to 8 such basalt units. These units constitute five aquifers in the individual microwatersheds. Springs at higher elevations in such watersheds bear testimony to the presence of such aquifers even at higher elevations. The compound basalts are usually quite weathered and sheet jointed. They are underlain by dense compact basalt subunits, which are commonly sub-vertically jointed in their upper portions. The sheet jointed portions of the compound basalt and the subvertically jointed upper part of the underlying compact basalt together form shallow unconfined aquifers in the Kolwan valley. The upper jointed parts of these compact basalts grade downward into more unjointed, dense portions marking the base of many hydrogeological units in the area.

Studies on the scale of the Walki river basin, the Chikhalgaon watershed and one check dam in the Chikhalgaon watershed were used to understand the scale and variability issues in artificial recharge studies in the Deccan basalt.

Monitoring network and data

Monitoring on the scale of the river basin included data from 6 rain gauge stations, 2 pan evaporimeters, 30 dug wells and 9 check dams, including two on the main river channel of the Walki. An automatic weather station consisting of a 12-channel data logger was installed at Chikhalgaon to record rainfall, air-temperature, relative humidity, solar radiation, evaporation and wind-velocity (Kulkarni et al., 2003). Additionally, staff gauges were installed at the check dams to record temporal changes in the stream or river stage in response to various factors such as rainfall, evaporation, infiltration, releases from irrigation dams upstream (for two of the check dams). A stilling well with a continuous water level recorder was installed to record high-frequency stage data for CD3, in addition to staff gauge data.

Using ACWADAM's experience in setting up hydrogeological monitoring processes in the Deccan basalt region as

well as in other parts of India, a systematic groundwater monitoring process was set up in the Kolwan valley using all dug wells and the network of boreholes close to CD3. The monitoring infrastructure was established on the basis of the AGRAR guidelines for fieldwork (Gale et al., 2003), with practical considerations and standards provided by Gunston (1998). Relevant data such as rainfall, evaporation, differences between inflows and overflows, aquifer transmissivity and specific yield, infiltration rates, water table contour maps etc. were used to derive water budgets on the three scales of the study. The data set from November 2003 onwards was used to estimate broad water budgets on the three scales of study. It is difficult to describe each of these in detail but some salient features of the study are presented here, with summarized budgets on the three scales. Figure 3, for instance shows the relationship between rainfall, water levels in the shallow aquifer (represented by borehole water level data) and the stage in CD3 between December 2003 and April 2005.
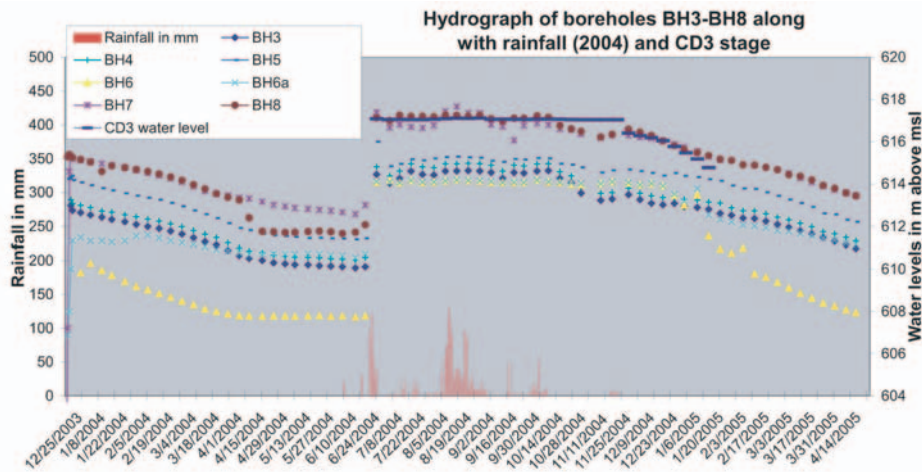
Figure 3. Time series data-rainfall, water levels in boreholes and CD3

RESULTS AND DISCUSSION

A comprehensive analysis of all data is under progress (Kulkarni et al., *in preparation*). We present here results from data that highlight how data on three different scales shows variability in various components of a water budget.

Recharge structure scale

Monitoring of the stage in the structure CD3, water levels in the observation boreholes, wet and dry season water budgets are discussed in a separate publication in this set of proceedings (Badarayani et al., *in press*). The structure CD3 fills up rapidly after the onset of monsoon. It takes two to three days, for the three check dams to fill up. They continue to top over throughout the rainy season, and hence, remain at flood level or maximum level for a period of over 100 days. Figure 4 shows the decline of water level in CD3 following the rainy season.

Tables 1 and 2 summarise the water budget on the scale of a single recharge structure, using wet and dry season results respectively. These results are a 'first cut' estimate of measured data and will be refined over the course of the next season when monitoring in the area continues. Nevertheless, they are indicative of what happens around the recharge structure (CD3) during the wet and dry seasons.

The annual infiltration of 111 mm of water from the structure may not necessarily imply that all of this is effective recharge to the aquifer. Evidence in the structure downstream, i.e. CD2, points to the possibility of a portion of this infiltration emerging as base flows in downstream areas (Kulkarni et al., *in preparation*), therefore clearly pointing to the necessity of understanding that infiltration of water from artificial recharge structures is far more complicated,

when it comes to quantifying 'effective recharge' to the aquifers, even in close proximity to individual recharge structures.
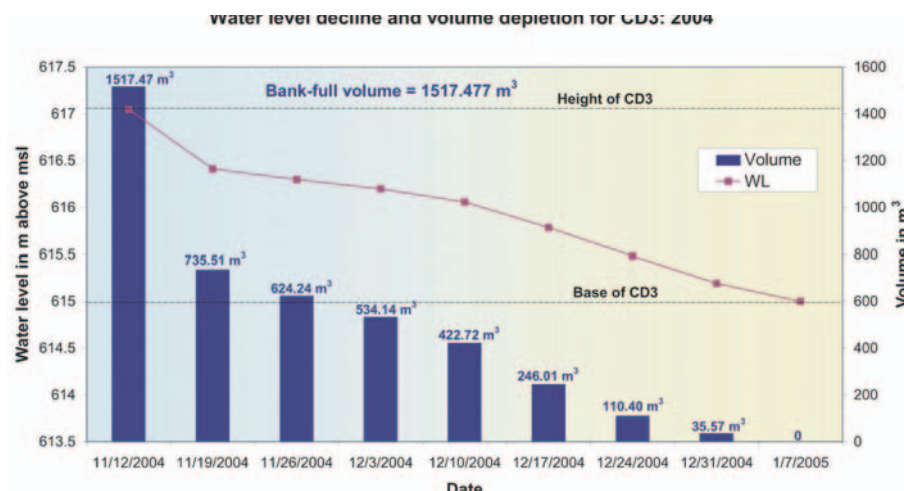
Figure 4. Water level decline in CD3 during the post-monsoon period of 2004

Table 1. Water budget for structure CD3 for the period of its overflow

Component	Quantity	Proportion to rainfall
Rainfall	1,860 mm	100%
Inflow	2,420 mm	130%
Overflow	2,087 mm	112%
Evaporation	222 mm	11.9%
Infiltration	110 mm	5.9%

Table 2. Water budget for structure CD3 for the period of water level decay

Component	Height of water column in structure (m)	Reduced to the common base of the catchment area of structure	Proportion to volume of structure	Proportion to rainfall
Storage	1.785	1.57 mm	100%	0.08%
Evaporation	0.315	0.28 mm	17.8%	0.015%
Leakage	0.2	0.18 mm	11.46%	0.0097%
Infiltration	1.27	1.11 mm	70%	0.06%

Microwatershed scale

Not all of the check dams in the Chikhalgaon watershed facilitate recharge because some of them are constructed in natural groundwater discharge areas. Of the three check dams, CD1 and CD2 are located in natural discharge areas whereas CD3 is located in the vicinity of a natural recharge area. Groundwater levels in the aquifer in the vicinity of such dams remain higher than the stage height of water in the dam throughout the year. However, if 2 or 3 such structures are located along a stream, groundwater recharge is likely to result from the structures located in natural recharge areas.

Table 2 is a water budget developed for the Chikhalgaon watershed using data collected on the microwatershed scale. The data included measurement of rainfall, flows over the check dams (Figure 5), evaporation, and water levels in dug wells and observation boreholes. The infiltration is a sum total of natural and artificial infiltration to the set of five aquifers exposed as a layered sequence of horizontal basalt units over the microwatershed area. Out of 200 mm, some 111 mm of infiltration occurs in the Chikhalgaon aquifer, whereas the remaining gets distributed in the other aquifers in the microwatershed (Badarayani et al., *in press*). The value of infiltration for the microwatershed is in agreement with the value of infiltration studies in basalt aquifers from other areas (Athavale and Rangarajan, 1990; Sukhija et al., 1996).
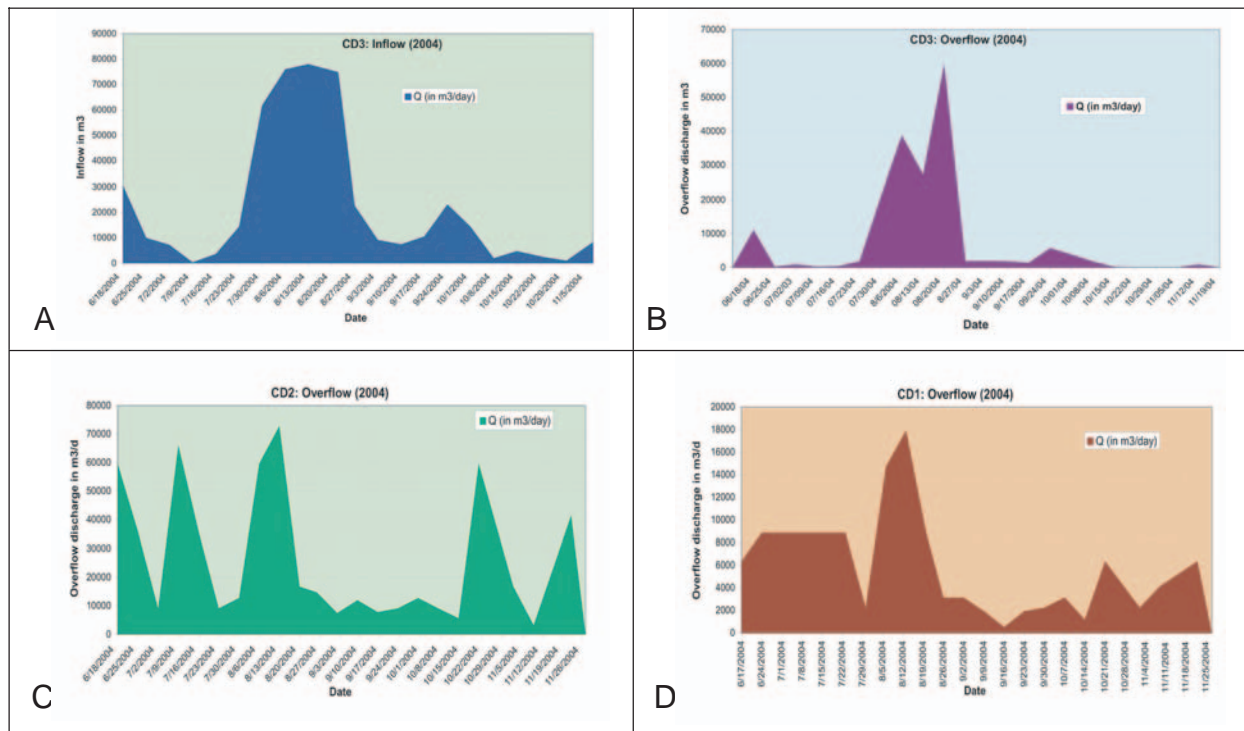
Figure 5. Flow over check dams in Chikhalgaon watershed. Inflow into CD3 is shown in A, whereas B,C and D shows overflows from CD3, CD2 and CD1 respectively

River basin scale

The Walki river basin receives high but variable rainfall. Rainfall varies from as much as 2,600 mm per year in the upper reaches of the basin to as less as 1,300 mm in the lower reaches, i.e. to the east. The average rainfall for Kolwan valley from the six raingauge stations was 1920mm during the period of continuous record, i.e. between May and November 2004. The spatial and temporal variability is illustrated through the two graphs presented in Figure 6. Table 1 presents a broad water budget for the Walki river basin, based upon results obtained from the data collection network described above. Despite the three minor irrigation tanks and about 10 other structures, surface runoff constitutes the single largest component of the water budget in the Kolwan valley.

Infiltration here includes infiltration from natural and artificial sources (based on the other scales of study). However, working on a river basin scale often requires generalizations. For instance, in this case, infiltration at various levels where favourable conditions exist have been assumed on the basis of direct measurements made at a few specific locations. The infiltration of 200 mm, which is more than 10% of the rainfall is slightly higher than that estimated for larger river basins of peninsular India (Athavale and Rangarajan, 1990).

A

B
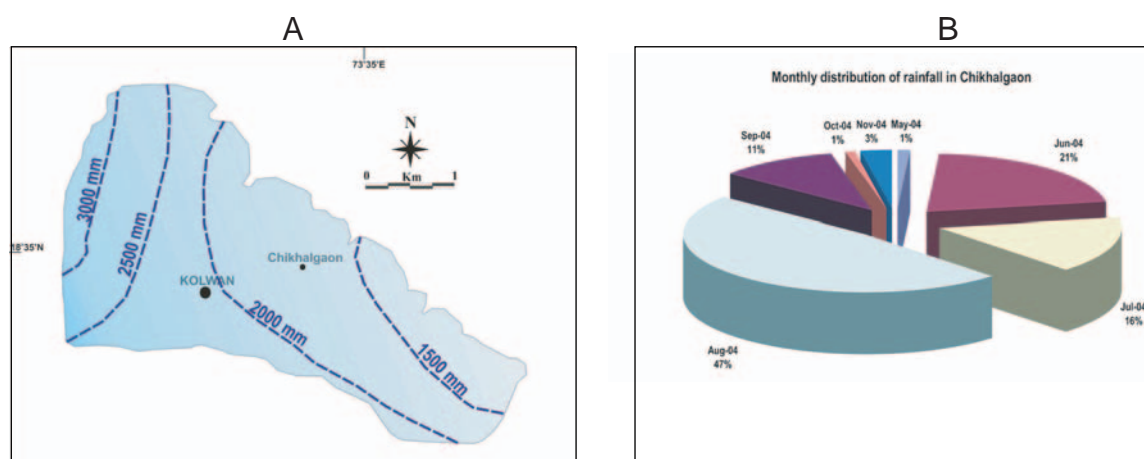
*Figure 6. Isohyets (A) for Kolwan valley showing annual rainfall variability
and monthly rainfall variability for the year 2004 (B)*

Table 3. Water budget for Walki river basin

Component	Quantity	Proportion to rainfall
Rainfall (average from 7 stations)	1,920 mm	100%
Soil moisture	50 mm	2.6%
Infiltration	200 mm	10.41%
Pumping (mainly from rivers)	100 mm	5.2%
Surface runoff and evapotranspiration	1,470 mm	76.56%
Average annual input to minor irrigation tank storages	110 mm	5.73%

CONCLUSIONS

The diversity in hydrological and hydrogeological conditions in the Kolwan valley, including variability in the rainfall across the river basin is clearly evident from the results of the study on artificial recharge on three different scales. Although understanding the processes of recharge on the three scales is not simple, studies at the aquifer or recharge structure scale reveal that artificial recharge takes place from some of the structures, although the magnitude of augmented water is limited. Surface runoff is the single largest component of water balances on all three scales, significantly different from water balances in other areas of the Deccan basalt where groundwater abstraction is the single largest component of the water balance (Macdonald et al., 1995). A comparison of these two extreme scenarios from the Deccan basalt is presented as conceptual diagrams (Figure 7).

Small structures meant for artificial recharge in microwatersheds contribute small quantities to the overall water budget of the microwatershed or river basin. However, in limited storage aquifers (such as the Chikhalgaon aquifer), where the total aquifer storage is less than 100 mm, this may be a significant contribution where groundwater resources are concerned, despite the relatively small proportion of infiltration to the rainfall or runoff.
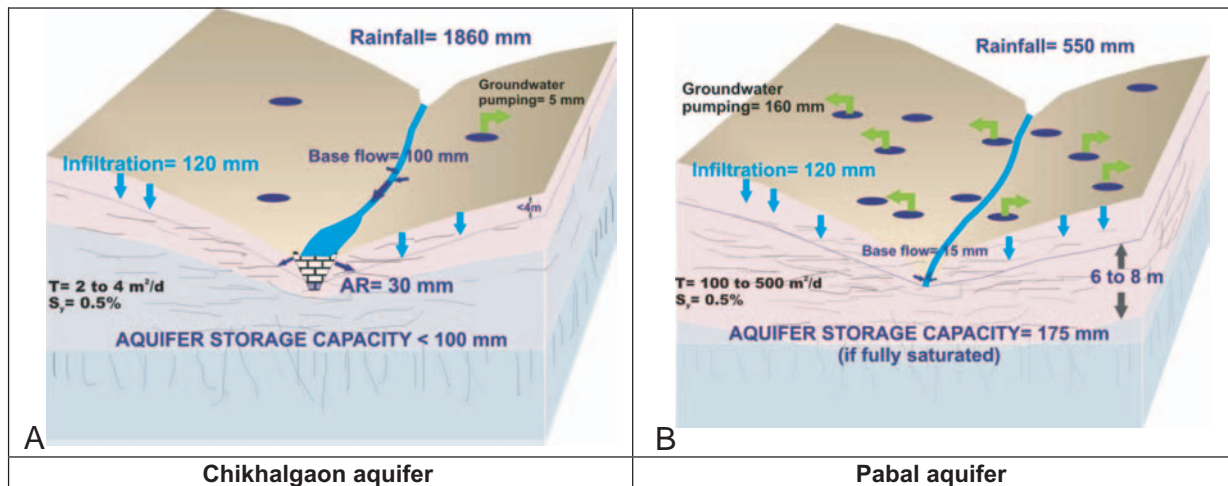
A — **Chikhalgaon aquifer**
- Limited groundwater abstraction;
- Infiltration is about 6% of annual rainfall;
- Infiltration results in recharge and increased baseflow.

(after Macdonald et al., 2005)

B — **Pabal aquifer**
- Case of groundwater overabstraction;
- Infiltration is some 25% of annual rainfall;
- Very minor baseflow.

Figure 7. Comparison between Chikhalgaon and Pabal aquifers from the Deccan basalt

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Evaluation of the strategies for the re-equilibrium of the groundwater balance of an overexploited aquifer (Prato, Italy)

F. Landini, G. Pranzini and M.E. Scardazzi

Abstract

The aim of this study is to understand the hydrologic mechanisms effecting the recharge of the over-exploited alluvial fan Prato aquifer (Tuscany, Italy) and to evaluate strategies for the re-equilibrium of the groundwater balance. The Prato aquifer plays a key role for the water supply of the Pistoia-Prato-Florence urban area, with a total drinking water demand of more than 100 millions of cubic meters per year. A series of strategies has been evaluated, acting on both the input and the output terms of the groundwater balance. Several field investigations have been carried out to acquire information for an artificial recharge trial program in the northern part of the fan body. As a tool to validate such experimental plan a physical model has been developed and tested both in steady state and transient conditions. The results of an artificial recharge scenario simulation (400 l/s for 8 months a year), for the next 10 years, show that such a recharge program will allow the recovery of the ground-water levels, although only 120 l/s are sustainable for the system at the recharge selected site. For this reason multiple recharge sites along with new strategies are becoming necessary to achieve the re-equilibrium of the balance.

INTRODUCTION

The intent of the project for the recovery of the groundwater levels of the Prato (Tuscany) (Figure 1) aquifer is to define the feasibility and the effectiveness of different strategies aiming to enhance the recharge of the aquifer by improving the positive term of the water balance, such as river bed losses and vertical infiltration. Several studies have been undertaken to plan and develop long term policies also for decreasing pipeline losses and for a 50% reduction of well abstractions by promoting the re-use of treated wastewater, for non civil purposes. These objectives are included within the 'Programs of the water resources protection' of the Water Protection Plan of the Region of Tuscany – to be adopted according to the UE Framework Directive 2000/60, the D.Lgs. 152/99 in force in Italy and the 'Master Plan' of the Regional Water Regulation Authority.

Figure 1. Area of study

The Prato aquifer is mainly formed by an alluvial fan body (Figure 2), which contains an upper unconfined layer, strongly overexploited (Figure 3), underlayed by gravel lenses of semi-confined or confined minor aquifers (Figure 4).
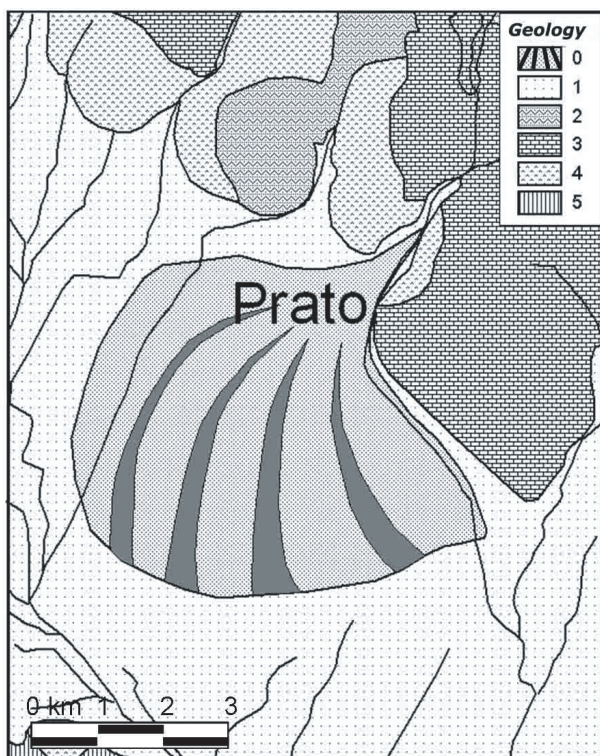
Figure 2. 0. Bisenzio river fan delta.
1. alluvial deposits; 2. ophiolitic deposits;
3. limestones; 4. claystones and sandstones;
5. turbidite sandstones.
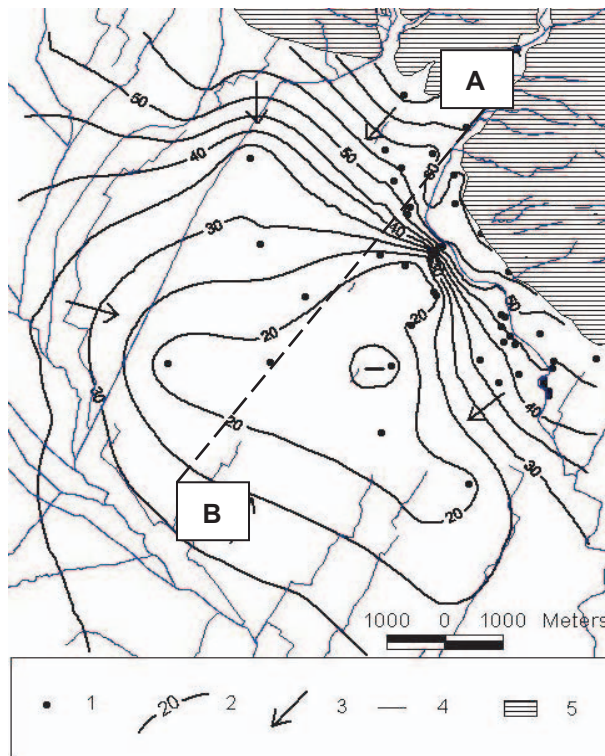
Figure 3. Piezometric surface (April 2004).
Legend: 1. Water level measure points;
2. Water level contour lines; 3. Flow path direction ;
4. hydrographic network.

Historically groundwater resources of the Prato aquifer were mainly exploited for industrial purposes being the area one of the largest Italian textile districts. Recently water demand for civil uses has grown very fast leading to a structural deficit on the hydrogeological balance with the result of increasing the pressure on water resources on our region. An essential role has also been played by the intense urbanization, reducing natural recharge.
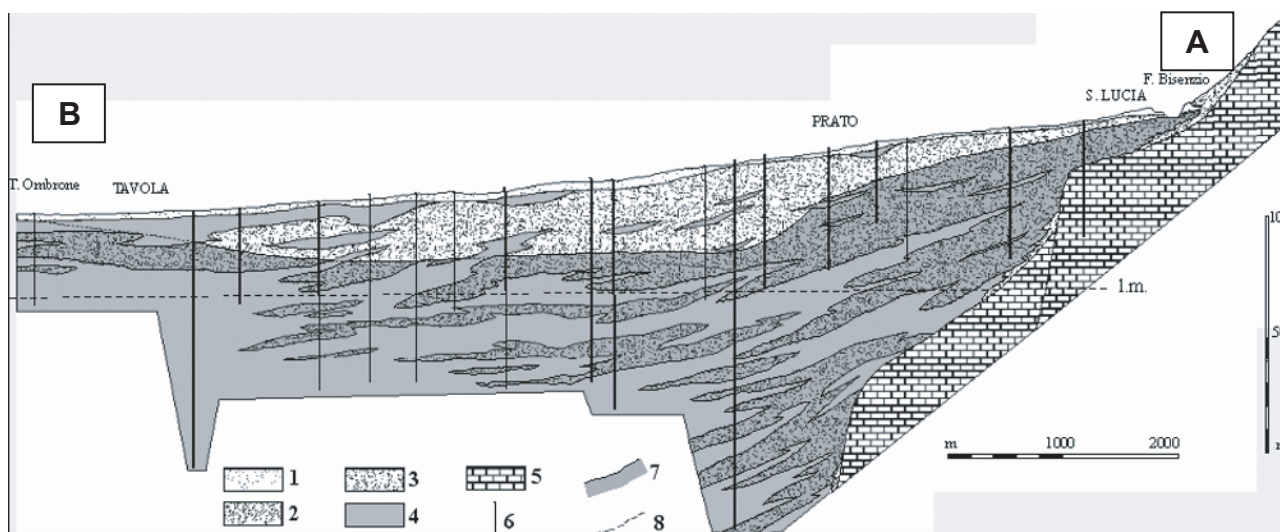
Figure 4. 1. sandy silt; 2. colluvium; 3. sandy-silty gravel; 4. clay and silt; 5. bedrock;
6. stratigraphic wells; 7. unconfinedgroundwater level; 8. semi-confined groundwater level.

Combining these aspects with the rising impact of climate changes, this cyclic piezometric levels (Figure 5) locally fall below the sea level reducing wells productivity, having a serious impact on the economical and social framework of the region.

ONGOING STRATEGIES

In order to overcome this crisis a series of actions have been carried out to improve the hydrogeological balance terms over the last 25 years.

Figure 5. Monitor 'Badie': water level time series 1959–2005

In a chronological order a first attempt to differentiate the resources for drinking purposes was to increase the use of surface water from the upper part of the river Bisenzio catchment-basin. Successfully reducing groundwater abstractions for industrial use guided the creation of a dedicated industrial aqueduct for the re-use of treated waste water. In parallel a series of small dams were built up to enhance the river losses into the aquifer, providing an important contribution to the groundwater balance. Most recently more water is used from the Florence aqueduct supplied by the Arno river – whose discharge is regulated by a large dam placed on the upper part of the basin –- has been made in connection with the Prato aqueduct.

Despite the mentioned investments and commitments the Prato aquifer has not yet shown a significant and persistent recovery.

HYDROGEOLOGICAL BALANCE

The groundwater balance of the Prato aquifer has been calculated for the hydrogeologic year 1st of October 2000 – 30th of September 2001, on the basis of an existing methodology proposed for the year 1988.

In the year of interest the amount of total rainfall has been about 25% higher than the long period average, thus an anomalous surplus of the balance has actually occurred.

The six terms of the water balance for the year 2000–2001 are: Hidden flow from the surrounding aquifers (As); Well abstractions (Du); Rainfall infiltration (Ia); River losses (If); Pipeline losses (Ri); Storage (dR).

Figure 6. Hydrogeological scheme of the Prato urban aquifer. All terms in millions of cubic meter

The equation is therefore structured as follow:

$$If + As + Ia + Ri = Du + dR$$

The conceptualization of an urban aquifer was adapted to our case study (Figure 6). The elements of complexity of the balance are mainly depending on several aspects such as the impact of well abstractions for civil and non civil uses, the variability of land use, with the dominance of low permeability areas obstructing infiltration, the presence of a complex system of pipelines and sewage channels, interfering with the groundwater level, locally recharging or draining the aquifer. The water balance structure follows the scheme of Figure 6, where the contribution of evapotranspiration is not taken into account, because it is negligible compared to the other terms such us industrial abstractions (error about 0.002%).

The total volume of water stored rose from 97 Mm^3 to 118 Mm^3 during the hydrologic year. Rejuvenation time (Castany, 1985) of the aquifer is 1.7 to 2 years.

EXPERIMENTAL PLAN

A feasibility study of an artificial recharge program was developed including a series of studies and field investigations. The geological and hydrogeological knowledge of the area was improved by geophysical investigations, along with core sampling and logging (Figure 7) and well tests on the aquifer properties. Additionally a continuous monitoring of the groundwater level nearby the location of the pilot plant was set up (Figure 8).

Transmissivity and hydraulic conductivity values were derived from a well pumping test (Figure 9a), while the same parameters for the non-saturated zone were provided by the recharge test (Figure 9b).

The values obtained for hydraulic conductivity are consistent throughout the saturated and the non-saturated zones, with a magnitude ranging between 4 to 9 m/d, according to the lithological composition of the core samples.
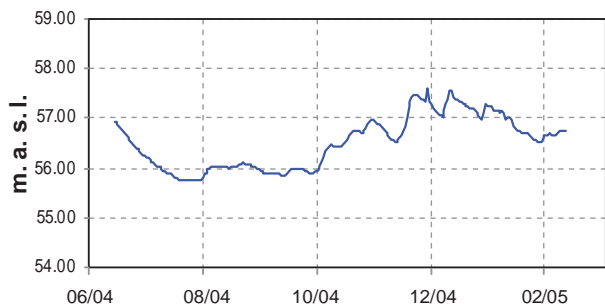
Figure 8. Continuous monitoring of the water level at the plant location
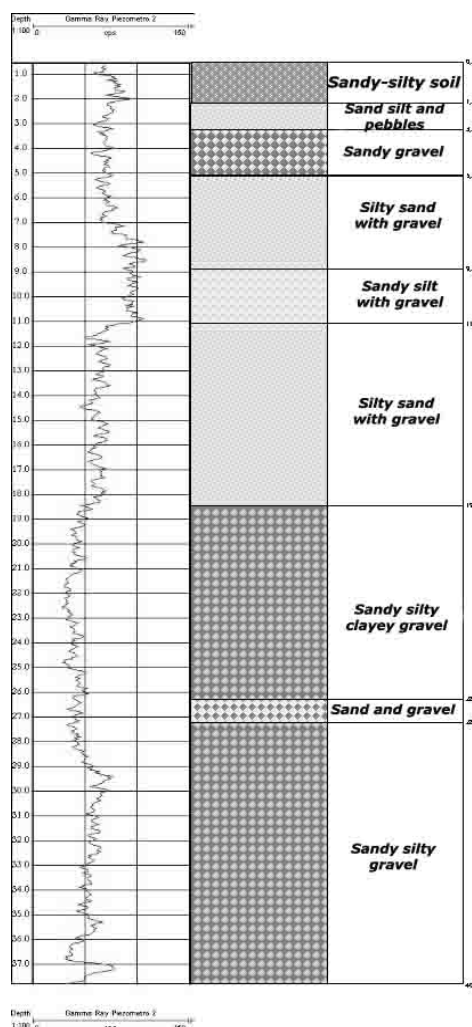
Figure 7. Gamma Ray Well log and stratigraphy derived from core sampling
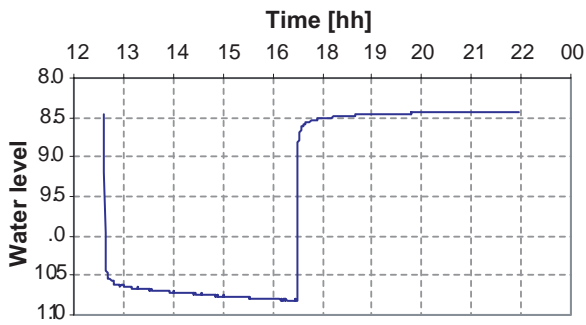
Figure 9a. Well pumping test
(constant discharge rate 3.6 l/s)
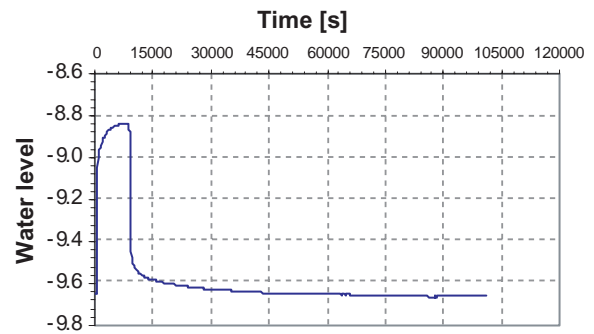
Figure 9b: well recharge test
(constant recharge rate 2 l/s)

As a tool to evaluate the effectiveness of an artificial recharge program of 400 l/s, corresponding to a recharge of 8 months a year, with a time duration of 10 years, a transient model of the aquifer was developed and calibrated for the time period 1960–2001. Although locally the model showed problems simulating single wells behaviour, likely because of poor reliability of industrial well abstractions rates, three different water management scenarios, for the next 10 years (2004–2014) were tested. Beside artificial recharge, a modulation of the river Bisenzio discharge was simulated, to predict the effects of improving river bed infiltration.

A third reference scenario was based on the actual data for the year 2001, keeping them constant during the 10-year simulation. Figure 10 shows the results on water levels for the three model scenarios at the monitoring well 'Badie'. The prediction model run with the values for 2001 ('Status Quo') indicates an attenuation on the effects of the over-exploitation with the consequent rising of the water level of 2 m over the 10-year period.
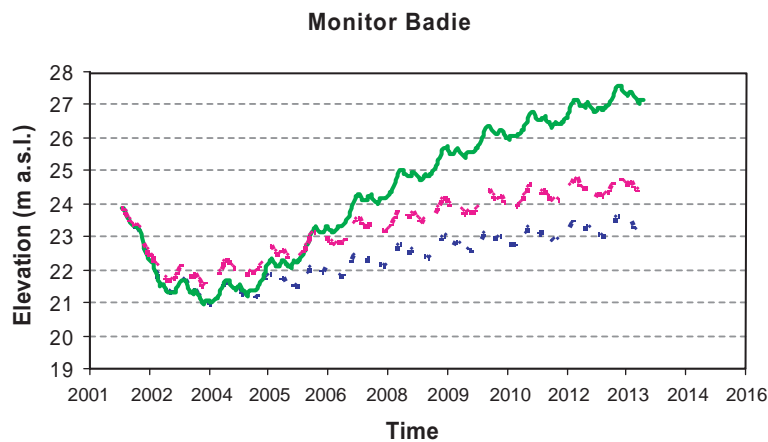
Figure 10. Monitor Badie: simulated groundwater
level trends for three different 2014 scenarios:
'Status Quo', 'Artificial Recharge', 'River Head Modification'

The influence of the river discharge modification ('Rivermod') provides an extra increment on the water level elevation of about 3 m the period to 2014. The most efficient scenario for the recovery of the aquifer is definitely represented by the artificial recharge ('AR'). In particular the recharge begins to be effective 6 months after the activation of the injection well field.

Figures 11a and 11b show the spatial distribution of the effects resulting 10 years later respectively for an artificial recharge scenario and for a river discharge modification scenario. The latter substantially improves the portion of the river infiltration on the hydrogeological balance.

The highest impact of this scenario occurs in the centre of the urban area, where the cone reaches the maximum of depression.

The effects of the artificial recharge are depending on the hydraulic conductivity, therefore on the hydraulic gradient generated at the plant location, where we have the most effectiveness of the recharge.

However such increment, generated by a recharge rate of 400 l/s for 8 months, raises the water level above the topography of the area. For such reason an additional model simulation with a 30% reduced recharge rate proved compatible with the site elevation (Figure 12).
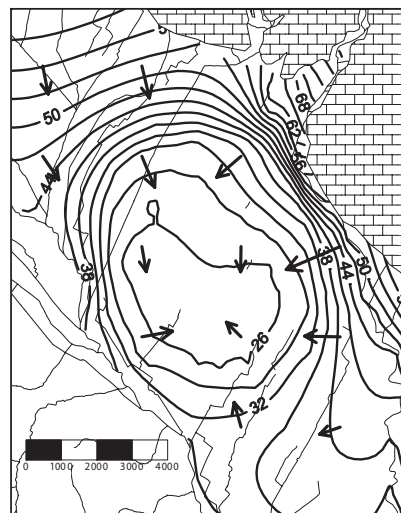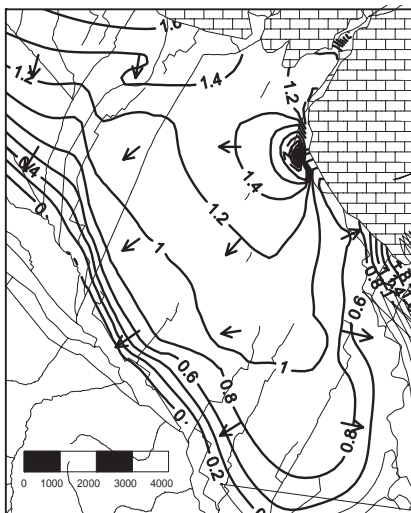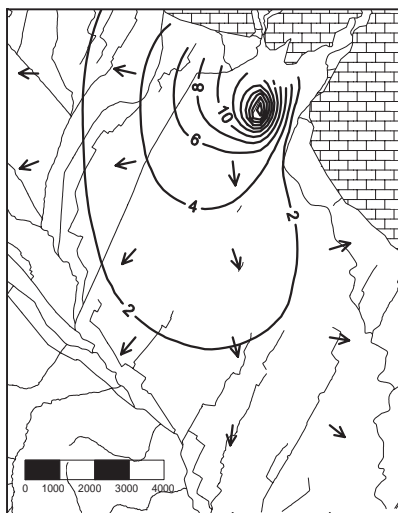
Figure 11. Water level increment contour lines simulated for 2014:
left) with a an artificial recharge;
right) with a river discharge modification

Figure 12

CONCLUSIONS

The hydrogeological studies and the groundwater balance of the Prato aquifer show that an artificial recharge by injection wells would provide a good strategy to reduce the water shortage and to increase water resouces in the area. A recharge plant in the north would not allow complete recovery of the aquiferin 10 years time at a maintainable rate of 120 l/s. Enhancing river bed infiltration and simultaneously reducing groundwater demand are necessary additional measures.

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Technical effectiveness of artificial recharge structures in hard rock area – A case study in Coimbatore District, India

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S. Gurunathan, T. Arivalagan and S. Thirumurthi

INTRODUCTION

The phenomenon of groundwater exploitation is much more pronounced in the state of Tamil Nadu, which depends heavily on ground water for domestic, industrial and irrigation purposes. This is mainly because of uneven and erratic distribution of rainfall. It has been observed from many studies that there has been a continuous decline in ground water table in the Tamil Nadu state. It necessitated the importance of augmenting groundwater resources particularly by artificial recharge structures. With this view the study was undertaken to test the effectiveness of artificial recharge (AR) structure in hard rock areas of Coimbatore district and the impact of AR on water quality and livelihood trends.

THE STUDY AREA

Coimbatore district is located in the western part of Tamil Nadu state. The district is bordered to the west by the mountains of the Western Ghats but is dominated by the plains to the east. The district is underline by crystalline basement rocks, typical of much of peninsular India. Kodangipalayam watershed is in the middle portion of the Noyyal river basin. The study was undertaken in this watershed covering two villages namely Karanampettai and Kodangipalayam (Latitude 11°02'00" to 11°04'00" N Longitude 77°01'00" to 77°14'00" E) of Palladam block of Palladam taluk of the Coimbatore district since, the area falls under the over exploited groundwater extraction category (>100%) (SG and SWRDC, 2002). The impact of artificial recharge on this terrain is critical in position both in terms of hydrogeological perspectives as well as in context to larger impacts that include livelihood changes that occur as a consequence of the Kodangipalayam watershed development project. The artificial recharge structure in the Karanampettai micro watershed covers a catchment area of 1,408 sq.km. The elevation ranged from 307.7 to 325.0 m above MSL. The study area underlined by a wide range of high grade metamorphic rocks of the peninsular gneissic complex. These rocks are extensively weathered (up to 15±5 m) and overlain by recent valley fills. The geological formations found in the area are charnockite overlying migmatite and banded gneiss cut by fractured pegmatite (Figure 1). Climatically, the area belongs to semi-arid climate. The annual rainfall is about 524±100 mm with bimodal distribution of summer (March to May) and north east monsoon (October to December).

METHODOLOGY

The recharge structure was constructed during 1978 in the confluence of two streams by forming embankments to store water with a spillway arrangement for excess rain water to drain. The runoff water from the catchment area of 1.408 sq.km gets collected during the monsoon period and stored only for the purpose of improving the groundwater storage. The recharge structure was renovated periodically to remove the silt accumulated in the water spread area and to augment the natural ground water storage. The water spread area of the recharge structure is 9,000 m^2
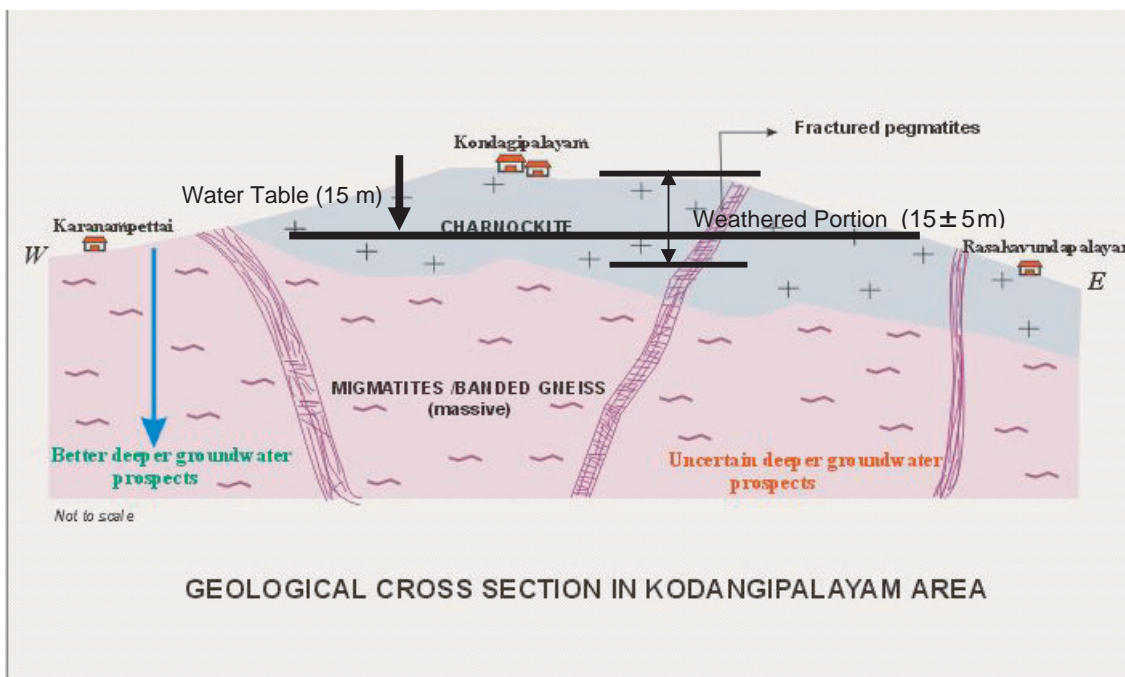
Figure.1. Geological cross section of Kodangipalayam watershed

and the depth is 2.40 m with the capacity of 15,000 m^3. Topographic survey was carried out to identify the surface water catchment of the recharge structure, to obtain the geodetic heights (datum) at observation points, to know the stage volume relationship of recharge structure. Automatic weather station was installed to record the weather parameters. Automatic water level recorder installed in the recharge structure to record daily water level in the
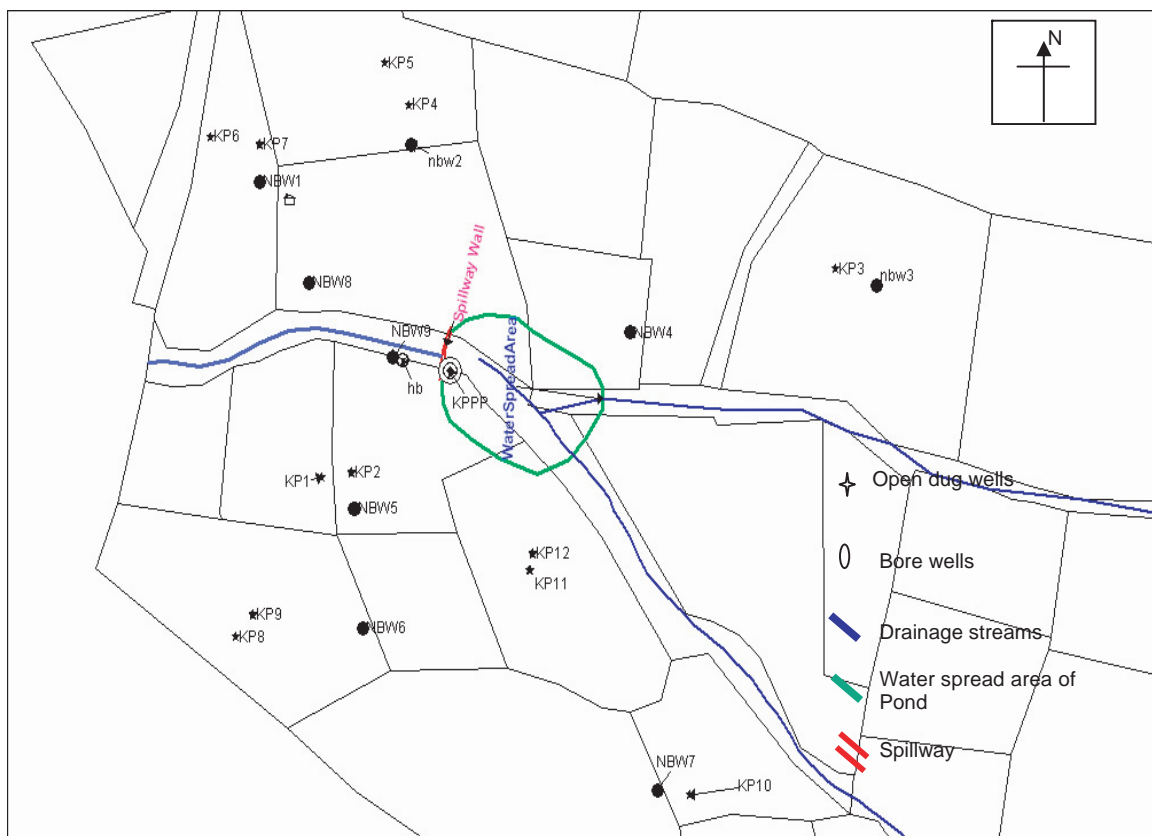
Figure.1. Geological cross section of Kodangipalayam watershed

pond. Eight bore wells (NBW 2 to 9) were drilled and twelve open wells (KP 1 to 12) were also selected (Figure 2) to monitor the hydro geochemistry in the micro watershed. Water sampling for quality assessment was done from August 2003 to October 2004 for seven times viz., 25.08.03, 11.09.03, 30.09.03, 28.10.03, 26.03.04, 10.06.04 and 12.10.04 and analysed for pH, EC, cations and anions using the standard procedure. The sampling periods also cover both before and after monsoon periods to explore the impacts of artificial recharge on ground water quality. Using the questionnaire method impact assessment on livelihood trends was assessed with 60 farm families (Gale et al. 2003).

RESULTS AND DISCUSSION

Recharge pattern

The results of the ground water level variations in the monitoring boreholes (NBW 2 to 9) showed that in all bore holes the influence of continuous recharge started on 29th days (29.10.2004) after the start of recharge structure filling (Table 1).The water table contours also confirmed the same trend (Figure 3a and 3b). The influence of

Table. 1. Recharge pattern in the observation boreholes of Karanampettai

Distance from the pond (m)		0	64	165	166	218	260	456	476
		KP-PP	NBW9	NBW8	NBW5	NBW2	NBW6	NBW7*	NBW3*
Date	Days from start of pond filling	Water level (m)		Water level rise or fall in the observation bore holes (m)					
01/10/2004	0	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00
08/10/2004	8	0.50	–0.31	–0.05	–0.09	–0.20	–0.04	0.10	0.22
15/10/2004	15	1.67	–0.51	–0.10	–0.15	–0.60	–0.09	0.18	0.30
22/10/2004	22	2.40	0.48	–0.28	–0.19	0.00	–0.13	0.46	0.87
29/10/2004	29	*2.15*	*2.11*	*0.91*	*2.60*	*1.76*	*2.06*	*1.78*	*1.75*
10/11/2004	41	*1.95*	3.93	2.90	4.92	3.78	5.28	2.70	2.73
17/11/2004	48	1.80	*4.04*	3.37	5.98	5.75	6.10	3.11	3.14
19/11/2004	50	1.75	4.01	3.43	6.16	6.08	6.35	3.34	3.23
30/11/2004	61	1.55	3.75	3.30	7.10	6.25	7.26	4.08	3.80
02/12/2004	63	1.47	3.77	3.24	7.19	6.34	7.55	4.23	3.87
03/12/2004	64	1.44	3.90	3.45	*7.31*	*6.75*	*7.66*	4.25	4.02
06/12/2004	67	1.35	4.00	*3.60*	7.13	6.56	7.45	4.15	3.96
27/12/2004	88	0.78	2.95	2.80	6.61	6.37	6.55	6.05	4.40
30/12/2004	91	0.66	2.45	2.53	6.20	6.72	6.47	*6.13*	*4.67*
03/01/2005	95	0.57	2.23	2.40	6.02	6.80	5.99	5.71	4.31
10/01/2005	102	0.30	1.75	2.26	5.91	6.90	5.82	5.33	4.25
17/01/2005	109	0.07	1.53	2.08	5.84	6.80	5.71	5.12	4.20

Note: Negative sign indicates the water level decrease from the from the reference date (01.10.2004); * bore hole location is lateral to the flow of streams (Figure 2).
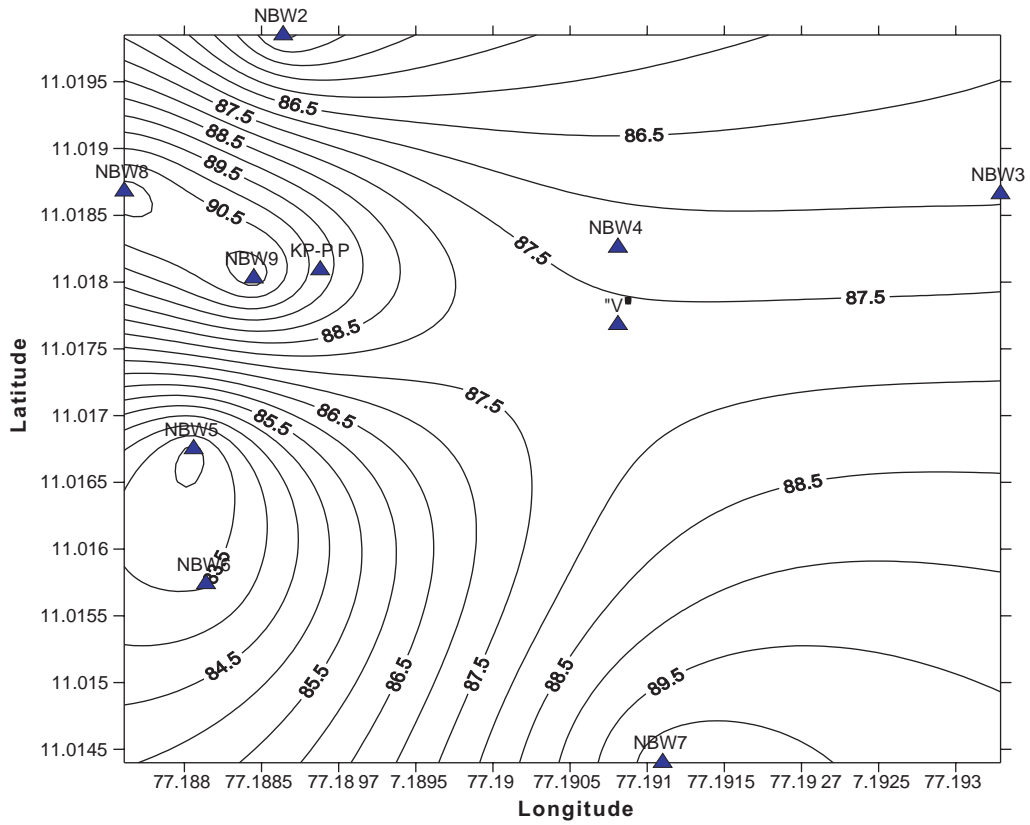
Figure 3a. Water level contours of bore wells on 01 Oct 2004
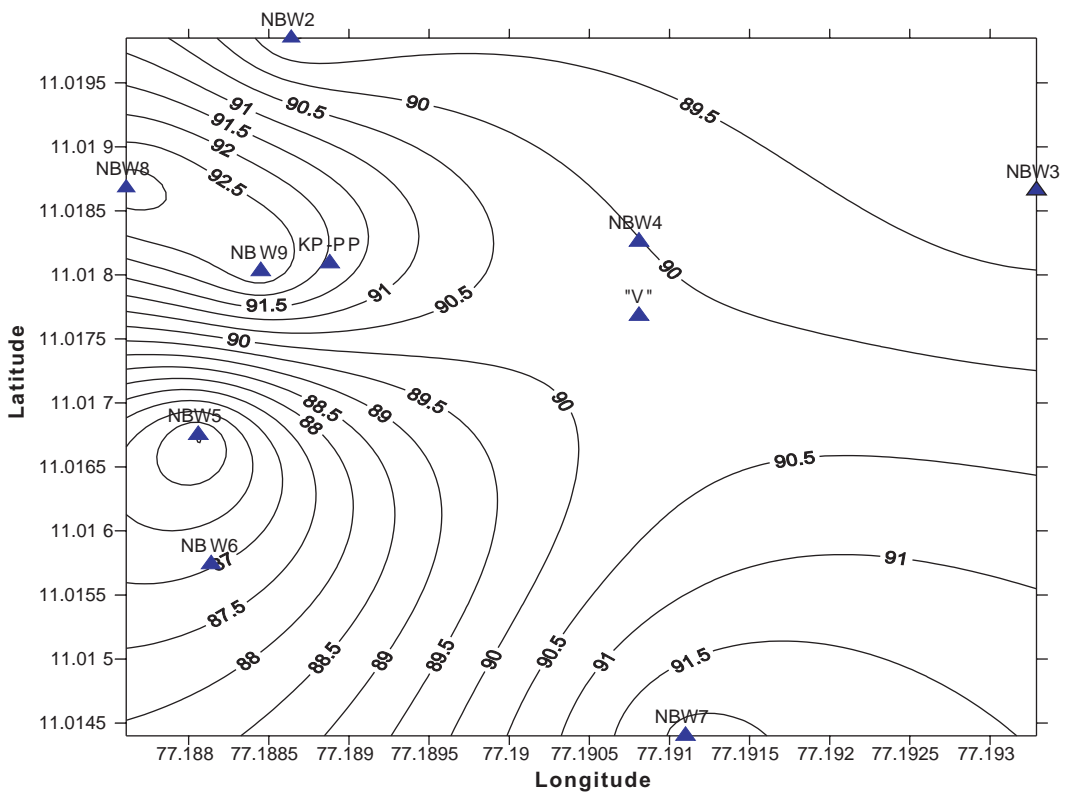
Figure 3b. Water level contours of bore wells on 29 Oct 2004

recharge continued up to three months from the start of recharge structure filling. The maximum water level rise found to be varying with the distance from the recharge structure. The nearest (64 m) bore well (NBW 9) from the recharge structure, recorded maximum water level rise on 48 days after the start of filling. In the far away (more than 400 m) bore wells NBW 3 and NBW 7, recorded maximum water level rise on 91 days after the start of filling the recharge structure. Bore wells situated in the radius of 165 to 217 m from the recharge structure, namely NBW 2, NBW 5, NBW 6 and NBW 8, takes about 60 to 65 days to record maximum water level rise (Table 1). Cross-section groundwater levels in the study area (Figure.4) indicated that the influence of recharge was much more higher in the closer wells (NBW 8 and 9), followed by NBW 2,5 and 6 and lower in distanced wells (NBW 3 and 7). This is supported by minimum water level fluctuation in the nearby wells and maximum fluctuation in distanced wells.
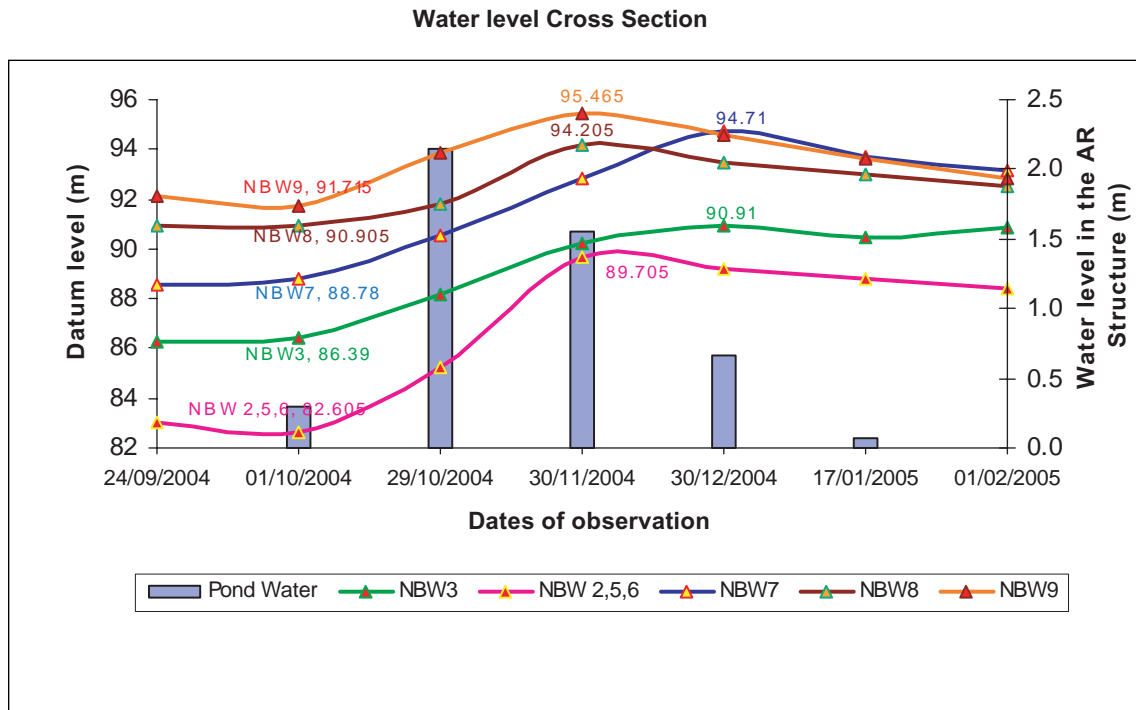
Figure 4. Water level cross section of observation wells

Water quality

Water quality assessment in open dug wells revealed that Electrical Conductivity (EC) was very high (>2.25 dSm^{-1}) in far away wells namely KP 9, KP 7 and KP 8 since the recharge water has to travel longer distance (more than 350 m) to reach these wells leads to higher salinity level because of high interaction with Na and Cl containing mineral material for substantially longer time (roughly 90 days). High level (0.75–2.25 dS^{-1}) of salinity observed in medium distanced wells namely KP 1, KP 3, KP 4, KP 5 and lateral wells KP 10, KP 11 and KP 12 because of medium level of recharge structure influence. Medium level (0.25–0.75 dSm^{-1}) of salinity observed in hand pump, KP-HP (regularly pumped for bathing and washing cloths), since this is the nearest monitoring point (60 m) which receive maximum benefit from the recharge structure leads to reduced salinity. Recharge structure (KP-PP) and KP 2 (Which collects only the rainfall run-off water) were of low (<0.25 dSm^{-1}) in salinity hazard (Figure 5). The chloride concentration also followed the same pattern with distance as Electrical Conductivity. The before and after recharge comparison (Figure .6) of water quality showed that the quality has improved slightly due to the artificial recharge.
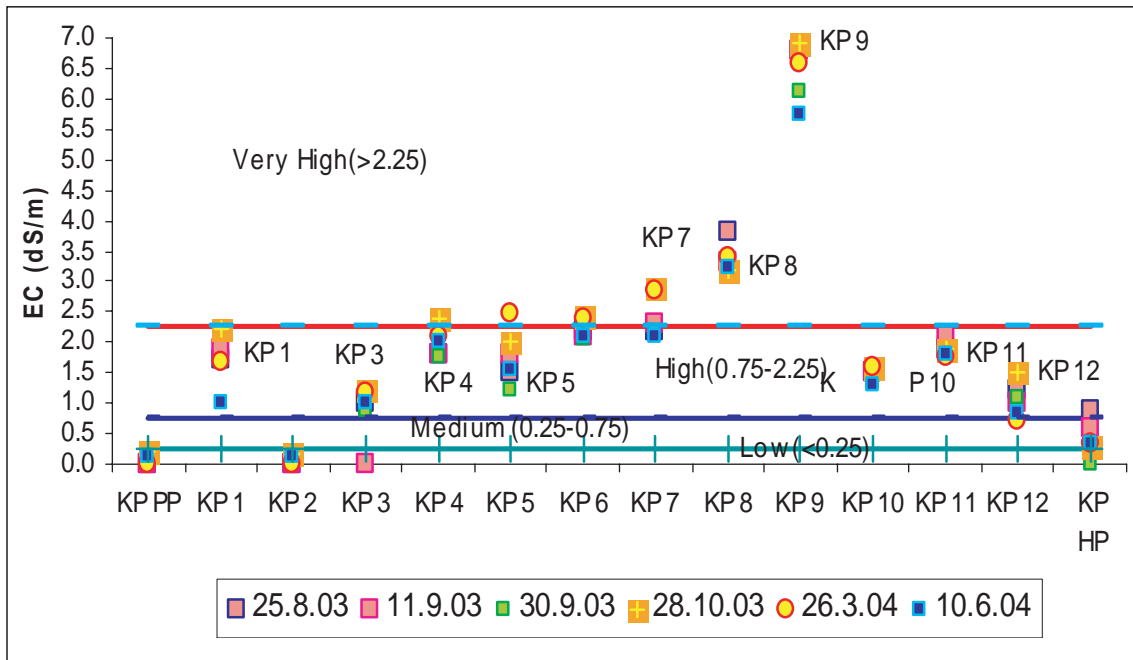
Figure 5. Electrical conductivity (dSm⁻¹) of observation open wells
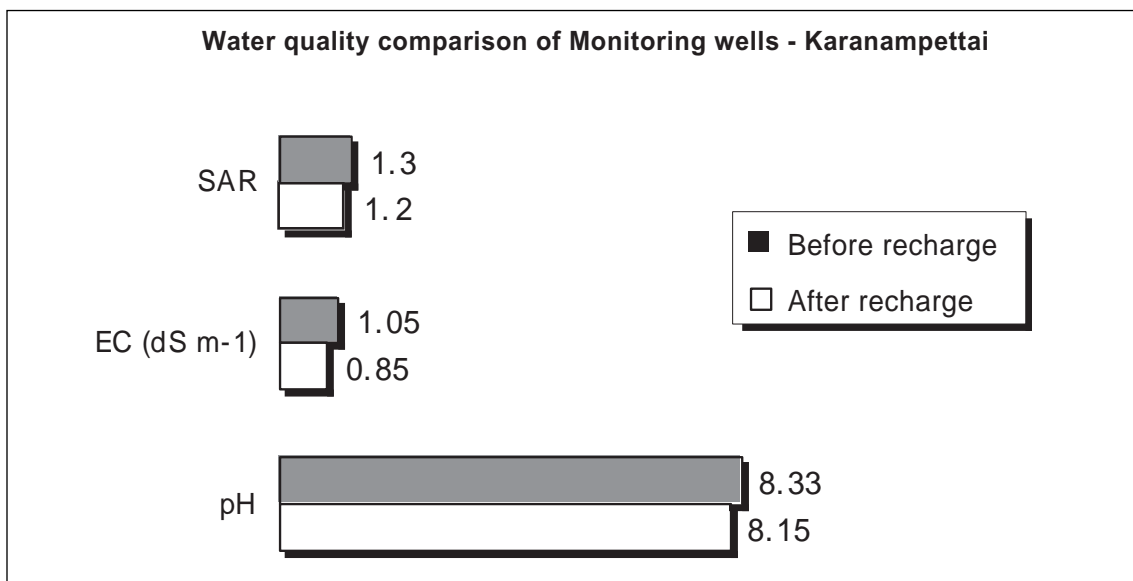
Figure 6. Water quality comparison of before and after recharge

Table 2. Impact of AR structure on livelihood trend

Sl.No	Details	Before	After
1.	Employment (Labour hours)		
	(a) On farm	147	160
	(b) Off farm	65	90
2.	Well failures %	12.5	5
3.	Pumping hours	< 2	3

Livelihood impacts

The results of the livelihood impact assessment (Table 2) showed that the on farm employment has increased due to the AR structure induced recharge of aquifer. Well failure percentage brought down to five percent and the hours of pumping increased from less than 2 hours to 3 hours, this makes farmers to go for relatively high value crops.

CONCLUSIONS

The influence of recharge started on 29th days after the start of recharge structure filling and continued up to three months from the start of filling. The maximum water level rise found to be varying with the distance from the main recharge structure. The nearest bore well recorded maximum water level rise on 48 days and far away (more than 400 m) bore wells, recorded maximum water level on 91 days after the start of filling the AR structure. The EC and chloride concentration affected with distance from the recharge structure. Very high salinity in far away wells, high salinity in medium distanced wells, medium salinity observed in hand pump (nearest monitoring point) which receive maximum benefit from the recharge structure leads to reduced salinity. On farm employment has increased due to the AR structure induced recharge of aquifer. Well failure percentage brought down to five percent and the hours of pumping increased from less than 2 hours to 3 hours, this makes farmers to go for relatively high value crops.

ACKNOWLEDGEMENT

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Experience of capturing flood water for artificial recharge of groundwater in North China

Sun Ronglin, Liang Xing, Wang Xianguo, Liu Qingyong

Abstract

The purpose of this paper is to summarize the experiences of capturing flood water for artificial recharge (AR) of groundwater in north China. Two cases, Huangshuihe artificial recharge system (ARS) in coastal area and Luohe ARS in inland area, were studied. Many works were built to capture flood water for AR, such as surface reservoir, dam, underground dam, infiltration well or trench, etc. The groundwater quantity of Huangshuihe ARS has increased by $11 \times 10^6 - 60 \times 10^6$ m^3/y and the pumping cones of Luohe ARS have been filled. However, as they have not been maintained well, some problems have arisen in the running of ARS. Infiltration rate of infiltration wells or trenches of Huangshuihe ARS has reduced by half because of clogging. The rise of groundwater level for Luohe ARS is threatening the safety of urban understructures. Based on the above experiences, this paper proposes to build the standardized project in the middle route of 'South-north Water Transfer Project' so that the similar problems can be solved.

Keywords

Artificial recharge of groundwater; clogging; flood water; maintenance; rise of groundwater level

INTRODUCTION

Water scarcity is a severe problem in north China, where the climate is mainly arid and semi-arid with continental monsoon. More than 70% of precipitation is concentrated between June and September (Pan and Zhang, 2001). As major runoff events always occur in short time, plentiful flood water discharges into the sea without being stored. Groundwater is the main water source in many cities. Excessive exploitation of groundwater has resulted in regional cones of depression, land subsidence, seawater intrusion, etc.

In order to solve the conflict between the supply and demand of water resources, capturing flood water in times of water surplus is necessary for the use when water is in short. The traditional way is to build surface reservoirs. But they have many disadvantages, such as huge cost, great evaporation loss, sediment accumulation, and adverse ecological, environmental, and socio-cultural effects, etc. Therefore, artificial recharge (AR) is preferred, for it can augment the amount of groundwater available through the works designed to increase percolation of surface waters into the groundwater aquifers. Moreover, AR can maintain sustainable use for aquifers with low cost.

At present, some artificial recharge system (ARS) projects are running in north China. Through the conjunct operation of surface water, groundwater and external transferred water, ARS has augmented the amount of groundwater resource and improved the environment. However, some common problems have arisen in the running of ARS, such as clogging of infiltration zones and water source pollution. In particular, the groundwater level rise is threatening the urban understructures which were built before the running of ARS. Two cases, Huangshuihe ARS in coastal area and Luohe ARS in inland area, were studied to summarize the experience of capturing flood water for AR of groundwater in north China.

HUANGSHUIHE ARS

Introduction of Huangshuihe ARS

Huangshuihe ARS is located on the middle and lower plain of Huangshuihe River, Longkou City, Shandong Province. Huangshuihe River is an ephemeral stream. One big surface reservoir is built on the middle branch with the storage capacity of 149×10^6 m^3, and the rest flood water of 98×10^6 m^3/y will be discharged rapidly into Bohai Sea. The excessive exploitation of groundwater resulted in water table declining and seawater intrusion, which caused lots of pumping wells to stop working. Huangshuihe ARS was built to capture flood water for AR of groundwater in 1995 (Liu, Zhang, et al., 2003; Liu, Meng, et al., 2003).

Slightly permeable bedrocks around the Huangshuihe ARS form a close boundary. The main aquifer is Quaternary phreatic aquifer composed of gravel-containing coarse sand and medium and fine sand lamellas. There is close interaction between aquifers and surface water.

Huangshuihe ARS is composed of the surface reservoir, aquifers and many works (Fig. 1)

(1) Six stepped dams. Each is about 200 m long and 2.5 m high. If the river level is higher than 2.6–2.8 m, dams will flap by themselves. The total backwater length is 12,690 m.

(2) Infiltration wells, trenches and basins. As the mild clay and sabulous clay of 5,800 m long and 3–17 m thick spread on the surface layer of the riverbed, it is necessary to build the infiltration zone. 448 infiltration trenches, 773 infiltration basins and 2,518 infiltration wells were built to increase the infiltration rate of flood water into aquifers.

(3) Underground dam. A big underground dam was built under the sixth downriver dam to capture the under-current that flows into Bohai Sea and prevent the seawater intrusion. It is 5,996 m long and 26.7 m high in average. The total storage capacity of groundwater is 53.6×10^6 m^3.
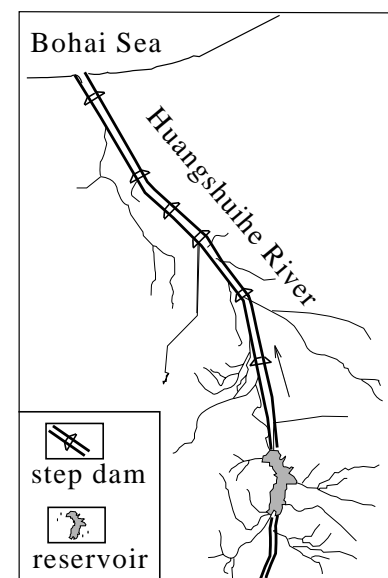
Figure 1. Plan of Huangshuihe ARS

Achievements

Two important successes have been obtained since Huangshuihe ARS began to run in 1995. The groundwater resource has increased by 11×10^6–60×10^6 m^3/y and the seawater intrusion has been prevented. The discarded agricultural and industrial wells have begun to work again. As sewages along the Huangshuihe River are discharged into Bohai Sea after disposal through special drainage pipelines, the groundwater is not polluted.

Problems

The main problem is that infiltration rate of the riverbed is reduced by half because of clogging. Many forklifts were used to shovel the surface accumulation in 1998, which was onerous and expensive. As the accurate locations of infiltration trenches, basins and wells have not been found, no better methods can be used to resume their infiltration rate.

LUOHE ARS

Introduction of Luohe ARS

Luohe ARS is located on the alluvial plain of Luohe River. Luohe River is 38,000 m long in Luoyang City with a runoff of $1,933 \times 10^6$ m^3/y and it has a branch called Jianhe River (Fig. 2). The shallow aquifer in the city is 10–80 m deep and composed of coarse sand and gravel, and it is the major water source for industry, agriculture and domestic use. The amount of groundwater mining increased from 33×10^6 m^3/y in 1962 to 220×10^6 m^3/y in 1998; many depression cones appeared in pumping zones and Luohe River recharged groundwater by vertical penetration at that time.

Luohe ARS is composed of stepped dams and the shallow aquifers. 5 stepped rubber dams were designed to capture flood water in 1999. From upriver to downriver, they are orderly Zhoushan Dam (4.5 m high, finished in April, 2004), Shangyanggong Dam (4 m high, finished in March, 2000), Tongleyuan Dam (3.5 m high, finished in June, 2001), Luoshenpu Dam (4 m high, finished in June, 2002) and Hualinyuan Dam (3.5 m, now in construction) (Fig. 2). They discharge water from July to September and store water in the other time. The total storage capacity of these dams is 19.23×10^6 m^3. As the hydraulic conductivity of the riverbed is 200 m/d, it is not necessary to build infiltration wells and trenches.
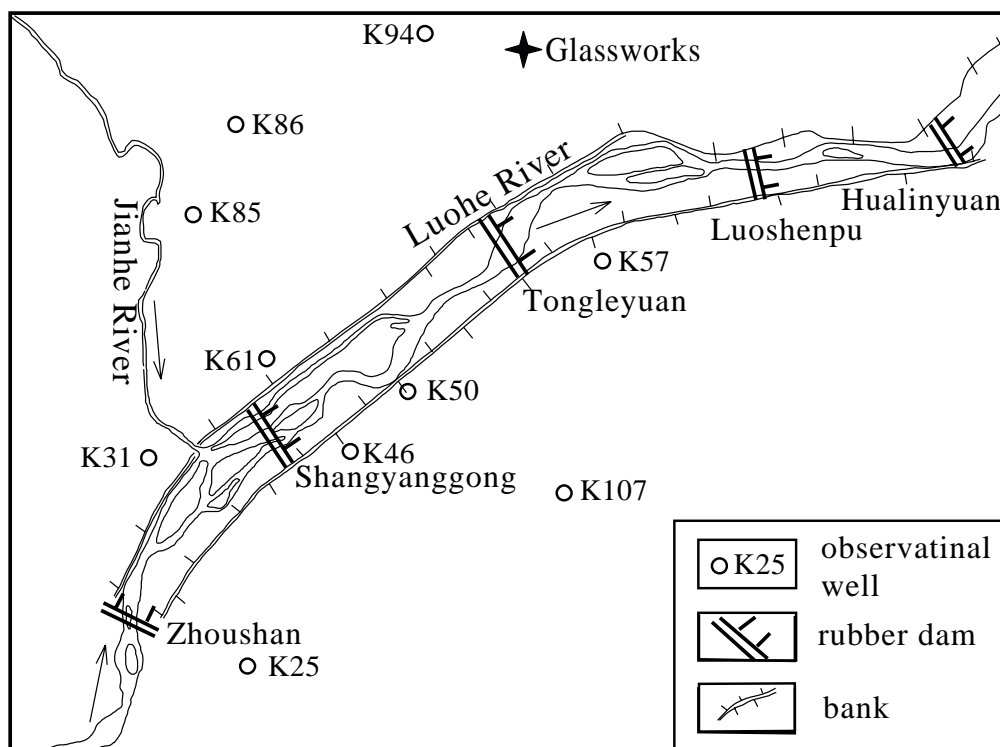
Figure 2. Plan of Luohe ARS

Achievement

Two important successes have been obtained since Louhe ARS began to run in 2000. (1) The amount of groundwater available has increased greatly and the groundwater level of observational wells risen (Table 1). The pumping cones along Luohe River have been filled. Luohe River recharges the shallow groundwater in the form of lateral penetration. (2) Luohe ARS becomes an important part of Luofu scenic spot. There are many kinds of flowers and beautiful gloriettes on the banks of Luohe River, and they help the tourism develop rapidly.

Table 1. Groundwater level of observational wells before and after ARS's running   (m)

Time	K25	K31	K46	K50	K57	K61	K85	K86	K94	K107
1999-5-25	3.67	17.00		10.62	5.80	14.12				
2000-5-25	3.05	13.26	9.43	10.51	7.12	11.24	24.06	23.94	22.06	8.75
2002-5-25	2.60	14.24	8.84	6.96	3.45	10.42	22.55	22.40	20.16	7.43
2003-5-25	3.60	11.00	7.94	8.98	2.95	9.30				

Problems

Two serious problems have occurred in the running of Luohe ARS. (1) The rise of groundwater level in some locations is threatening the safety of the urban understructures within the area of 70.5 km^2. Many understructures were built above the buried depth of groundwater level, which was deeper because of the excessive exploitation of groundwater before the running of ARS. When groundwater level rises higher than understructure bottom, large amount of water will rush into underground shelters and foundation pits. For example, the water yield is increasing from the initial 240 m^3/d to 1,630 m^3/d in the bomb shelter of the City Glassworks (Fig. 1). As the groundwater level was about 127.5 m before the running of ARS and becomes 129.5 m after the running of ARS, it is 0.4 m higher than the bottom of bomb shelter. Some buildings are even distorted because of uneven subsidence. Moreover, the super-shallow water table depth will result in the waterlogging and salination of the lowlying areas. Therefore, it is necessary to make measures to control the amount of captured flood water and the continual rise of groundwater level. (2) Concentrations of Cr^{6+}, Pb^{2+}, Cu^{2+} and NH_4^+ in shallow groundwater are increasing because Jianhe River and the upriver water of Luohe River have been polluted and the industrial wastewater is directly discharged into Luohe River.

RESULTS AND DISCUSSION

There are many other cases which demonstrate that capturing flood water for AR is successful. However, if it is not well planned before the construction or if it is not maintained well after its running, the ARS will not work effectively. It is very important to remember the following experience of capturing flood water for AR.

Thorough survey

It is important to carry out a thorough survey prior to deciding recharge methods. All the factors related to ARS should be surveyed in detail as follows: climate record (rainfall, humidity, evaporation rates), hydrogeological conditions (reasonable buried depth and storage capacity of aquifers, pore structure), the depth of understructures, water supply and demand with economic development, existing drainage networks, etc. Thorough survey is important to ensure safe running of ARS. Especially the consequence of storing waters should be predicted. Otherwise, adverse problems will appear which are not expected. For example, too high rising of groundwater level has threatened the safety of understructures. For large-scale ARS, it is necessary to carry out trial testing of small-scale artificial recharge system.

Methods of capturing flood water for AR

Works. During capturing flood water for AR of groundwater, it is necessary to build some works, such as barrages (or dams), infiltration wells or trenches. According to specific conditions, underground dams, storage reservoirs, penetration tanks, pretreatment tanks, diversion canals and other works may be built for ARS.

Site selection. ARS can be built beside the riverbed or on the riverbed, and this is based on the quality of flood water and the penetration capacity of the riverbed (Bouwer, 2002; Asano, 1985). The ARS beside the riverbed applies to rivers with great runoff and more suspended solids or without penetrable riverbed. It is necessary to find some penetrable soils where penetration tanks are built. As the ARS beside the riverbed needs other supporting works to capture and lead flood water, it is very expensive and therefore rarely used in China.

ARS on the riverbed applies to rivers with small runoff and penetrable riverbed. It only needs dams on the riverbed to capture flood water. If the local riverbed is not penetrable, the infiltration zone needs to be built. However, it is expensive and onerous to build large numbers of infiltration wells or trenches on the riverbed and recovery their infiltration rate. Compared with ARS beside the riverbed, it needs no places outside the riverbed and many supporting works. As it is cheaper and easier to be maintained, ARS on the riverbed is often used in China.

Maintenance of ARS

The maintenance of ARS is very important, because it can decide whether the running of ARS will be successful. Three techniques should be given attention to.

Recovery of infiltration rate. The main problem in infiltration systems for AR of groundwater is the clogging of the infiltrating surface (penetrable riverbed, infiltration wells or trenches), and the resulting reduction in infiltration rates (Bouwer, 2002). During capturing flood water for ARS, the suspended solids are the main cause of clogging. Many methods have been used to recovery the infiltration rate of the single well or trench. However, for large numbers of infiltration wells or trenches, it is onerous and costly. Thus, the infiltration systems should be shallow so that they can readily be dried for cleaning, disking, or other maintenance procedure to restore the infiltration rate. While deep facilities are good for temporary storage of floodwater, they are not good for infiltration and recharge because of sediment accumulation (Bouwer and Rice, 2001).

Good water quality. In particular, industrial and domestic sewages around the ARS should be treated before they are discharged into rivers. If the water source is contaminated, the water already stored in the aquifer could be contaminated, thus lowering the value of the water, especially for the drinking water supply.

Timely groundwater storage and recovery. The aquifers of ARS must ensure adequate storage room before flood water comes and enough groundwater reserve to be used in dry periods. Groundwater level rising must be controlled in a reasonable range. Otherwise, it will cause such adverse effect as that in Luohe ARS. So it is necessary to make dynamic plan for artificial storage and recovery. Besides the above-mentioned techniques, a good water market should be built in China.

CONCLUSIONS AND SUGGESTION

Through the conjunctive use of surface water and groundwater, capturing flood water for AR can augment the account of groundwater resource. However, there are still some problems in the running of AR. The design and management of ARS involve geological, geochemical, hydrological, biological, and engineering factors. If it is not well planned or maintained, the ARS will not work effectively. Currently, many ARS projects are planned to be built in north China. However, there are not systemic theory and responding criterions in China. In order to resolve and prevent these problems, it is necessary to build a standardized ARS.

The middle route of 'South-north Water Transfer Project' has begun in China. As the flood season and dry period of Hanshui River, as the supply area, are always coinstantaneous with those of the target area, it is necessary to store local flood water and water transferred from Hanshui River in flood seasons or high-flow years. On the other hand, many pumping cones have occurred in many cities along the middle route of the water transfer project, and they

provide enough room for AR. Moreover, many large surface reservoirs are distributed along this route. It is a good opportunity to build ARS in the middle route of 'South-north Water Transfer Project'.

Through studying existing problems and carrying out standardized ARS projects, we can summarize the theory and technology of AR and establish responding criteria and regulations.

ACKNOWLEDGMENTS

Hydraulic and geochemical characterization of Ajali Sandstone Aquifer, SE – Nigeria: Implication for groundwater recharge process

Moshood N. Tijani

Abstract

This study presents the hydraulic and geochemical characteristics of the regional aquiferous Ajali Sandstone Formation (ASF) in SE-Nigeria with emphasis on textural controls on the hydraulic conductivity (K) and assessment of possible geochemical controls of the aquifer materials on the groundwater quality. The results of the textural grain-size analyses indicate fine to medium sand (>75% sand fraction) with mean grain sizes of 0.23 to 0.53 mm and coefficient of uniformity (C_u) values of 1.58 to 5.25 (av. 2.75) while standard sorting values of 0.56ϕ to 1.24ϕ implies moderately well sorted materials. The order of estimated K values are $K_{permeameter} > K_{Beyer} > K_{Hazen} > K_{Kozeny-Carmen} > K_{Fair-Hatch}$ with average values of 1.4×10^{-3}, 4.4×10^{-4}, 3.8×10^{-4}, 2.2×10^{-4} and 8.1×10^{-5} m/sec respectively. These values conform to the specification for recharge systems and hence indication of the high infiltration rate within the outcrop zones of the ASF. Furthermore, the estimated porosity of 18.3 to 32.8% confirms the potentials of the ASF. However, the average value of 7.9% and 2.4% for Al_2O_3 and Fe_2O_3 respectively and elevated trace metal concentrations (av. 3.5 mg/kg Cu, 12.3 mg/kg Pb, 11.6 mg/kg Zn, 9.3 mg/kg Ni and 42.9 mg/kg Cr) in the ferruginized samples call for concern, due to the attendant negative impacts of flow-induced geochemical processes on the groundwater quality and aquifer management.

Keywords

Ajali sandstone; Anambra Basin; geochemical characteristic; grain size analysis, hydraulic conductivity; permeability test.

INTRODUCTION

Long-term sustainable groundwater management in terms of quantity and quality, in unconsolidated sandy aquifers requires reliable knowledge of the hydraulic conductivity (K) and textural characteristics as well as possible geochemical interactions between the aquifer matrix and groundwater. For such hydraulic characterization of aquiferous sedimentary units, several laboratory and field methods are usually employed (Batu, 1999). Although *in-situ* field determination of K are more accurate and highly reliable in site-specific investigations, laboratory and empirical approaches are usually preferred, especially during preliminary hydrogeological investigations, in order to cut down on cost and time on one hand. On the other hand, such approaches are seen as low-cost alternative, especially in developing countries like Nigeria, where lack of funds for field scale operations constitutes serious limitation.

This study, therefore, highlights; (a) the distribution of hydraulic conductivity (K) in the regional aquiferous Ajali Sandstone Formation (ASF) in southeastern Nigeria (Fig. 1) through comparative assessment of estimations from grain size analysis (GSA) and laboratory permeameter tests (b) the geochemical characteristics of ASF and implication of recharge/infiltration induced geochemical interactions between the groundwater system and the aquifer materials. The target ASF is located within the Cretaceous Anambra Basin, SE-Nigeria.

Hydrologically, ASF is an extensive regional stratigraphic unit also referred to as the false-bedded sandstone, due to its cross-stratification in places. The thickness ranges from about 350–450 m in places and thin southward to few

tens of meters around Okigwe area (Hogue and Ezepue, 1977) while field study revealed ferruginization of the upper sections of the formation in many outcrop locations. The summary of geology and lithostratigraphic units within the basin and sample locations within the ASF are shown in Fig. 1, while details on stratigraphy and sedimentation history can be found in Ojoh, 1990.
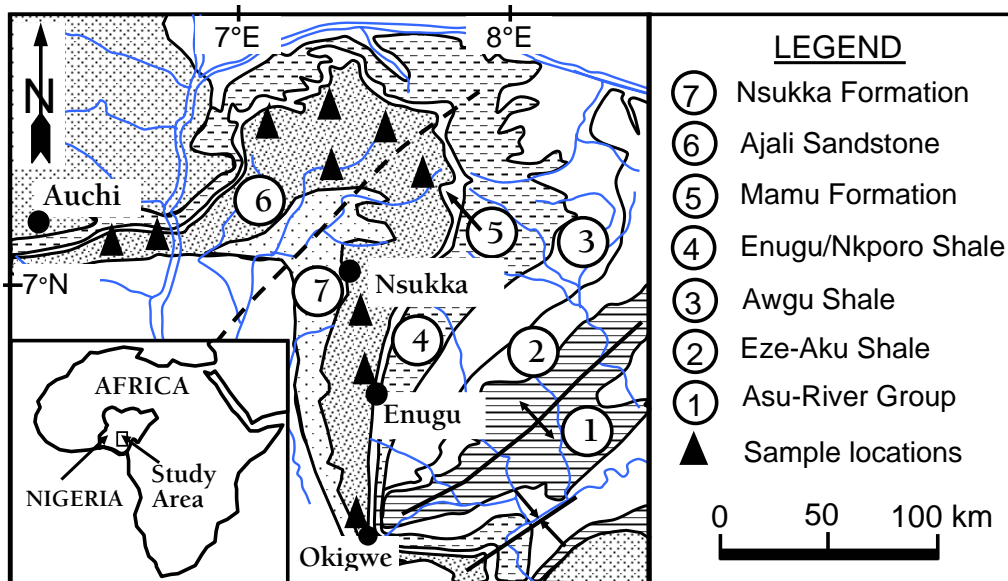
Figure 1. Location and geological map showing sampling points within ASF

METHODS

Subsequent to field sampling operations and pre-treatments (drying and disaggregating), laboratory GSA and constant head permeameter tests were carried on the ASF samples following standard laboratory procedures (ASTM D-422 and D-2434). In addition geochemical analyses of major and trace elements compositions were undertaking using X-ray fluorescence (XRF) method (model *Rigaku ZSX*) at the Department of Earth and Planetary Science Systems, Hiroshima University, Japan.

For data evaluation, cumulative curves were plotted from the GSA results and values of selected grain sizes (D_{10}, D_{30}, D_{60} etc in mm and phi) were extracted for subsequent estimations of some textural indices (uniformity coefficient and coefficient of curvature) and statistical parameters (sorting, skewness, kurtosis, etc) using the relations of Folk and Ward (1957). Using the laboratory permeameter results, the respective hydraulic conductivity values (K) were calculated using the well-known Darcy relation, while empirical estimations of K based on grain size parameters were determined using *Hazen, Beyer, Fair-Hatch and Kozeny-Carmen* relations. Further details on these relations in respect of their applications and limitations can be found in Uma et al., (1989) and Lee, (1998).

RESULTS AND DISCUSSIONS

Textural – Statistical indices

The results of the estimated textural and statistical indices as presented in Table 1 show that the ASF falls predominantly within the medium sand range. However, the estimated inclusive graphic standard deviation (sorting) ranges from 0.55 to1.29ϕ suggesting a wider range from poorly sorted to moderately well sorted material. Also varied values of inclusive graphic skewness from −0.24 to +0.20 (av. 0.02) imply nearly symmetrical to coarse skewed arrangement while graphic kurtosis (KG) ranges from 0.94 to 1.51 (av. 1.20) and suggest mesokurtic through

leptokurtic with majority of the samples having leptokurtic population. For the textural indices, the uniformity coefficient (C_u), which provides a measure of uniformity of the grain sizes ranges from 1.58 to 3.48 while the coefficient of curvature (C_c) ranges between 0.89 and 1.26 (with exception of Uturu-Okigwe sample with C_u=5.25 and C_c=1.57). These values are consistent with C_u<4 and C_c of 1 to 3 implying uniform grading with enhanced drainage/permeability and well-graded granular material respectively (Budhu, 1999).

Table 1. Summary of textural-statistical indices and hydraulic conductivity

Textural-statistical parameters				*Hydraulic conductivities (m/day)*			
Parameters	*Min.*	*Max.*	*Mean*	*Parameters*	*Min.*	*Max.*	*Mean*
G-Mean (φ)	0.98	2.26	1.50	$K_{Permeameter}$	26.90	258.8	118.8
IG-Std. dev. (φ)	0.55	1.29	0.88	K_{Beyer}	6.43	75.60	35.48
IG-Skewness	−0.24	0.20	0.02	K_{Hazen}	6.64	64.80	31.05
G-Kurtosis	0.76	1.51	1.20	$K_{Kozeny-Carmen}$	3.65	36.45	17.84
Cu	1.58	5.25	2.73	$K_{Fair-Hatch}$	2.47	9.71	6.58
Cc	0.89	1.57	1.11	K_{Darcy}	7.23	55.50	31.45
%Sand	75.10	99.70	95.40	Porosity (%)	25.50	32.80	29.35

Determination of hydraulic conductivity

Results of hydraulic conductivity (m/day) estimations from permeameter tests and from a number of empirical relations are also presented in Table 1. The empirical estimations revealed a minimum range of 2.5−9.7 m/d (Fair-Hatch, method) and a maximum range of 6.4−75.6 m/d (Beyer method) in slight contrast to 26.9−258.8 m/d obtained from the laboratory constant head permeameter tests. The relatively higher K-values from the permeameter tests can be attributed to the effects of sample repacking and loss/removal of fines with drainage water during the tests. Nonetheless, among the empirical estimations, only Fair-Hatch method yielded values comparable to the field pump test data (0.24−5.8 m/dy) compiled from Okagbue, (1988). This can be attributed to the fact that the Fair-Hatch relation incorporated different grain size fractions and shape factor, which seems to be a fair approximation of the field packing condition under the influence of the overburden pressures. Generally, the observed trends of different estimations is $K_{permeameter} > K_{Beyer} > K_{Hazen} > K_{Kozeny-Carmen} > K_{Fair-Hatch}$ with corresponding average values of 118.8, 35.5, 31.1, 17.8 and 6.6 m/day respectively. By and large, a comparative presentation of the different K-value estimations and permeability evaluation as presented in Fig. 2, revealed that the ASF falls within a range of 10^{-1} to 10^3 m/day for clean sands and interpreted as permeable to highly permeable unit.
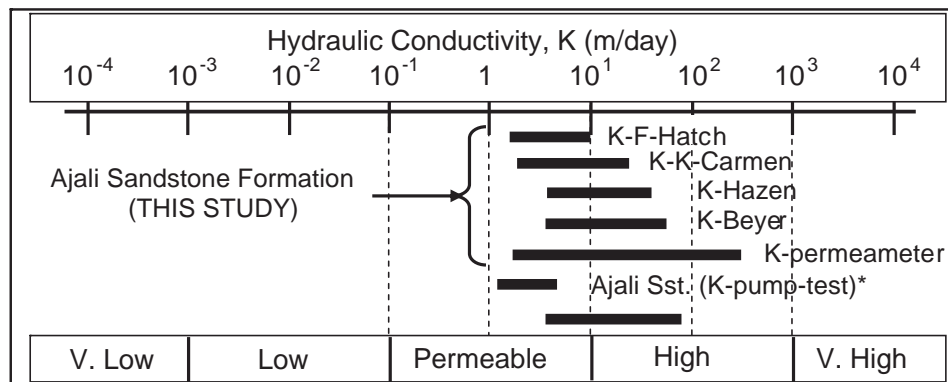
* Benin Formation. *Data source:* Okagbue, (1988).

Figure 2. Profiles of hydraulic conductivity estimations and permeability evaluation

Furthermore, the hydraulic conductivity profile is comparable to similar aquiferous unit of Benin Formation (Okagbue, 1988). In addition, statistical correlation of the data revealed positive dependence of the different K estimations (r = 0.30 – 0.90) on graphic mean size, percentage sand content and porosity. This clearly underlies the significant association between the grain-size distribution and the porosity/permeability of granular materials like the ASF.

Geochemical characterization

The summaries of the geochemical analyses results of major and trace elements are presented in Table 2. As expected, the major element geochemistry shows that the ASF is highly enriched in quartz (94.8–99.0 wt.% of SiO_2) while other major oxides (not shown in Table 2) are generally below 1.0 wt.% in the fresh samples. The similarity in elemental composition for all the analyzed samples is a reflection of the homogeneous nature of the ASF over the different outcrop localities within the Anambra Basin.

Table 2. Summary of geochemical analyses results of the major and trace elements

Parameters	Fresh sarmples (N=12)					Ferruginized samples (N=4)				
	Min.	Max.	Mean	Median	St.Dev.	Min.	Max.	Mean	Median	St.Dev.
SiO_2 %	94.82	98.95	97.21	97.28	1.20	76.79	91.86	83.80	80.19	7.24
Al_2O_3 %	0.33	2.56	1.16	0.97	0.61	3.56	11.57	7.93	10.17	3.92
Fe_2O_3 %	0.03	0.59	0.23	0.19	0.17	1.80	3.57	2.70	2.96	0.79
Cu (ppm)	0.01	1.16	0.27	0.01	0.47	3.07	3.73	3.48	3.64	0.36
Pb (ppm)	2.05	4.43	3.05	3.11	0.84	8.01	15.62	12.31	13.29	3.90
Zn (ppm)	2.66	6.78	4.37	4.10	1.45	7.20	15.18	11.62	12.49	4.06
Ni (ppm)	1.52	3.66	2.53	2.33	0.70	5.41	12.36	9.33	10.23	3.56
Co (ppm)	0.01	0.66	0.22	0.08	0.29	0.78	2.79	1.85	1.97	1.01
Cr (ppm)	5.09	12.83	9.42	11.14	3.28	17.89	58.48	42.92	52.38	21.89
V (ppm)	3.75	15.39	10.64	12.30	4.73	26.94	107.1	77.41	98.24	43.93
Sc (ppm)	0.01	1.80	0.71	0.81	0.67	2.14	8.18	6.17	8.18	3.49

However, the ferruginized samples have similar chemical trend but with elevated concentrations of Al_2O_3 (3.50–11.60 wt.%) and Fe_2O_3 (1.80–3.60 wt.%), which are clear indications of weathering impacts. Furthermore, the ternary, Quartz–Sesquioxides-Alkalis plot of the chemical composition of the ASF samples as presented in Fig. 3 shows that all samples lie at the silica corner indicating severe depletion of the sesquioxide and the Alkali. This is attributed to the removal of ferromagnesian minerals and feldspars through reworking and transport735ation of the source materials during the sedimentary process. As shown in Table 2, there are marked differences between the trace elements concentrations in the fresh and ferruginized samples. However, while the trace metals profile in the ferruginized samples is consistent with trace metals association in typical ferruginous regolith (V, Cr, Nb, Co, Cu, As, Pb, Zn, Mn, and Mo) as outlined in Taylor and Eggleton, (2001), it also clearly shows the significance of weathering process in the release of trace/heavy metals into the environment.
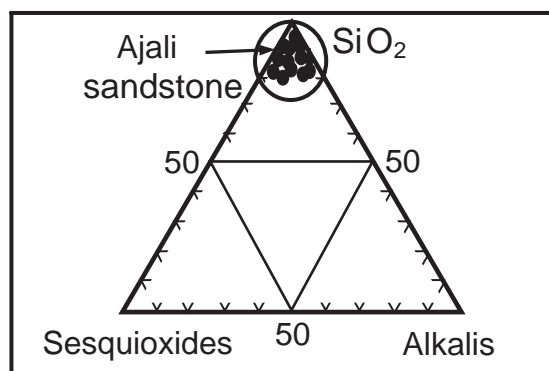
Figure 3. *Quartz–Sesquioxides-Alkalis plot*
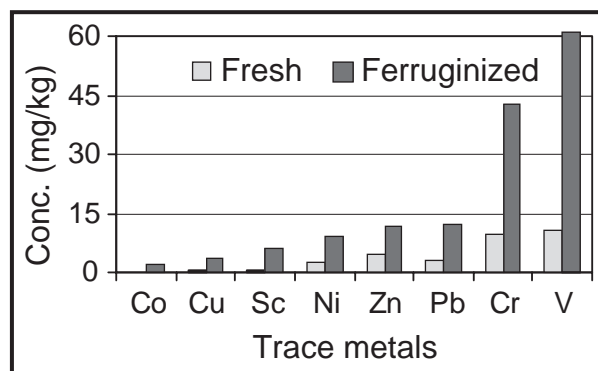
Figure 4. *Trace metals profiles*

Implication for aquifer management

The implication of weathering-induced release of trace/heavy metals in respect of aquifer management calls for concern. As shown in Fig. 4, the weathering process resulted in 2 to 6-folds increase in all the analyzed trace metals compared to the initial concentrations in the fresh samples. Taylor and Eggleton, (2001) noted that the formation of Fe-oxides and oxyhydroxides as shown below dominates weathering process under most regolith conditions and alongside with Mn-oxides and Al-hydroxil complex (in the clay minerals) serve as sinks for wide range of trace/heavy metals demonstrated in this study.

a) Under oxidizing condition (i.e. unsaturated/phreatic zone)

$$Fe^{2+} + \tfrac{1}{2} O_2 + H_2O = Fe(OH)_2$$
$$4Fe^{2+} + O_2 + 10 H_2O = 4Fe(OH)_3 + H^+$$
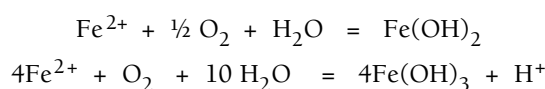
b) Under reducing condition (i.e. saturated zone)

$$3Fe^{2+} + 4H_2O = Fe_3O_4 + 4H_2$$
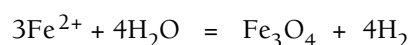
The overall implication is two folds; first in terms of groundwater quality, the recharge induced geochemical process can enhance the dissolution and remobilization of adsorbed trace/heavy metals into the groundwater with attendant negative environmental and human health impacts. Secondly, in terms of aquifer management, the formation of Al_2O_3 and Fe_2O_3 alongside with clay minerals during weathering will lead to reduction in the permeability of the aquifer and filter layers around the borehole on one hand. On the other hand, groundwater flow induced redox reactions, involving such Fe-Mn-oxyhydroxides, can lead to precipitation, encrustation and corrosion of the well (borehole) materials.

SUMMARY AND CONCLUSION

The comparative assessment of empirical K-estimations presented in this study revealed similar range of K-values comparable to laboratory and field results and signifies the reliability of the empirical estimates. However, while the textural and hydraulic characteristics exert positive impacts on groundwater recharge and storage, the geochemical and mineralogical components of the ASF in respect of the ferruginization process signify negative impacts of groundwater flow-induced geochemical processes in terms of (a) groundwater contamination, through mobilization/dissolution of trace/heavy metals (b) water well (borehole) deterioration, through encrustation and clogging of the effective interstitial pore spaces. These underlie the need for adequate hydraulic and geochemical characterization of aquifer as presented in this study.

ACKNOWLEDGEMENTS

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Evaluation of infiltration velocity changes induced by sediment accumulation on the artificial infiltration basin: A case study at Pingtung Plain, Taiwan

C.-S. Ting , M.K. Jean and Y.P. Huang

Abstract

The objective of the on-site groundwater artificial recharge experiment described in this study is to observe and understand the variation of the water infiltration rate and the groundwater mound once recharge begins and to determine the mutual impact of the water recharging volume and the specific aquifer storage capacity. The duration of the current experiment is 17 days, of which 14 days are spend on the general recharge test and 3 days are spent on a sand addition test. In the recharge test, the average recharge volume is 10,000 CMD, the basin water level is maintained at a depth of 1.7 m, and the average infiltration rate is 15.2740 m/day. Following the sand addition test, it is found that the infiltration rate reduces to 5.3740 m/day as a result of the gravel layer becoming clogged by fine clay. By means of a soil characteristic curve, it is found that the fine sediment penetration depth is greater than 35 cm in a gravel filter. The use of a grain size filter during the groundwater recharge operation provides an efficient means of trapping sediment while maintaining the permeability of the underlying aquifer. The experimental results indicate that the factors influencing the infiltration rate include the sediment charge, the water level of the infiltration basin, the capacity of the basin, the distance between the groundwater level and the basin level, and the Sodium Adsorption Ratio.

Keywords

Infiltration rate, groundwater level, sand addition test, soil characteristic curve.

INTRODUCTION

Due to its geographic location and particular meteorological conditions, Taiwan experiences precipitation predominantly during the period from May through October. As a result of hypsographical variation, its water resources are not readily stored on the land surface, but tend to flow rapidly into the ocean. Consequently, the utilization rate of surface water resources in Taiwan is very low. However, the thriving aquaculture found along the southwestern coastal areas of Taiwan is dependent on an abundant supply of freshwater. In most cases, this freshwater has been supplied by the intensive pumping of groundwater. This has led to a drop in the local groundwater level, and in severe cases has resulted in land subsidence and seawater intrusion.

It has been suggested that infiltration basin systems provide an effective means of enhancing groundwater recharging. Many practical groundwater recharge systems using infiltration basins are in operation around the world nowadays. Yamagata City in Japan has used recharge basins and recharge pits for groundwater recharging since 1986. According to Hida (1990), this infiltration basin system provides an excellent recharge effect and has been instrumental in raising the groundwater level. A number of groundwater recharge plans have also been implemented in America. As in Japan, these recharge basins demonstrate an excellent recharge performance. For example, in one recharge basin studied in California, the annual groundwater recharge volume was found to be as high as 1.56×107 m^3 (Bouwer, 1974).

In 1999, Pingtung County local government instructed the CTCI Corporation, Taiwan to identify a suitable location

for a manmade lake in the Pingtung Plain (CTCI, 2003). The CTCI identified Wanlong Farm, located at an upstream position on the Linpien River, as the most suitable site. Several pilot experiments and studies were performed a few years later, and highly favorable results were obtained.

ENVIRONMENT OUTLINE OF STUDY AREA

In accordance with the recommendations of CTCI, this study takes Wanlong Farm, Wanluan Town, Pingtung County, Taiwan, as its study area. The location of the study area is indicated in Figure 1. The groundwater level in this area varies by approximately 30 m over the course of the year, i.e. from approximately 30 m below the surface during the flood season to 58 m below the surface during the dry season.

The specific criteria considered when choosing Wanlong Farm as the study area are summarized in the paragraphs below.
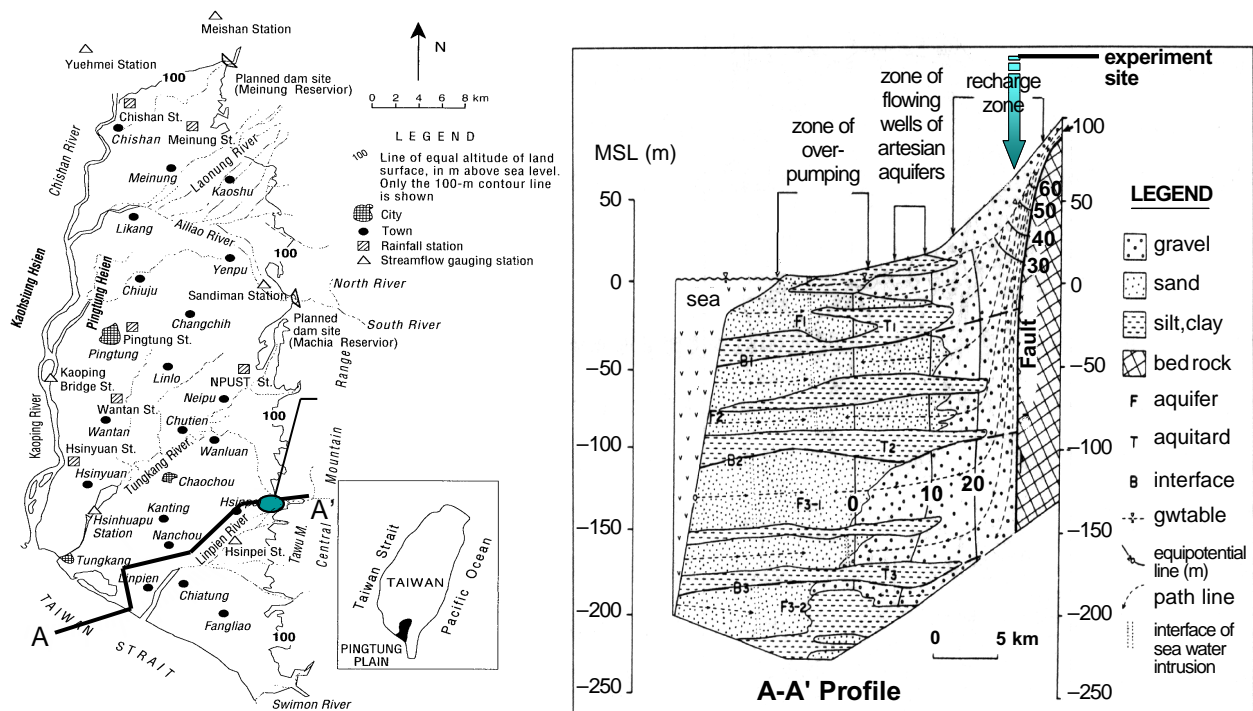
Figure 1. Location of study area in Pingtung Plain, Taiwan (Ting et al., 2002)

Hydrogeological conditions

Wanlong Farm is located at the top of the Linpien River alluvial plain, which consists mainly of coarse pebbles. The natural recharge function of this region is quite reasonable. According to pumping test data, the hydraulic conductivity (T) is 18,818 m^2/day, the specific yield (S) is 0.13, and the permeability coefficient (K) is 344 m/day.

Watershed conditions

Linpien River is located in the southern region of Taiwan. Its length is approximately 42 km and its slope is 1:15, making it the shortest and steepest river in Taiwan. The annual average precipitation is 3,330 mm/year. 97% of the river runoff accumulates during the wet season while the remaining 3% is distributed during the dry season. This temporal imbalance presents water planners with a considerable challenge when attempting to conserve and utilize the river's water resources.

Water intake considerations

In order to achieve water sources for the groundwater recharge of the current experiment, an abundant and stable water supply is needed. The study area is located close to the Er-Fong irrigation channel, which is charged primarily by submerged water flowing through the region and has been used as a local agricultural water supply since 1923. On-site investigations and calculations have indicated that the Er-Fong irrigation channel has a water supply capacity of more than 100,000 m^3/day during the wet season.

METHODOLOGY

The objective of the present study is to use a variety of on-site experimental data, e.g. the infiltration basin water level, the recharge amount, the basin area, etc., to analyze the interrelationship between the manmade lake and the groundwater level. These experiments permit investigation of the variation of the groundwater recharge amount with the natural groundwater level, the moisture capacity of the soil under different loading pressures, the aggraded depth of soil etc. The current study methodology is stated as follows:

Analysis of water SAR parameter

The water quality has a profound influence on the recharge effect. Clogging by waterborne sediment is a fundamental problem in groundwater recharging.

Bouwer (1974) proposed the Sodium Adsorption Ratio (SAR) as a parameter for water quality assessment. The SAR value is a function of the concentrations of Na+, Ca2+ and Mg2+ in the inflow water, where these concentrations are expressed as mEq/L. Accordingly, the SAR parameter is recognized as a key factor in determining the infiltration rate during the recharge process.

Analysis of soil moisture

The movement and storage of soil moisture is strongly influenced by the structure of the soil. A soil-moisture characteristic curve can be drawn by calculating the soil specific retention for different soil types. This curve shows the difference in the water retention capacity caused by the suction effects of different soil textures. In the current study, the variation in the water volume retention observed at different soil depths is analyzed using the soil-moisture characteristic curve in order to estimate the penetration depth of fine sand into a gravel filter.

Mechanisms affection of infiltration rate

Analyzing the Infiltration Rate Standard Chart shown in Figure 2, issued by Pérez-Paricio and Carrera (1998), the initial fall in the infiltration rate is caused by the sediment being expended and then disintegrated after moisturization. The infiltration rate subsequently increases as a result of pore air dissipation as air is dissolved in the water. Finally, the infiltration rate falls gradually as a result of sediment pore clogging. This clogging may be a result of the deposition of sediment carried
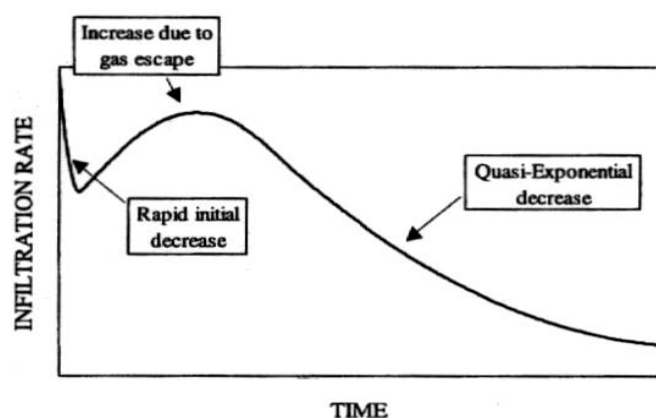
Figure 2. Infiltration Rate Standard Chart (Pérez-Paricio and Carrera, 1998)

by the water in suspension, or may be caused by algal growth, sediment colloidal swelling, microbial activity, or soil dispersion, etc.

Calculation of infiltration using the water-budget method

According to the Law of Continuity, the water-budget method can be expressed as:

$$dV/dt = I - O - E + P - I$$

where dV/dt is the variation of the infiltration basin storage, I is the volume of the inflow, O is the volume of the outflow, E is the volume of evaporation, P is the volume of precipitation, and I is the volume of leakage.

EXPERIMENTAL DESIGN

The current experiment was conducted over a continuous period of 17 days. The general recharge experiment was executed during the initial 14 days, and the sand addition test was then performed in order to investigate the effect of different sand contents on the infiltration rate. Finally, the study compared the results of the indoor sand tank experiment to gain the better understanding of groundwater recharge processes.

On-site recharge experiment

As discussed in Section 2, the experimental site was located inside Wanlong Farm, Pingtung County, Taiwan. Figure 3 shows the on-site arrangement of the infiltration basin and the observation wells. As shown, the basin bottom measured 20 m × 20 m, the basin top measured 30 m × 30 m, the basin depth was 2.5 m, and the slope of the banks was 1:2. Four observation wells were disposed in and around the basin, designated as MW-1, MW-2, MW-3 and MW-4, respectively. A pumping well was also implemented, annotated as SGW-1.
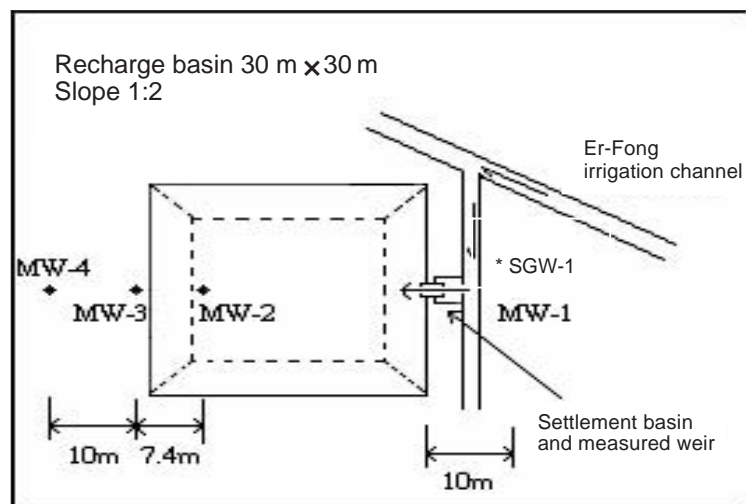
Figure 3. On-site arrangement of infiltration basin and observation wells

Instruments:
(1) Groundwater level recorder: to record the groundwater level in each observation well every hour.
(2) V-notch weir (Triangular weir): to record the water depth above the crotch head every hour in order to establish the variation of the water inflow.
(3) Basin water level: to ensure the water level remained at a constant depth of 1.7 m.
(4) Microclimate station: located at the southeastern side of the study area and designed to record the precipitation, evaporation, temperature, humidity and atmospheric pressure data automatically.

DATA ANALYSIS AND DISCUSSION

On-site recharge experiment and sand addition test

Infiltration velocity

The observation results are presented in Figure 4. It can be seen that the initial infiltration rate was 13.11 m/day. The infiltration rate then stabilized at 15.61 m/day. However, following sand addition, the infiltration rate fell gradually to a value of 5.25 m/day, a reduction of approximately 67%. Hence, grading size of the sand is a key factor influencing the groundwater recharge effect.
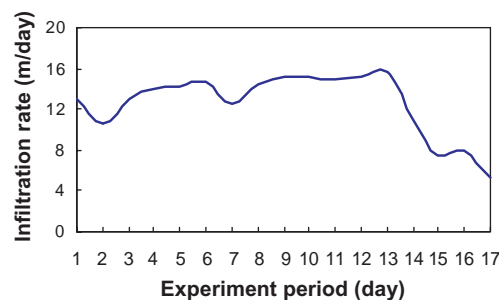
Figure 4. Temporal variation of infiltration speed

Groundwater level variation

Following commencement of the infiltration test, the groundwater levels in wells MW-1, MW-2 and MW-3, located in and around the infiltration basin, were seen to rise gradually. However, during the same period, the groundwater level in MW-4 remained constant at approximately 38–32 m below the surface. Figure 5 shows the temporal variation of the groundwater levels in the

Figure 5. Temporal variation in groundwater levels in four observation wells

four wells. It can be seen that the water levels in MW-1 and MW-2 increased significantly after 24 hours. In the final stages of the experiment, it is observed that even the groundwater level in well MW-4 increased by approximately 5 m. These results suggest that following groundwater recharging, the infiltration water becomes shaped into a so-called water bank around the experimental site.

Discussion of infiltration mechanisms

This study compared seven factors known to influence the infiltration rate, namely the basin water level, the difference between the basin water level and the groundwater level, the basin area, the storage volume, the recharge inflow, the sand content, and the water quality. The primary estimation results for the correlation of each factor to the infiltration rate are presented in Table 1.

Figure 6 shows the variation over time of the sand content at the bottom of the basin. It can be seen that the maximum value of the sand content is 252.3 ppm, but that the minimum value is only 6.4 ppm.

The water quality was sampled every 12 hours during the sand addition test. Equation 1 was applied to estimate the value of the SAR. This study investigated the influence on the water infiltration rate of each of the parameters listed in Table 1 by means of regression equations. The infiltration and sand aggregation regression equations are presented in Tables 2 and 3 below.

Table 1. Correlation coefficient for each factor relating to the infiltration rate

Item	Correlation coefficient
Basin water level, **h**	0.57
Water level underground, **H**	0.41
Infiltration area, **A**	0.23
Storage volume, **V**	0.54
Recharge amount, **Q**	0.16
Sand content, **S**	-0.68
Water quality, *SAR*	0.39

Figure 6. Temporal variation of sand content at basin bottom
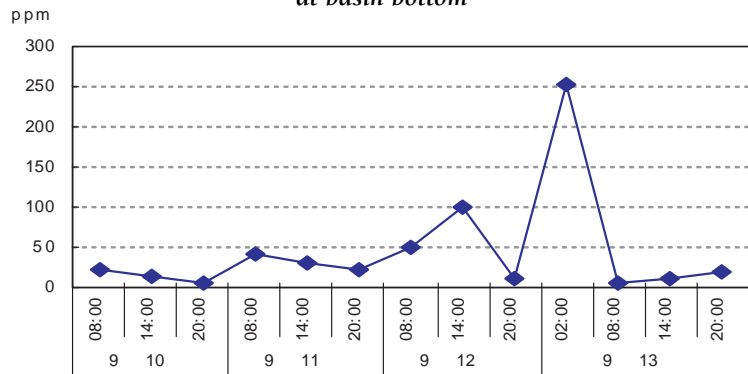
Table 2. General infiltration regression analysis

Parameters	Linear regression equation	R^2
h, A, V, Q	$I = 4.8h + 0.0005A - 0.014V + 0.04Q$	0.92
h, V, Q	$I = 4.89h - 0.014V + 0.04Q$	0.92
h, V	$I = 27.93h - 0.045V$	0.41
h	$I = 1.57h + 11.95$	0.27

Table 3. Sand aggregation regression analysis

Parameters	Linear regression equation	R^2
SAR and S	$I = 12.02SAR - 0.001S$	0.39
S	$I = -0.0081S + 9.98$	0.41
SAR	$II = 13.27SAR - 4.47$	0.39

Indoor experiments

Sedimentation depth calculation

A pit measuring 30 cm in length and width and 50 cm in depth was dug in the bottom of the basin. For soil sampling purposes, the pit was separated into three layers. Figure 8 shows the experimental grain size distribution curves obtained at each depth. The percentage content of clay, silt, sand, and gravel in the soil samples was then calculated according to the 'Triangular Soil Classification'.
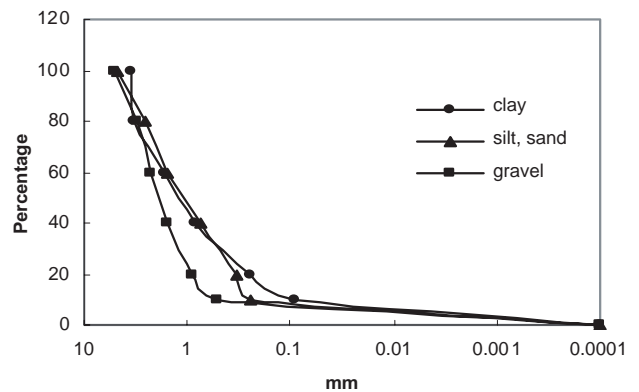
Figure 8. Grain size distribution curves at three pit layers

The saturated hydraulic conductivity and porosity of the soil samples were also measured. Darcy's Law was applied to calculate the saturated hydraulic conductivity K, K=QL/HA, where Q represents the water inflow volume during a specific time (cm^3/sec), L represents the soil sample column length (cm); H represents the water head difference (cm), and A represents the soil sample column section area (cm^2). The calculated results are presented in Table 4.

Porosity calculation

The porosity of the soil sample column was calculated on the basis of its bulk density and particle density. Hence, it was easily measured by indoor experiment. The porosity results are also presented in Table 4.

Table 4. Saturated hydraulic conductivity and porosity of different soil layers

Soil Layer	Saturated hydraulic conductivity (m/sec)	Porosity	Soil characteristic
1st layer (0–15cm)	0.429×10^{-4}	0.48	Sandy clay
2nd layer (15–30cm)	0.572×10^{-3}	0.34	Sand loam
3rd layer (30–50cm)	0.294×10^{-2}	0.32	Sand

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

(1) Based on the present experimental results, the water infiltration rate is determined primarily by the sand content of the water inflow. The infiltration rate is also influenced by the basin's hydrological disposition and the SAR value of the charge water.

(2) Following completion of the sand addition test, a sediment depth of 6.4 cm was measured on the bottom of the basin. The infiltration rate slowed from 15 m/day to 5 m/day over the course of the test. The main reason for this reduction was algal growth and eutrophication of the water quality.

(3) The groundwater level records of the four observation wells disposed around the study area indicated a high water infiltration rate in the chosen study area. Hence, the hydrological conditions of Wanlong Farm render this region an eminently suitable site for the implementation of an artificial groundwater recharge scheme.

Recommendations

(1) The selection of the recharge basin site must consider the geological permeability of the proposed site.

(2) To prevent sediment clogging, thereby maintaining a normal infiltration process, silt scraping of the basin bottom should be performed at regular intervals. Additionally, the use of a relative sediment material grading research and thickness design is recommended.

(3) Researching the water quality variation before and after recharge is an important issue in order to establish whether the quality is influenced by the local geological conditions. In the current study, the experiment extended for a relatively short period of time, and hence the effect of the water quality on the infiltration rate was not obvious. Hence, it is recommended that a longer-term pilot study of groundwater recharge be conducted as part of a future study.

ACKNOWLEDGEMENTS

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Infiltration mechanism
of artificial recharge of groundwater –
A case study at Pingtung Plain, Taiwan

C.-S. Ting, C.H. Lee, C.Y. Lin, S.H. Chen and K.C. Chang

Abstract

This paper discusses the artificial groundwater recharge effect of high-infiltration basins. Hydrogeological parameters of the study area are collected in order to construct a conceptualized physical model. Numerical simulation software, TOUGH2, is then used to simulate the infiltration behavior of an artificial recharge into an underground aquifer. Four wells (MW-1, MW-2, MW-3, and MW-4) are observed at the field site and the groundwater levels are compared with the simulation results. It is found that good agreement exists between the observed and numerical data for wells MW-1 and MW-2. However, the observed groundwater level in MW-3 is higher than the simulated level. MW-3 is located at the edge of the artificial recharge lake, and the high ground-water level may well be the result of a portion of the infiltration load following the well border into the well screen. Conversely, the groundwater level in MW-4 is found to be lower than in the simulated well due to local permeability in the well location. Finally, the numerical results predict that the groundwater level will attain a steady state at approximately 47 hours after the beginning of infiltration.

Keywords

Infiltration, artificial recharge of groundwater, Taiwan.

INTRODUCTION

Taiwan is located in a sub-tropical region and hence receives a very high annual rainfall. However, the amount of rainfall received varies significantly over the course of the year. According to measurement data, 90% of the annual precipitation falls during the rainy season (i.e. May to September) while only 10% of the rainfall occurs during the dry season (October to April). This imbalance in the monthly rainfall presents planners involved in the conservation and utilization of water resources in Taiwan with a major problem.

The flourishing development of inland aquaculture along the coastal areas of southwestern Taiwan relies on an abundant freshwater supply. Groundwater represents the primary source of fresh water for aquacultural water diluting and flushing. Furthermore, the swelling population in these regions has also led to an increase in the domestic water consumption. To meet these increased water demands, groundwater pumping schemes have been implemented extensively along the coastal areas. However, this has resulted in a significant fall in the groundwater level, and in severe cases has been directly responsible for environmental and geological hazards, including land subsidence, soil salinization, and seawater intrusion. In order to address these problems, local government initiatives have been launched to investigate the potential for exploiting new water resources and developing artificial groundwater recharge schemes.

Artificial groundwater recharging has developed into a major element of Taiwan's integrated water resource exploitation policy over recent decades. However, much work remains to be done in developing effective means to transfer excess surface water into an aquifer during the rainy season in order that this water can be conserved for use in the dry season.

This study chooses Wanlong Farm in the southern region of Taiwan as a pilot research area. Wanlong Farm is located in an upstream position on the Linpien River in the Pingtung Plain. The object of this study is to use **TOUGH2** numerical software to simulate the groundwater infiltration situation in this area and to compare the numerical results with observed data.

Model theory

TOUGH2 (Pruess, 1999) is a general-purpose numerical program for multi-phase fluid flow in a porous and fractured stratum. This model uses the integral finite difference method for space discretization, and first-order fully implicit time differencing. The basic mass and energy balance equations solved by TOUGH2 can be written in the general form as (Pruess, 1999):

$$\frac{d}{dt} \int_{V_n} M^k dV_n = \int_{\Gamma_n} F^k \bullet n d\Gamma_n + \int_{V_n} q_m^k dV_n \tag{1}$$

where V_n is any arbitrary sub-domain of the flow system, and is bounded by a closed surface Γ_n; M^k is the accumulation term of the mass or energy per volume, with $k = 1, 2... NK$, $k = NK + 1$, labeling the mass components; F^k is the mass and energy flux; q denotes the mass sinks and sources; and n is the normal vector of the surface element $d\Gamma_n$.

Capillary pressure and relative permeability are the two fundamental control parameters in the **TOUGH2** simulation of an unsaturated stratum. The relative water content S_e is usually used to describe the capillary pressure of the soil. The relative water content is a function of the actual versus potential maximum moisture content and can be expressed as:

$$S = \frac{\theta - \theta_r}{\theta_s - \theta_r} \tag{2}$$

where θ is the fresh weight, θ_s is the turgid weight and θ_r is the dry weight. This study uses the water retention equation proposed by van Genuchten (1980), i.e.

$$S_e - \left[(1 + \alpha\psi)^n\right]^m, m - 1 - \frac{1}{n} \tag{3}$$

where S_e is the relative water content; α is the empirical fitting parameter; Ψ is the soil capillary pressure; n is the pore index; and m is the fitting parameter. Furthermore, the relative permeability of the soil is simulated by the van Genuchten & Mualem relative permeability model incorporated in **TOUGH2** as:

$$K_{rl} = \sqrt{S_e} \left\{ 1 - \left(1 - [S_e]^{1/m} \right)^m \right\}^2 \tag{4}$$

where K_{rl} is the relative permeability of soil in a liquid status; S_e is the relative water content; and m is a fitting parameter.

ON-SITE CASE STUDY AND SIMULATION

Study area introduction

Linpien River is located in the southern area of the Pingtung Plain in Taiwan. With a total length of 42 km and an average slope of 1:15, Linpien River is one of the shortest and steepest of the 21 main rivers in Taiwan. The average

annual precipitation in this region is 3,330 mm, the area of the watershed is approximately 344 km^2, and the average annual river discharge is 864×10^6 M^3. However, the distribution of the annual precipitation is severely unbalanced over the year. As a result, 97% of the annual river discharge occurs during the wet season while only 3% is provided in the dry season. Hence, fully exploiting and utilizing the water resources provided by this river is highly challenging. Accordingly, in 2002 the Taiwan CTCI Corporation proposed the construction of a manmade lake beside the river and conducted a small-scale recharge experiment in situ. The stated intention of the proposed lake was to intercept the abundant surface water during the wet season and to recharge the water into an underground aquifer for later use during the dry season.
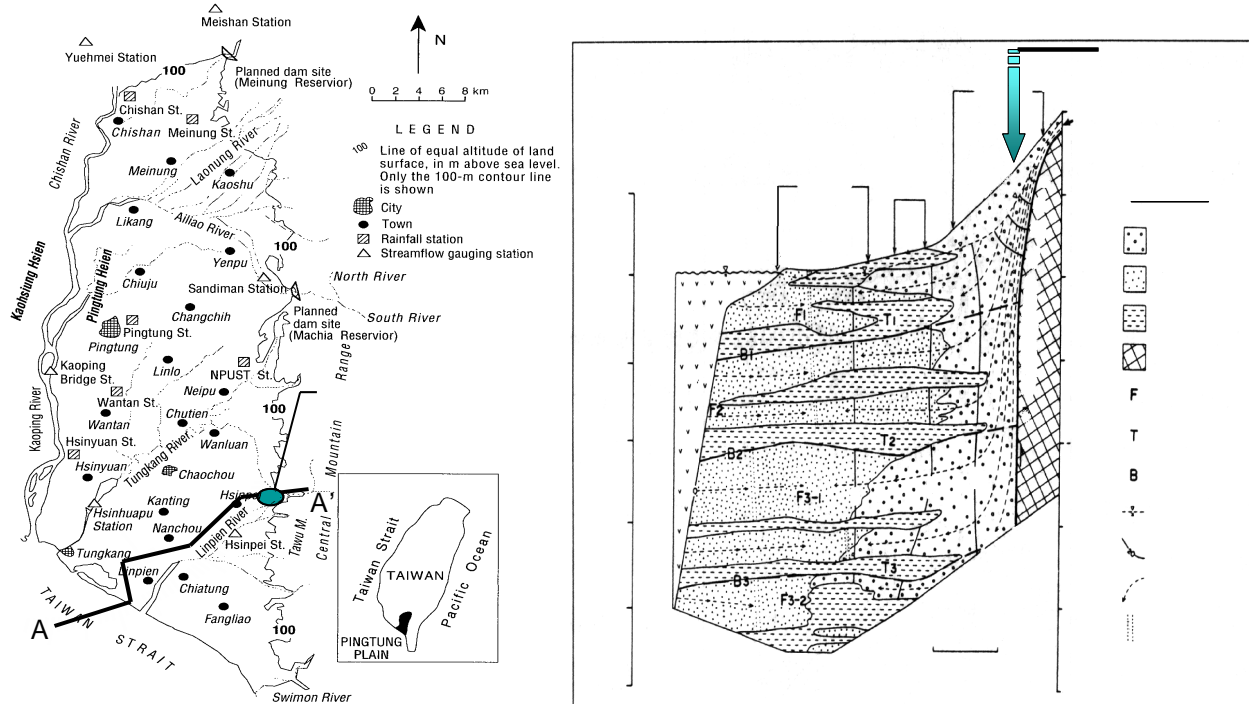
Figure 1. Location of study area in Pingtung Plain, Taiwan (Ting et al., 2002)

Pilot study site description

(I) Pilot test location selection

Bouwer (1999) recommended that the initial stages of a groundwater recharge plan be kept small and simple. He stated that the plan could subsequently be enlarged to a more realistic operational scale according to the identified future needs, such as the target recharge volume, for example. A small-scale on-site pilot test is a suitable first step in implementing a groundwater recharge scheme.

Previous hydrogeological data analysis and long term investigations identified Wanlong Township on the Linpien River in southern Taiwan as a major groundwater natural recharge area. Hence, this study chose this area as a suitable pilot study site for an infiltration test.

(II) Observation well disposition

CTCI (2002) constructed four groundwater level observation wells in situ. As shown in Figure 2, the wells were constructed along the direction of the water flow. The first well (MW-1) was located in the upstream region of the pilot test basin, MW-2 was located actually in the test basin, and the remainder (MW-3~4) were located downstream

of the basin. The observation wells were all embedded 64 m underground. Each well was constructed with a 30 m well screen located 32 m ~ 62 m underground, as shown in Figure 3.
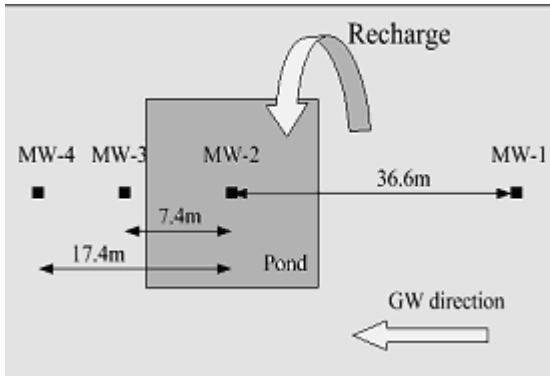
Figure 2. Disposition of groundwater level observation wells
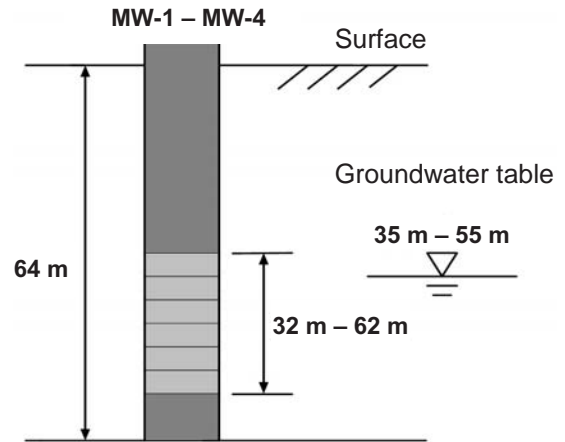
Figure 3. Position of well screen

Construction of on-site numerical model

(I) Model boundary conditions

The current simulation area was aligned in the direction of the observation well array. As shown in Figure 4, the vertical section of the simulation area measured 80 × 50 m (length × depth). The infiltration basin was positioned in the central lower area and measured 10 × 1 m (length × depth). The imposed boundary conditions are indicated in Figure 4. This study assumed that no lateral flow existed in the unsaturated layer, and neglected the precipitation inflow. Hence, the upper and lower sides of the section, and the two lateral sides of the unsaturated section, were specified as no flow – Neuman boundaries. However, the two lateral sides of the saturated section were each assigned a constant head – Dirichlet boundary condition.
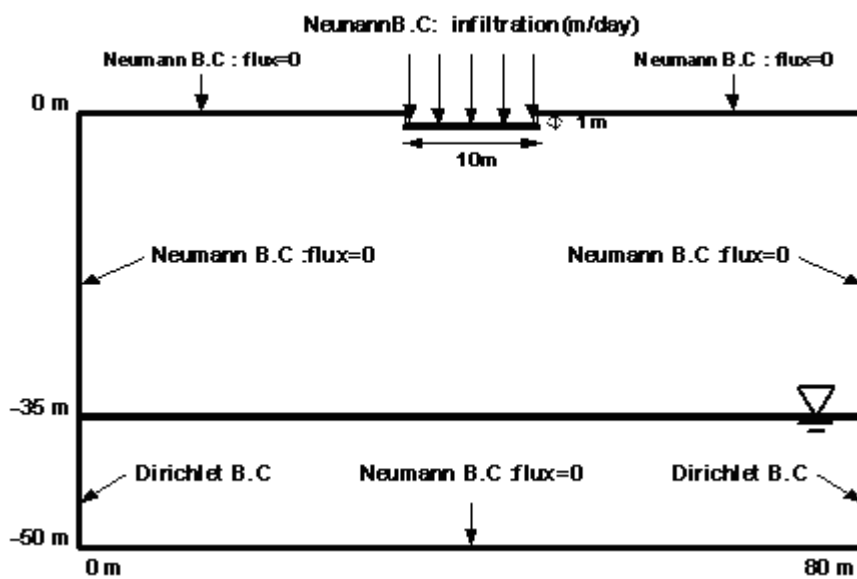
Figure 4. Simulation model and boundary conditions

(II) Hydrogeological parameters

(1) Pilot infiltration basin scale and recharge description

As shown in Figure 5, the dimensions of the physical infiltration basin were 11.5 × 11.5 × 2.7 (length × width × height). The lateral sides were inclined with a 1:0.97 slope. The pilot test commenced at 18:00 PM on the 8th of October 2001 and continued until 14:00 PM on the 16th of October. Hence, the total continuous testing time was 7 days and 20 hours. During this period, the average volume of water inflow was 2129 CMD and the water level in the basin was maintained at a height of 1.7 m above the bottom of the basin.
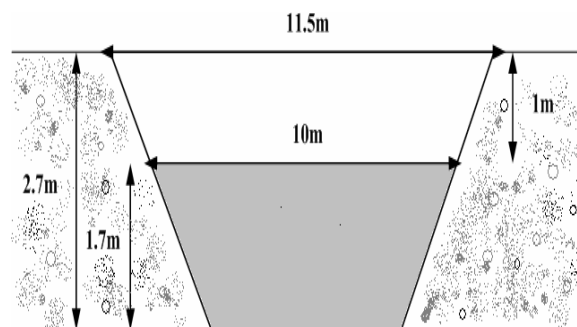
Figure 5. Dimensions of physical infiltration basin

(2) Infiltration rate

Figure 6 shows the temporal variation of the water infiltration rate in the downward direction. It is observed that the average value of the infiltration rate is 22.76 m/day.
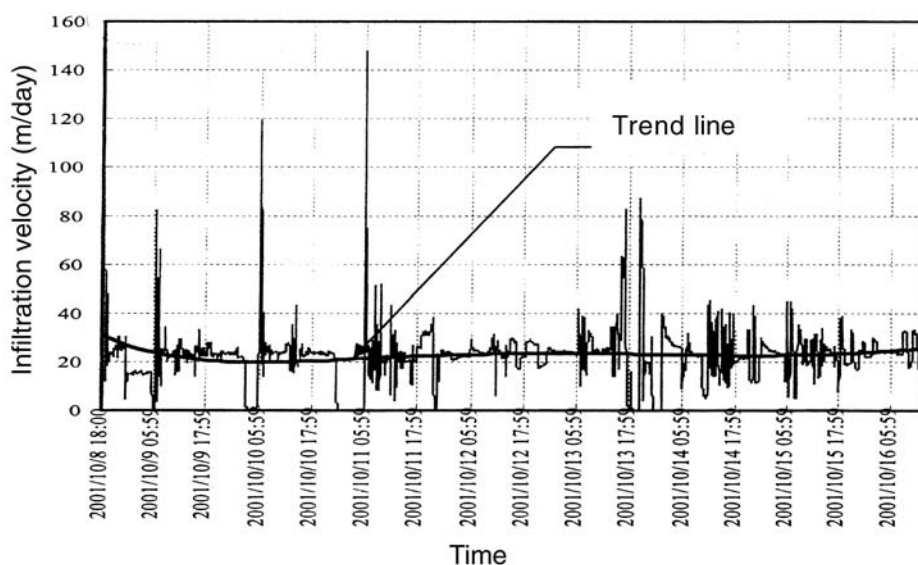
Figure 6. Temporal variation of water infiltration rate over study period

(3) Porosity

The study region was characterized by the presence of sand and gravel. Accordingly, in the present simulations, the average porosity parameter was specified as 0.4.

(4) Van Genuchten soil characteristic parameters

Carsel and Parrish (1998) evaluated 12 different soil types and established the corresponding soil characteristic parameters, as shown in Table 1. On-site investigation showed that most of the soil in the present study region was sand, and hence the current simulations specified the Van Genuchten soil characteristic parameters as: α = 0.145 cm^{-1}, n = 2.68, m = 1–(1/n) = 0.626, residual volumetric moisture content = 0.045 and saturated moisture content = 0.43.

Table 1. Characteristic parameters of different soil types (Carsel and Parrish, 1988)

Soil Texture	θ_r	θ_s	α [1/cm]	n
Sand	0.045	0.43	0.145	2.68
Loamy Sand	0.057	0.41	0.124	2.28
Sandy Loam	0.065	0.41	0.075	1.89
Loam	0.078	0.43	0.036	1.56
Silt	0.034	0.46	0.016	1.37
Silt Loam	0.067	0.45	0.020	1.41
Sandy Clay Loam	0.100	0.39	0.059	1.48
Clay Loam	0.095	0.41	0.019	1.31
Silt Clay Loam	0.089	0.43	0.010	1.23
Sandy Clay	0.100	0.38	0.027	1.23
Silty Clay	0.070	0.36	0.005	1.09
Clay	0.068	0.38	0.008	1.09

(5) Groundwater level and hydraulic gradient

Based on the on-site recorded data, the simulations set the initial groundwater level as 35 m underground and applied a hydraulic gradient of 0.0125.

(III) Simulation results and data analysis

The parameter settings described above were input to the TOUGH2 simulation software. The corresponding simulation results are presented in Figures 7 with 8. Figure 8 shows that once the infiltration stabilizes, the simulated groundwater levels in wells MW-1 and MW-2 are in good agreement with the observed water levels. However, the simulation results for the MW-3 and MW-4 water levels are significantly different from those observed in-situ. For example, the actual groundwater level in MW-3 is higher than the predicted result. This particular well is located at the edge of the infiltration basin, and the higher groundwater level may be the result of water in the basin flowing
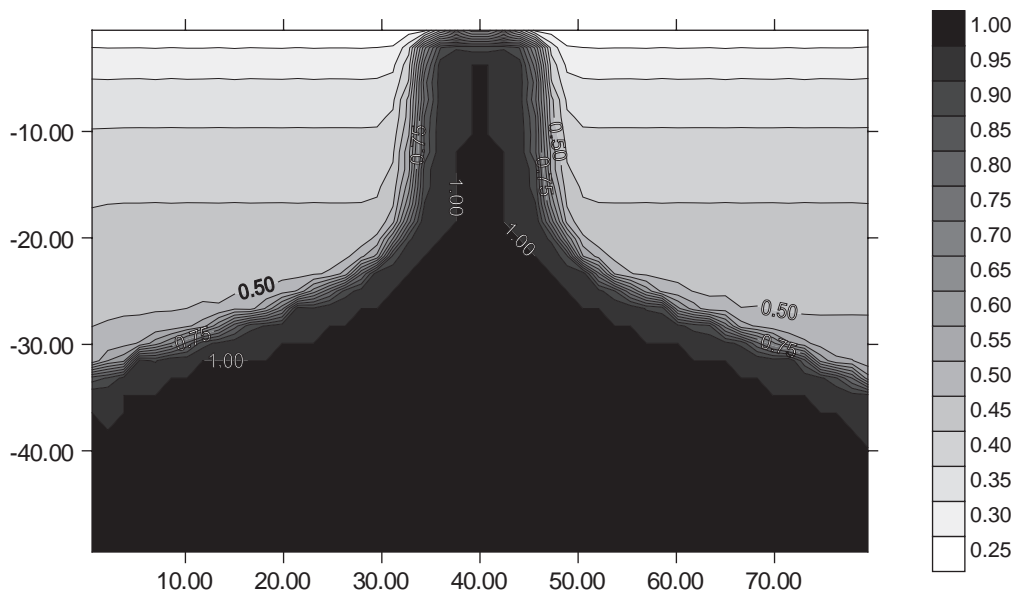
Figure 7. Simulation results for water infiltration
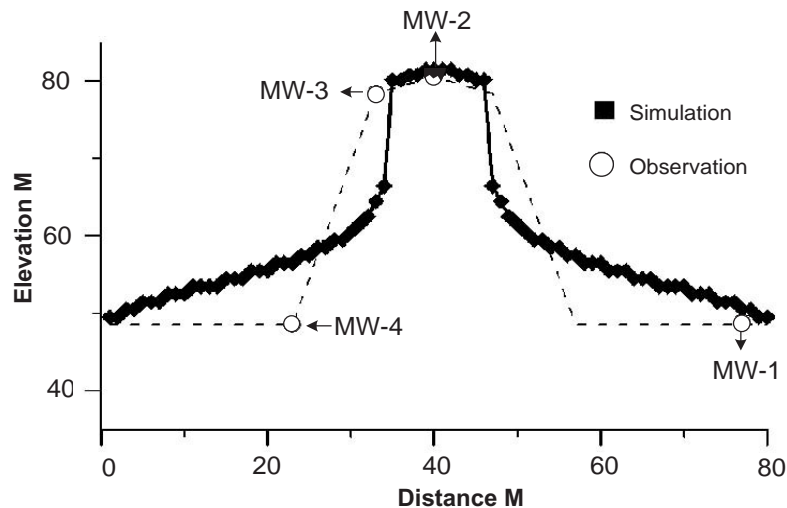
Figure 8. Comparison of simulation and observed results for groundwater levels

into the well screen along the well during the course of the experiment. In the case of well MW-4, the observed groundwater level is lower than the predicted value. The discrepancy between the two sets of results can be attributed to the high permeability of the local area around this particular well, which causes the well to have a low storage capacity with the present simulations take no account of water flowing into the well screen or of local permeability characteristics.

On-site investigation showed that the water levels in wells MW-2 and MW-3 stabilized 50 minutes and 240 minutes, respectively, after the start of water infiltration. The speed with which the water level in MW-2 stabilizes may well be because this well is located immediately below the infiltration basin. Conversely, MW-3 is located at the edge of the infiltration basin, and hence water infiltration is a more gradual process. The water levels in wells MW-1 and 4 showed very little change over the course of the infiltration experiment since these two wells are located outside of the impact radius of the infiltration basin. Finally, the simulated time for the infiltration to become stable was 47 hours. This was longer than the observed time.

CONCLUSIONS

- The simulation results have shown that once infiltration achieves a stable condition, the simulated groundwater levels in the MW-1 and MW-2 wells are consistent with the observed data. MW-3 is located at the edge of the infiltration basin, and hence during the course of infiltration, the water in the basin not only percolates downward, but also flows into the well screen along the wall of MW-3. Hence, the observed water level in MW-3 is higher than the predicted result. Conversely, MW-4 has a high local permeability and hence its water holding capacity is poor. Therefore, the simulated water level in MW-4 is higher than that observed in-situ.

- MW-2 is located immediately below the infiltration basin and therefore its water level rises much more rapidly than the level in MW-3, which is located at the edge of the infiltration basin.

- The water levels in MW-1 and MW-4 remain virtually unchanged over the course of the infiltration experiment because they are positioned outside of the impact radius of the infiltration basin.

- The total simulation time required for the infiltration to stabilize is 47 hours. This is longer than the time observed in-situ.

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Groundwater recharge:
Results from deep injection tests in Achaia aquifer systems, SW Greece

X. Stavropoulos and K. Voudouris

Abstract

This paper presents the results of groundwater recharge tests in Achaia aquifer systems, through deep boreholes. The study area consists of Post-alpine clastic deposits, flysch and carbonate formations. The main aquifers are developed in terrestrial deposits and carbonate rocks. Groundwater recharge tests were carried out, through six boreholes, during 2002. The depth of boreholes was greater than 180 m. The winter runoff of torrents and water from springs were the source of recharge water used in the experiments, without any treatment. The duration of the recharge tests ranged from 18 to 63 days. During the entire recharge period, the total volume of recharged water in six boreholes was 137,000 m^3. The water level in the test boreholes was rising after the end of recharge experiment at a distance about 500 m from recharge boreholes. The maximum rise of groundwater level at the end of the tests was 45 m in the aquifer of the clastic deposits and the minimum 3.26 m in carbonate aquifer. The investigations have shown that the groundwater recharge through deep boreholes is one of the options available for increasing the groundwater reserves and improving the groundwater quality in this area.

Keywords

Achaia. Greece. groundwater recharge. injection borehole. field test.

INTRODUCTION

Groundwater is the main source of water for irrigation and domestic use in NW Achaia. The area is intensively used for agriculture. The land is mainly used for the cultivation of citrus fruits, olives and vineyards, especially in the lowlands. Due to overexploitation, the aquifer systems show signs of depletion and quality deterioration. Sufficient water support is problematic, especially in the dry periods. In order to minimize the negative effects of overexploitation, aquifer recharge tests have been carried out, by using surface water. Direct recharge is used, where permeable soils and/or sufficient land area for surface infiltration are not available and aquifers are deep and/or confined (Peters, 1985; Bouwer, 1996). A number of artificial recharge projects have been carried out in North Peloponnesus e.g. Fleet and Voudouris (1995); Koumantakis et al. (1999); Voudouris et al. (2002). The results of these projects have been very encouraging and demonstrated the feasibility of recharging aquifer systems by injection boreholes. According to Murray and Tredoux (2002), aquifer recharge method via boreholes is not only applicable to large scale, but it can be used effectively in small scale operations.

This paper deals with the results of groundwater recharge field experiments in NW Achaia, via boreholes, during 2002. Previous investigation on hydrogeology in the study area, including geological and hydrogeological information (depth and type of geological formations, aquifer, water level measurements, hydraulic parameters, direction of groundwater flow etc), has been done by Stavropoulos (1992) and Voudouris (1995).
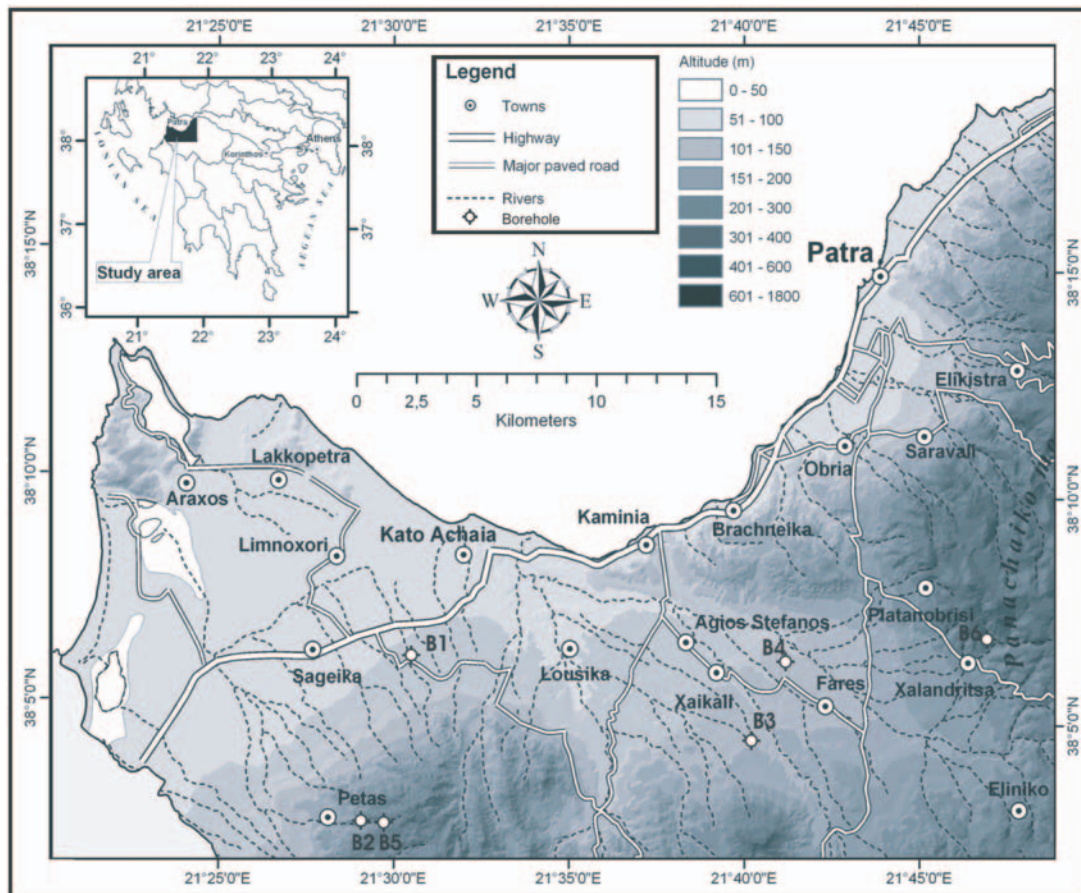
Figure 1. Location of the study area, showing the position of the recharge boreholes

GEOLOGY-HYDROGEOLOGY

The study area is located in the northwestern part of Achaia prefecture (Fig. 1) covering the Larisos and Piros basins. The mean annual precipitation of the area is 660 mm and the average annual temperature is 17.8 °C (Voudouris, 1995).

Geologically, the area consists of Plio-Quaternary deposits (conglomerates, sands, sandstones, silty sand, clays), which form one multiple aquifer system. From bottom to top, the following principal lithofacies have been recognized (Zelilidis et al., 1988): A) Marine deposits (very fine grained sand, silt, silty clay), B)Lacustrine deposits (sand, sandy silt and silt), C) Terrestrial fluvial deposits (conglomerate, sand, silty sand), D) Alluvial deposits (conglomerate with matrix sand), and E) Terrestrial fluvial deposits (conglomerate, sand). Flysch deposits crop out in the southern part of the study area, comprising of beds of sandstone and marls, alternating with conglomerates. Flysch sediments overlie the limestones (Voudouris 1995).

The terrestrial facies of the Plio-Quaternary deposits form a multiple aquifer system, which is at least 200 m thick. The confined aquifer of these deposits is the main source, that covers the water demands, by numerous deep boreholes. The water level within these deposits was at an average depth of about 10–51 m below ground surface (b.g.s.). Last decades, groundwater extraction for irrigation use, has caused a significant decline in the piezometric level (Voudouris et al., 2002).

The direction of groundwater flow is generally from South to North-East at a hydraulic gradient of 1‰ −1.7%, as

measured from the compiled piezometric maps (Stavropoulos, 1992, Voudouris, 1995). The mean hydraulic conductivity was estimated to be 1.5 x 10^{-5} m/s and the S-coefficient in Plio-Quaternary deposits about 1.5 x 10^{-4}.

FIELD EXPERIMENTS

Six (6) new boreholes were drilled (Fig. 1): three in Larisos basin (B1, B2, B3), and three in Piros basin (B4, B5 and B6). The selection of the location was done by using criteria like availability of land, high permeability, gravity flow from water source to recharge borehole, etc. After each borehole had been drilled, pumping tests were carried out, in order to estimate the critical yield (Table 1).

Table 1. Drilling data of recharge boreholes.

Borehole	Geological formation	Depth (m)	Critical yield (m3/h)	Groundwater level (m b.g.s.)
B1	Sandstones and conglomerates (flysch deposits)	242	75	51.08 (1/3/02)
B2	Sandstones and conglomerates (flysch deposits)	241	35	23.46 (10/4/02)
B3	Sands, gravels, conglomerates	215	100	9.61 (4/4/02)
B4	Sands, gravels, conglomerates	196	100	22.61 (8/4/02)
B5	Sandstones and conglomerates (flysch deposits)	202	25	23.08 (6/3/02)
B6	Limestones	180	30	6.21 (18/4/02)

Water from surface runoff and springs, without any treatment, was used during the aquifer recharge experiments in the boreholes, (Table 2). The water was transferred at each site via the existing irrigation network. Measurements of groundwater level were made in the piezometers of the recharge boreholes, as well as in the neighbouring observation boreholes.

Table 2. Results of the conducted groundwater recharge tests

Borehole	Recharge water	Period/Duration (days)	Recharge flow rate (m3/h)	Water volume (m³)	Water level rise (m)
B1	Surface water	1/3/2002–2/5/2002 63 days	42	63,500	44.93
B2	Surface water	10/4/2002–30/4/2002 20 days	13	6,200	19.83
B3	Discharge of springs	4/4/2002–4/5/2002 30 days	10	7,200	4.16
B4	Irrigation canal	8/4/2002–3/5/2002 26 days	75	46,800	8.51
B5	Surface water	6/3/2002–6/4/2002 32 days	12	9,200	13.61
B6	Discharge of springs	18/4/2002–5/5/2002 18 days	8	3,500	3.26

The recharge water was of low conductivity, Ca-HCO$_3$ type. Problems were associated with the high turbidity values of recharge water from torrents. According to American Society of Civil Engineers (ASCE, 1998), the turbidity of recharge water for groundwater recharge should not exceed 2–5 NTU.

RESULTS

The results of groundwater recharge tests are presented in Table 2. Recharge duration ranges from 18 days (B6) to 63 days (B1). Irrigation starts from the beginning of May; thus duration of recharge tests was not longer.

The recharge flow rate ranges between 8 m^3/h (B1) and 75 m^3/h (B4). The optimal recharge rate is one-third or half of the maximum discharge rate (Giao and Nutalaya, 1998). The quantity (m^3) of injected water was: 63,500 m^3 (B1), 6,200 m^3 (B2), 7,200 m^3 (B3), 46,800 m^3 (B4), 9,200 m^3 (B5) and 3,500 m^3 (B6). The total volume of recharge water in six boreholes was 137,000 m^3.

The rise of the groundwater level at the end of the test ranges from 3.26 m (B6) to 44.93 (B1), assuming that the water level due to natural recharge was negligible. The groundwater level rise was decreasing with the distance from recharge borehole. A longitudinal cone formed as a result of the groundwater recharge, oriented with the groundwater flow.

Figure 2 shows the fluctuation of the water level in the recharge borehole B3 and in the observation boreholes (150–700 m distance). The groundwater level rise is extended from the recharge borehole at a distance greater than 500 m.
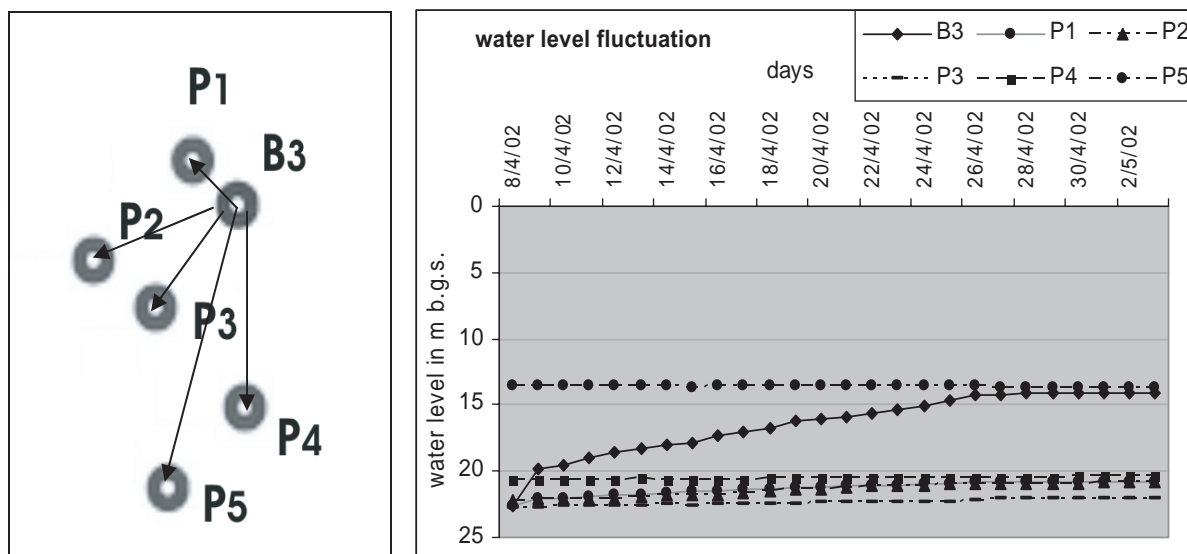
*Figure 2. Impact of recharge on groundwater level in borehole B3 and observation boreholes
(B3P1 = 150 m, B3P2 = 350 m, B3P3 = 300 m, B3P4 = 450 m, B3P5 = 700 m)*

Figure 3 shows the fluctuation of the water level in the recharge borehole B6, which has been drilled in carbonate rocks. The groundwater level rise in the recharge borehole was 3.26 m at the end of the recharge period. It has not been recorded groundwater level rise in two neighbouring observation boreholes at distance 400 and 1,600 m, respectively.

Based on chemical analyses, a small decrease of the electrical conductivity (EC) has been observed at the end of the recharge tests (Fig. 4). Prior to recharge the EC values had been about 735 µS/cm. After injection the EC values
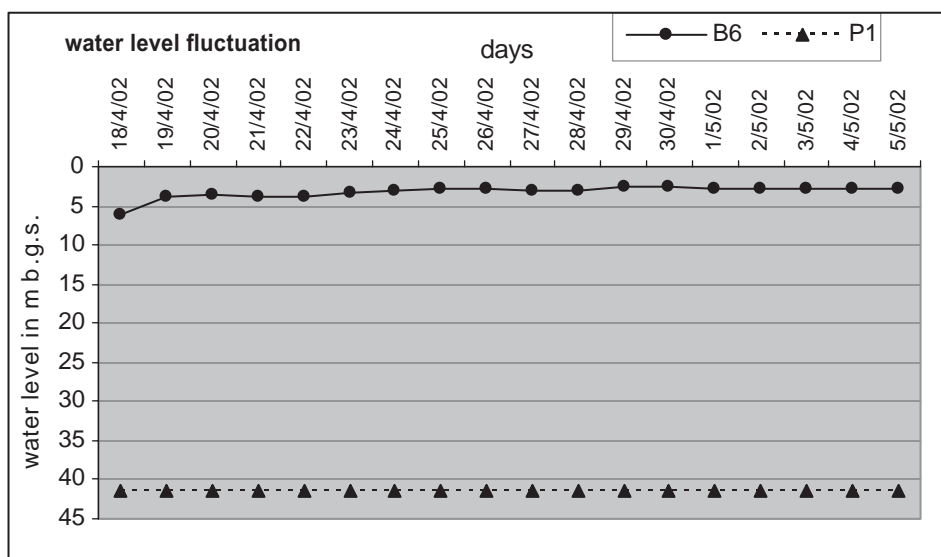
Figure 3. Impact of recharge on groundwater level in borehole B6 and piezometer P1 (B6P1=400m)
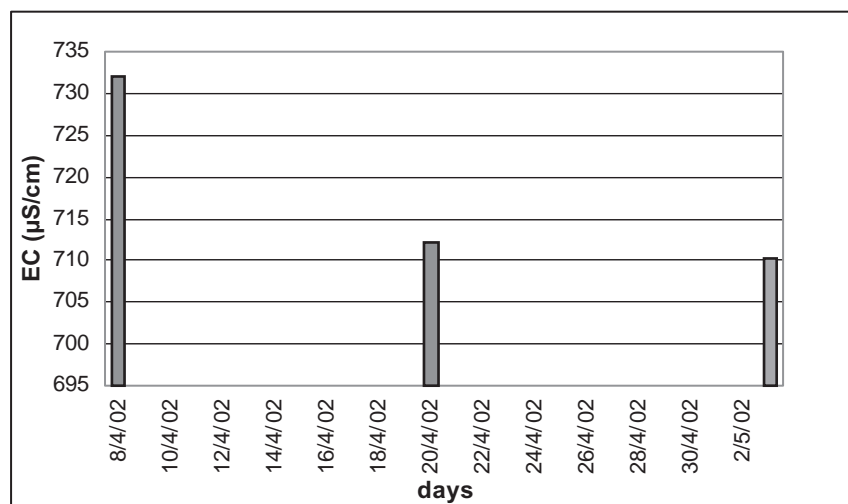
Figure 4. Electrical conductivity fluctuation (µS/cm) of groundwater in piezometer P2, during the recharge test in borehole B3. Location P2 as shown in Fig. 2

were less than 715 µS/cm. In the early stages of the recharge test, EC increases as a result of the unsaturated zone washout induced by the water level rise (Hionidi et al., 2001). An additional benefit of recharging freshwater to the aquifer is that, it improves the groundwater quality.

The great difficulty of using recharge boreholes is their rapid clogging (Peters, 1985). Because of the limited recharge water volume and duration of the recharge tests in Achaia, the effect of clogging was insignificant. On a regular base, the recharge water will be pretreated to improve its physical-chemical parameters.

CONCLUSIONS

Six recharge tests through deep boreholes were carried out in Achaia area. These field tests showed that, recharge rates were about 10–70 m^3/h. The total volume of recharge water in six boreholes was 137,000 m^3. The maximum

rise of the groundwater level at the end of the tests was 45 m and the minimum was 4.16 m. Water levels rose up to 500 m from injection boreholes in clastic deposits. In recharge borehole, which has been drilled in carbonate rocks the groundwater level rise was 3.26 m at the end of the 18-days recharge test.

Field tests have shown that, groundwater recharge via deep boreholes can be an effective means of augmentation of water budget, in the future. The tests also have improved the groundwater quality, especially the electrical conductivity. The technique is fairly effective and feasible; the environmental impact can be considered less severe and may be applied as part of a sustainable water resources management plan, based on surface and groundwater exploitation, simultaneously.

Experiments will be continued in the study area in order to determine the best locations with high permeability to make the aquifer recharge. Results would benefit by computer modeling to simulate the water cycle, the groundwater flow and the groundwater levels before and after the aquifer recharge.

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Aquifer re-injection as a low impact groundwater investigation tool – A case study from the Pilbara region, Western Australia

J. Youngs and D.M. Brown

Abstract

A case study presenting an innovative use of aquifer re-injection as a low impact groundwater investigation tool is described. The Marandoo Iron Ore mine is bounded by National Park in an area of ecological and cultural sensitivity within the Pilbara Region of Western Australia. A program involving prolonged aquifer pumping with re-injection was adopted as a low-impact method of temporarily stressing a sensitive groundwater system and providing definitive, quantitative, information on the system.

A trial involving prolonged pumping from bores on a mining lease with re-injection of discharge into an adjacent dolomitic aquifer underlying the surrounding National Park has been completed. This approach i) facilitated large-scale 'reverse' hydraulic testing in the National Park without the risk of drawdown impacts; ii) conserved the fresh groundwater resource and prevented unseasonal discharge of water to ephemeral drainage; and iii) allowed the technical feasibility of aquifer re-injection as a water management option to be assessed.

The combined pumping and re-injection approach has proved to be an effective method of obtaining good hydrogeological data on a major groundwater system with minimal environmental impact. It is considered that the technique could have wider application.

Keywords

Re-injection, test-pumping, ecological impacts, water management.

INTRODUCTION

Background

Iron Ore has been mined above the water table (AWT) at Pilbara Iron's Marandoo Mine since 1994. The mine lies in the Pilbara region of Western Australia, on a mine lease excised from the Karijini National Park (KNP). The mine lease encompasses a significant below water table (BWT) ore reserve. The constraints and implications of dewatering the BWT ore reserve are major factors in determining whether the BWT ore body will be mined. To this end, the scale of dewatering and potential environmental impacts have been the subject of hydrogeological investigation by the authors since 2001.

There are a number of uncertainties relating to the conceptual hydrogeological understanding of the Marandoo groundwater system. *The most important of these relate to the degree of connectivity between the major aquifers of the area.* This uncertainty has implications for: a) the estimation of pumping volumes and rates required to dewater the ore body in advance of mining; and b) the prediction of potential impacts of prolonged dewatering on the surrounding environment.

Environmental concerns are focused on the potential for lowering water levels in a regional shallow Calcrete groundwater system. The major concern is the potential for detrimental impacts on an ecologically significant stand of Coolibah trees which occurs 2 km north of the boundary of the mining lease (refer to Figure 1).
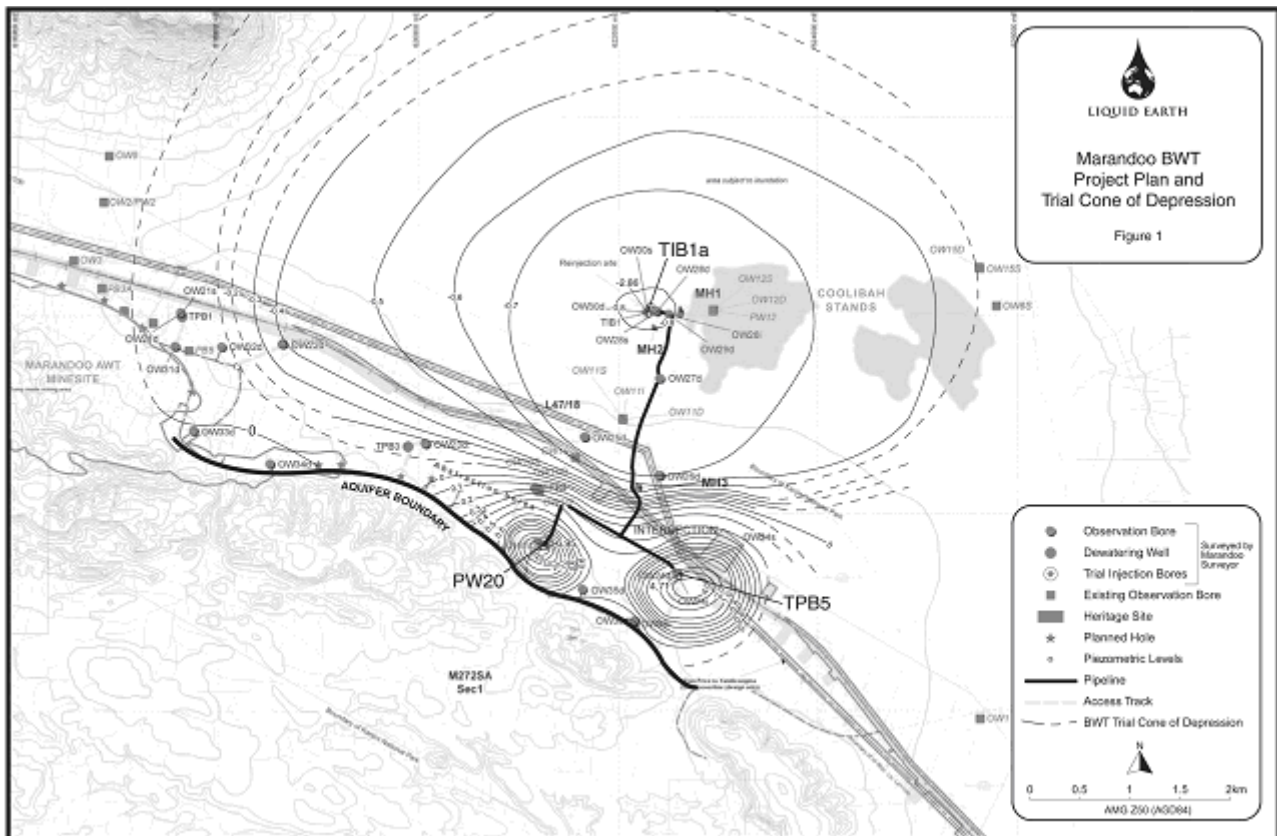
Figure 1. Marandoo BWT project plan and trial cone of depression

Rationale for the BWT pumping and re-injection trial

An extensive program of 'conventional' hydraulic testing was conducted at Marandoo between 2002 and 2003. Given the significant scale of the aquifers under investigation it was recognised that short term pumping tests would not provide sufficient certainty for a definitive assessment of dewatering volumes and impacts. To improve the certainty it was identified to be necessary to 'stress' the groundwater system over an extended period and use the resulting monitoring data to calibrate a predictive numerical model of the Marandoo groundwater system. Conventional long-term test-pumping with discharge to the surface environment could not be undertaken due to strict regulatory restrictions on unseasonal discharge to the National Park and the potential for prolonged discharge to impact on the findings of the trial through recharge to the critical shallow groundwater system.

The Marandoo pumping and re-injection trial was designed to overcome the constraints of traditional testing methodologies by providing a low-impact method of temporarily stressing a sensitive groundwater system. Two high-yielding bores in the Marra Mamba aquifer on the mine lease were linked and pumped simultaneously. The combined discharge from these production bores was piped approximately 2km north where it was reinjected into the Wittenoom Dolomite aquifer beneath the Coolibah stand. Water levels were observed in an extensive multi-aquifer monitoring network. Figure 1 presents the layout of the Marandoo pumping and re-injection trial infra-structure.

The advantages of the adopted approach were that it:

i) facilitated large-scale 'reverse' hydraulic testing in the National Park without the risk of drawdown impact on the unconfined groundwater system;

ii) conserved the fresh groundwater resource and prevented unseasonal discharge of water to ephemeral drainage; and

iii) allowed the technical feasibility of aquifer re-injection as a water management option to be assessed.

Conceptual hydrogeology

The Marandoo operation sits on the southern margin of a major, flat lying, internally draining basin known as the Mt Bruce Flats, which forms the upper reaches of the Southern Fortescue River catchment. The Marra Mamba iron ore outcrops as a curvi-linear feature known as the Marandoo Ridge which bounds the basin to the south. The region is semi-arid with rainfall dominated by seasonal cyclonic events. There are no significant permanent water bodies or well-developed creek systems in the basin.

The stratigraphic sequence at Marandoo is summarised in the schematic section presented in Figure 2.
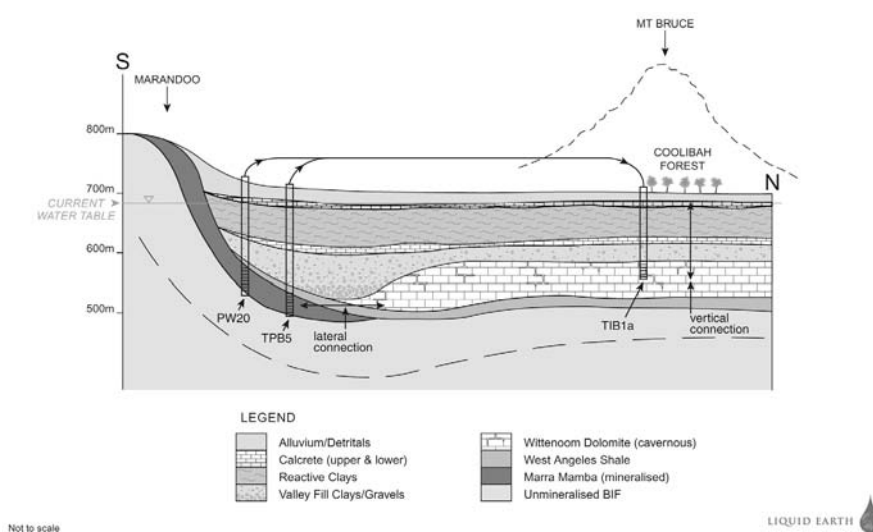
Figure 2. Marandoo BWT trial schematic

The primary groundwater systems comprise a shallow system, hosted by a calcrete horizon within a Cainozoic valley fill sequence, and a deeper Proterozoic bedrock groundwater system hosted by mineralised Marra Mamba and karstic Wittenoom Dolomite Formations. The shallow and deep groundwater systems are separated by a thick sequence of valley fill clays. The deep bedrock aquifer is bisected by a layer of weathered West Angelas Shale which is believed to act, at least locally, as an aquitard between the dolomite and Marra Mamba aquifers.

The major uncertainties related to the hydrogeological understanding of the Marandoo area have been the degree of connection between the shallow calcrete and deep bedrock aquifers (a function of the clay layer properties), and the degree of connection within the deeper groundwater system between the mineralised Marra Mamba aquifer and karstic Wittenoom Dolomite aquifers (a function of the shale layer properties).

METHODOLOGY

A series of production and monitoring bores, including the bores used in the pumping/re-injection trial, were constructed on the lease and in the adjacent National Park during 2002/2003 (refer Figure 1). Bore construction methods were adapted to combat the challenging swelling and fretting behaviour of the valley fill clay beds. Originally it was intended to construct the re-injection bores using sections of round-wire screens to maximise

efficiency, however on drilling the dolomite was found to hold open and develop well so the deeper sections of the injection bores were left open-hole. Figure 3 presents a schematic log of re-injection bore TIB1a, showing the geology, bore construction and re-injection infrastructure.

Submersible pumps at the production bores were powered by two on-site generators per bore to ensure uninterrupted pumping. Water was directed along a thermo-welded polypipe pipeline equipped with periodic air relief valves and scour valves. The infrastructure at the reijection end of the system was refined during commissioning, to determine the simplest arrangement capable of achieving good results.

The initial arrangement incorporated a simple re-injection tube to direct water from the pipeline to below the water table. The system was sealed apart from the two-way air release valve at the headworks. Under this arrangement there was considerable air entrainment and the water level in the bore rapidly (within five minutes) approached surface. The loss of full pipe flow due to flow acceleration over the pipe bend also prevented accurate operation of the mechanical flow meter upstream of the well-head. Closure of the air valve on the wellhead reduced the air entrainment, but did not address the problems of cavitation at the surface or loss of flow meter operation, and introduced the possibility of collapse of the injection tube due to the resulting low pressure.

Figure 32. Reinjection site schematic

Inverted cones were constructed from worked polypipe and retrofitted in re-injection bores TIB1 and TIB1a to combat the problem of negative pressure at the surface. The cones were suspended on cables from tripods above each bore, and seated at the bottom of the injection tubes. The cone impedes the flow such that water in the injection tube backs up to surface under design flow rates, producing positive pressure at the surface. Calibration of the cones to the size of the injection tubes proved to be difficult, and ultimately standby injection bore TIB1a was found to perform much better than the intended injection bore TIB1.

The pumping and re-injection trial was run continuously for 44 days during October/November 2004. Pumping rates from both bores were raised slightly during the trial. For the final two weeks of the trial TPB5 was pumped at 2.0 ML/d, PW20 delivered 1.7 ML/d, and the combined re-injection rate was 3.7 ML/d.

RESULTS AND DISCUSSION

Equipment performance

Possible clogging by air entrainment was recognised as the major risk to the performance of the trial equipment. This was reinforced during initial testing of the equipment when water levels rose dramatically while flow was allowed to cascade into the bore.

Plugging by pumped sediments, chemical precipitation and biological growth were also potentially important. The pipeline was pumped to scour for several hours after construction to flush sediments out before downhole re-injection began. Chemical typing of the source and host waters showed the waters were chemically almost identical, resulting in a low likelihood of precipitation or dissolution. The risk of significant biological growth over the time-frame of the trial was considered to be low.
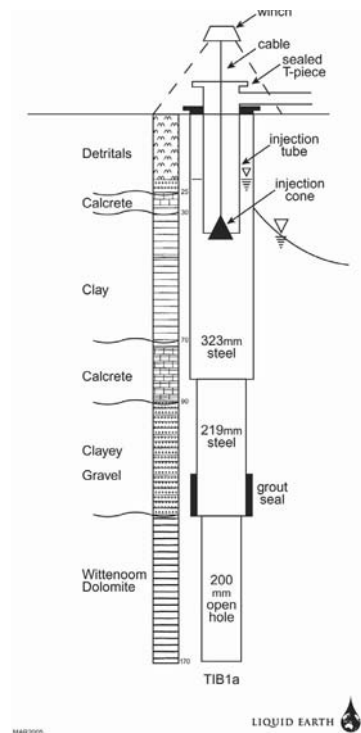
Figure 4 shows the water level responses in the re-injection bore, two nearby dolomite monitor bores and one shallow calcrete monitor bore during the trial. When the levels are compared to the type curves developed by Pyne (1995) for air entrainment and suspended solids, there is no evidence of either clogging mechanism. However the rate of drawup rate in the injection bore was found to increase after two days of operation, with no corresponding increase in water level in the observation bores to suggest an aquifer boundary influence. This reaction may indicate the possibility of a low degree of clogging by air, bacterial growth or chemical precipitation during the trial.
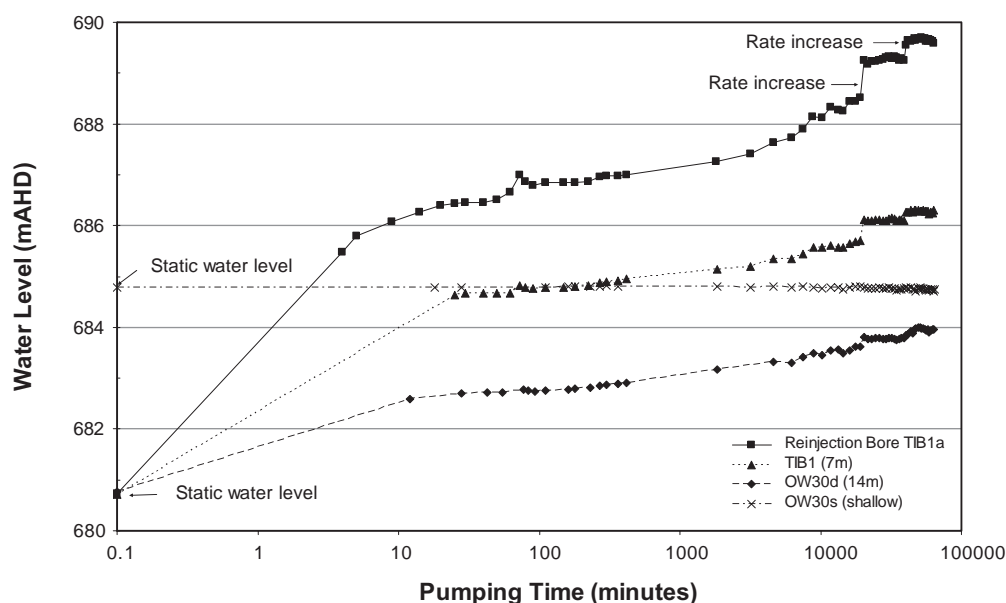
Figure 4. Marandoo BWT trial reinjction site water levels

Injection bore TIB1 bore has essentially the same construction as TIB1a, and the two were designed to operate in tandem if required. TIB1 was test-pumped at a rate of 3.0 ML/d in May 2003. The drawdown after 1 hour was 7.2 m. By comparison, the 'drawup' in TIB1a after 1 hour of re-injection at a rate of 3.7 ML/d was 5.9 m. This comparison suggests that in this case the re-injection operation may be slightly more hydraulically efficient than pumping. The open-hole construction of the re-injection bores undoubtedly influenced the efficiency of the injection process.

The effectiveness of the injection tube and downhole cone arrangement to manage air entrainment was demonstrated by the trial. This low-cost arrangement is to be refined during an upcoming multi bore re injection project also at Marandoo where the re-injection tubes will be sized to the maximum design flow rate, and cones will be used to manage lower flow rates.

The trial required a full-time (10 hours per day) presence on-site throughout the duration of the 44 days of operation. A technician was retained to refuel and maintain generators, maintain pumping rates, monitor water levels and inspect the pipeline pressures and general condition. The only notable infrastructure problems were three generator/electrical failures, which caused a combined individual pump downtime of 23 hours. In general the infrastructure operated successfully and enabled collection of a good dataset.

Assessment of aquifer interconnection

The extent of lateral connection across the deep groundwater system can be seen in the cones of depression/ impression overlayed on Figure 1. The cones display net 'drawup' in some Marra Mamba monitor bores on the mine

lease. This demonstrates clear qualitative evidence of significant connection between the Wittenoom Dolomite and Marra Mamba aquifers. Contour spacing is much higher in the dolomite than in the Marra Mamba, demonstrating the higher regional transmissivity and storage in the karstic aquifer. These findings confirmed the conceptual understanding of bedrock aquifer characteristics which could not be definitively evaluated through conventional short term pumping tests.

In terms of quantifying the dewatering requirements of Marandoo, the most important outcome of the trial is the dataset it provides for calibration of the groundwater model and for determining the influence of the West Angelas Shale aquitard. The shale layer is the major control on the contribution of the dolomite aquifer to Marandoo dewatering. Earlier calibration of the model was relatively insensitive to this layer.

Water levels, pumping rates and reinjection rates during the 44 day Trial formed a dataset which was used to calibrate the shale layer properties. The calibration returned a horizontal hydraulic conductivity of approximately 0.75 m/day and a storage value of 0.0001 for the shale. These results represent a significant tightening of the shale properties by comparison to earlier work. The calibration process with the Trial dataset was relatively sensitive to the shale properties, and also highlighted the need for further geological exploration work to improve delineation of the shale in some areas. The confidence in Marra Mamba dewatering estimates has been significantly improved by provision of the Trial dataset to the modelling.

Vertical connection across the clay layer was investigated by review of the shallow bore hydrographs, particularly at the pumping and re-injection sites. The influence of the trial was not seen in any of the shallow aquifer hydrographs across the project area.

The vertical gradient at pumping bore TPB5 was increased markedly by the abstraction (from 0.053 m/m to 0.15 m/m). There was no break in the stable water level trend in shallow monitor bore OW24s, which is 5 m from the pumping bore. Re-injection at TIB1a reduced the prevailing local vertical hydraulic gradient from strongly downward to weakly downward (from 0.10 m/m to 0.019 m/m). The shallow monitor bore OW30s lies 5 m from the re-injection bore and displayed a long-term downward trend which was uninterrupted by the trial.

It can now be concluded with confidence that the degree of widespread connection between the deep and shallow aquifers is extremely low.

The 'reverse-pumping test'

Aquifer parameters at TIB1a were derived by simply treating 'draw-up' as 'drawdown', and applying standard analysis techniques to water level responses. The results were similar to results derived from earlier test-pumping of the adjacent TIB1 bore:

2003 test-pumping: transmissivity = 3000 m^3/d/m, storativity = 2.8 x 10^{-3}
2004 re-injection: transmissivity = 2200 m^3/d/m, storativity = 1.2 x 10^{-4}

This technique must be applied carefully to ensure that re-injection results can be extrapolated to pumping performance. It could not, for instance, be applied accurately to an unconfined aquifer, where the saturated thickness will increase under re-injection as opposed to a decrease under pumping. This situation could also introduce inaccuracies arising from vertical inhomogeneity.

CONCLUSIONS

The completion of a prolonged large-scale pumping/re-injection trial at Marandoo has provided definitive infor-mation on the characteristics of a complex multi aquifer groundwater system. The trial has provided valuable information relating to aquifer interconnection in an environment where traditional short-term tests were incon-clusive and long term test pumping has been severely limited by discharge constraints.

The data from the trial has been used to improve the calibration of a numerical model and subsequently predict dewatering estimates and potential impacts. As a result of the Marandoo trial, aquifer re-injection is now being con-sidered as a tool for investigation and long term excess water management at a number of sites across the Pilbara region of WA.

ACKNOWLEDGEMENTS

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Sustainability of managing recharge systems

MAR strategies

Arid zone water management

Water re-use for agriculture

Sustainability of managing recharge systems: The case of the Chad Basin transboundary aquifers

Oladepo Adenle

Abstract

The Chad Basin transboundary aquifers are located within the geographical basin of the Lake Chad, which is between latitudes 6 and 24 degrees north and longitudes 10 and 23 east. This is a closed lake basin containing a freshwater lake. Greater than two-thirds of this basin is located in an arid zone, the Sahelian climatic belt of West/Central Africa.

Of all the principal aquifers of the Chad Basin, the only ones exploited are the Quaternary, the Early Pliocene and occasionally the Continental Terminal because of the latter's depth and water quality. This paper does not delve into the controversies concerning the hydrogeology of the basin but uses the Barber and Jones (1960) designations – Upper, Middle, and Lower Zones of the Chad Formation as equivalents for the Phreactic, Middle and Lower zones of Hanidu et al. (1989) respectively.

The paper discusses factors that affect how the recharge systems in this basin can be managed sustainably by reviewing the works of several authors with respect to surface water-groundwater interactions along the two main drainage basins that contribute flows into the Lake Chad – the Komadugu-Yobe and the Chari-Logone as well as around and under the lake. The identified factors that affect sustainable management of the recharge into the Chad Basin aquifers are institutional, management approach and practices as well as climate, i.e. climatic seasons and climate change.

Keywords

Chad Basin; transboundary aquifers; recharge.

BACKGROUND

The Chad Basin is located in the eastern part of the Sahel region of Africa at the southern edge of the Sahara desert. The hydrological basin of the Lake Chad lies between latitudes 6 and 24 degrees north and the longitudes 10 and 23 degrees east. Lake Chad is located in a closed basin covering approximately 2.5 million km^2 (Figure 1). In contrast to the closed basin lakes of, for example, the southwest U.S.A. the Lake Chad water is fresh (84−801 mg/l TDS) Isiorho et al., 1996. The lake is shallow generally less than 7 meter deep.

The Lake Chad Basin is characterized by extensive floodplains. It is believed that the basin has one of the largest wetlands in the Sahelian region of Africa with over 10 million hectares in Chad alone. In terms of their ecological importance three floodplains in the Chad Basin are worthy of mention – Chari, Logone and the Yobe rivers.

On a regional basis the Lake Chad Basin Commission manages the water resources of the Lake Chad Basin (LCBC). In 1964, the four countries (Nigeria, Niger, Cameroon and Chad) bordering the lake created the LCBC to handle the water resources management in an area referred to as the 'conventional basin' which covered approximately 427,300 Km^2. In 1994 membership of the Commission increased to five when the Central African Republic (CAR) joined the commission and the conventional basin area increased to 984,455 Km^2. When Sudan membership is

fully actualised LCBC member states will number six and the area of the conventional basin will increase to 1,053,455 Km^2.

Figure 1

Source: Lake Chad Basin Commission
FEWS, May 1997
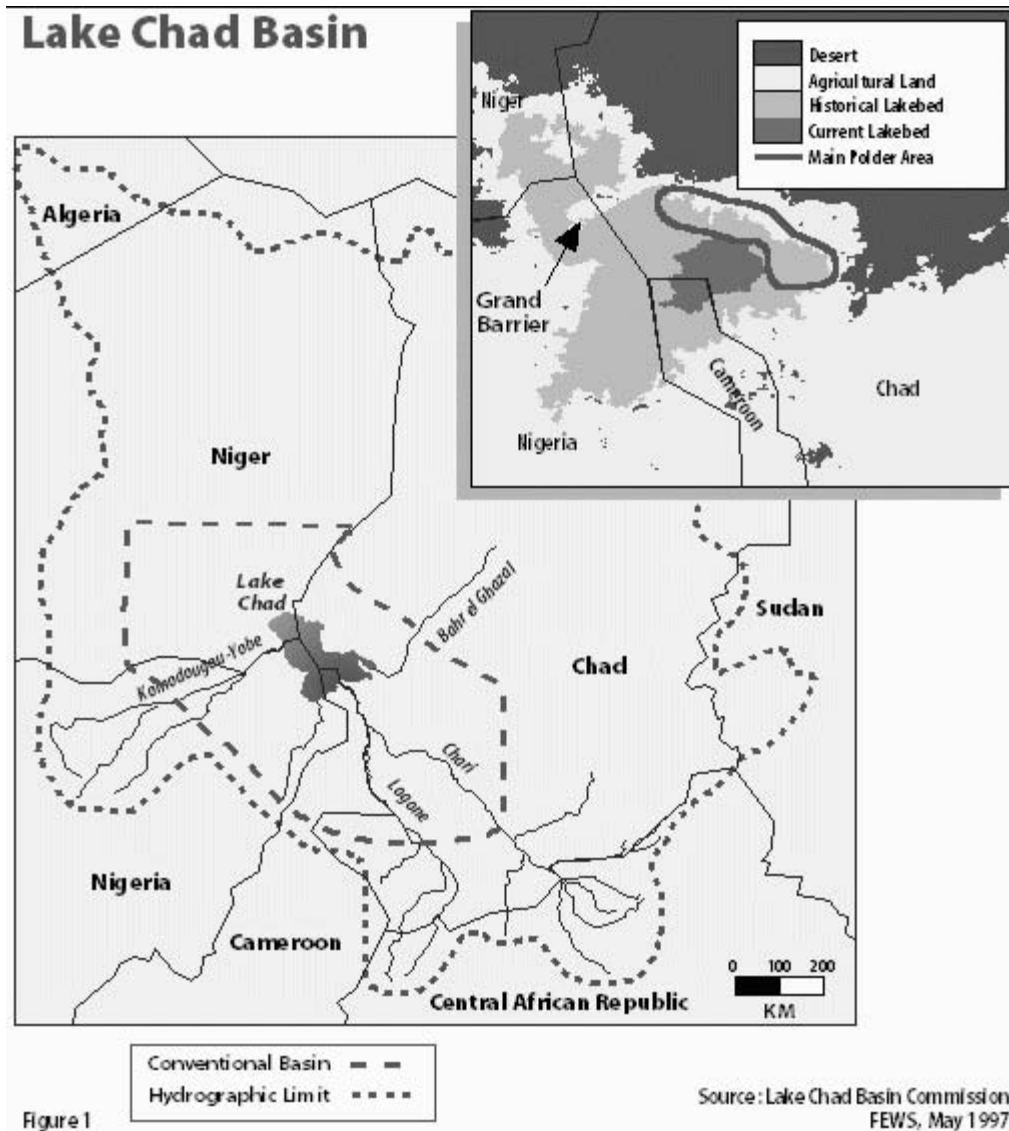
PRINCIPAL AQUIFERS

The main hydrologic units identified in the Chad Basin are Lacustrine actual Quaternary, Dunary Quaternary, Quaternary, Middle-Upper Pliocene, Early Pliocene, Continental Terminal, Continental Hamadian, Marine Cretaceous, Continental Intercalaire, III and IV Volcanism and Crystalline basement. Of these the only ones actually exploited are the Quaternary, the Early Pliocene and occasionally the Continental Terminal because of its depth and water quality. All of the aquifers below the Quaternary are confined and artesian in the center of the basin.

The Quaternary Aquifer (Upper aquifer of the Chad Formations)

This Upper aquifer is phreatic and is made up of fine-grained sediments approximately 30 m thick, and is hydraulically connected to Lake Chad (Isiorho et al., 1996). The phreatic aquifer is not continuous all over the basin area, and recharge conditions are poor. Natural recharge occurs primarily by influent seepage from seasonal streams, perennial rivers and the Lake Chad bed. Recharge also occurs in the floodplains.

The Confined Aquifer of the Early Pliocene (Middle Aquifer of the Chad Formations).

The upper aquifer is separated from the underlying middle aquifer by the lower Pliocene aquifer found at depths of between 150 and 400 m, and is approximately 200 m of clay-rich sediment (Kindler et al. 1990). In some parts of the Basin, this aquifer is artesian. The sands of the Middle aquifer may be superposed directly on the Miocene (CT), particularly in the Kanem region, to form a single aquifer with a thickness that may exceed 275 m.

The middle Aquifer is a Continental Terminal aquifer that essentially comprises an alternation of sandstone and clay encountered between 450 and 620 m from the surface, and extends from Niger and Nigeria into Cameroon and Chad (Kindler et al. 1990). Just like the Quaternary aquifer the piezometry of this aquifer has a gradient that is towards the plains of the low land in the northeast where it becomes sub-outcropping and continuous with the Quaternary aquifer. In the central part of the basin, under the lake, it is artesian with pressures exceeding up to 20 m.

The Confined Aquifer of the Continental Terminal (CT) (Lower Aquifer of the Chad Formations; Oligocene-Miocene)

This system is for the most part sandy and around 100m thick but it can be thicker than 600m in the rifts like the one in Doba.

The two other aquifers are Confined Aquifer of the Continental Hamadian (Middle Cretaceous-Maastrichian)and the Confined Aquifer of the Continental Intercalaire (Dinanian-Early Cretaceous).

RECHARGE CHARACTERISTICS

Goes and Offie (1999) listed three possible mechanisms of groundwater recharge in the Hadejia-Jama'are-Yobe River Basin as (1) direct infiltration of rainwater into the ground; (2) Seepage through stream/lake beds (influent streams/lake) and (3) Infiltration through inundated floodplains or wetlands.

Apparently recharge of infiltrating rainwater in this basin, which is virtually the Nigerian part of the Chad Basin, affects only the upper part of the phreatic aquifer. According to Madubuchi et al. (2003) stable isotope compositions of the three aquifers (the Quaternary (Upper aqufer), the Middle and lower aquifers) appear to be of common origin, except the upper part of the Quaternary tapped by shallow wells which is clearly of recent origin. Furthermore, Maduabuchi et al. concluded that stable isotope suggests that the waters of the three aquifers are paleo-waters and that the high tritium values are recorded in the shallow boreholes in the recharge areas in parts of Jigawa and Bauchi states, about 400 km SW of Maiduguri.

- *Direct infiltration of rain water*
  Goess and Offie noted that ground water recharge in arid and semi arid regions of the world is about 1–30% of the local rainfall in such regions. They found that in a sand dune area 60 km northeast of Nguru (Kaska) there was a mean annual water table variation of 0.13 m in 11 piezometers during a one-year period. They reported Carter et al.'s estimate of recharge for this sand dune area as being 49 mm which is 17% of the local annual rainfall. They concluded that local rainfall is the most likely recharge source in this area and that the relatively high recharge rate is due to a high infiltration rate in the sand dune area which has low vegetation density, and that rain-fed recharge in the floodplain areas and near major river channels will be much less due to the dense vegetation cover.

- *Infiltration through riverbeds*
  Goess and Offie noted that in the Hadejia-Jama'are-Yobe River Basin, outside the Basement Complex area, the shallow groundwater tables dip away from rivers. They reported that the estimate of recharge along the Yobe River between Gashua and Lake Chad (286 km) in 1984 is $17 \times 10^3 \times 10^3$ m^3, and that this estimate is based on a survey that was carried out in 1984 which was in the middle of a low river flow period (1982–1985). They

further noted that the annual discharge in 1984 of the Yobe River at Gashua was the lowest ever (1963–1998) and that therefore flooding and floodplain recharge was probably negligible during 1984. The recharge influencing the gradient that year is therefore assumed to be due to riverbed recharge only. Recharge of the of the phreatic aquifer through seepge from the lakebed was also confirmed by Isiorho et al. (1996).

- *Infiltration through inundated floodplains*
  The presence or absence of a uniform clay cover in the floodplains determines whether the annual floods recharge the groundwater and if the shallow aquifer is confined or unconfined. Though Alkali (1995) concluded that actual recharge is limited to the riverbed, due to the clayey floodplain and mainly confined condition of the shallow aquifer, Thompson's (1995) study is a strong indication that the shallow aquifers along the Hadejia River System in the Hadejia-Nguru Wetlands (HNW) are predominantly unconfined.

  Hanidu et al. quoting Scet International noted that the Kerri-Kerri recharges the Lower Pliocene Aquifer and the Continental Terminal and work has to be done to prove or disprove this assertion.

  Hanidu et al. further noted that the Continental Terminal aquifer outcrop has not been found in Nigeria and so it is unlikely that it is being recharged from the surface anywhere in Nigeria. However, extensive outcrops occur in Southern Cameroon, southern Chad (Zone de Koros), northern Chad and east Niger and it is likely that recharge occurs there.

FACTORS THAT AFFECT THE RECHARGE SYSTEMS IN THE BASIN

Extent of the wetlands/recharge areas

Recharge areas along stream courses – shortened stream courses, for example the Yobe River which used to discharge into the Lake Chad and which forms the international boundary between Nigeria and Niger does not reach the lake any more. This means reduction in the area at which recharge could occur, lakeshores and lakebeds. If one takes into consideration the reduction in the extent of Lake Chad (based on NASA satellite imagery the Lake Chad has been reduced by 90% in surface area between the 1960s and 1990s) then the reduction in the magnitude of recharge could be imagined.

The hydrologic characteristics of recharge areas in aquifer outcrops

Recharge in aquifer outcrops is a function of both the extent of the outcrop and nature of the outcrop at any point in time with respect to presence of vegetation cover and the occurrence of swelling clays that enhance surface runoff at the expense of recharge (USGS, 2001).

Recharge areas in wetlands

It has been observed that the Hadejia-Nguru wetlands have declined by 210,000–230,000 ha. and that the most important environmental function of the Hadejia-Nguru wetlands is its role in recharging the groundwater aquifer of the Chad Formation. Barbier et al. (1997) claimed that Hadejia-Nguru Floodpalins have shrunk to an estimated 70,000 to 90,000 ha. Evidence presented by Hollis et al. (1993) shows that a reduction in floodplain inundation leads to a lower rate of groundwater recharge. Since 1983, when the extent of flooding dropped appreciably, groundwater recharge fell by an estimated aggregate amount of 5,000 km^3. Flooding could reach approximately 2,000 km^2 in the Hadejia-Nguru plain prior to construction of dams in Kano area. Now it covers no more than 1,000 km^2. The SEMRY project resulted in 30% decrease in the flooded area of the Waza-Logone Floodplains (LCBC, 2002). Water diversion for the project combined with drought has eliminated the flooding of some 59,000 ha of floodplain and seriously reduced another 150,00 ha. The Yaeres has dried up by 30% of its former extent.

Excessive evapotranspiration

Excessive evaporation from large open water surfaces created as a result of construction of large dams (19 of them around Kano) in the upper reaches of the Komadugu-Yobe drainage system reduces the volume of potentially available water for recharge along stream courses, in seasonally inundated plains.

FACTORS THAT AFFECT SUSTAINABLE MANAGEMENT OF RECHARGE IN THE BASIN

Institutional factors

The institutional arrangement in the basin is such that there are no clear linkages between the various institutions. The LCBC agreement's statutes are not mandatory with respect to proper management of groundwater resources. This has resulted in upstream water agencies in the Basin, for example, in Nigeria not consulting with downstream agencies in matters of mutual interest. Furthermore, there is more than one River Basin Development Authority within the same catchment of the Komadugu-Yobe drainage system as well as five state water boards in Nigeria. Each of these stakeholders acts independently and this has resulted in conflict. The development plans of these institutions are not harmonized. Even if Nigeria alone is considered, the development plans of the two RBDAs (HJRBDA and CBDBA) are not harmonized let alone those of the five State Ministries of Water Resources and their parastatals the state water boards which hardly have development plans because their programmes are dictated by political exigencies. Sustainable management of the recharge in the basin depends on harmonizing the development plans of all actors within the Basin.

Management approach

Groundwater resources management in the Basin is uncoordinated. Dams are constructed without their impacts on stream flows and duration and extent of inundation considered. Gaps exist in data collection and monitoring of the resource is not comprehensive and at times depends on network of open water wells which are not good enough compared to infiltrometers to distinguish between water level changes due to precipitation and other causes such as water abstraction trends (Thompson). Sustainable management of groundwater resources, recharge system included, relies on maintaining reliable groundwater database and efficient monitoring. It is not possible to manage a resource about which very little information is available apart from approximations.

Land-use practice plays a role in the sustainability of recharge. There is uncontrolled grazing and poor management of the watershed which leads to desertification and increase in overland flow as well as evapotranspiration. Furthermore, the way surface water resources is managed affects recharge sustainability. Upstream construction of dams without due consultation with the downstream users and without consideration for the downstream uses has adversely affected volume of recharge – e.g. the 19 dams constructed in the upper reaches of the Komadugu-Yobe drainage basin.

Climate

Climate plays a significant role in whether the recharge in the Chad Basin can be managed sustainably. The persistent drought of the past two decades have resulted in reduction of the Lake Chad area to 10% of its original size, that is 2,500 square kilometres as well as in the extent of flooding and duration of flooding. It also affects the volume of water available for infiltration. Desertification caused by persistent drought results in little or no vegetal cover and high surface runoff as well as reduced recharge. The climatic modification of rainfall conditions observed

since the early 1970s, and the resulting shift in the isohyets 180 km further south, combined with human activities have helped to deplete the plant cover. It has also affected how much water is available for recharge in areas affected by the isohyetal shift.

CONCLUSION

This paper discussed the recharge characteristics of the aquifers in the Chad Basin with special reference to the Nigerian part of the Basin, the factors affecting recharge and those that influence the sustainable management of the recharge system. The lack of coordination of water resources management activities within the Basin, which in turn is linked to poor institutional arrangement, is one of the major factors that could jeopardize sustainable management of the recharge system in the Chad Basin because of the impact the surface water management of the upstream users have on the extent of flooding and the wetlands area. However, a key factor that could militate against managing the recharge in this Basin sustainably is the poor state of groundwater resources database as well as the practice of having to rely on approximations.

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Application of GIS to aquifer retention time, well recharge capacity, and river depletion calculations to determine optimum locations for artificial recharge

Malcolm Anderson, Michael Jones, Keith Baxter and Derek Gamble

Abstract

In aquifers where flows to and from surface water are significant, river depletion during groundwater abstraction and loss of aquifer storage to rivers during artificial recharge are important environmental and engineering considerations. These issues can affect the financial feasibility of artificial recharge operations. In complex aquifer systems, optimum artificial recharge locations are not always located at the greatest distance from surface waters. For instance in south London other factors that significantly affect the optimum recharge and abstraction locations include the dominant fracture orientation and style, local variance in aquifer transmissivity and storage, and the vertical hydraulic conductivity of leaky layers separating the target aquifer and surface water features. The complex interplay of these factors can be counter-intuitive and prevent identification of optimum recharge and abstraction locations by simple inspection methods. This can be addressed using GIS to carry out spatial analysis of key parameters, for example distance to a river or spring spill point, transmissivity, storage and vertical hydraulic conductivity of intervening layers, and then calculating spatial variations in diagnostic parameters of aquifer retention time, river flow depletion, borehole injection capacity and abstraction capacity. The most favourable artificial recharge locations can then be identified where the diagnostic parameters are optimised.

Keywords

Abstraction and injection capacity; aquifer retention time; artificial recharge; river flow depletion; chalk; London Basin.

INTRODUCTION

Following the success of the North London Artificial Recharge Scheme (NLARS), Thames Water is investigating the potential for a South London Artificial Recharge Scheme (SLARS) in the southern area of the London Basin (Figure 1). In contrast to the NLARS area, the southern basin is a significantly more challenging environment with complex hydrogeology, localised hydraulic connections from the Chalk aquifer to surface water courses, and substantial variation in storage capacity, transmissivity, recharge and abstraction capacity (Jones et al., in press). Furthermore, selection of artificial recharge test sites was limited to locations where sufficient surplus water main capacity was available, whilst the built environment of London restricted test sites to existing water works or parkland. Thus the test locations had to be selected on the basis of expediency rather than locations of optimum artificial recharge potential. From the outset it was understood that site selection would be significantly more successful if the artificial recharge potential of each test location could be assessed in the context of the substantial and localised variance in artificial recharge potential across the south London basin as a whole. The geographical information system (GIS) calculation method presented in this paper was developed as a solution to resolve this issue.

METHOD

The mathematical function of a GIS is a very powerful and generally underused tool and can be used to calculate spatial distributions of more useful parameters that cannot be directly measured. As each vector (numerical value) map essentially defines the distribution of one term of an equation, spatial distributions of useful parameters can be calculated using an appropriate equation and separate maps defining the spatial distribution of each term in the equation. Here we show how this approach can produce useful spatial distributions of the three hydraulic and hydrological factors critical to identifying both feasible and optimum artificial recharge locations.
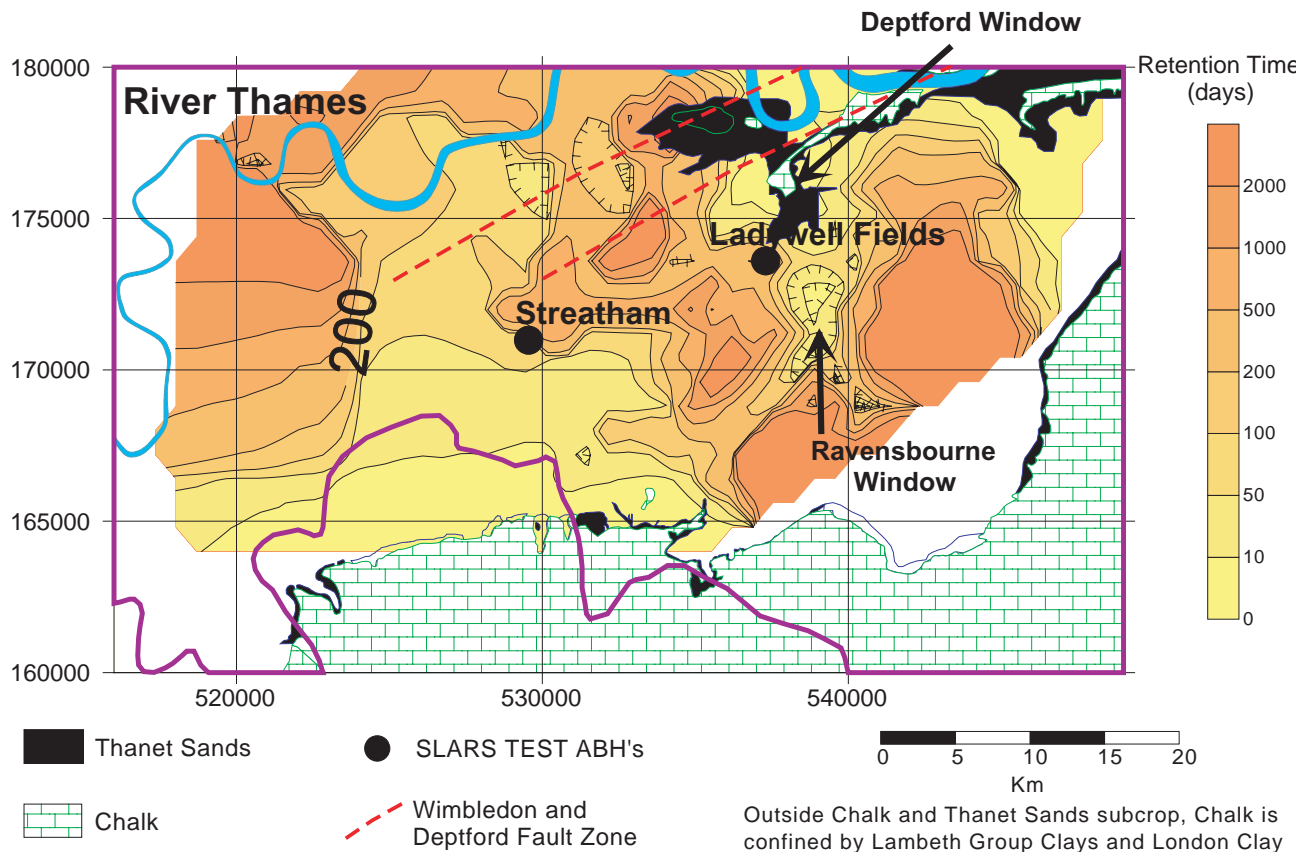
Figure 1. Map of retention time in the SLARS study area

Hydraulic constraints on artificial recharge potential are essentially limited to just three fundamental factors; 1) borehole recharge capacity; 2) borehole abstraction capacity; and 3) recharge retention time. Optimum artificial recharge conditions occur at locations where all three factors are maximised. Borehole recharge and abstraction capacity are important factors because the greater the capacity of each borehole, the greater the economies of scale that can be achieved. Target design recharge and abstraction rates of 5 to 10 Ml/day were selected on the basis of Thames Water's previous and current abstraction operations in the SLARS area.

Retention time is a new concept developed by MWH to define the significance to artificial recharge operations of aquifer-surface water connections, where stream flow depletion can occur during aquifer abstraction and loss of injected water can occur during artificial recharge. However, losses and gains from the aquifer to the connected surface watercourse do not occur instantaneously once recharge or abstraction begins. If there is a significant time delay before unacceptable losses occur then artificial recharge is feasible. Where the purpose of artificial recharge is to allow increased abstraction to support seasonal peak demand, retention times of six months or more are required.

Where artificial recharge is required to support drought water demand, then longer retention times are required. In the Thames Valley during the last 100 years the longest drought period was 16 months and therefore this was considered as the appropriate drought retention time requirement.

Retention time

Conservative retention time values can be calculated very simply by rearranging the partial derivative of the straight-line distance-drawdown approximation equation (Cooper and Jacob, 1946) in terms of time to give Equation 1:

$$t = r_o^2 \, S \, / \, 2.25T \qquad \qquad \text{Eqn. (1)}$$

where t, r_o, S and T are respectively retention time, radial distance between the surface water course and the abstraction or injection borehole, aquifer storage coefficient and aquifer transmissivity. Equation 1 defines the time at which drawdown in the aquifer beneath the surface watercourse first occurs and is suitable where no stream flow depletion or injected water losses are acceptable.

Typically Equation 1 is appropriate where the hydraulic conductivity of the stream bed, or other potential aquitards between the aquifer and the stream bed are relatively high. However, where limited stream flow depletion or loss of injected water is acceptable, and the hydraulic conductivity of the stream bed or other aquitards is sufficiently low, a different calculation method is appropriate. Although the equation for stream flow depletion through an aquitard layer (Hunt, 1999) has recently been developed, rearrangement of the equation in terms of time requires further mathematical development and was outside the scope of the current study. Furthermore hydraulic connection between the Chalk aquifer and the rivers of the SLARS area occurs in localised 'windows' where the effectively impermeable London Clay and or Lambeth Clay have been removed by erosion; a geometry for which the Hunt equation is inadequate. Instead, stream flow depletion can be accounted for in the retention time calculation. Firstly, the straight-line time-drawdown approximation equation, (Cooper and Jacob, 1946) must be rearranged in terms of time:

$$t = [\, r^2 \, S \, / \, T \,] \, e^{\, [\, 4 \pi T s / Q \,]} \qquad \qquad \text{Eqn. (2)}$$

where the terms are the same as in Equation 1, e is the exponent, s is the drawdown and Q is the pumping rate. Then Darcy's Equation (Darcy, 1856) may be rewritten in terms of s to describe the vertical groundwater flow from a river to the aquifer (or vice-versa) within a 'window':

$$s = q \, b' \, / \, (K' \, W \, L) \qquad \qquad \text{Eqn. (3)}$$

where q is the maximum acceptable flow loss from the stream, K' and b' are respectively the vertical hydraulic conductivity and thickness of the stream bed or other stream depletion limiting aquitard layer. L and W are respectively the length and width of the stream bed through which stream depletion is occurring. Finally, substitution of Equation 3 into Equation 2 gives Equation 4:

$$t = [\, r^2 \, S \, / \, T \,] \, * \, e^{\, [\, 4 \pi T q \, b' / (Q \, K' \, W \, L) \,]} \qquad \qquad \text{Eqn. (4)}$$

Borehole recharge and abstraction capacity

The equation for borehole recharge and abstraction capacity is the product of the borehole specific capacity and the allowable maximum (and minimum) head in the well. For the borehole recharge capacity calculation, ground level was selected as the appropriate maximum allowable head in the SLARS area. This level is essentially a compromise. Although higher heads are feasible by engineering boreholes for significant positive pressures during injection,

other factors such as landfills and engineering structures (multi-storey building basements and underground railway tunnels) may locally limit the groundwater level draw-up cone at locations adjacent to the pumping borehole (Jones *et al.*, in press). Borehole recharge capacity was calculated from Equation 5.

$$RC = (GWLMax - RGWL) * S_c \qquad \text{Eqn. (5)}$$

Where RC is the recharge capacity, S_c is the specific capacity of the borehole, RGWL is the rest groundwater level and GWLMax is the maximum allowable head (i.e. the ground surface level). Borehole abstraction capacity is also a difficult parameter to calculate because the minimum allowable head is limited by a number of factors that change across the study area. These include existing borehole pump depths (derogation issues); minimum groundwater levels acceptable to regulators that are yet to be defined; and local water quality issues associated with dewatering the Basal Sands (a leaky layer that includes the Thanet Sands at its base). However as significant borehole hydraulic tests are available from the SLARS area, abstraction capacity was determined from test results and values contoured in the GIS accordingly. Alternatively where borehole abstraction capacity data are not available, abstraction capacity may be calculated using Equation 6.

$$AC = (RGWL - GWLMin) * S_c \qquad \text{Eqn. (6)}$$

Where AC is abstraction capacity and RGWL and GWLMin are the rest groundwater level and minimum allowable groundwater level respectively. However in order to use Equations 5 and 6, values for specific capacity are required. In this study reciprocal specific capacity values were determined from Equation 7 below (Bierschenk, 1963).

$$1/S_c = s_w/Q = B + CQ \qquad \text{Eqn. (7)}$$

Where s_w is the drawdown in a pumped borehole, B is the time variable linear well loss coefficient and C is the non-linear well loss coefficient that is independent of time. Theis (1967) rearranged the Cooper and Jacob (1946) equation to show reciprocal specific capacity as a function of the transmissivity and time, but it is clear from Bierschenk's analysis that the Theis equation describes only the linear loss component of the specific capacity identified in Equation 7. This gives the equation for the linear loss coefficient B, (Equation 8). It should also be noted that this equation assumes full penetration and an effective radius that is equal to the actual borehole radius. In the dual porosity Chalk of the SLARS area, this normally occurs after elapsed pumping (or recharge) times of 100 to 1,000 minutes.

$$B = 2.3 \log [(2.25 \ T \ t) / (r^2 \ S)] / (4 \ \pi \ T) \qquad \text{Eqn. (8)}$$

Mace (1997) showed that specific capacity is a function of the non-linear loss coefficient, C, and transmissivity, T. However C is also a function of T. The mathematical function is most easily determined from best-fit empirical data using available pumping test data. In the SLARS study area, only data from boreholes with similar construction that may be considered to conform to the Thames Water standard for borehole construction were used. This empirical relationship is shown on Figure 2 and is defined in Equation 9.

$$C = 0.0004 \ T^{-1.3625} \qquad \text{Eqn. (9)}$$
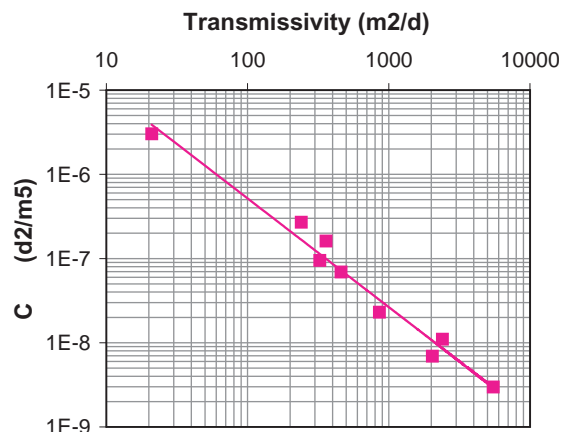
Figure 2. Relationship of C to T

Equation 9 includes data from both boreholes and wide diameter wells and is valid providing the boreholes are confined and the borehole (or well) diameter is greater than 600 mm. Combined together equations 7, 8 and 9 give an effective calculation method for specific capacity.

Application using a GIS

In order to use Equations 1 to 9 in a GIS, spatial distribution maps of the terms that can be directly measured (such as transmissivity and storage) must first be prepared. Most GIS have a number of contouring functions that vary in suitability for purpose. Invariably, trigonometric rather than kriging or averaging functions were found to be the most appropriate, as the latter two do not produce distributions that are consistent with the measured data. Smoothing of trigonometric contours can in most cases produce a more presentable distribution equivalent to kriging but which is also consistent with the measured values. Maps of some parameters such as distance to nearest spill point may need to be produced with an alternative method. As distance to nearest spill point (r) is a function of geometry, this map can be produced by geometrical construction. GIS software do not currently have a map function capable of doing this, so this map must be produced by hand. This is done by digitising the distance to a spill point for a sufficient number of locations and then using the trigonometric function to produce the finished GIS map. Aquifer storage proved to be an exceptional case in the SLARS study area because storage values from reported pumping tests tend to be significantly in error due to inconsistent analysis of the significant dual porosity in the Chalk, (Anderson et al, in press). Consequently, a storage map where the storage values were assigned according to the confined, semi-confined or unconfined status of the aquifer at that location, was found to locally produce a more reliable storage distribution than a trigonometric contour map of storage values reported from pumping test data. Thus the assigned value method was used to produce the GIS map of storage values in this study. The issues with the reported SLARS storage values highlight the need to carry out a quality assurance check following map production. This process is required to confirm the contouring method is a) accurate b) fit for purpose and c) is consistent with the conceptual model of the parameter distribution.

RESULTS AND DISCUSSION

Using the mathematical equations described here it proved feasible to produce retention time and recharge capacity maps for the SLARS study area. The recharge capacity method was well established prior to the SLARS study and required little iterative technical development, whilst the required data (transmissivity, groundwater rest levels and ground levels) were all available prior to the start of the SLARS investigations. Maps of recharge capacity produced in the initial period of the study were subsequently proven during the recharge testing investigations to provide reasonable estimates of recharge capacity in the SLARS area.

As an entirely new concept, a number of iterations were required to develop the retention time technique during the study. Initial retention time maps relied on Equation 1 only. As expected, when these results were compared with the (Hunt, 1999) stream depletion calculation (adjusted for local 'window' geometry), the results were found to be too conservative. However as the method does not require values for the vertical hydraulic conductivity of intervening leaky layers, and this data was not available at the time, the method proved very useful. Although the method produces conservative values of retention time and therefore can not be used to reliably distinguish locations suitable for either drought or seasonal peak demand schemes, it did identify the spatial variance in relative suitability for artificial recharge. It was also possible to confirm that the two SLARS test locations; Streatham and Ladywell Fields, (Figure 1); that were selected on the basis of other considerations; were also good locations to undertake artificial recharge investigations. At the Streatham site a recharge capacity of 14 Ml/d and a conservative retention time of 50 days together were close to the optimum conditions in the west SLARS area of the River Wandle catchment. In contrast the Ladywell Fields site was close to the River Ravensbourne, and was likely to be a good location to investigate the significance and behaviour of leakage to and from the river.

The second iteration of the retention time method used Equation 2 and a fixed drawdown of 4 m for the value (s). This approach was found to be consistent with the (Hunt, 1999) stream depletion calculation (adjusted for local 'window' geometry), using the only reliable K_v value then available from the study area, obtained from pumping test and river flow measurements in the Ravensbourne catchment. These results suggest the K_v value of the Thanet Sands is locally 0.001 m/d and the Thanet Sands and not the River Alluvium locally limit vertical leakage in the Ravensbourne window, (Figure 1). However subsequently the leaky hydraulic response of the Ladywell Fields pumping test indicated a new reliable K_v value of 0.03 m/d for the Alluvium in the Deptford window, (Figure 1), where the River Alluvium rests directly on the Chalk. As a result it became evident that a single drawdown value was inappropriate and an adjustment for spatial variation in K_v was necessary. Figure 1 shows the improved calculated spatial variation in retention time in the SLARS study area using Equation 4. This final result was validated against operational recharge data from NLARS, the SLARS recharge pumping test investigations and the (Hunt 1999) river depletion equations, and found to be sufficiently accurate to be an effective method.

CONCLUSION

The mathematical function of a GIS is a very powerful and generally underused quantitative tool. It can be used to calculate spatial distributions of retention time and recharge capacity that can not be directly measured. Spatial distributions of recharge capacity and retention time can be used to determine the feasibility of both drought and seasonal peak lopping solutions at each location in a study area. In a study area where artificial recharge potential varies substantially, this is probably the only method of determining the context of results from limited recharge test locations.

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A strategy for optimizing groundwater recharging by flood water in the northwestern coastal zone of the Gulf of Suez area, Egypt

H.H. Elewa

Abstract

In the northwestern coastal zone of the Gulf of Suez region, Egypt, the groundwater is being excessively pumped thorough a number of domestic and commercial wells. For this sake, recent groundwater recharging by direct rainfall and surface runoff water needs to be maximized and optimized.

The hydrographic basins of the study area are distinguished into five hydrographic basins. These basins are W. Ghweibba, W. Badaa, W. Hagul, W. Hammtih and W. South Hagul. The construction of retardation dams in some selected locations will enhance the groundwater recharging, or at least minimize the flood damage with the concomitant increase in seepage/runoff ratio. The sites selection of these dams was determined according to several criteria, e.g. soil characteristics, soil infiltration capacity, slope factors, morphometric characteristics and flood mitigation measurements.

The water seepage rate in Wadi South Hagul is 0.08×10^6 m^3/h, which gives good chance for a large part of flood water to percolate through the surface soil to the under ground. The trunk channel of W. Hammtih basin is characterized by high runoff rate, when it is compared with the seepage rate (0.17×10^6 m^3/h). Wadi Hagul basin is characterized by low seepage rate, where the high flooding episode of 3.25×10^6 m^3 occurred in 1990 with runoff rate of 2.25×10^6 m^3/h and seepage rate of 0.71×10^6 m^3/h. For Wadi Badaa basin, the seepage/runoff relationship indicates that W. Badaa is moderate in the accumulation of floods water, where, the high flood rate of 5.05×10^6 m^3/hour happened in 1990 with seepage rate of 2.93×10^6 m^3/hour. Finally, W. Ghweibba basin has high runoff rate of 22.8×10^6 m^3/hour in 1990 with seepage rate of 9.70×10^6 m^3/h.

A flood-harvesting plan was prepared to maximize the groundwater recharging by surface runoff water depending on the previously discussed techniques and measurements.

Keywords

Groundwater recharging, flood water harvesting, flood hazards mitigations, drainage basins.

INTRODUCTION

In the last few years, West Gulf of Suez area, one of the most important areas, took considerable attention by the Egyptian Government to become one of the mega national projects regions. The water resources in this area will play an increasing important role in providing a source of potable water for land using and construction of new settlements. Optimizing the recharging processes of groundwater by surface runoff water is an urgent need for this economical region. This policy affects the aquifer dependability as a continuous resource of water and may eventually minimize the deterioration of its water quality due to the liability of the coastal zone to seawater intrusion problems.

Location of study area

The study area, Northwest Gulf of Suez, is a desert region located between Gebel Al Galala Al Baharyia and Gebel Ataqa at Cairo-Suez district, Egypt (Figure 1). The area extends from 29°20'00' to 30°00'00"N and from 31°50'00" to 32°30'00' E and covers about 4,750 km^2 and includes five main hydrographic basins (W. Ghweibba, W. Badaa, W. Hagul, W. Hammtih and W. South Hagul) (Figure 2).
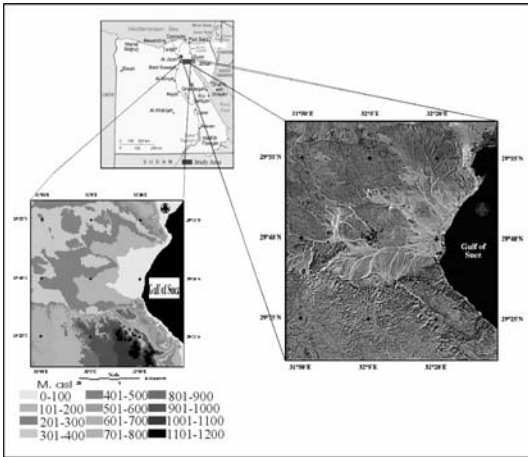
Figure 1. Location map, Landsat ETM+ image (bands 7 4 2) and colored digital elevation model (DEM) of the study area
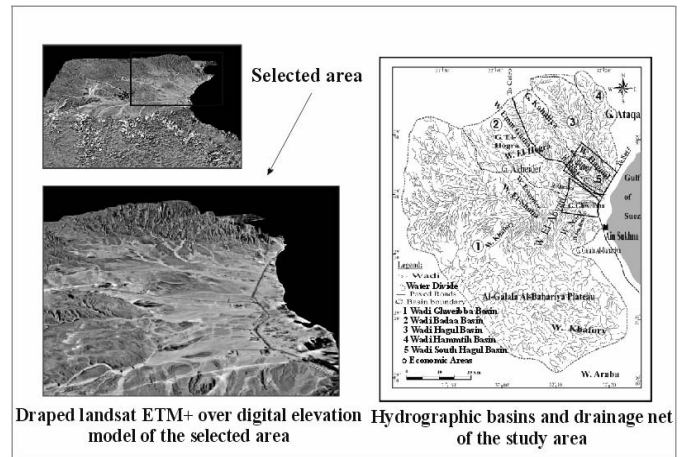
Figure 2. Physiographic features and hydrographic basins of the study area

RUNOFF CALCULATIONS

Soil infiltration capacity

The major abstraction from rainfall during a significant runoff storm is the infiltration water through the soil into the underground. The relation between the infiltration rate and hydraulic gradient is not a simple one (Konknke, 1968 and Hillel, 1980 and Skaggs and Kaleel, 1982). The watershed area of the study region is large and for a good representation of infiltration tests, twelve sites were selected for representing the different soils and calculation the amount of water percolation from the surface runoff to the groundwater aquifers in the study area (Figure 3). The infiltrations were carried out using the double ring infiltration as described by Black (1973). The values of cumulative infiltration and infiltration rate as a function of time is illustrated graphically (Figure 4). The calculation of the vertical infiltration was carried out by using a formula developed by Philip, 1957.

The infiltration capacity is increasing at the upstream of the main wadis with a maximum value of 9.58 m/day
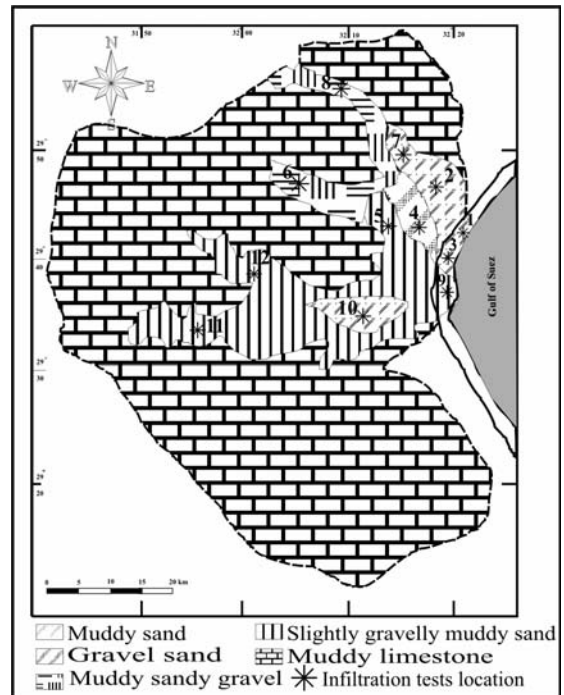
Figure 3. Soil texture map of the study area and locations of infiltration field tests

at the Test no. 8 (at the upstream of Wadi Hagul), whereas the minimum value is 0.38 m/day at the Test no. 2 (in the downstream of Wadi Hagul) (Figures 3 and 4). The infiltration rate is decreasing in the northeastern part of the area between Gebel Ataqa and Gebel El Galala El Baharyia (at the outlet of Wadi Hagul, Wadi South Hagul and the outlet of Wadi Badaa). From this point of view, the infiltrated water from surface runoff is the main source for groundwater aquifer recharge. Thus, it is expected to be increased at the area where there is an increase for infiltration rate (i. e. at the southwest, northwest and south of the study area). According to the infiltration tests and soil texture, the map of soil distribution in the study area is constructed (Figure 3).
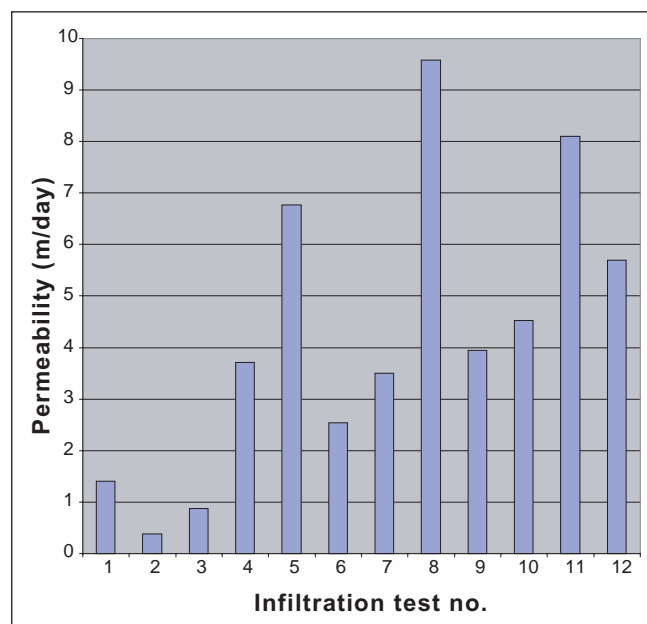
Figure 4. Results of hydraulic conductivities of twelve infiltration tests

Surface runoff

The surface runoff is influenced by the catchment's characteristics and climatic factors. These climatic factors include precipitation and its properties, humidity, evaporation, wind speed and temperature. The main hydrologic (physical) processes in any catchments are according to the basis of rainfall-runoff relationships (Soil Conservation Service 'SCS', 1986).

Runoff calculation methods

There are many methods widely used for estimating peak discharge of surface runoff. Several methods are recommended for wadis hydrology in arid watershed areas depending on the available data. The used methods are Ball, 1937 ($Q = 750A (i-8)$, where Q is the runoff rate (m^3), A is the drainage area (km^2) and i is average rainfall (mm) when i > 8 mm; and Rational method (McCuen, Richard H. 1998) ($Q = 0.278 C i A$, where Q is the runoff rate (m^3), A is the drainage area (km^2), i is average rainfall (mm) and C is the runoff coefficient constant based on the watershed conditions (0.25 – 0.50).

Floods potential

Based on the findings revealed from the present analytical study, the desert floods in the study area are inherently of low duration (few hours to some days) and are commonly characterized by sharp peak discharge. The time to peak discharge is variable and may be in the range of 0.91 to 1.37 hour.

The high seasonal runoff accumulation occurred in 1990 with rainfall depth of 22.8 mm. The low seasonal runoff accumulation occurred in 1993 with rainfall depth of 0.7 mm. W. Ghweibba have high annual volume and rate of surface runoff, whereas the volume is decreasing to the middle of the study area at W. Badaa, W. Hagul and W. S. Hagul. The volume and rate of floods increases at the north of the study area at W. Hammtih, where it is characterized by high difference between upstream and downstream.

The hydrographic basins in the study area are of high potentiality of flash flooding and short time interval between the beginning of rise and peak discharge. The flash floods in the study area are expected in future seasonal rainfall.

Floods hazard mitigation

The industrial projects and settlements at basins outlets are threatened by flash floods. (e.g. Economic area at North West Gulf of Suez, tourist projects & the main desert highways).

For this sake, precautionary measures should be taken into account to prohibit or at least minimize the flood damage with the concomitant increase in seepage/runoff ratio. The following is a brief description and discussion of the flood damage mitigation measurements (Figure 5).

1. *Wadi South Hagul basin*

   This basin occupies a rather circular shape with an area of about 56.1 km^2. The basin drains low terraces with low slope ratio. The high flash flood events in Wadi South Hagul occurred in 1988, 1990, 1991 and 1998 with runoff rates of 0.47, 0.43, 0.40, and 0.35 millions m^3/h and a volume of water accumulation of about 0.36, 0.62, 0.39 and 0.35 millions m^3. The water seepage rate is 0.08 x 10^6 m^3/h, which gives good chance for a large part of flood water to percolate through the surface soil to the under ground. W. South Hagul have high surface area of soil (about 100%) (Figure 3), the runoff water can be exploited by the construction of small boulder detention dams (Grigg and Helmeg, 1975); to replenish the groundwater aquifers (Figures 5 and 6).

2. *Wadi Hammtih basin*

   This basin is of the fifth order; it has an area of about 207 km^2 and drains G. Attaqa high tableland. It is of hexagonal shape and has high difference of elevation between the upstream and downstream. The trunk of W. Hammtih flows easterly to south southeasterly towards the Gulf of Suez. Massive Eocene Limestone with only small outcrops of Miocene dominates the lithology of its floor, whereas the Quaternary deposits (variable soil) expose near the mouth (Abu El Nasr, 1979). The seepage and surface runoff relationship shows the high water accumulation in Wadi Hammtih of about 0.24, 0.57, 1.36, 1.46, 1.78 and 2.22 millions m^3 in 1997, 1992, 1988, 1991, 1998, and 1990, respectively. Wadi Hammtih is characterized by high runoff rate when it is compared with the seepage rate (0.17 x 10^6 m^3/h). (Figure 5) shows the possibilities of the floods in all last seasons in the period from 1978 to 1998 (Figure 5). To protect those leading tributaries, detention dams should be constructed to increase the seepage/runoff ratio (Figure 6).

3. *Wadi Hagul basin*

   Wadi Hagul is hexagonal to circular in shape. It is of fifth order and has an area of about 293 km^2. Rugged mountainous relief (up to 450– 900 m) and sharp slope on the main stream characterize the basin.
      The main trunk flows to Gulf of Suez at the southeast. Wadi Hagul also is characterized by low seepage rate and the base floor is composed of Pliocene clays and limy mud. High flooding episode of 3.25 x 10^6 m^3 occurred in 1990 with runoff of rate 2.25 x 10^6 m^3/h, where the seepage rate is 0.71 x 10^6 m^3/h (Figure 5). This in turn increases the flash floods potentiality with the low groundwater replenishment. Construction of artificial dykes, dams and open lakes is an urgent demand for storing water (Figure 6). This water will recover the deficiency in groundwater.

4. *Wadi Badaa basin*

   This basin has an area of about 658 km2 and high percentage of soil area of about 24.31% of its total area

(Figure 3). The basin drains the moderately elevated mountains with steep slopes and elevations up to 700 m. The basin is asymmetric with fan shape and discharge to the southeastern direction towards the Gulf of Suez. The lithology of W. Badaa is composed of Eocene limestone in large part of the catchments area, Miocene limestone, Pliocene Clay and Quaternary deposits. The seepage/ runoff relationship indicates that W. Badaa is moderate in the accumulation of floods water (Figure 5).

The high seasonal floods occurred in 1988, 1990, 1991, and 1998 had volume of water of about 4.3,7.3, 4.6 and 5.67 millions m^3, respectively. The high flood rate of 5.05 x 10^6 m^3/hour happened in 1990 with seepage rate of 2.93 x 10^6 m^3/hour (Figure 5). Wadi Badaa comprises large part of the economic area and highways (Qattamyia-Ain Sukhna Road). The floods frequently cross the road (1990 and 1998). Construction of detention dams will be of twofold purposes; optimize the groundwater recharging by surface runoff water and provide protection from flood hazards (Figure 6).

5. *Wadi Ghweibba basin*

This is a large basin of the North West Gulf of Suez and has area of about 2,978 km^2. It is circular to hexagonal in shape and has large area of surface soil (Figure 3). The main trunk of W. Ghweibba flows easterly toward the Gulf of Suez and its large tributaries are mostly oriented northeasterly and southeasterly.

The lithology is dominated by massive Eocene limestone with only small outcrops of Oligocene, Miocene, Pliocene and Quaternary deposits. The high volumes floods event occurred in 1988, 1990, 1991 and 1998 with water accumulations of 19.2, 33.00, 20.9 and 25.6 million m^3 in respectively (Figure 5). The high runoff rate is 22.8 x 10^6 m^3/hour in 1990 with seepage rate of 9.70 x 10^6 m^3/h. Wadi Ghweibba includes large part of Economic area of north west Gulf of Suez and this floods accumulation is very dangerous, especially during the high depth rainfall storm. As previously mentioned, the construction of rocky retardation dams to harvest surface water and provide flood hazard protection, is an advantageous practical procedures in the study area (Figure 6)

CONCLUSION

The geographic and hydrographic features of the study area reveal promising flooding potentialities that could be of enormous benefit for enhancing the groundwater and hydrogeological conditions. The runoff calculations using the specific Ball equation and the widely used rational method revealed such a good flooding possibilities and quantities. The flood mitigative measurements resulted in the determination of runoff/seepage relationships for each major hydrographic basin of the study area. These measurements were graphically represented for complete historical period of rainfall and runoff episodes. Detention dams for capturing runoff and alleviating flood hazards were proposed as a flood harvesting plan. These dams were proposed according to the stream ordering and flood mitigative measurements. Dams' locations were determined at the outlets of the upstream third-order basins.

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A methodology for wetland classification attending to the capacity for artificial aquifer recharge. Application to the Coca-Olmedo wetlands, Duero Basin (Spain)

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Abstract

A specific system to classify wetlands attending to their suitability for aquifer recharge-based restoration is proposed. The system is based on several approaches, highlighting wetland's relationship with the aquifer, current state of conservation, typology of environmental impacts that have affected the wetland, hydrogeological conditions, presence of water and viability of aquifer recharge. This paper arises from the PhD of the first author.

INTRODUCTION

Interactions between the human being and aquatic ecosystems date from ancestral times. Anthropic actions have contributed to the high dynamism of water landscapes, resulting in very diverse environments. In this respect, Spain is the country of the EU that presents the greatest diversity wetland types (Casado and Montes, 1995).

Wetlands are particularly valuable from the scientific point of view, since they represent 'living laboratories' for the study of multiple natural processes. In addition, wetlands are important for ecological and socio-economic reasons. However, the reality of wetlands is often misunderstood by human beings, who at times find it difficult to adequate themselves to aquatic ecosystems (González, 1989, 1992 a, b, c.).

Wetland recovery through artificial aquifer recharge is a relatively new idea, or at least, one that has generally been hinted but not implemented (MAPA, 1999, Galán et al, 2001; Fdez. Escalante and López, 2002). Artificial aquifer recharge is currently applied in Holland and the United States, among other countries (Stuyfzand and Mosch, 2002), and has the potential to contribute to wetland restoration. However, artificial recharge often implies a change in water quality, due to the different nature of local and imported waters (Fdez. Escalante, 2005).

This paper presents a method for wetland classification attending to their capacity to be regenerated through artificial aquifer recharge. The application of this methodology to a set of wetlands in Spain is described in detail. The paper arises from the PhD of the first author (Fdez. Escalante, 2005).

OBJECTIVES

This paper assesses the current state of some of the Coca-Olmedo wetlands. Approaches to wetland characterization based on specifically-designed artificial parameters have been applied.

After such characterization, a brand new method for wetland classification is presented and applied. This classification indentifies the suitability of wetlands for regeneration by means of artificial aquifer recharge.

(i) Characterization and inventory

Characterization based on the new classification has been carried out in three of the 17 wetlands with bibliographical references (Rey Benayas, 1991). In all the cases (those 17 and 68 additional inventoried in Fdez. Escalante, 2002 and 2005), field measurements were taken for morfometric characterization, with the ultimate objective of generating numerical data to allow the study of their future evolution.

Wetlands have been classified according to three approaches:

A) Those that are under legal protection, like the lagoons of Caballo Alba (SG-1), La Iglesia (SG-3) and Las Eras (SG-2).
B) Those that maintain a certain 'lavajo' (a local name for a type of wetland) in spite of human attempts of drainage. This is the case of the Valderruedas or Valdeperiñan wetlands, both of which present significant alterations.
C) Those that have been deeply transformed (plowed areas, urbanized or with an undue use).

The objective of the inventory and characterization is to study the viability of regenerating of the wetlands by means of artificial aquifer recharge, provided that technical and socio-economic conditions are favorable.

Table 1 shows the characterization approaches and their application to a given wetland. This is intended as an example of how all the necessary knowledge can be integrated in a single table, so that coordinated action can be taken, particularly in regard to environmental issues.

Data presented in Table 1 refers to August 2003, and is partially based on Gonzalez (1988 and 1989) ideas on environmental evaluation.

(ii) A brand new wetlands classification proposal

The inventory of the previous section has been incorporated to a documental database of wetlands potentially elligible for artificial recharge (provided that economic, environmental and political conditions allow).

This classification keeps in mind the conservation state and functioning of the system in natural conditions, that is, before anthropogenic impacts occur (Fdez. Escalante, 2002, 2005; García, 2003) (Table 2).

Four wetlands typologies have been identified according to their functioning. Each of these is assigned one color:

Red: Currently extinct wetlands, detected by means of indirect techniques: old aerial photographs, interviews to local population, tests detected in field, etc. These have been termed 'indicial wetlands'.
Orange: Strongly degraded wetlands. Recovery is difficult because due to the variety of impacts.
Yellow: Wetlands associated to net runoff. These are generally endorreic, and unrelated to groundwater: their origin is not conditioned by the phreatic level of aquifer, but rather to superficial drainage networks. These wetlands may remain dry during the drought periods and hardly lose their ecological value (high resilience).

Their recovery is relatively easy if artificial recharge water does not have a significant impact on water quality.

Blue: Groundwater-dependent wetlands (Custodio, 2000). Their recovery is more complicated than in the previous case, since they require a qualitative modification of recharge water, using techniques to replicate the processes of soil-water interaction.

Suitability of wetlands to be restored by means of operations of artificial recharge corresponds in general to blue or yellow types.

Given the importance of water availability for recharge, each colour is accompanied by a relative index to the presence of water and of agricultural data.

The presence of water is largely conditioned by the date the inventory was carried out, as well as by the type of hydrological year. Nevertheless, regenerative works are envisioned for the wetlands degraded due to the disappearance of their natural water sources (though irrigation, drainages, derivations of streams, canalisations, etc.), and not due to the absence of precipitations during dry periods.

Weights assigned to each type of wetland in relation their state of conservation are based on the following criteria:

1) Absence of water for most of the year. Very strong affection. The wetland and their surrounding land have been subjected to changes in use.
2) Total absence of water. Affection is substantial. Derived impacts of the terrain ploughing and drainage, cropping activity is usually present in the area.
3) Widespread absence water. Affection is remarkable. Recovery is feasible by increasing water inputs to the system and correcting certain additional significant impacts (drainages or undue uses). Agricultural activity usually present in the area.
4) Absence of water. Wetland recovery is possible by increasing water inputs to the system. Additional environmental impacts are of scarce intensity. No agricultural activities in wetland perimeter fringe nor in the influence area. Hydrochemical quality conditioned by the presence of manure in the surroundings areas.
5) Presence of water the whole year. No agricultural activity. Narrow variation in water quality. Suitability for protection perimeters, legal protection, etc, in order to safeguard their current state and to avoid deterioration.

Types 1 and 2 correspond to indicial wetlands in most cases. Techniques fo detection are diverse. Chemical composition of soils generally prevails, except in the case of wetlands associated to the superficial hydrology, whose preservation index is variable.

Types 4 and 5 correspond to reasonably well preserved wetlands, while type 3 corresponds to an intermediate state.

An application of the new classification is presented in Table 3. Most of the inventoried wetlands classified as 'blue' are susceptible of a rational regeneration by means of induced artificial recharge. In contrast, red and orange wetlands correspond to a high degree of deterioration.

CONCLUSIONS

A significant part of the wetlands in the area of study present a fair degree of conservation and a low ecosystem value. Some wetlands are quite well preserved and in many cases they are under legal protection.

Implementation of artificial recharge would allow the use of excess water for environmental restoration. The

abundance of degraded wetlands suggests this may be one of the most recommended uses for excess water. However, it is necessary to devise a system to prioritize which wetlands should be regenerated in the light of their own technical and political viability.

Within the Coca-Olmedo wetlands, there is a second typology that corresponds to unrecoverable cases. These refer to wetlands that have been dramatically modified, generally due to anthropogenic action. These wetlands, classified as 'red', have been excluded from Table 3.

Although there is a degree of subjectivity in assigning weights to 'state of conservation', wetland characterization and monitoring in time will allow to appreciate the changes and act accordingly.

Most wetlands tend to experience an increase in their numerical index with time, although not necessarily all end up with an index 5. Legal protection could be aimed at achieving a degree of connectivity, so that wetland regeneration may have a positive impact on the rest of the system Rey Benayas (1991).

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Table 1. An example of characterization suitable for a set of wetlands

CHARACTERIZATION TEMPLATE: 'LA IGLESIA' WETLAND. AUGUST OF 2003	
IDENTIFICATION	**CHECK-UP**
WETLAND CODE	Nº 57. SG-3
TOPONOMY	'LAGUNA DE LA IGLESIA'
ADMINISTRATIVE UNIT	COCA
AUTONOMY REGION	CASTILLA-LEÓN
TOWN/ MAP 50.000	VILLAGONZALO DE COCA/ 455 ARÉVALO
BASIN	DUERO
SUBASIN	2229 CEDEX. Eresma Duero (Voltoya-Adaja)
UTM COORDINATES (Huse 30): X:	367434
Y:	4562545
Z:	798,1
TYPE	ENDORREIC AND SALINE
FIGURE OF PROTECTION	WETLAND CLASSIFIED AREA SG-3
AEREAL PHOTO	8726 (1957); 428F (1987)
ORTOIMAGEN	428 S-3 (JCL 2000).
IMAGE SATELLITE	Spot ORBIT 031, Spot 2 de orbit 032 y Landsat 202
PHOTOGRAFY	57.jpg (annexes 4.1.2. y 4.2.2.).
RESOLUTION MDT AVAILABLE	1 METER
ACCESS	JUST BESIDE COCA TO VILLAGONZALO RD
CHARACTERISTICS	
GENETIC ORIGIN	TOPOGRAPHICAL BASIN
HYDRODEPENDENCY	NO
RELATED AQUIFERS	NO
OVEREXPLOITATION YES/NO	YES
CONNECTION WITH FLUVIAL NET	NO
SOIL ENVIRONMENT	LOAMY AND SALINE LAND
HYDROCHEMICAL FACIES	DURING 2002 AND 2004 WAS DRY. SMALL PUDDLE OCASSIONALLY
CHEMICAL ANALYSIS	NO
PERiMETER OF PROTECTION	NO
VULNERABYLITY	HIGH
LANDSCAPE ASSESMENT	4
PRESENCE OF HIDROFILIC VEGETATION	HALOPHITIC
PRESENCE OF NITRIC VEGETATION	YES. PERIMETRAL FRANGE

Table 1 (contd.)

MORPHOMETRY	
BIGGER AXIS	482.6 m
SMALLER AXIS	227.5 m
LIMNIMETER	NO
LIMNIMETER BENCH MARK	NO
MAXIMUM DEPTH OF RECIPIENT BASIN	1,4 m
MAXIMUM DEPTH	0 (2004)
Maximum perimeter (m)	1209
Maximum wide (m)	227.5
Longitude Maximum (m)	482.6
Maximum Surface (km^2)	0,794
References	Rey Benayas, 1991
NATURAL PROTECTED AREAS (ENP)	
LIC (PLACES OF UE INTERES)	NO
Spanish Historical Patrimony	NO
ZEPA SPECIAL AREA OF BIRD PROTECTION	NO
RAMSAR	NO
OTHER	CLASSIFIED HUMID AREA SG-3
WETLAND PERFORMANCES	
(TYPE OF AFFECTION)	DOWNDRAWN OF WATER LEVEL DUE TO EXTRACTIONS BY IRRIGATION PURPOSES
FILLED (YEAR)	
DRAINAGE (YEAR)	YES / NO
CROPS	NO
SURROUNDED by	CEREALS
DAMMED (YEAR)	NO
DUG	NO
DREDGED (YEAR)	NO
ARID EXTRACTIONS	NO
REGULATION	YES. DREIN IN SOUTH AREA
WATER EXTRACTIONS	NO
SUSCEPTIBILITY TO WASTES	
URBAN	NO
INDUSTRIAL	NO
AGRICULTURAL	YES. DIFFUSE CONTAMINATION
AGROCHEMICAL	YES.
PESTICIDES	YES BY LIXIVIATE
USES	
SHEPHERDING	OCCASIONAL
WATER FOR CATTLE	NO
CITIES SUPPLY	NO
EXTRACTIONS AND OUTLETS	NO
IDUSTRIAL USES	NO.
OTHER USES	AESTHETIC, LANDSCAPE

Table 1 (contd.)

PUBLIC USE	
RECREATION	NO
ACUICULTURE	YES (NEGATIVE))
URBANIZED	NO
HUNTERY	YES
FISHERY	NO
EDUCATIONAL	NO
RECREATIONAL	NO
MEDICINAL	NOT AT THE CURRENT TIME
VEGETATION USE	NO
OTHERS	NO
OBSERVATIONS	SALINE SOIL WITH POSIBLE INTEREST
ADMINISTRATIVE DATA	
PROPERTY	COCA TOWN HALL
ADMINISTRATION	JCL
CONSERVATION DEGREE	4/BLUE TYPE (Fdez. Escalante, 2005)
INSTRUCTOR/SOURCE	Fernández Escalante, 2005
OBSERVATIONS	IMPROVEMENT BY MEAN OF MAR

Table 2. Classification

Conservation state

5. Water. It can improve easily.
4. Without water. It can recover. No agricultural activity nearby.
3. Without water. Affected. Agricultural activity nearby.
2. Without water. Substantially affected. Ploughed, cultivated.
1. Without water. Substantially affected. Changes in use.

Table 3. Wetland inventory and characterization

Nº	Type	Cons. state	Name	Locality	Date	UTM Coordinates X	Y	Z	Morphometry Area (m²)	Lenght, Perim. (m²)	Width, Max (m)	Depth, Max (m)	Max (cm)	Influence area morphometry Area (m²)	Lenght, Perim. (m)	Width, Max (m)	Max (m)	BIB Scale	BIB
1	RED	1		AGUASAL	31-07-03	363023	4574355	750.5						1223	148	58,6	30,8	N	N
2	BLUE	4	Laguna de las Eras	AGUASAL	31-07-03	363298	4574254	750.6	680	315	25.0	20.0						N	N
3	BLUE	3		AGUASAL	31-07-03	363272	4574192	750.6	746	104	33.0	33.0						N	N
4	RED	1		AGUASAL	31-07-03	363055	4574177	750.1						423	77	27,6	18,9	N	N
5	BLUE	5		AGUASAL	31-07-03	363178	4574046	750.3	557	88								N	N
6	BLUE	5	Juncarral	AGUASAL	31-07-03	362876	4574044	749.0	745	125	49,7	20,8	35					N	N
7	BLUE	4	Bodón de la Liebre	AGUASAL	31-09-03	363580	4571974	754.5	380	77	27,5	20,4						N	N
8	BLUE	4	Bodón de la Crijota	AGUASAL	31-07-03	363181	4571973	753.6	675	300	15.0	12.0						N	N
9	BLUE	3		AGUASAL	31-07-03	363308	4571934	753.9										N	N
10	BLUE	3		AGUASAL	31-07-03	363648	4571809	754.4	2836	247	77,7	63,3						N	N
11	BLUE	3		LLANO DE OLMEDO	31-07-03	364202	4571656	755.8	358	70	24,2	19.0						N	N
12	BLUE	4	Bodón Guarrero	LLANO DE OLMEDO	31-07-03	364467	4571575	757.0	1619	149	49,8	47.0	0					N	N
13	RED	1	El Horno	LLANO DE OLMEDO	31-07-03	366213	4570738	761.1										N	N
14	RED	1		LLANO DE OLMEDO	31-07-03	366747	4570506	762.4										N	N
15	BLUE	4	Bodón de la Vega	LLANO DE OLMEDO	31-07-03	366407	4570320	762.5	2265	234	84,1	38,4						N	N
16	RED	1	AGUA CABALLO	VILLEGUILLO	31-07-03	367496	4570188	764.7										N	N
17	BLUE	4	La Cárcaba	LLANO DE OLMEDO	31-07-03	365458	4570158	762.6	16837	565	229,4	112,9	48					N	N
18	ORANGE	2		LLANO DE OLMEDO	31-07-03	365095	4570039	772.2	1891	174						67,9	39,4	N	N
19	BLUE	5	Los Majuelos	LLANO DE OLMEDO	31-07-03	364390	4570049	778.2	4610	310	110.0	75,4	22					N	N
20	RED	1		VILLEGUILLO	31-07-03	367740	4570024	765.3						8772	376	113,8	103,6	N	N
21	RED	1	La Cochina	LLANO DE OLMEDO	31-07-03	365783	4570020	763.8						89759	1766	476,5	282,2	N	N
22	ORANGE	2	Ermita Llano de Olmedo	LLANO DE OLMEDO	31-07-03	364825	4569739	779.7	2438	216						79,7	40,2	N	N
23	BLUE	4	CARCAVA GRANDE	AGUASAL	31-07-03	361753	4570906	758.0	3638	276	89,4	40,3	55					N	N
24	ORANGE	2		AGUASAL	31-07-03	361573	4570906	758.0	1400	159						62.0	31,4	N	N
25	YELLOW	2		AGUASAL	31-07-03	361410	4571053	759.0										N	N
26	BLUE	3		AGUASAL	31-07-03	362997	4571729	759.0	675	310	19.0	15.0						N	N
27	BLUE	4		AGUASAL	31-07-03	361671	4570562	759.0	1404	170	67,1	34,8	15					N	N
28	BLUE	4	CARCAVA CHICA	AGUASAL	06-08-03	361733	4570619	759.0	4463	331	121,4	53,5	18					N	N
29	BLUE	4	El medianero	VILLEGUILLO	06-08-03	366739	4569680	764.6	87	34	11,4	9,9						N	N
30	YELLOW	1	Las Dehesillas	VILLEGUILLO	06-08-03	368349	4569105	767.6	650	300	17.0	16.0						N	N
31	ORANGE	3		VILLEGUILLO	06-08-03	367194	4569144	767.7	161	57						24,8	9,4	N	N
32	YELLOW	3	EL ARENERO	VILLEGUILLO	06-08-03	367129	4568786	776.0	1238	206						73,6	27.0	N	N
33	ORANGE	2		VILLEGUILLO	06-08-03	366283	4568485	772.0										N	N
34	RED	1		VILLEGUILLO	06-08-03	367714	4568197	769.8						6308	410	185,4	50,9	N	N
35	YELLOW	1		VILLEGUILLO	06-08-03	366335	4568177	768.5	549	86						30,7	23,2	N	N
36	ORANGE	2	BODON REDONDO	VILLEGUILLO	06-08-03	369045	4567899	770.7	605	124						47,5	22,9	N	N
37	ORANGE	1		VILLEGUILLO	06-08-03	369611	4567868	771.5	7067	402						149,6	67,2	N	N
38	BLUE	3	LAGUNA DE VALDEPERILLÁN	LLANO DE OLMEDO	06-08-03	364791	4567848	762.8	246601	3268	1220,4	335,8						N	N
39	YELLOW	3		VILLEGUILLO	06-08-03	365917	4567840	768.3	30678	995						347,8	127,6	N	N
40	BLUE	3	LAGUNA DE CABALLO ALBA	VILLEGUILLO	06-08-03	365529	4567306	768.7	197200	2139	555,1	552,7						N	a
41	YELLOW	2		VILLEGUILLO	06-08-03	367367	4566909	788.3	3763	261						97,9	53,9	N	N
42	YELLOW	2		VILLEGUILLO	06-08-03	366217	4566602	772.9	59331	2281						942,9	115,1	N	N
43	BLUE	3		FUENTE-OLMEDO	06-08-03	364150	4566613	774.6	56112	910	311,9	300,7						N	N
44	YELLOW	2	LOS POZUELOS	FUENTE DE SANTA CRUZ	06-08-03	365167	4566375	772.4	79008	1407						383,8	249,1	N	b
45	BLUE	3		CIRUELOS DE COCA	06-08-03	367263	4565494	794.9	2392	206	78,6	37,4	22					N	N
46	BLUE	3		CIRUELOS DE COCA	06-08-03	367272	4565455	794.8					20					N	N
47	YELLOW	2		CIRUELOS DE COCA	06-08-03	366162	4565362	775.4	149115	3535						899,6	289,9	N	N
48	YELLOW	2		CIRUELOS DE COCA	06-08-03	367390	4565180	795.7	8331	476						204,4	55,8	N	N
49	BLUE	4		CIRUELOS DE COCA	06-08-03	367354	4565070	795.6	2250	220	69,1	46,2	18					N	N
50	BLUE	3		CIRUELOS DE COCA	06-08-03	367451	4564778	797.0	5029	341	98,4	92,2						N	N

Table 3. (contd.)

Nº	Type	Cons. state	Name	Locality	Date	UTM Coordinates			Morphometry					Influence area morphometry				BIB Scale	BIB
						X	Y	Z	Area (m²)	Lenght, Perim. (m²)	Width, Max (m)	Depth, Max (m)	Max (cm)	Area (m²)	Lenght, Perim. (m)	Width, Max (m)	Max (m)		
51	BLUE	3		CIRUELOS DE COCA	06-08-03	367525	4564520	798.2	10750	551	179,5	78,8						N	N
52	RED	1		COCA	06-08-03	369709	4563554	783.0						3856	278	111,8	49,6	N	N
53	ORANGE	2	LAGUNA DE VALDERUEDAS	VILLAGONZALO DE COCA	06-08-03	367332	4563161	799.0	45185	1382						637,4	143,9	N	b
54	RED	1		VILLAGONZALO DE COCA	06-08-03	368141	4563197	802.1										N	N
55	RED	1		VILLAGONZALO DE COCA	06-08-03	367991	4563036	803.8										N	N
56	BLUE	4	LAGUNA DE LA IGLESIA	VILLAGONZALO DE COCA	06-08-03	368570	4562726	786.9	79439	1209	482,6	227,5	0					N	c
57	BLUE	5	Laguna de las Eras	VILLAGONZALO DE COCA	06-08-03	367434	4562545	798.1	51562	1149	468,9	164,9	60					N	c
58	ORANGE	3		VILLAGONZALO DE COCA	06-08-03	367601	4561954	808.2	3123	264						98,5	59,2	N	N
59	ORANGE	2		VILLAGONZALO DE COCA	06-08-03	366912	4561831	807.3	158	49						15,7	12,2	N	N
60	YELLOW	2	EL BODO	BERNUY DE COCA	06-08-03	365036	4561291	811.3	358	77						30,6	19,27	N	b
61	RED	1		VILLAGONZALO DE COCA	06-08-03	367182	4561024	817.9						279	60	19,6	18,1	N	N
62	BLUE	3	FUENTE MIÑOR	VILLAGONZALO DE COCA	06-08-03	367446	4560584	812.8	62394	1334	559,8	169,1						N	b
63	YELLOW	2		BERNUY DE COCA	06-08-03	365378	4560000	820.3	178	49						14,2	14,8	N	N
64	YELLOW	2		BERNUY DE COCA	06-08-03	365240	4559907	824.5	7334	489						226,5	58,5	N	N
65	YELLOW	3		SANTIUSTE DE SAN J & B	06-08-03	368301	4559261	801.7	382	71						24.0	20,4	N	N
66	BLUE	4	BODÓN DE LA HIRUELA	SANTIUSTE DE SAN J & B	06-08-03	368426	4559078	802.6	2172	184	66,2	39,5	40					N	N
67	BLUE	5	HUMEDALES DE BERNUY	SANTIUSTE DE SAN J & B	06-08-03	366880	4558862	815.4	23044	600	234,1	67,6	50	3442	271			N	b
68	RED	2		SANTIUSTE DE SAN J & B	06-08-03	367351	4558400	824.5								110,1	60,2	N	N
69	RED			AGUASAL	31-07-03	362806	4572371	757.0										N	N
70	RED			AGUASAL	31-07-03	363358	4572241	757.0										N	N
71	RED			AGUASAL	31-07-03	362936	4571963	757.0										N	N
72	RED			AGUASAL	31-07-03	363651	4571992	757.0										N	N
73	RED			AGUASAL	31-07-03	363152	4571742	757.0										N	N
74	RED			LLANO DE OLMEDO	31-07-03	364021	4571449	762.0										N	N
75	RED			LLANO DE OLMEDO	31-07-03	364366	4571257	763.0										N	N
76	RED			LLANO DE OLMEDO	31-07-03	364558	4571080	762.0										N	N
77	RED			AGUASAL	31-07-03	364909	4573024	750.0										N	N
78	RED			AGUASAL	31-07-03	361227	4571651	750.0										N	N
79	RED			AGUASAL	31-07-03	361280	4571161	750.0										N	N
80	RED			AGUASAL	31-07-03	361083	4570830	750.0										N	N
81	RED			AGUASAL	31-07-03	363426	4574444	750.0										N	N
82	RED			AGUASAL	31-07-03	363502	4574420	750.0										N	N
83	RED			AGUASAL	31-07-03	363776	4574166	750.0										N	N

Conservation state
    5 Water. It can improve easily.
    4 Without water. It can recover. No agricultural activity nearby.
    3 Without water. Affected. Agricultural activity nearby.
    2 Without water. Substantially affected. Ploughed, cultivated.
    1 Without water. Substantially affected. Changes in use.

Functioning
    Groundwater-dependent wetlands.
    Wetlands associated to net runoff. Unrelated to groundwater/Endorreic.
    Strongly degraded wetlands. Recovery is difficult.
    Currently extinct wetlands. Interviews, etc.

a   ENP JCL, Rey Benayas, 1990, Montes, 1995
b   Benayas, 1990
c   ENP JCL, Rey Benayas, 1990
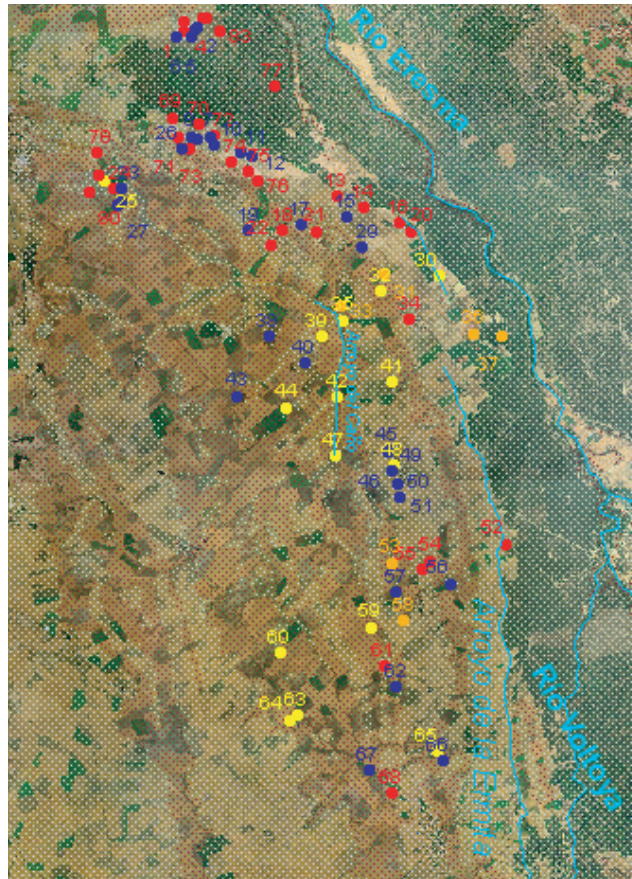
Operation

- Hydrodependent wetland.
- Wetland in connection with the net of surface runoff. No Hydrodependent /endorreic.
- Wetland strongly damaged with difficult recovery.
- Indicial wetland missing.

Figure 1. Location of the wetlands inventoried of the Coca-Olmedo Complex and application of the classification for their regeneration by means of artificial aquifer recharge techniques. Scale (aprox): 1:150.000

Use of environmental indicators in a multi-criteria analysis of the impact of MAR on groundwater dependent wetlands

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Abstract

The following pages present an attempt to design a system of environmental indicators to monitor the evolution of an aquifer by means of recent artificial recharge works. In this way, it is possible to value its effectiveness and to follow the evolution and the interaction between the different technical, economical and environmental aspects. For the study of its evolution and the degree of interaction, a system of ranges-weights and a multi-criteria evaluation polygon have been designed. The application of the proposed methodology allows to know the 'state of pressure', to correct adverse environmental impacts and to systematically improve the efficiency of technical operations.

Keywords

Artificial recharge of aquifers, environmental indicators, evaluation of environmental impact, PSR system, Coca-Olmedo wetlands complex.

INTRODUCTION

This paper is based on the experience acquired during the artificial recharge of the Cubeta of Santiuste (Segovia) aquifer, promoted by the General Secretary of Rural Development of the Spanish agriculture ministry (MAPA). It has been applied in two selected wetlands of the Coca-Olmedo Complex (Rey Benayas, 1991) susceptible of regeneration by means of artificial recharge: the lagoons of Las Eras and La Iglesia, in Villagonzalo de Coca (Segovia) (González, 1989, Fernández Escalante, 2005). The following pages present a method for environmental impact evaluation, management and control of artificial aquifer recharge operations (AR), particularly in cases were wetland restoration is possible. We propose an environmental indicators system (most of own design), and a multi-criteria evaluation polygon to tackle and synthesize the evaluation and monitoring processes. The method allows to reflect quantitative, qualitative, evolutionary, ecological aspects, etc., as well as the qualitative evolution of all kind of waters in the study area.

DESCRIPTION OF THE SYSTEM

The Santiuste basin is a small aquifer with 85 km^2 in extension that is limited by the Voltoya and Eresma rivers to the East, and low permeability Tertiary outcrops to the West. Saline wetlands abound. Despite its reduced size, it is considered an important irrigation area that depends heavily on groundwater (see Figure 1). Intensive pumping led to a significant drop in the water table and in turn to severe wetlands degradation. In 2002, an artificial recharge project for the aquifer was implemented by government initiative. The scheme is based on a small dam that taps the Voltoya river, carrying the water from a altitude of 817 m above sea level, by gravity down to a transferring station located at 814 m above sea level. A 10 km long pipeline starts from this station, carrying the water down the slope

to a 36 m^3 deposit. The main recharge ditch, dug out through the 20 per cent of the old watercourse of the Ermita stream, starts from this deposit to the surroundings of the Eresma river, where the irrigation area ends. The trace of the ditch goes down 30 m of height difference and 0.28% slope through a 10.7 km distance. This ditch has 54 stopping devices with the aim of facilitating the infiltration and decants processes. A parallel service road is being built 4 m away from the ditch that gets elevated 50 cm above the terrain height and will serve as a barrier against possible overflows.

The infiltration surface is over 33,300 m^2. The maximum directed flow of water surplus from the Voltoya river is about 0,5 m^3/s between November and April, although it may decrease or even get stopped according to river flow and characteristics of the considered hydrological year. The sheet of water inside the ditch will range between 50 and 100 cm in height but it will tend to be 80 cm. Four infiltration ponds were built in 2004, for a total surface of over 200m^2. Net infiltration is estimated at 1.5 Mm^3 per cycle.

New water resources have both triggered an increase in irrigated area and a new disponibility environmental flow. Without these resources, wetlands would have probably been lost completely. Over the 2005 summer, remedial measures have been applied to recover La Iglesia and Las Eras wetlands by diverting water from the artificial recharge scheme to the western sector of the aquifer. In view of new irrigation developments, MAR has turned into the only viable alternative to reduce environmental impacts and to reach more sustainable agricultural practices. MAR allows to monitor relevant parameters (controllable features) as to how the 'pressure state' evolves with time. This in turn facilitates setting social and environmental target states. However, give the complexity of the system, evaluation must be open to uncontrolled variables.
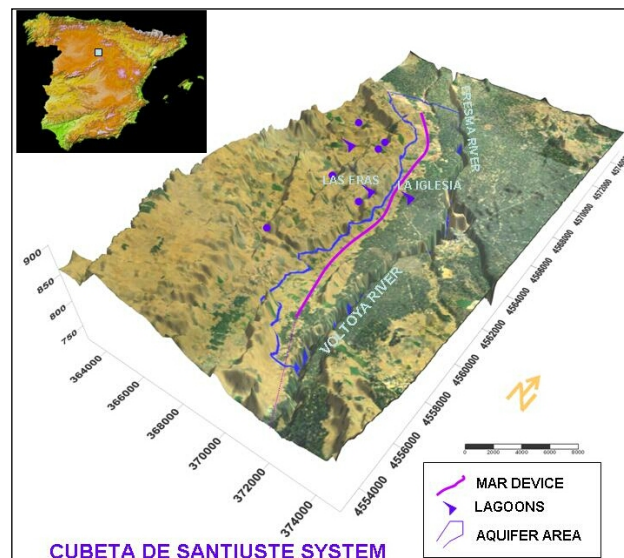
Figure 1. Representation of the Cubeta de Santiuste aquifer showing MAR device and the most significant lagoons

METHOD

The proposed system is consistent with the rules of the Spanish and European legislation for the elaboration of Environmental Impact Assessments (EsIA), as per the Law RD. 6/2001, and is based on environmental engineering approaches. The design of environmental indicators is based on the PSR Framework (Friends and Raport, 1979), which distinguishes between Pressure, State and Response indicators. The multi-criteria evaluation polygon (or variogram) that we present is original and based on the experimental control of a wetlands system along five years

of development of the first author's PhD Thesis (Fernández Escalante, 2005). A more detailed explanation may be found in Fernández Escalante et al. 2005.

RESULTS: DESIGN OF ENVIRONMENTAL INDICATORS

Pressure indicators

A set of ten pressure indicators has been designed, for the purpose of controlling the direct and indirect impacts of human activities. These are evaluated consequently by the importance and the intensity of those human activities that can generate environmental stresses or impacts. Their determination is carried out in the field (direct measures). To reach the 'state of pressure' and to get an accurate impact assessment, a system of ranges-weights must be applied so as to homogenize the intensity and scale of the different concurrent environmental impacts. This is carried out by means of correction factors added to those 'less expressive' indicators. Accumulative impacts and synergies are introduced to the system by means of assessment factors. For example the impact of a three-year drought is more significant than twice the stress of a two-year one. This *modus operandi* is similar for the remaining groups of PSR Framework indicators. The system is to be applies to each wetland or key element under restoration. An assessment should be carried out at least once a year, following adequate data-collection campaigns. This allows for an evaluation of how the restoration process is affecting wetlands. Dealing with all individual wetlands in a joint manner allows for an assessment of the aquifer situation.

The following list outlines the main pressure indicators. Their detailed definition, ranges-weights values applied and corrective factors for accumulative impacts and synergies (e.g. inter-year climatic variations) should be consulted in the bibliographical references (Fernández Escalante, 2005; Fernández Escalante el al, 2005). The result is a numerical value whose variability allows evaluating progress over time. This may provide information on environmental aspects (whether environmental impacts have been reduced between two consecutive measurements) or social and economic conditions (whether agricultural production has increased). The evolution of the total 'state of pressure' corresponds to aggregating the evolution of the first two indicator types (state and pressure) and the third one (response) once anthropic action has begun.

List 1: Main pressure indicator
1. Aquifer overexploitation by irrigation,
2. Total organic carbon (TOC) in the waters of AR,
3. Modernization and improvements of AR and irrigation devices.
4. Efficiency in water use,
5. Socioeconomic evaluation,
6. Political background of the activity,
7. Proximity to the device of AR (*Tt*),
8. Area of influence (m),
9. Presence of groundwater and temperature- dependent ecosystems,
10. Relationship of wetlands with other aquifers, springs, wetlands, etc.

State indicators

These describe the quality of the environment and of the natural resources associated to processes of socioeconomic exploitation. In addition, these also reflect the changes caused in the environment. Their evaluation is based on analytic methods which allow for quantification of 'performance response' (MIMAM, 1997).

Within the PSR system, these are the most adequate indicators for the application of specific techniques to control parameter variations over time. Results suggest that the most viable option is factorial analysis by means of a correlation matrix such that those correlated are independent and not redundant and that duplicities may be omitted (Rey Benayas et al., 2003).

List 2 summarizes the main ten state indicators. Five of those are of general application and five designed *ex profeso*. In accordance with the previous operating system, the scales of ranges–weights and corrective factors would be applied.

List 2: State indicator

1. Aquifer contaminated by nitrates (A1[1] MIMAM, 1997),
2. Rivers and wetlands with good quality according to the biotic index 'bmwp' (A3 MIMAM, 1997),
3. General quality index (ICG, A4 MIMAM, 1997),
4. Characterization of the vulnerability of diffuse contamination (CRIPTAS),
5. Aquifers affected by continental saline intrusion (A2 MIMAM, 1997 modified),
6. Groundwater salinization,
7. Turbidity and total of dissolved solids (TSD) in the water of AR,
8. Groundwater table in observation wells,
9. Difference of average table between the phreatic level and AR level in each 'hydroenvironment unit' (Fernández Escalante, 2005),
10. Soil clay percentage. Sample taken from 15 cm deep (AR channel clogging indicator).

Once completed the 'characterization template' which includes most of the precise data to apply the system of indicators, the environmental stress evaluation in the intervention area (space) referred to each cycle of artificial recharge (time) is carried out. As a result, a specific EIA index can be obtained, taking into account that an arithmetic average is applied to all those that span several-years. A particularly significant example is the indicator adopted for clog control (channel clogging is considered one of the biggest impacts for the implementation of AR). The study of the cake with binocular glass after the first year of artificial recharge shows that the initial sandy texture is strongly altered. Fines are located around the surface of the grains. Seeds and grains of pollen, indicial tests of the influence of the organic activity in the aquifer, are also observed. The presence of particles blackened by oxide suggests possible influences of the dissolved oxygen concentrations in the system. Some clogging indicators have been applied previously, like the Index of Failure in Membrane (MFI), whose initial valuation for the study area in its first cycle of artificial recharge was from 25 to 30 s/l[2] (MFI units).

As a consequence of the huge technical difficulty to control the stress effect of several impacts (physical, biological and indirectly chemical clogging for surface systems of artificial recharge influenced by total organic carbon (TOC); the percentage of initial clay; infiltration surface; time of artificial recharge; infiltrated volume, and so on) we have proposed a specific indictor for the study area and similar scenarios that corresponds to variations in the percentage of fines in the soil and its evolution along each artificial recharge year/cycle. This evaluation is carried out by means of a granulometric analysis, using a 200 ASTM sieve. Monthly samples are taken at the surface level and at a depth of 20 cm in the different stations of control of the AR channel. The percentage of fines (by weight) with regard to the initial value indicates the stress change. The difference of weights is subjected to a corrective factor to enlarge the value due to its great importance. In principle, we propose to divide the laboratory value by the number of days of artificial recharge from the beginning of the cycle until the date of sampling. This value is multiplied by 100, to get an assessment level in the order of the remaining indicators. The average value is used as a global indicator at the end of the cycle of artificial recharge. For the test site this has varied from 7 to 17 units in the first cycle. Table 1

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1. A1, A2, A3 etc are codes given by MIMAM, 1997 to some indicators already designed by themselves.

shows the application of this system to La Iglesia lagoon. In August 2003, the 'pressure state' was estimated at 2035. By August 2005, MAR had resulted in a significant decrease of pressure (1835) despite severe drought conditions.

Response indicators

These indicate the level of social and political involvement in environmental matters and resources, corresponding to the policies and actions that the economic and environmental agents carry out to protect the environment. Proposed indicators have provisional character, and are subject to checking the response of the performances in the long term. List 3 presents a group of fifteen response indicators and their calculation method, according to PSR framework (Friends and Raport, 1979), ranges and weights.

List 3: Response indicators
1. Evolution of the dimensions of the AR device,
2. Sand deposit of channels, dams, natural or artificial riverbeds, etc.,
3. Increase of the erosion in the banks, slopes and influence areas,
4. Bench marks and slopes differences,
5. Changes in hydrogeological parameters,
6. Devices clogging and changes in channels bed permeability,
7. Dissolved oxygen concentration in observation wells,
8. Evolution of groundwater quality due to manures and nitrogen fertilizers uses,
9. Total carbonate concentration in AR water,
10. Crops affection inside the influence area,
11. Native vegetation affection inside the influence area,
12. Changes of ecological conditions in wetlands,
13. Variations in the water level in wetlands and specially those below the ground surface,
14. Reduction of undue consumption of water,
15. Nutrients balance in irrigation areas related to AR activities.

DESIGN OF A POLYGON OF MULTI-CRITERIA SPECIFIC EVALUATION

A multi-criteria polygon is a graphic and parametric variogram that reflects in an intuitive, visual, quick and peda-gogic way the evolution of the quantitative, qualitative and environmental aspects during and after each artificial recharge cycle, as well as their incidence in directly related factors (Bascones, 2000; Custodio, 2000; Barbier et al., 1997; Fernández Escalante and Cordero, 2002). Assigning a weight to each parameter with incidence in the state of the evaluated elements allows for the use of a variogram like matrix when relating factors with processes. This is in contrast with the record of initial characterization, which only identifies impacts. The evaluation of environmental impacts and their control is presented in two simultaneous ways: numerical and graphical:

*   *Numerical*
    This is the result of applying system ranges-weights and corrective factors. It allows for assigning a global value to all the concurrent impacts in each cycle or period of measurement. It can be used as a global or integrated stress evaluation. The differences between numerical calculations can be also used as an achievement of the target indicator.

*   *Graphical*
    This consists on a variogram where a given set of fields, (whose stress evaluation values has been determined based on empiric experiences or technical approaches), is encircled by an 'envelope' line. The chosen graphical representation for the polygon of evaluation multi-criteria has been a histogram of horizontal bars. Each bar

includes from 4 to 6 fields corresponding to the final assessment level of each impact. The enveloping line works as an affection degree indicator for each time.

The proposed methodology stages are:

1. Pick the best indicators for each specific situation.
2. Hierarchization of the indicators according to their magnitude.
3. Range intervals are established in an arbitrary way, based on technical concepts and an empirical base. Therefore, these can be modified in other scenarios different to the Coca-Olmedo wetlands Complex.
4. Attribution of the weight to each indicator range. In different scenarios, this attribution is carried out by the people in charge of evaluation.
5. Attribution of an evaluation or correction factor to each indicator in function of their index of incidence, i.e. the relative importance of each indicator for the proposed objective. In the practice, the corrective factor is a multiplier for the ranges-weights result (see Fernández Escalante et al. 2005).
6. Multiplication of the ranges of each indicator by their weight, application of a function factor (especially for accumulative impacts and synergies) and calculation of all the variables.
7. Attribution of a category or level of final evaluation to each numerical result of the proposed qualitative calculation of ranges-weights. Six affection degrees are considered: worthless (1), fair (2), moderate (3), intermediate (4), severe (5) and high (6).

With these results, graphical representation can be carried out in three successive stages:

1. Shade the categories until the maximum evaluation level. The scales can be redefined for each category in other scenarios or corrected for the case of defining the 'state of pressure' for each specific setting.
2. Layout of the enveloping line encircling the shady fields (or affection degrees) of the multi-criteria evaluation polygon for each scenario.
3. Study of the variation of this final product along the time during the control program. Relative variations allow to appreciate if the stress is increasing (migration of any field of any impact to the right or elevation of the final numerical value) or the corrective measures work appropriately (the enveloping line has been displaced to the left in any field or there is a decrease of the global value of the multi-criteria evaluation).

An example of the final design of the multi-criteria evaluation polygon is presented in the Table 1 a) and b). Although this initially takes pressure and state indicators into account, it leaves space to for aggregation of the response indicators.

The final result is a numerical value that corresponds to the evolutionary state of the activity, wetland or key element, and an enveloping line or 'polyline' encircling fields shadowed in the diagram (fields included in each affection degree). This 'polyline' facilitates its use as indicator by comparison between different years/cycles. The final numeric value is lower and, consequently, MAR is achieving its objectives in wetlands recovering. Environmental impacts have been reduced particularly for the first and second state indicators and the ninth and tenth response indicators.

CONCLUSIONS

The application of the proposed methodology can be used to monitor the evolution of the different parameters, allowing to correct some environmental adverse impacts and to systematically improve the efficiency of technical operations. The evolution of the proposed systems, specially the variogram, shows that it is necessary to change the location of certain AR devices, in order to improve the maintenance program and to apply specific SAT technologies. The system of indicators constitutes an important novelty for the control of the restoration activities, whose application may allow operators to know whether their performance is appropriate. The system of environmental

Table 1 a) and b). Variograms of La Iglesia wetland. Villagonzalo de Coca, Segovia, Spain.
First: 2003, August; Second: 2005, August. The encircling line is marked in purple colour.
The comparison between both polylines determines the stress difference. The final numeric value
is lower (2035>>1835) and, consequently, MAR is achieving its objectives in wetlands recovering.

APPLICATION TO "LA IGLESIA" WETLAND EXAMPLE. FEBRUARY OF 2004

STATE/PRESSURE INDICATORS	<	ASSESMENT LEVEL				>
1) Rivers and wetlands with good quality according with the biotic index (BMWP).					100	
2) Quality General Index (ICG).					100	
3) Aquifers contaminated by nitrates.		112				
4) Characterization of the vulnerability to diffuse contamination (CRIPTAS).			151			
5) Salinization of aquifers by continental saline intrusion.			458			
6) Groundwater salinization.			458			
7) Assesment of turbidence and TDS in AR waters.	< 25					
8) Level of water in observation wells.	3,2					
9) Difference of levels between water table and the level of water for artificial recharge.			2,8			
10) Soil fines percentage. Initial index of clogging.	14%					
1) Aquifer overexploitation by irrigation.		50****				
2) Nutrient balance in AR waters.		50****				
3) Modernization and improvements of the devices.		50****				
4) Efficiency in water use.		50****				
5) Socioeconomic evaluation.		50****				
6) Political backdrop of the activity.		50****				
7) Proximity to the AR facilities.		50****				
8) Area of influence.		50****				
9) Presence of hydrodependences or termodependences ecosystems.				100		TOTAL
10) Relationship of the wetlands with other wetlands, springs, lagoons, etc.				100		2035

APPLICATION TO "LAS ERAS" WETLAND EXAMPLE. FEBRUARY OF 2004

STATE/PRESSURE INDICATORS	<	ASSESMENT LEVEL				>
1) Rivers and wetlands with good quality according with the biotic index (BMWP).		50****				
2) Quality General Index (ICG).		50****				
3) Aquifers contaminated by nitrates.		112				
4) Characterization of the vulnerability to diffuse contamination (CRIPTAS).			151			
5) Salinization of aquifers by continental saline intrusion.			458			
6) Groundwater salinization.			458			
7) Assesment of turbidence and TDS in AR waters.	< 25					
8) Level of water in observation wells.	3,2					
9) Difference of levels between water table and the level of water for artificial recharge.			2,8			
10) Soil fines percentage. Initial index of clogging.	14%					
1) Aquifer overexploitation by irrigation.		50****				
2) Nutrient balance in AR waters.		50****				
3) Modernization and improvements of the devices.		50****				
4) Efficiency in water use.		50****				
5) Socioeconomic evaluation.		50****				
6) Political backdrop of the activity.		50****				
7) Proximity to to the AR facilities.		50****				
8) Area of influence.		50****				
9) Presence of hydrodependences or termodependences ecosystems.		50				TOTAL
10) Relationship of the wetlands with other wetlands, springs, lagoons, etc.		50				1835

indicators seeks to grant to each impact its fair evaluation. This is achieved by applying a system of ranges-weights elaborated on empirical experiences during a five years period. Accumulative impacts and synergies are subject to correction factors. Up to six fields or affection degrees (each assigned a specific colour) are described in this regard. These environmental indicators may enhance aquifer monitoring, control of the magnitude and scale of the environmental impacts that operate on artificial recharge systems and their quantitative influence on associated ecosystems. The system of ranges-weights has a matrix character and requires a continuous update as new information becomes available and further field campaigns yield new data. The variogram works in turn as an indicator of achievement of the objective (where the objective is to get an operative system of artificial recharge under conditions of minimum environmental impact). The variogram can be obtained by applying a macro-based computer program that allows for a regular update and to evaluate the type of appropriate performance for the correct administration of the system. The macro is available in Internet for its general access and employment on the part of the interested technicians. The one designed by the authors can be obtained from www.uax.es/publicaciones/tecnologia.htm.

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Analysis of feasibility and effects of artificial recharge in some aquifers. Modelling of integrated management in the Medio Vinalopó basin (Alicante, Spain)

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Abstract

The main objective of the study is the optimization of water management in the Medio Vinalopó basin. A management simulation code (AQUATOOL-SIMGES) has been applied to different management scenarios. It also tries to test the effects on aquifers exploitation rates and the feasibility and impacts of artificial recharge with external excess water from the Jucar system, as a possible alternative.

The most probable scenario corresponding to the National Hydrologic Plan includes a transfer of 80 Mm^3/y from river Jucar distributed over six months, and substitution of a part of the pumped groundwater with transferred surface water. With this scheme, groundwater exploitation would be highly reduced. However, overdraft would remain in several aquifers. The proposed scheme could not guarantee proper summertime supply, while winter excesses would be lost.

But overdraft could be corrected in some aquifers using those winter surpluses for artificial recharge and keeping in operation current urban supply wells. Moreover safe yields would be increased, which would allow a higher pumping rate than in other scenarios. Aquifers could then be exploited even during summertime, when there is no water transfer to meet urban demands. It would mean an increase of guaranteed supply and a reduction of deficits with regard to the planned scenario.

Keywords

Artificial recharge; conjunctive use; management; modelling; Spain.

INTRODUCTION

The Vinalopó Valley (South-eastern Spain) has been suffering from a water deficit since the development of new irrigated farmlands in the second half of the 20th century. The possibility of exploiting new groundwater resources made farming improve. But it also meant an increase of the irrigated area and a higher water demand. Moreover, urban and industrial demands increased too. As a consequence, local aquifers started being overexploited and groundwater quality also began to deteriorate (Rico, 1994).

A small part of the problem has been solved by transferring surface water from a nearby basin. It is used for urban supply of some municipalities in Medio Vinalopó, Bajo Vinalopó and Campo de Alicante.

Treated wastewater reuse for irrigation has also increased, though the volume currently used is still very far from meeting farming demands. There are still high possibilities of reuse in Medio Vinalopó and Alacantí, with surpluses available from the Alicante treatment plants.

However, those solutions have met just some of the demands on the system. Aquifers are still the main water source in the region and they keep being overexploited.

In this context, the National Hydrologic Plan has proposed to import exceeding water from the Júcar system as a solution. The Júcar Plan settles a maximum of 80 Mm^3/y to be transferred. It could be increased up to 120 Mm^3/y in the future, after improving the irrigation system of the source basin.

The water to be imported will substitute part of the groundwater currently pumped. Some production wells should be abandoned. However, some other wells will be kept working to support the new supply system.

In this framework, the Province Authority and the IGME have carried out a project to optimize water management in the Medio Vinalopó system. This involves a management model with different alternative schemes. The model tests the effects on the aquifers exploitation rates and on guaranteed supplies, and also the effects of artificial recharge in some aquifers. It is the first step to analyse the feasibility and management of such techniques as a possible solution for local aquifers recovery.

APPLIED METHOD

The AQUATOOL code (Andreu *et al*. 1992) has been used to set up the model. This software consists of a pre- and post-processing tool for the code SIMGES, which is also included. SIMGES is a modelling code for integrated management simulation of hydrologic basins (López-Geta et al.. 2001).

It is able to simulate complex hydrologic systems, including reservoirs, aquifers, runoff, recharge (natural and artificial), exploitation and transport facilities, demands, etc. An order of priority can be defined to meet demands, and also operation rules can be established for reservoirs.

Different water management schemes can be optimised with this tool. The guaranteed supply which is obtained then shows the best alternative. Vice versa, the capacity of reservoirs, pipes and pumps can be calculated for pre-determined demands and guaranteed supply.

The time step for simulation is one month, and runoff is calculated applying mass balance conservation. On the other hand, aquifers can be simulated with different models: from the most simple (tank type) to the most complex (distributed parameters), with intermediate types like aggregated parameters models (single or multiple cell).

INTEGRATED MANAGEMENT MODEL

The volume of water currently used in the Medio Vinalopó system reaches 66.3 Mm^3/y. It supports the economic development of the region: agricultural, urban and industrial. The main demands are for farming with 68.2 Mm^3/y, which means 80% of the total. They are followed by the urban demand with 16.6 Mm^3/y which represents the remaining 20%.

Demand is currently met with groundwater from the Medio Vinalopó and Alto Vinalopó aquifers. They provide 90% of the total supply. The higher volumes come from the Serral-Salinas aquifer with 18.75 Mm^3/y, Jumilla-Villena with 12.23 Mm^3/y, and Solana with 9.36 Mm^3/y.

There is also a transfer of 1 Mm^3/y from the nearby Segura basin. It meets the urban demand of two municipalities.

Table 1. Supply sources in the Medio Vinalopó system (m³)

| Demands | Supply sources | | | |
	Aquifers	Reuse	Transfers	Total
Urban	15,043,946	0	1,000,000	16,043,946
Farming	44,882,111	5,384,074	0	50,266,186
Total	59,926,057	5,384,074	1,000,000	66,310,132
Percentage	90.4	8.1	1.5	100
Percentage	90.4	8.1	1.5	100

Surface water is scarce in the basin. It is also irregular depending on the rainfall pattern. Furthermore its quality is very poor due to wastewater disposal, which often constitutes the whole stream flow. It also has a high degree of salinization due to dissolution of salts from the river bed. These features make runoff useless, therefore no regulation infrastructures have been built.

Some farmers use treated wastewater for irrigation of grapes and other crops. It comes from some regional treatment plants, reaching an annual volume of 5.38 Mm³. This means 8% of the total supply of the system, and 11% of the satisfied agricultural demand. Wastewater reuse may still increase.

A total of 16 aquifers have been considered in the hydrologic system. They have been simulated by simple models provided by SIMGES. The available data has determined the simulation period, which is from 1995 to 2001.

Table 2 shows the ratio of pumping over average natural recharge for each aquifer, which means aquifer exploitation rates. We can conclude that:

Table 2. Current average exploitation rates

Aquifer	Pumping over natural recharge
Serral-Salinas	571.3%
Madara	110.6%
Umbría	229.0%
Argallet	123.2%
Sª Crevillente	601.3%
Jumilla-Villena	153.3%
Solana	212.6%
Sª Mariola (Cabranta)	385.0%
Peña Rubia	98.7%
Argueña	191.6%
Caballo-Fraile	19.1%
Sª Cid	114.4%
Tibi	105.6%
Ventós-Castellar	103.2%
TOTAL	205.8%

The main aquifers of the system are heavily overexploited. Sª de Crevillente, Serral-Salinas, Solana and Umbría can be highlighted, with exploitation rates of 601%, 571%, 212% y 229% respectively.

The remainder of the considered aquifers also show pumping rates higher than natural recharge or near balance.

Seven different scenarios have been tested, including the current management scheme and six alternatives (Gómez et al. 2004). Table 3 shows pumping calculated in all cases.

The results of the current scheme simulation (SIM1) show an average pumping of 123.3 Mm^3/y in the whole system. 54 % of them (66.9 Mm^3/y) goes to Medio Vinalopó demands, which are estimated at 84.8 Mm^3/yy. The

Table 3. Pumping in the main aquifers for different scenarios

Aquifer	Pumping (Mm^3/y)						
	SIM1	SIM2	SIM3	SIM4	SIM5	SIM6	SIM7
Serral-Salinas	19.79	12.77	12.09	12.88	13.18	12.77	1.76
Jumilla-Villena	26.05	14.44	12.62	13.18	17.17	13.14	10.61
Solana	45.94	22.43	22.43	23.92	24.58	15.05	20.83
Sª Crevillente	9.02	6.32	6.05	7.19	7.20	7.19	1.51
Others	22.5	12.27	12.12	13.93	13.77	13.5	7.42
TOTAL	123.30	68.23	65.31	71.10	75.90	61.65	42.13

average annual deficit is 16.7 Mm^3/y. The rest of the pumping goes to Alto Vinalopó, Bajo Vinalopó and Alacantí areas.

Scenarios 2 and 3 (SIM2 y SIM3) include a 80 Mm^3/y transfer from the Jucar basin, distributed over 10 months (except July and August). Overdraft is highly reduced with these two schemes, but more with SIM3 which involves an increase of wastewater reuse. However, the Sierra de Crevillente, Serral-Salinas and Umbría aquifers would remain overexploited. Guaranteed supplies would also clearly be improve.

Table 4. Average exploitation rates for different scenarios

Aquifer	Pumping over natural recharge						
	SIM1	SIM2	SIM3	SIM4	SIM5	SIM6	SIM7
Serral-Salinas	571.3%	369.8%	350.1%	372.9%	98.2%	369.8%	51.1%
Jumilla-Villena	153.3%	85.0%	74.2%	77.6%	68.0%	77.3%	62.5%
Solana	212.6%	103.8%	103.8%	110.7%	99.0%	69.7%	96.4%
Sª Crevillente	601.3%	421.5%	403.6%	479.5%	480.5%	479.5%	101.1%
TOTAL	205.8%	113.9%	109.0%	118.7%	93.3%	102.9%	70.3%

The scheme proposed by the Jucar Plan (SIM4), includes a 80 Mm^3/y transfer distributed over six months (October to March), and involves substitution of part of the groundwater pumped with transferred water. This scenario also reduces current exploitation, but less than alternatives 2 and 3. Overdraft would remain in the Sierra de Crevillente, Serral-Salinas and Umbría aquifers, as in the previous cases. Whilst agricultural guaranteed supplies would clearly improve, urban ones would deteriorate. This is due to the obligation to close some wells and to the low regulation capacity of the planned infrastructure. It cannot properly guarantee summer supplies, while winter surpluses would be lost.

Instead of that scheme, we propose to use winter excesses for artificial recharge in some aquifers (Serral-Salinas, Solana and Jumilla-Villena), and to keep current wells in operation. This option contitutes scenario 5 (SIM5). It would eliminate overdraft in those three aquifers. But it would also increase aquifer yields and make higher pumping rates possible. That means better guarantees and lower deficit than the planned scenario.

This alternative is similar to SIM4, including the Júcar-Vinalopó transfer (80 Mm^3/y) and its regulation reservoir, with the same time distribution. It also includes an increase of treated wastewater reuse up to 20% of each farming demand.

The change of this alternative consists of applying artificial recharge to the Solana, Serral-Salinas and Jumilla-Villena aquifers. The water for this purpose would be the part of the transfer that cannot be regulated with the planned scheme. Table 5 shows the proposed distribution.

Table 5. Proposed time distribution for artificial recharge.

Aquifer	Volume of recharge (Mm^3)					
	November	December	January	February	March	Total
Serral-Salinas	1.975	2.000	2.000	2.000	2.000	9.975
Jumilla-Villena	0.517	0.726	0.817	3.100	3.100	8.260
Solana	0.162	0	0	2.725	0.335	3.222

The management optimisation obtained for the simulating period (October 1995 to September 2001), means that the average monthly guaranteed supply would reach 91.6%, while average annual guaranteed supply would be 84.9%, with an average annual deficit of 3.227 Mm^3.

The annual guarantee applied by SIMGES means a supply default in two cases:
• if a monthly deficit is higher than 30% of the monthly demand in any month,
• if the annual deficit is higher than 15% the annual demand.

Then the average annual guarantee for urban demand would be 100%, and 66.7% for farming demand. This means an important improvement compared to the current management scheme, both for urban and agricultural demands.

Pumping would decrease in the whole system compared to the current scenario, but it could be kept higher than that of simulations 2, 3 and 4. It would be 75.9 Mm^3/y, which means 61% of current pumping.

In this way the Serral-Salinas and Solana aquifers would have a balanced budget, and the Jumilla-Villena aquifer would begin to recover. However the Sierra de Crevillente aquifer would still be overexploited, decreasing its pumping/recharge rate to 480%. The Umbria aquifer would also maintain a pumping rate of 148%. And the rest of the aquifers would be balanced or recovering.

We have also determined the volume and time distribution of a transfer that would completely meet demands completely and would permit aquifers recovery (SIM7). In that case, the transfer would be regulated in the source basin and distributed all year long. It would reach up to 104.25 Mm^3, which means an increase of 24.25 Mm^3/y with respect to the maximum planned transfer.

On the other hand, groundwater quality is usually better than that of surface water. It would therefore be advisable to keep groundwater as the main urban supply source.

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Developing regulatory controls for stormwater discharge to a potable aquifer in regional South Australia

P. Gorey and H. King

Abstract

Urban stormwater generated within the regional city of Mount Gambier, South Australia, has historically been discharged via drainage wells to an unconfined karstic carbonate aquifer that feeds into the Blue Lake. This practice is now widespread, and the South Australian Environment Protection Authority (SAEPA) has a key role in establishing regulatory controls to ensure that these stormwater discharges do not compromise the beneficial uses of the lake, which are currently as the city's main potable water supply and as a major tourism attraction.

The approach adopted by the SAEPA includes five key components that are considered necessary to achieve protection of the receiving groundwater: 1) a hazard analysis and risk assessment to quantify the degree of risk posed by stormwater discharge, 2) the development of stormwater treatment design standards that are relevant and applicable to the city, 3) the development of legislative tools that enforce stormwater design standards, 4) a communications program and 5) commitment and partnerships with other regional government agencies and scientific organisations.

The experiences of the SAEPA in adopting this approach have been positive, and demonstrate that a balanced consideration can provide for greater environmental protection outcomes.

Keywords

Blue Lake; Mount Gambier; regulation; stormwater.

INTRODUCTION

The regional city of Mount Gambier, located in the lower South East region of South Australia, is established on the northern side of a small maar. The freshwater crater lake is called the Blue Lake and is the only known example in the world of a lake that annually changes its colour in the blue spectrum (Telfer 2000). The lake is therefore a major tourism attraction for the region. The steep walls of the Blue Lake result in a surface catchment area (86 hectares) only slightly larger than the area of the lake itself (60 hectares) (Telfer 2000). This means that the rainfall runoff contribution to the recharge of the Blue Lake is very minimal, with most of the water in the lake being generated by groundwater inflows. Because the water quality is generally good, with total dissolved solids being less than 400 mg L^{-1}, the Blue Lake, in addition to being a tourism attraction, is also the main potable water supply for the city of Mount Gambier.

The protection of the water quality in the Blue Lake is an area of focus for state and local government authorities. There are a variety of land use practices within the vicinity of the lake that need to be controlled in order to protect groundwater quality; in particular, the practice of stormwater discharge to groundwater within the Mount Gambier area has been specifically identified as a potential threat to groundwater quality (Blue Lake Management Committee 2001; Emmett 1985; Emmett & Telfer 1994; Telfer 1994; Waterhouse 1977).

The development of regulatory controls for stormwater management within the Mount Gambier area has been complicated by the environmental setting and the established urban development. The SAEPA, the peak environmental regulator for South Australia, has adopted an approach to establish regulatory controls that deliver a high level of protection for the aquifer and are economically achievable in the region.

Environmental setting

The Blue Lake and surrounding area is located within the Gambier Embayment of the Otway Basin. The main geological units in order from youngest to oldest are Holocene volcanic deposits, Bridgewater Formation (calcareous sands), Gambier Limestone and a confined Dilwyn Formation.

The Gambier Limestone consists of alternating beds of bryozoal limestone and marl overlying the dolomitic Camelback member. Karstic features, which are common in the Gambier Limestone, result in both a primary and a secondary porosity of the aquifer.

The Gambier Limestone is an important unconfined aquifer throughout the region. It is the primary source for the Blue Lake, and the water level in the lake is a reflection of the groundwater level. It is suggested that the groundwater inflow into the lake is dominated (90%) by inflows from the unconfined aquifer (Ramamurthy et al. 1995). The dolomitic Camelback member is the dominant source for flow into the Blue Lake and also the target for drainage wells constructed within Mount Gambier. It is for this reason that protection of the water quality in the unconfined aquifer is considered critical for the protection of water quality in the Blue Lake.

The local geology has made it difficult to accurately define a groundwater 'capture zone' for the Blue Lake. However, the accepted 'Blue Lake Capture Zone', the location of the city of Mount Gambier and the inferred flow paths of groundwater in the unconfined aquifer are illustrated in Figure 1.

Stormwater and wastewater management in Mount Gambier

The topography in Mount Gambier is gently undulating with no defined surface drainage networks. This absence of watercourses, coupled with the presence of well-draining soils and rock units, meant that the early developers in the region adopted the option of constructing drainage wells that intersected the unconfined aquifer as a practical and cost-effective solution for the disposal of stormwater and wastewater. Although the discharge of wastewater is now prohibited, drainage wells are still used for the disposal of stormwater from both public and private land.

In many cases there are only rudimentary, or an absence of any, treatment measures prior to disposal of stormwater to the aquifer. Upgrading of the existing stormwater infrastructure is a major challenge, with over 800 stormwater drainage wells estimated to be located within the Blue Lake Capture Zone. The concentration of stormwater drainage wells within the urbanised areas of the city is illustrated in Figure 1.

Roles of the SAEPA and other government organisations

At the national level the Australian Government establishes baseline conditional targets for all water resources in order to provide for these resources to be maintained for beneficial end uses. This approach occurs under the structure of the National Water Quality Management Strategy (NWQMS), and includes suggested water quality criteria for water resources that are being managed to deliver potable and aquatic ecosystems.

At the state level the majority of regulatory and management functions relating to the protection of the Blue Lake and the surrounding aquifers are undertaken and coordinated from within a variety of separate departments of the South Australian Government. The focus for the SAEPA is the protection of water quality, with other government
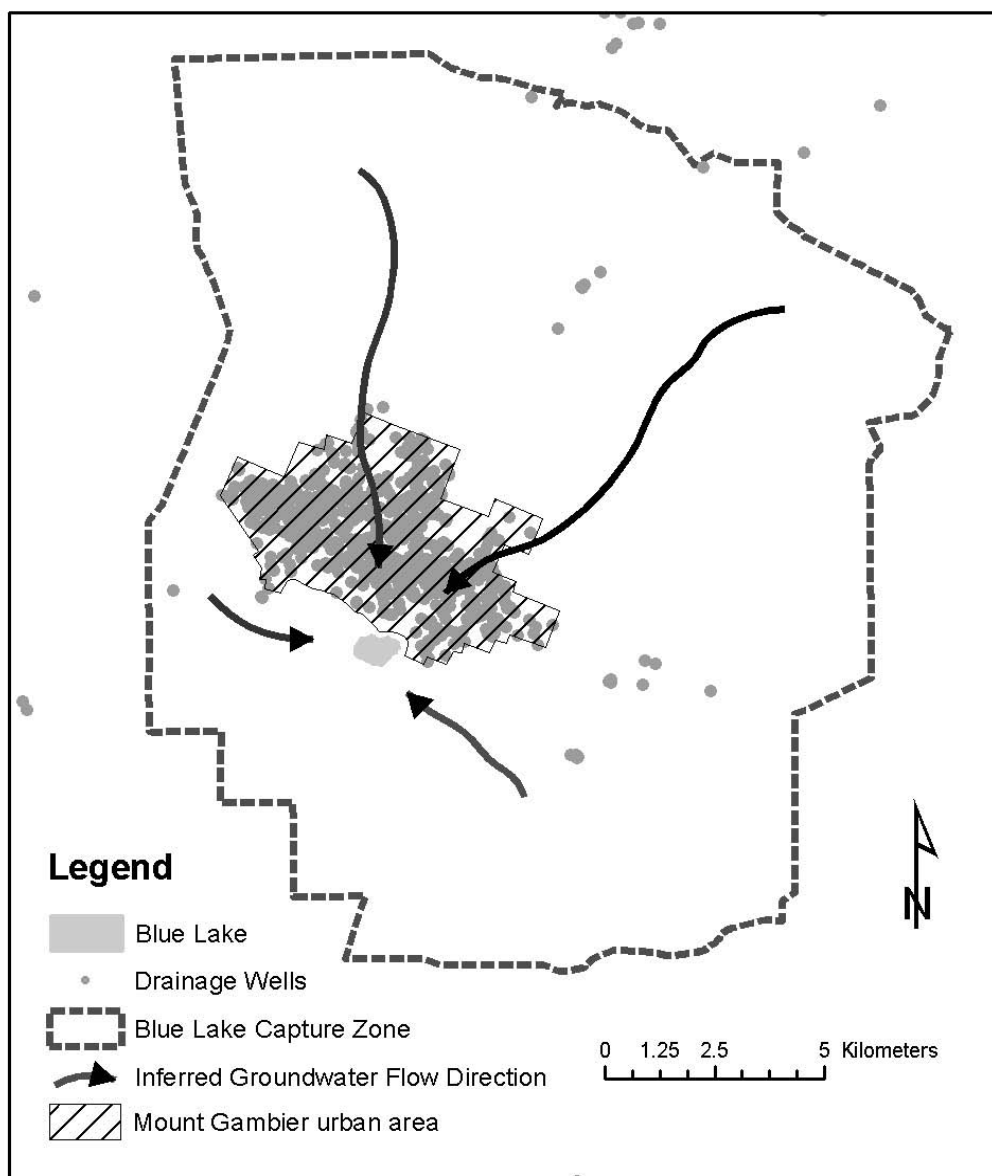
Figure 1: Blue Lake, Mount Gambier urban area and drainage well locations

departments being responsible for the management of sustainable water allocations, the pursuit of water sensitive urban design and the management of the reticulated water supply.

The SAEPA has recently completed a process to allow the principles of the NWQMS to be incorporated into legislation in South Australia. The *Environment Protection (Water Quality) Policy 2003* (WQEPP) is the legislative instrument that has become the pivotal regulatory tool allowing the SAEPA to progress towards improved stormwater management for discharges to the aquifer in Mount Gambier.

Local councils are the third tier of government that have an important role in stormwater management in Mount Gambier, and the Blue Lake Capture Zone encompasses two local council areas. These authorities are integral to the protection and management of the aquifer and the Blue Lake as they are responsible for land use planning and the administration of municipal stormwater management systems.

Challenges and considerations

The development of acceptable water quality criteria for the discharge of stormwater to the aquifer has been a major hurdle for the SAEPA and other regional government organisations. The critical aspect of this problem has been to maintain the water quality of the aquifer under the city to potable standards while recognising that the karstic nature of the aquifer limits the ability to quantify the attenuation capacity of the aquifer, or to be confident in predicting dilution or mixing zones. Adopting a precautionary approach in this situation continually led to the decision that the water quality criteria for stormwater being discharged to the aquifer needed to be equivalent to the target standard for the aquifer, namely potable quality or better. The issue is further complicated as a result of recent work by Telfer (2000) and Turoczy (2002), which indicated that the colour change process in the Blue Lake is significantly influenced by very low concentrations of (non-calcite) suspended particulates in the water column. These outcomes suggest that the target water quality criteria for the Blue Lake may need to be significantly better than drinking water quality in order to maintain the colour change characteristic.

A variety of pollutants including hydrocarbons, pesticides, nutrients and metals have been identified in stormwater within Mount Gambier (Emmett 1985). However, after over 100 years of stormwater discharge to the aquifer, the water quality in the Blue Lake remains good. An increasing trend in nitrate concentrations reflects a similar regional trend of nitrate in groundwater, and is difficult to attribute to stormwater discharge.

An approach was required that would deliver a practical and achievable solution to provide the maximum possible level of protection for the underlying aquifer and the Blue Lake, and yet not compromise the colour change characteristics or the potable end use.

METHODS

A variety of approaches were considered in determining the manner in which stormwater management would be regulated within the Blue Lake Capture Zone. As a result of internal and external deliberations, the SAEPA developed a regulatory approach that is integrated with the approaches being undertaken by other government authorities, and builds upon the non-regulatory mechanisms that are also being adopted.

The approach taken has the following key components, which are considered to have been fundamental to the success of developing the regulatory approach.

A hazard analysis and risk assessment to quantify the degree of risk posed by stormwater discharge. In order to understand the scale of stormwater impact on the receiving aquifer, and therefore on the Blue Lake, it was necessary to ascertain the source, transport and fate of stormwater contaminants, and to quantify the potential risks of stormwater discharge. Opportunistically, the majority of this work was able to be undertaken through an international research project 'Assessing and Improving the Sustainability of Urban Water Resources and Systems'. This project is supported by the European Commission and coordinated within Australia by the Commonwealth Scientific and Industrial Research Organisation. The initial outcome of this assessment has been that, in addition to the dilution of contaminants in the aquifer, stormwater pollution is having a lower than expected impact upon the underground aquifer and the Blue Lake through both an attenuation and/or adsorption capacity within the aquifer matrix and in-lake processes. It follows that although there is the risk of contamination of groundwater from stormwater in Mount Gambier, it is not necessarily justified to assume there is no attenuation capacity within the aquifer matrix. Quantification of this capacity remains to be established.

The development of stormwater treatment design standards that are relevant and applicable to the city. The SAEPA considered that the approach of directly implementing stormwater quality discharge criteria to the Mount Gambier area

would be ineffective as it would provide insufficient guidance to urban planners, landowners and the regulating agencies. The SAEPA therefore developed stormwater treatment guidelines in order to document the minimum design standards for stormwater treatment systems that were considered to be best available technology economically achievable. Importantly, these standards were developed to be appropriate to the environmental setting. The design standards do not prohibit discharge of stormwater to the aquifer, but provide a range of options suitable for different urban developments.

Developing legislative tools that enforce stormwater design standards. The hierarchical structure of the legislative tools available to the SAEPA provide options for enforcing stormwater treatment principles. The SAEPA is currently considering the development of a code of practice that will provide general guidance regarding urban stormwater management, and, importantly, could require compliance with stormwater design standards under the provisions of the new WQEPP. Once developed and linked to the WQEPP this code of practice could be enforced through the issuing of an Environment Protection Order.

A communications program. Although the local community has an affinity with the Blue Lake, experience indicates that many in the community do not understand that stormwater quality is likely to have an impact upon the water quality of the underlying aquifer or the Blue Lake. An ongoing communication program is therefore critical to publicise the need to protect stormwater. Other government partners have already established a successful stormwater pollution prevention program that has substantially delivered the roles of improved community understanding of stormwater management in Mount Gambier.

Commitment and partnerships with other government agencies and scientific organisations. The most significant component of achieving improved regulation in stormwater discharge has been the high level of cooperation that exists at the regional level between state government departments, the regional water management authority, local government authorities and peak scientific research agencies. This cooperative arrangement has enabled coordination of activities, assurance that research is aligned to resource managers' needs, efficiency savings, and a platform for robust discussion leading to mutually acceptable outcomes instead of confrontational dialogue. Further, this cooperation is important in recognising that although regulatory arrangements may be established by the SAEPA, the responsibility for stormwater management does not rest with the SAEPA, and therefore there is a need to ensure a commitment to implementation.

DISCUSSION

The concerns of adverse impacts of stormwater discharge in Mount Gambier have been raised for a number of years, and although there have been many improvements, significant work still needs to be done for there to be confidence in the protection of water quality in the aquifer and the Blue Lake. The situation existed that significant improvements in stormwater management were difficult in the absence of legislative tools and a clear understanding of the risks that stormwater discharge in the city posed. The fundamental decision for the SAEPA has been to adopt stormwater treatment management practices as standards instead of defining stormwater discharge criteria. The benefits of this approach are that landowners are provided with clear guidelines of the measures that need to be established, and that the SAEPA and other organisations have already considered what is reasonable and practical. Based on the scientific research being undertaken, the SAEPA is able to proceed with this approach with confidence.

Although regulatory controls are important, the development of such controls can be difficult and requires the careful balance of both the long- and short-term economic, social and environmental aspects of stormwater management. The SAEPA has demonstrated that in the Mount Gambier area, the adoption of a combination of approaches has been successful in achieving significant gains over a short period of time.

CONCLUSION

The process of implementation of this regulatory approach is still continuing, and it is likely that full implementation will not occur for many years. However, the lessons being learnt already are that a balanced approach is needed and cooperation among state and local government authorities is critical.

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The Streatham groundwater source: an analogue for the development of recharge enhanced groundwater resource management in the London basin

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Abstract

Abstraction from the confined Chalk aquifer of the London Basin began in the late 18th century and increased throughout the 19th century, with the Streatham groundwater source beginning abstraction in 1882. Abstraction throughout the London Basin increased until the 1940s causing over-exploitation and declining groundwater levels, the consequence at Streatham was a progressive reduction in yield until it ceased operation in 1954. Groundwater levels continued to decline until the mid 1960s, but subsequently recovered as abstraction reduced. It was in the period of declining groundwater levels that much of London's underground infrastructure developed. The later rise in groundwater levels produced a risk of structural damage and flooding, and since the late 1990s mitigation has entailed a progressive increase in abstraction; a new borehole at Streatham was one of several constructed across London for this purpose.

In the 21st century artificial recharge testing at Streatham has demonstrated significant borehole injection rates, and the site is at the forefront in the development of a South London Artificial Recharge Scheme. The challenge is to define an operating strategy for storing injected water to ensure that enhanced resources are available to meet seasonal peak and drought demands, while ensuring underground infrastructure is not impacted and abstraction remains sustainable.

Keywords

Artificial recharge, chalk, groundwater resource management, London basin, SLARS.

INTRODUCTION

Currently, Thames Water supplies up to a maximum of 2,600 million litres of water per day (Ml/d) to London, of which about 650 Ml/d is groundwater abstracted from the Chalk aquifer. In the future, however, demand for water will increase; in east London alone a population increase of around 800,000 is expected by 2016. Although a twin track approach of demand management and water resource development is being followed to maintain a supply-demand balance, groundwater abstraction in the London Basin is close to its sustainable limit. As a result, Thames Water is investigating several groundwater options, such as use of deeper aquifers, utilising poorer quality brackish groundwater and enhanced use of artificial recharge in the Chalk. In investigating these options it is informative to consider the historical development of groundwater resources in the London Basin. For the development of a South London Artificial Recharge Scheme (SLARS), the Streatham groundwater source can be used as an analogue for both the historic resource development in the London Basin, as well as the future development of recharge enhanced groundwater resource management.

HYDROGEOLOGY OF THE LONDON BASIN

The London Basin beneath metropolitan London is a syncline, dissected by a network of faults and fractures,

within which the Chalk forms the principal aquifer (Figure 1). The Chalk aquifer crops out in the hills of the Chilterns and the North Downs, to the north and south respectively, forming the main areas of natural recharge, with groundwater flow converging generally in the confined Chalk under central London. Within the Chalk, groundwater flow is controlled by its fracture-matrix, dual porosity characteristics. Beneath much of London the Chalk is overlain by the Basal Sands, which owing to their leaky hydraulic connection, form an important aquifer system, This Chalk-Basal Sands aquifer is confined by the Lambeth Group and the London Clay, consisting predominantly of clay units. Overlying these clays, a sand, gravel and clay sequence deposited by the River Thames and its tributaries forms a minor aquifer that is hydraulically separated from the Chalk over most of the Basin. However, the confined Chalk aquifer beneath London is more complicated than described above:

- Higher transmissivity aligned with N-S oriented valleys but also along the predominant NE-SW fracture orientation, potentially resulting in zones of preferential flow;
- Significantly lower transmissivity perpendicular to the predominant NE-SW fracture orientation, i.e. across major fault and fold structures, that influence hydraulic gradients; and
- Erosion to form 'windows' through the confining layer beneath the tidal R. Thames, and some of its tributaries in south east London, resulting in aquifer-river hydraulic connection.

Streatham is located in south west London where the Chalk is about 75 m below ground level, and is overlain by 10 m of Basal Sands and then 65 m of the Lambeth Group and London Clay. The Streatham groundwater source therefore abstracts from the confined Chalk aquifer. It is located in a zone of fracture enhanced transmissivity, aligned roughly N-S along the Wandle river valley, extending from the Chalk outcrop south of Streatham to the River Thames.
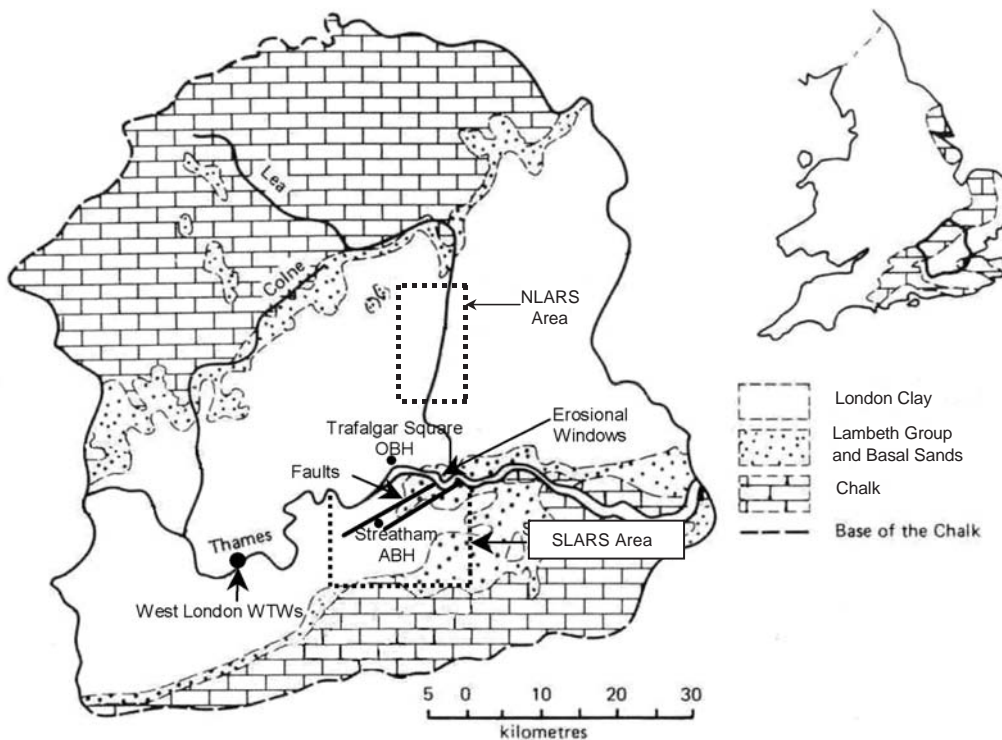

Figure 1. Hydrogeology of the London Basin

HISTORY OF GROUNDWATER DEVELOPMENT

Abstraction began from the confined Chalk aquifer of the London Basin in the late 18th century, with many of the wells and boreholes initially being artesian. Abstraction increased steadily throughout the Industrial Revolution into

the 19th century and groundwater levels declined, requiring deeper wells and deeper, more powerful pumps. It was in 1882 that the Southwark and Vauxhall Water Company commenced construction of the original Streatham groundwater source, producing around 5.5–6 Ml/d by 1884 from the Chalk. The Streatham well is about 2.3 m in diameter, lined with brickwork and cast iron segments to the top of the Chalk, extending through the Chalk as an open hole of 0.75 m diameter to a depth of 220 m. In around 1884 a borehole was drilled through the base of the well in an attempt to enhance abstraction, penetrating to a depth of 385 m and reaching the Devonian basement, but it did not encounter a significant additional aquifer.

Widespread abstraction from the Chalk continued to increase until the 1940s resulting in over-exploitation of the aquifer and declining ground water levels. This pattern is well illustrated by the groundwater hydrograph from the Trafalgar Square observation borehole in the centre of the London Basin (Figure 2).

The consequence of the declining groundwater levels was a reduction in the yield of many wells and boreholes. At Streatham a progressive reduc-

Figure 2. Chalk groundwater level evolution at Trafalgar Square

tion in base load (i.e. continuous abstraction) yield occurred, from 6 Ml/d to around 3.5 Ml/d, followed by a change to summer, abstraction only by the time it ceased operation in 1954. Test pumping in 1962 demonstrated a yield of just less than 3 Ml/d. The general decline in groundwater abstraction resulting from declining groundwater levels and yields was also accompanied by a decline in industrial and manufacturing operations within metropolitan London after the 1939–45 war. Despite this decline in abstraction, Chalk groundwater levels throughout the London Basin continued to decline until around 1965, reaching a pseudo steady state condition at around that time (Figure 2). Groundwater levels were drawn down to almost 90 m below sea level in the centre of the London Basin, and to around 20 m below sea level at Streatham. Subsequently, groundwater levels have recovered dramatically, rising at maximum rates of 3 metres per year.
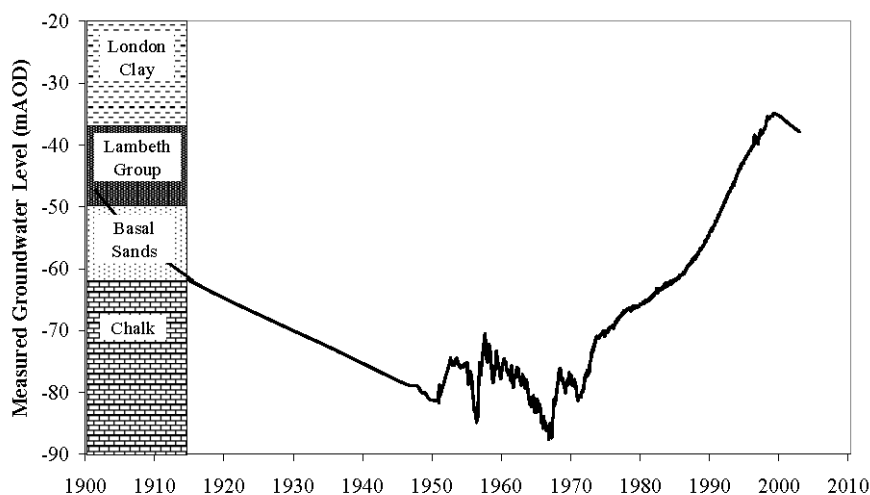
The threat from rising groundwater levels

It was in the period of declining groundwater levels that a large part of London's underground infrastructure was developed, including tall buildings with deep basements and underground railways. The consequence was that the subsequent rise in groundwater levels (Figure 2) posed a significant risk of structural damage, via changed geo-technical properties of founding clays and settlement, as well as the risk of direct flooding (CIRIA, 1989). The risk was such that a mitigation strategy of increased abstraction was promoted by GARDIT (General Aquifer Research, Development and Investigation Team), led by Thames Water, London Underground and the Environment Agency. Since the late 1990s implementation of the GARDIT Strategy has entailed drilling new boreholes, including a new borehole drilled at the Streatham site in 1993, and the re-commissioning of existing, disused wells across London. The Streatham borehole penetrates 128 m below ground level with an open hole, producing section of 0.72 m diameter in the Chalk. In 1999, after 45 years of disuse, the Streatham site began again to produce water for public supply, and is now licensed to abstract up to a peak of 9 Ml/d with an equivalent annual average of 5 Ml/d. Originally a contributor to Chalk groundwater over-exploitation and declining groundwater levels, Streatham is now active in controlling the rise of groundwater levels.

Current trends in groundwater level

As abstraction has increased as part of the GARDIT Strategy the rate of groundwater level rise has decreased from its maximum of 3 m/year to around 1 m/year, but in the centre of the London Basin groundwater levels are now static or beginning to fall. Notably however, in south west London groundwater levels are falling at rates of around 4 m/year owing to abstraction from Streatham and other groundwater sources commissioned as a contribution to the GARDIT Strategy. As a result of this, the possibility of a new, more dynamic role for sources such as Streatham has been identified. The concept involves managing groundwater storage to ensure that maximum groundwater levels do not impact subsurface infrastructure, and minimum levels define a water resource limit and/or a level at which other abstractors are derogated. Artificial recharge, using dual abstraction-injection boreholes, is fundamental in the development of this groundwater resource management concept.

ARTIFICIAL RECHARGE IN THE LONDON BASIN

The artificial recharge of the Chalk in the London Basin is not new, with Thames Water and its predecessor organisations having much experience in north London. This began with an initial trial in the 1890s, continued with concerted trials in the 1950s to 1970s, and resulted in the full commissioning of the North London Artificial Recharge Scheme (NLARS) in 1995 (Figure 1). NLARS is essentially designed for drought abstraction, nominally 1 in every 8 years, and is recharged using potable water to maximise storage in the Chalk-Basal Sands aquifer (O'Shea and Sage, 1999; Harris et al., in press). In investigating the viability of SLARS, whereby increased abstraction would be compensated by artificial recharge, it is clear that the hydrostratigraphy is the same as that in the NLARS area, but there are distinct hydrogeological differences:

- The Chalk and Basal Sands in the SLARS area are saturated, and thus there is less aquifer storage available to fill;
- The existence in south London of deep infrastructure at risk from high groundwater levels, which defines a maximum groundwater level and thus storage volume;
- In the SLARS area, there are zones where artificial recharge may be lost, e.g. spring discharges from the unconfined Basal Sands and Chalk, as well as erosional 'windows', thus retention times for the recharged water is lower in south London (Anderson et al., in press);
- Abstraction from the confined Chalk in south London is close to a sustainable limit, thus an abstraction increase may cause environmental impact in the unconfined Chalk or derogation of other abstractors.

In spite of these differences, it is considered that SLARS can enhance groundwater storage to meet summer peak demands, perhaps balancing abstraction and recharge on an annual basis. SLARS could also have potential to support water demand throughout extended drought periods, provided that either, (a) sufficient volumes of recharge water can be stored, pre-drought, without significant losses to springs or enhanced risk from high groundwater levels, or (b) high artificial recharge rates are viable, post-drought, to recover loss of aquifer storage.

SLARS artificial recharge potential

To test the viability of artificial recharge, a series of tests was carried out at Streatham in 2002–03. These achieved sustainable recharge rates of up to 14 Ml/d and abstraction rates of 12 Ml/d. The recharge water was derived from the River Thames in west London (see Figure 1); it was treated to potable standards at an existing works then transmitted almost 20 km eastwards via existing pipes.

In general borehole hydraulic performance during abstraction and recharge testing was similar, but there are important differences that could affect SLARS operation and need to be investigated. Specifically, the Streatham borehole hydraulic performance improved (i.e. lower turbulent head losses) between the time of the original testing in 1993

and 2002 due to operational abstraction. This suggests long term clearance of fractures and enhancement of fracture permeability. However, after significant artificial recharge, turbulent head losses increased suggesting that aquifer clogging was occurring. In fact, turbidity increased significantly in water abstracted following recharge, further supporting the occurrence of aquifer clogging. One cause of the turbid water, and thus clogging, is precipitation of $CaCO_3$ resulting from mixing of natural groundwater (minimum pH7.1) with river derived recharge water (pH7.5–8.1). Such clogging could cause operational problems and affect SLARS viability, but after only a day of abstraction the turbidity reduced and borehole hydraulic performance improved. This demonstrates that clogging is reversible and can be managed by pumping to waste (i.e. sewer) before pumping into supply.

GROUNDWATER RESOURCE MANAGEMENT USING SLARS

Having demonstrated artificial recharge potential at Streatham, the regional viability of SLARS has been assessed using the 3-D London Basin Groundwater model. This has considered increased abstraction and artificial recharge at several potential SLARS sites, and an operating strategy of:

- Summer (i.e. May to October) abstraction during non-drought years, with a combination of artificial recharge and no abstraction during the remainder of the year;
- Continuous abstraction throughout extended droughts, representing droughts such as 1976;
- A minimum regional groundwater level coincident with the top of the Basal Sands; and
- A maximum groundwater level where key underground infrastructure is at risk.

Summer abstractions of 26–38 Ml/d and continuous drought abstractions of 38 Ml/d were considered, with artificial recharge rates of 27 Ml/d. The top of the Basal Sands is set as the minimum groundwater level because of a groundwater quality risk. At low groundwater levels in the past, the Basal Sands were partially dewatered and the pyrite they contain was oxidised; on re-saturation groundwater with elevated iron and sulphate was produced (Kinniburgh et al., 1994). Although this did not occur at Streatham, and some other potential SLARS locations, inducing dewatering by increased SLARS abstraction poses a significant risk to groundwater quality.

Figure 3 shows that without any artificial recharge the additional abstraction from SLARS (i.e. about 17 Ml/d in non-drought summers) increases summer drawdown compared to background abstraction, but there is greater recovery as the SLARS summer abstraction period is shorter. With artificial recharge, drawdown is reduced to that predicted for the background abstraction scenario but the Basal Sands are still partially dewatered during summer abstraction periods. (Note: high groundwater levels produced by recharge are caused by a change to confined storage conditions.) These modelling results are for the area around the Streatham source and not the regional minimum groundwater levels; at around 2 km from the
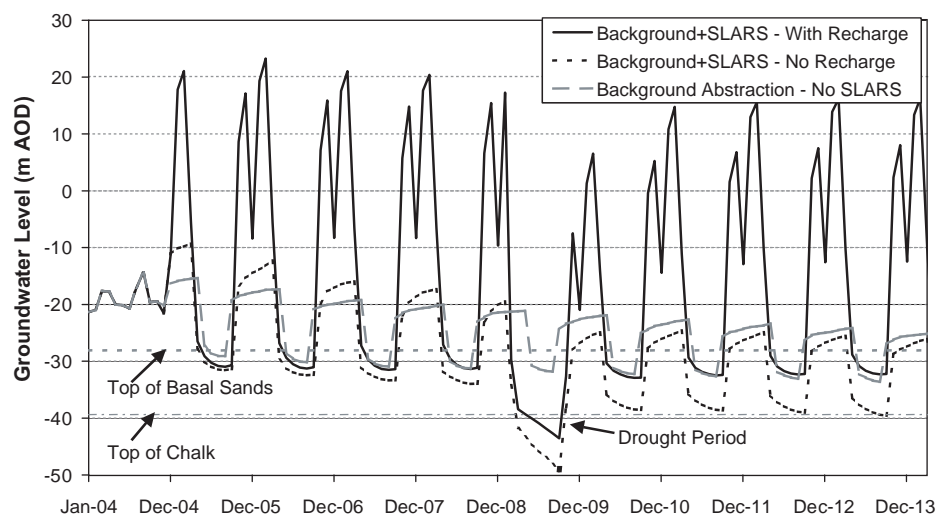

Figure 3. Simulated groundwater level responses to SLARS operation at Streatham

source observed drawdown is reduced by around 80%, and thus Basal Sands dewatering would not occur. However during droughts, predictions indicate the Basal Sands around the source might be completely dewatered creating a water quality risk.

Modelling predictions suggest that pre-drought artificial recharge does not store sufficient water to sustain base load abstraction through extended droughts. This deficiency in aquifer storage appears to be exacerbated by loss of recharge water to springs during pre-drought recharge. A loss of about 1.5 Ml/d is predicted pre-drought, but immediately post-drought there is no loss of artificial recharge to springs. Figure 3 shows that post-drought recharge is sufficient to balance the additional 17 Ml/d of SLARS summer abstraction, when compared with background abstraction. In addition, a slow progressive increase in storage is predicted as groundwater levels rise post-drought. These are initial results from a consideration of one possible operating strategy, but they provide a further demonstration of SLARS viability. Investigations are continuing and there are plans to drill and test new boreholes, as well as carry out further groundwater modelling. This work will allow the optimum operating strategy to be defined as well as the benefits to London's water supply.

CONCLUSIONS

For 120 years after commissioning, the evolution of the Streatham groundwater source reflected the wider evolution of the confined Chalk aquifer beneath London. It has evolved through phases of abstraction and disuse, followed by re-commissioning with a key role in reversing groundwater level rise. In the 21st century the viability of artificial recharge at Streatham has been demonstrated, and Streatham is in the forefront of SLARS investigations. By managing storage between maximum and minimum groundwater levels an operating strategy can be developed to assist meeting seasonal peak and drought demands, while ensuring underground infrastructure is not impacted and abstraction remains sustainable.

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Hydrogeology and water treatment performance of the Dösebacka artificial recharge plant – the basis for an efficient system for early warning

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Abstract

The EC-project ARTDEMO seeks to demonstrate an early-warning system for AR-plants, and this paper is a presentation of the initial work of setting up such a system at the Dösebacka plant in southwestern Sweden. Geophysical investigations, bore-hole drilling, geochemical characterisation, and water-treatment performance have been used to characterize the geology and to understand the character of the AR system. A geological and hydrogeological model of the Dösebacka aquifer is suggested. Geochemical analysis reveals two different water types, and an analysis of treatment performance demonstrates problems and solutions with regard to salt and turbidity.

INTRODUCTION

Municipal water supply from a plant using artificial recharge (AR) requires the ability of the aquifer to retain organic matter, pathogenic organisms, and other substances affecting water quality. However, the quality of the water generated by the AR system also depends on the quality of the infiltrated water, and it is important that the opeation can be shut down in cases of sudden changes in water quality. This calls for an alert early-warning system. The Dösebacka AR-plant, located along the Göta Älv river near Kungälv, Sweden, participates in the EC-project ARTDEMO with the main objective of demonstrating just such an early-warning system for monitoring and management. The focus of this paper is the understanding of the geology, hydrogeology and water-treatment performance of the aquifer as this information provides a basis for the creation of an effective early-warning system.

GEOLOGY AND GENESIS

The bedrock geology of the region is dominated by Precambrian granite and gneiss broken by two prominent north-south fault/fracture zones; one parallel to the valley occupied by the Göta Älv river, and the other, called the Mylonite Zone, about 15 km to the east (Samuelsson, 1985). The bedrock was eroded to a peneplain by the end of the Precambrian and covered with sedimentary rocks during the early Palaeozoic. These sedimentary rocks were eroded completely away during the Tertiary, and the re-exposed granite and gneiss was deeply weathered under a tropical climate. The gneiss and granite weathered more quickly and to greater depths along the fracture zones and numerous associated joints. The present topography, consisting of bedrock knobs separated by linear valleys, is merely the result of the removal of this weathered material during the Quaternary Period and the latter part of the Tertiary Period.

The numerous glaciations of the Quaternary Period did surprisingly little to shape the landscape. Following retreat of the ice, areas now below 100 m above sea level were inundated by the ocean, and marine clay was deposited in thicknesses up to 80 m in some of the bedrock valleys. The glacier left only a thin layer of till, and bedrock is commonly exposed in upland areas. The ridge above the AR-plant at Dösebacka is an exception and represents one of several scattered drumlin-like ridges in southwest Sweden. The Dösebacka drumlin consist of 30 m of till from the last glaciation overlying a thick sequence of older gravely glacial-stream sediment, some older till, permafrost features, and even some mammoth bones (Hillefors, 1974). The drumlin was formed up-ice from an exposed bedrock knob, and the till accumulated as the ice was forced to flow over the knob.

The aquifer used by the AR-plant consists of coarse sand and gravel, likely of glacial-stream-sediment origin (Bryllert, 2005). This sand and gravel rests on bedrock (or perhaps on till on bedrock) and is overlain by glacial and post-glacial marine clay that is thick on the river side of the AR-plant, but which wedges out west towards the valley side. The gravely sediment below the clay is similar in character to the sub-till layers exposed up slope in the drumlin, but it is unlikely that these gravely units represent the same layer. Alin and Sandegren (1947) indicate that the layers in the drumlin are truncated by erosion, implying that the sand and gravel in the aquifer is younger than the sediment exposed in the drumlin.

Two hypotheses are offered for the origin of the coarse aquifer sediment.

(1) The aquifer sediment represents a delta-like deposit that formed at the ice margin during deglaciation when the margin passed over the Dösebacka area. A nearby exposure of the aquifer sediment shows it to be bedded and containing well-rounded cobbles, similar to ice-marginal delta deposits at nearby Gråbo and Fjärås Bräcka. The topography of the Göta Älv valley would have allowed for the formation of such a delta, because the subglacial waters necessary for the formation of the delta would have most likely followed the valley trend.

(2) The aquifer sediment may represent wave-washed sediment that was eroded from the drumlin sediments. Following deglaciation, when this area was isostatically rising above the sea, waves could have washed coarse sediment from the drumlin into the Göta Älv valley. This explains why the aquifer lies immediately next to the drumlin. However, this site occupies a lee position and wave activity may not have been intense enough to produce the coarse sediment. Additionally, the aquifer sediment appears to be coarser than other wave-washed sediment in the region.

PLANT DESCRIPTION

Dösebacka Waterworks is located 5 km north of Kungälv, Sweden on the west shore of the Göta Älv river just below the Dösebacka drumlin. Raw water from the Göta Älv is pumped from an intake basin ('1' in Fig.1) to a sedimentation basin ('2' in Fig. 1). The water is distributed to nine infiltration ponds (letters 'A' through 'I' in Fig. 1). The water infiltrates the filter bed and is transported in the subsurface under unsaturated and saturated conditions. The aquifer water is withdrawn in fifteen abstraction wells (GRP 1-15, Fig. 1) and pumped to a reservoir, where it is pH-adjusted for corrosion prevention. It is also possible to chlorinate the water, but this is rarely done because the water in general has low levels of bacteria. Water from two of the wells (GRP 9 and 11) have high turbidity, and this water is treated chemically to create a precipitate and filtered through sand. The production of drinking water at the plant is 70-80 l/s.

METHODS

Geophysical examination was performed using Continuous Vertical Electrical Sounding (CVES) and refraction sounding. CVES was run in three lines parallel to and three lines perpendicular to the Göta Älv river (Rhodin, 2003). Refraction sounding was performed along two lines (Fig. 1)(Bryllert, 2005). Three boreholes were drilled to
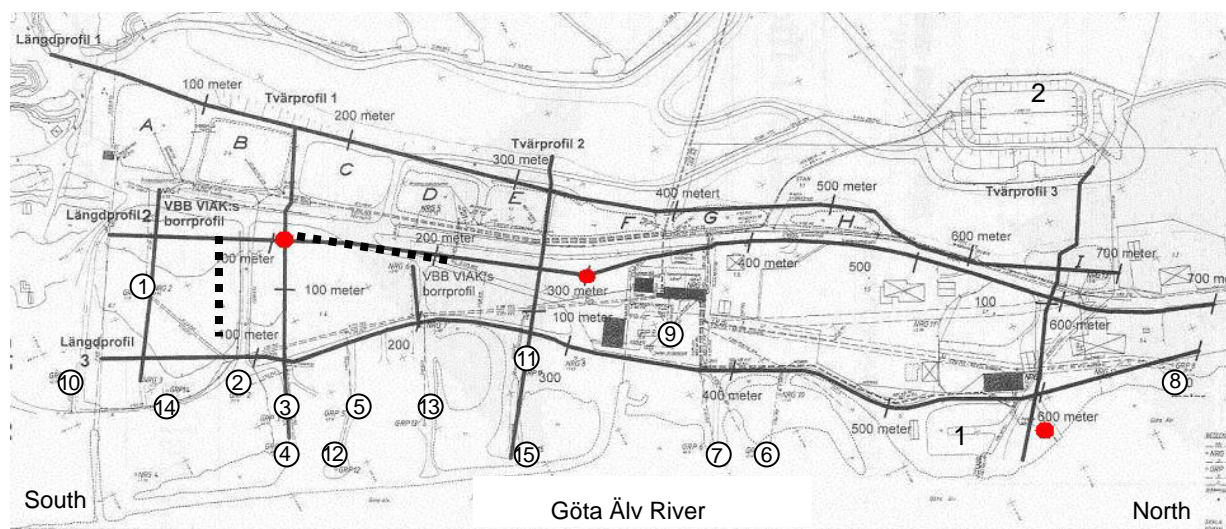
Figure 1. Map over the Dösebacka AR-plant; resistivity soundings (solid lines), drill holes (red dots), refraction soundings (dotted lines) and wells (numbered circles). Upper case letters A-I are infiltration ponds, and the numerals 1 and 2 are referred to in the text.

bedrock (Fig. 1), and samples were brought to the surface at every meter by air-induced flushing. The grain size of the sediment was determined using standard sieve and hydrometer methods. Sand-grain counts and X-ray diffraction (XRD) were used to determine the mineral content (Bryllert, 2005).

The hydrogeology and a groundwater-flow model are based on the known geology, groundwater logging, and temperature measurements in abstraction wells once a week during 2003-2004. Flow path were generated through numerical calculation with MIKE SHE (a product of DHI, Sweden). Residence time for abstracted water in GRP 9 was determined with a tracer experiment using NaCl that was added in Basin F (see below 'Tracer Experiment'). Analyses of Na^+, Cl^-, and electrical conductivity were performed in GRP 9. Residence time from the temperature measurement and the tracer experiment were compared, giving dampening rates. The rates were then used for conversion of the temperature-related residence times into probable residence time for each well.

The hydrochemistry at the Dösebacka site was analysed from water-quality data taken from 1992−2004. A Piper diagram (Piper, 1944) was produced from the water-quality data, giving information on the geochemical character of the groundwater in the abstraction wells. Stable isotopes (^{18}O and ^2H) and $^{87}Sr/^{86}Sr$ isotopic ratios were analysed at three different times in 2004 to infer the mixture between surface water and different groundwater bodies. Water-treatment efficiency was evaluated from water-quality data measured once a month during a period of nine months from August 2003 to April 2004.

Tracer experiment

The tracer experiment was conducted using sodium chloride (NaCl). The tracer is easy to handle, harmless to humans, cheap and the Cl^- ion is regarded as conservative in the aquifer material. Furthermore, the background Cl^- content in GRP 9 is fairly low (Table 3). The amount of sodium chloride was chosen so that the water-quality limits of 100 mg/l for Na^+ as well as for Cl^- would not be exceeded in water produced from the wells.

The tracer experiment started March 29, 2004. A total amount of 350 kg NaCl was mixed with 4 m^3 water in a tank. This concentrated solution was added to infiltration Basin F (Fig. 1). The water level in the basin was low (only a few decimetres) because the infiltration-water supply had been temporarily stopped. All the NaCl-rich water in the basin was allowed to infiltrate before the water supply to the basin was opened again.

Water was sampled from pumping well GRP 9 using two automatic water samplers. The sampling rate was one sample per day to start with, which was increased to two samples per day when the concentration peak was expected. After the peak passed, the sampling rate was again one sample per day, and, after some time, one sample every three days.

The calculated chloride concentration in Basin F on March 29th was 646 mg Cl⁻/l. This took 10 hours to infiltrate; the infiltration velocity was 31 mm/hr. By comparison, the maximum concentration in well GRP 9 which was 21 mg Cl⁻/l. The chloride peak was rather broad and it took more than twenty days for it to pass. Approximately 80% of the tracer was detected in GRP 9.

Table 1. Infiltration properties (2003-05-08)

	Pond A	Pond B	Pond C	Pond D	Pond E	Pond F	Pond G	Pond H	Pond I
Area (m^2)	2404	1967	1884	1301	1031	1096	787	563	1134
Infiltration rate (cm/h)	1.4	1.3	2.8	2.6	0.8	4.3	1.7	1.9	–

RESULTS AND DISCUSSION

Geophysics, drilling, stratigraphy, and mineralogy

CVES (Rhodin, 2003) nicely displays the extension of the glacial/post-glacial clay layer (Fig. 2). The clay layer is 20–40 m in the south and close to the river, but it wedges out up slope towards the ponds. Clay and silt seem to be absent in the northern part (Fig. 2). Drilling (red dots, Fig. 1) (Bryllert 2005) indicate only one aquifer, composed of stratified sand and coarse gravel. Clay or till were not found during drilling, but this does not exclude the possible existence of local lenses of clay and thin beds of till. The grain-size distribution of till in the region is somewhat similar to glacial-stream sediment, and it is difficult to differentiate the two in drill-hole samples.

The depth to bedrock is 15 meters close to the infiltration ponds (Bryllert, 2005), and an unpublished CVES measurement in basin B indicates bedrock 3 m below the pond bottom. Initial unpublished measurements with a gravimeter indicate a depth to bedrock of 50–60 m close to Göta Älv. Analysis of the mineral content shows that the sand and gravel consists of normal granitic minerals, with a high content of quartz and potassium feldspar (Bryllert, 2005).
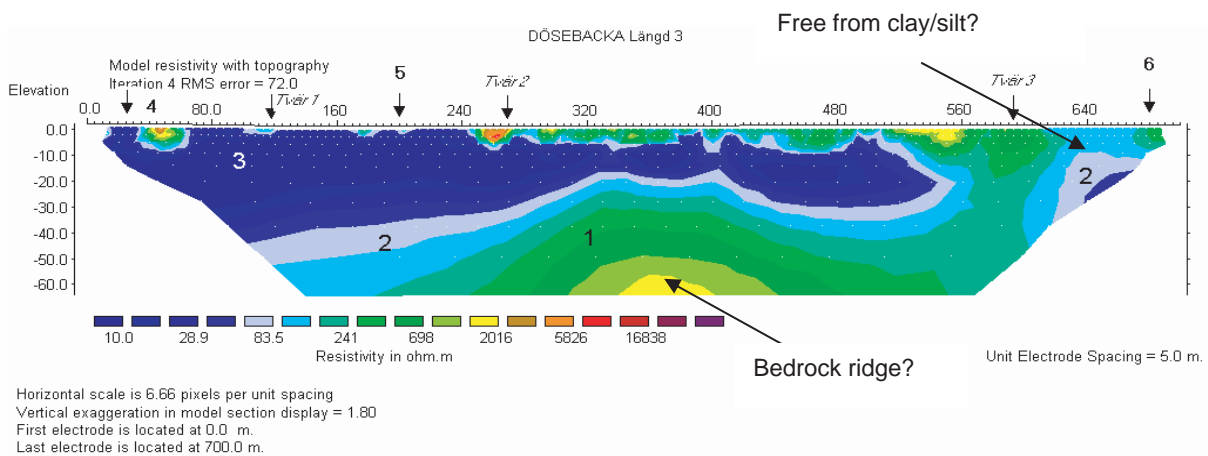

Figure 2. CVES profile 3 from Rhodin (2003)('Längdprofil 3' in Fig. 1). The profile runs parallel to and approximately 50 m west of Göta Älv. South is to the left in the figure. Low resistivity material (3) is interpreted as clay, (2) as sand and gravel and (1) as bedrock.

Groundwater flow and residence time

A computer simulation with MIKE SHE (Fig. 3) displays flow paths from the infiltration ponds to the Göta Älv river. Both hydraulic data and seismic investigations suggest a divide at pond C and D, directing the flow into one path from pond D-H towards GRP 9 and 11, and another from A-D towards the wells in the south of the area. This divide is likely made of bedrock or perhaps till. Fig. 3 also reveals a gradient from the river suggesting a transport of water from below the river. In the north, the absence of a confining layer allows induced infiltration from the river.

The bottoms of the infiltration ponds are approximately 4.7–7.7 m above sea level. The water table is about 0–2 m just east of the ponds, implying that the thickness of the unsaturated zone varies between 0 to 7.7 m. The presence of shallow bedrock in pond B and high groundwater levels for A-C (Fig. 3)

Figure 3. The flow paths at the Dösebacka AR plant generated with numerical calculation in MIKE SHE (with kind permission of DHI, Sweden)

might indicate that there is no unsaturated zone below some ponds. The crest of the divide beneath ponds C and D plunges deeply (50 m) towards the river (Fig. 2) and probably has less influence on the water paths close to the river than higher up towards the ponds.

The tracer experiment displayed a first response at 5 days and 95% of the total observed mass of tracer had been detected after 25 days. Assuming 20% effective porosity, a residence time of 11 days for GRP 9 and a hydraulic gradient of 0.0186 m/m between pond F and GRP 9, the average hydraulic conductivity is calculated to 8.4 x 10^{-4} m/s within a range from 1.7 x 10^{-3} m/s (5% percentile) to 3.7 x 10^{-4} m/s (95% percentile). The hydraulic gradient could not be exactly determined but a performed sensitivity analysis indicated a range of 3.0 x 10^{-3} to 4.9 x 10^{-4} m/s for a gradient variation of 0.0186 ± 0.0134 m/m. The infiltration rates varied between 0.8 to 4.3 cm/h, which gave an infiltration of approximately 62 l/s (Table 1), thus 15–20% of the abstracted water seems to come from other sources. The low rate in pond E might imply a hydraulic feature such as a divide. The residence time for GRP 9 is 11 days according to the tracer experiment and 35 days according to the temperature, giving a conversion factor of 3.2, which suggest residence times less than 75 days for the plant in general (Table 2).

Groundwater hydrochemistry

Stable isotope compositions of groundwater and springs plot close to the global meteoric water line (Craig, 1961) (Fig. 4). River water from Göta Älv and water samples from abstraction wells are significantly enriched in both isotopes and plot well below the meteoric water line, indicating isotopic modification as a result of evaporation. River water from Göta Älv and groundwater collected from the wells GRP 1, 2, 3, 5, 9, 10, 11, 13 and 14, are all characterized by mixed water type without dominance of any particularly ions. Groundwater from GRP 4, 6, 7, 8, 12, and 15 are of the Na-Cl/Na-Cl-Ca-HCO$_3$ types indicating some influence from a saline water body (Fig. 5). The
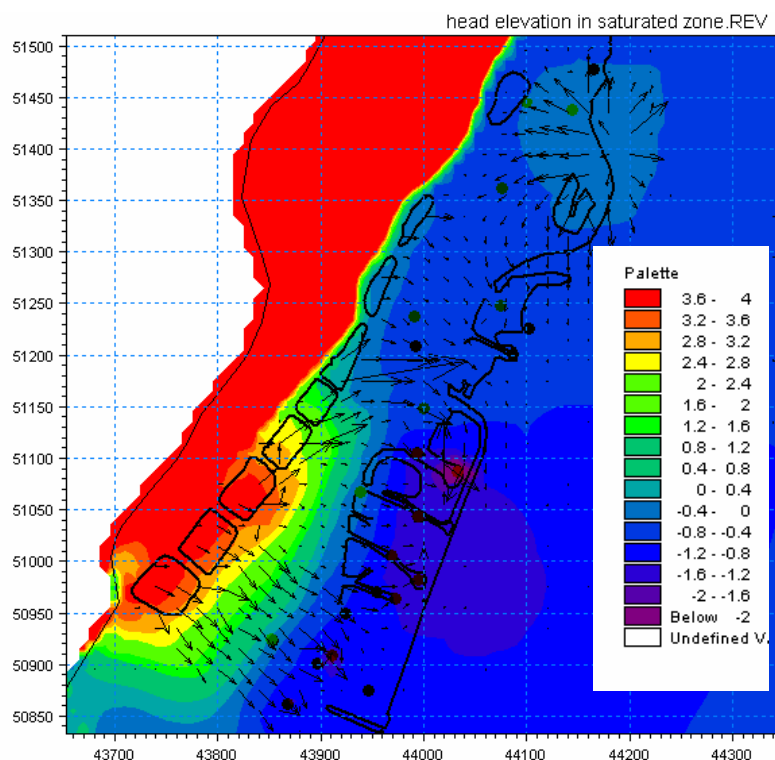
Table 2. Converted temperature-based residence time and pumping rates in Dösebacka

Well	Pumping rate (l/s) 2003-06-10 'Typical setting'	Temperature response (d) (Summer peak in raw water - heat peak in well)	Probable residence time (d) Dampening factor 3.2
1	2.5	70	22
2	1.4	180	57
3	1.6	170	53
6	6.8	190	59
7	9.5	140	44
8	3.7	150	47
9	17.9	35	11
10	6.3	240	75
11	6.1	80	25
12	0.9	180	56
13	6.4	175	55
14	7.6	125	39
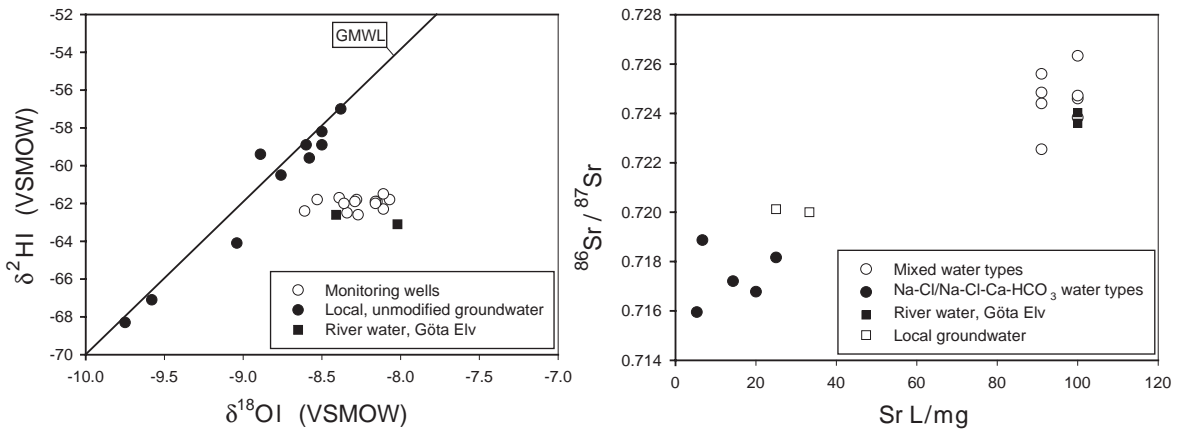
Figure 4. Left: Stable isotope compositions of water types from the Dösebacka AR plant. GMWL: Global meteoric water line (Craig, 1961). Right: Sr isotope (87Sr/86Sr) compositions of water types from the Dösebacka AR plant.
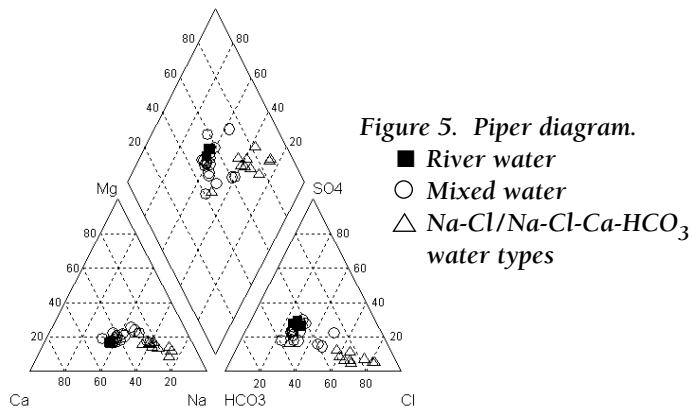
Figure 5. Piper diagram.
■ River water
○ Mixed water
△ Na-Cl/Na-Cl-Ca-HCO$_3$ water types

distribution of the electrical conductivity ranges from 90 to 1,650 µS/cm and confirms the relatively high concentrations of Na and Cl in these boreholes. Sr isotope analyses reveal two distinct groups that correspond to the well grouping on the basis of the hydrochemistry (Fig. 4). River water and the mixed water type are characterized by low Sr contents and relatively high [87]Sr/[86]Sr isotopic ratios, whereas the water types dominated by Na and Cl show high Sr contents and relatively low [87]Sr/[86]Sr isotopic ratios.

Water-quality problems

Turbidity and salt are the two main water-quality problems at the Dösebacka site. Turbidity is exemplified by GRP 9 (Table 3), which shows high turbidity (but similar organic content as the other wells). The problem is probably related to fines that follow the abstracted water. The problem with salt is believed to do with captured relict water from the period after the deglaciation when the area was inundated by the sea prior to and during the land rise. The problem is exemplified by GRP 7, which has problems not only with chloride but also with manganese and occasionally nitrite, probably connected to low oxygen concentrations in the relict water. Other wells with similar salt character are GRP 6, 8, 12 and 15. The pretreatment consist of sedimentation for peaks of high turbidity in Göta Älv. This is, however, not observeable in the measurement presented in Table 3. Posttreatment consist of chemical precipitation with Al-sulphate for the wells with high turbidity. This results in reduced turbidity, colour and organic matter (COD) in the finally treated water. There is possibly a higher risk of microbiological breakthrough in GRP 9 and 11 due to the shorter residence time, and in fact, coliforms have been detected in GRP 11, but the chemical precipitation constitutes a barrier in that sense. GRP 3 has also indicated coliforms, but this is probably due to leakage into the well during times of flooding of Göta Älv.

Table 3. Average Water Quality for some parameters (from 4–9 samples over the period) in Dösebacka, May 2003–May 2004 (: detection level < 2 cfu/100 ml)*

	Unit	Göta Älv	Pond A	GRP 1	GRP 3	GRP 7	GRP 9	Drink W.
Slowly growing bacteria	cfu/ml	3,100	> 3,100	93–100	0–10.0	48–50	5.0–13	33–38
Microorganisms, 22°C 3d	cfu/ml	1,400	> 2,400	95–100	0–10.0	0–10.0	2.5–10.0	5.5–5.8
Coliform bacteria 35°	cfu/100 ml	> 340	> 350	0–1.0	0.22–1.0	0–1.0	0–1.1*	0–1.0
E.coli	cfu/100 ml	> 46	> 38	0–1.0	0–1.0	0–1.0	0–1.1*	0–1.0
Hardness	°dH	1.5	1.4	1.5	1.5	7.7	1.7	3.3
Fe (++)	mg/l	0.12–0.13	0.12–0.13	0.014–0.053	0.0067–0.051	0.086	0.046–0.071	0–0.050
Cl (-)	mg/l	8.3	8	7.7	8.3	180	9.5	56
EC	mS/m	10	9.9	9.9	11	82	11	33
Mn (++)	mg/l	0.0083–0.025	0.0029–0.020	0–0.020	0–0.020	0.23	0–0.020	0.034
Nitrite nitrogen	mg/l	0.0028	0.003	0.00078–0.0014	0.00044–0.0012	0.0076	0.00020–0.0011	0.011
Ammonium nitrogen	mg/l	0.025	0.029	0–0.0078	0–0.0078	0.62	0.0010–0.0080	0.07
pH		7.2	7.2	6.9	7.1	7.8	6.9	8.1
Turbidity	FNU	5.2	4.9	0.85	0.72	1.1	1.6	0.54
Alkalinity	mg/l	20	20	21	27	120	22	61
COD-Mn	mg/l	3.7	3.7–3.9	0.56–1.0	0–1.0	0.22–1.0	0.70–1.1	0.20–1.0
Colour		22	22	5.0–5.6	5.0–8.3	5	9	2.0–5.0

CONCLUSIONS

A suggested model of the Dösebacka aquifer is a 10-15 m single aquifer of sand and gravel, which is sloping from 3-15 m below surface, under the infiltration ponds, to 50-60 m below the surface at the river. The aquifer is partly confined with a layer of clay; thin near the ponds and increasing to 20 to 40 m close to the river. In the north the clay is absent. Below the aquifer lies bedrock, probably heavily fractured (Fig 6).

The hydrology of the aquifer is governed by the gradient from the elevated infiltration ponds down to the river, a groundwater divide, and the induced infiltration in the north. The hydrochemistry reveal two different water types. One with the dominance of infiltrated river water and another that reflects prevalence of local groundwater and influence of saline formation water from the subsurface, which is the case for the wells close to the river. Because of high producing wells and existing flow paths, an explanation could be that the infiltrated water does not reach these wells.

From an early-warning point of view, the wells in the north might constitute a critical problem as abstracted water seemingly originates from direct infiltration from Göta Älv and needs swift actions in case of a detected contaminant. Another reflection concerns the groundwater divide, that might enable the operation of half of the plant in case of an incident, ensuring a production, even though reduced. Fecal contamination (coliform) in the wells has been indicated a couple of times for specific wells but the use of chemical precipitation constitutes a barrier for the critical wells, and thus a regular monitoring of the delivered water,

Figure 6. A cross section perpendicular to the river showing the general geological structure under the Dösebacka plant

related to the shortest residence time in the aquifer, should be enough. Salt ought to be closely correlated to pumping and requires more or less on-line measurement of flow and EC for critical wells.
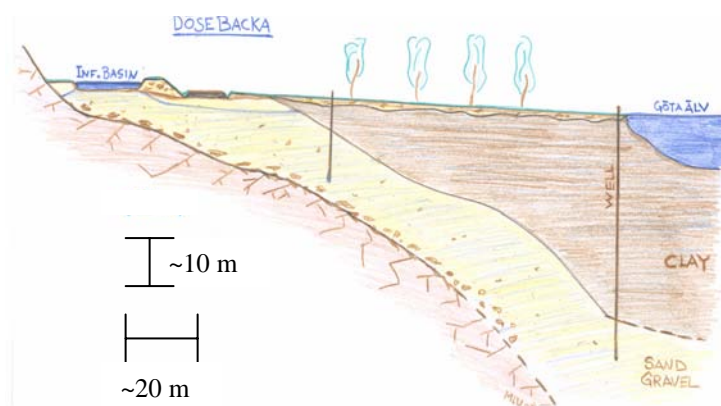
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Inexpensive soil amendments to reduce salty water intrusion into the aquifer

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ABSTRACT

Apatite, bentonite, zeolite, beringite, zero valent iron (ZVI), iron oxide and alkaline biosolid were evaluated as inexpensive and abundant stabilizing agents in metal contaminated soils by batch, column, soil TLC and bioavailability studies. Results from the laboratory study were used to design the greenhouse study in which amendments were mixed separately with the contaminated soil at two rates: 25 and 50 g kg^{-1} for growimg pearl millet and green gram. The influence of the stabilizing agents on the mobility, bioavailability and toxicity of Cr, Mn, Ni, Cu, Zn, Cd and Pb were evaluated using newly developed availability indices such as the modified distribution coefficient (K_d), bioavailablity factor (BF), recalcitrant factor (RF) and transfer factor (TF), all of which give us information about the amount of a metal contaminant that remains in the soil vs. the amount that moves into solution or the food chain. The amendments significantly reduced the mobility of metals through soil into the aquifer and availability of metals to plants. However, the effectiveness of these amendments varied. For example, iron oxide was most effective for soils contaminated with arsenic, whereas apatite was best at reducing the mobility of lead, cadmium, and zinc. The alkaline biosolid played an important role in stabilization of copper and nickel. Zeolite stabilized cadmium, zinc, lead, copper, and nickel in soil, especially when metal levels were low, but its efficacy might be questionable. Such changes among the soil quality indices indicate success of the remediation technique, and may have relevance in risk assessment, monitoring and checking salty water intrusion into the aquifer.

INTRODUCTION

The contamination of environmental resources particularly soil and water by hazardous wastes is a problem around the globe that poses a serious threat to life. Soil contaminants are anthropogenic from domestic, urban, industrial, agricultural, mining and automobile traffic activities. They leach down to the aquifer, and are consumed up by the crops. In this way enter the food chain and harm human health. Major anthropogenic inorganic soil pollutants with their important oxidation states are Be(II), F(–1), Cr(III-VI), Ni(II-III), Zn(II), Cu(II), As(III-V), Cd(II), Hg(0-I-II) and Pb(II-IV). Fertilizers and pesticides applied to crops are largely retained as recalcitrant contaminants by the soil, and also seep down to the aquifer.

At present, a variety of approaches have been suggested for soil remediation (Knox et al., 2003; Lombi et al., 2002; Oste et al., 2002; Basta et al., 2001). The most appropriate process depends on the forms of a contaminant, their concentration, pH, other constituents of the matrix and the desired standard for the matrix to be discharged into the environment. Traditional techniques of metal removal like coagulation with alum or iron salts, lime precipitation and flocculation are not very effective and form sludge whose disposal is a problem. The advanced treatment processes such as ion exchange, reverse osmosis, membrane separation, electrodialysis, chemical precipitation and activated carbon adsorption are prohibitively expensive. The use of microbes such as bacteria, fungi and algae in treating contaminated wastewaters is today an attractive technique but it is slow and as yet not suitable for applications on a large scale. Studies reveal adsorption to be the most promising technique because it is highly effective, cheap, easy and ecofriendly method among all physicochemical processes. The goal of this research was to evaluate the feasibility of using apatite (AP), bentonite (BT), zeolite (ZT), beringite (BG, a calcined Paleozioc schist, an alka-

line alumino silicate), zero valent iron (ZVI), iron oxide (Fe_2O_3) and alkaline biosolid (AB, lime stabilized Fe-rich municipal waste) as efficient, versatile, selective, cheap and green immobilizing media for Zn, Cu, Ni, Mn, Cr, Pb and Cd in contaminated soils.

MATERIALS AND METHODS

Two contaminated soil samples from the Shahadra landfill (S1) and St. John's roadside (S2) were collected from the top horizons (0–20 cm deep) and deep horizon (1.8–2 m) were air dried and sieved through a 2-mm screen, and then subjected to different physicochemical analyses including analysis of Cr, Mn, Fe, Ni, Cu, Zn, Pb and Cd with a Perkin-Elmer AAnalyst 100 AAS as per standard methods (APHA, 1998). The working standards of 0.1, 1.0, 10.0 and 100.0 mg/l metal solutions were used to calibrate the instrument. Batch test was conducted by placing a constant ratio of soil to wash liquid (1: 5 = 20 g/100 ml with 1 g of the sequestering agent as pellets) at room temperature. A control without stabilizer was taken for each concentration. Eight types of the wash liquids for 20 g soil were 100 ml DDW, and seven 100 ml DDW+ 1 g each of the seven amendments in pellet form. The flask was shaken by hand for about 30 seconds to disperse the soil particles, and then in a rotary shaker at 120 rpm for 2 h. The amendment pellets were filtered off with a standard 2-mm sieve. The soil suspension was centrifuged at 4,000 rpm for 10 min and the residue was extracted with perchloric acid. The extract was analysed for metals by AAS. A series of 4 washings was performed by washing the soil for 2 h each time with a fresh lot of same amount of the wash liquid containing 1 g stabilizer. The final concentration for a washing became the initial metal concentration for the next washing. The metal uptake on amendments at different concentrations was thus calculated. The data were analysed using the Langmuir constants and K_d values.

In order to establish leachability of contaminants through soil in natural conditions, downflow experiments were conducted using PVC columns (100 cm height x 2.5 cm i.d.) for each contaminant with and without stabilizers. A series of columns to provide different bed contact times (EBCTs of 10, 20, 30, 40 and 50 min) and bed heights (BHs of 10, 25, 40, 60, 80, 120, 160 and 200 cm) was used. 100 ml of each 1, 10, 50 and 100 mg/l contaminant solutions was passed through previously analysed soil sample at 1 ml/min under different conditions. The eluant was collected and analysed for the contaminant.

To study mobility by soil TLC, the contaminated soil was amended with varying quantities of immobilizers to study their mobility and effect on the migration of others. A soil slurry (1 soil : 2 water) was applied on a glass plate (25 x 5 cm) to give 0.5-mm thick layer. 0.1 M metal nitrate or chloride solution was spotted on the baseline with a micropipette. The contaminants were detected by spraying a suitable reagent and their mobility in terms Rf was calculated.

For bioavailability study amendments were mixed separately with the contaminated soil at two rates: 25 and 50 g kg^{-1}. There were four replicates for each treatment, and the 1-kg pots were arranged in a completely randomized design. After four weeks of soil equilibration, pearl millet (PM or *Pennisetum glaucum*) was planted and then harvested after six weeks of growth. After PM, green gram (GG or *Phaseolus mungo*) was planted and grown to maturity. While culturing, the plants were fertilized with dissolved N–P–K (1:0.7:0.7) (applied as KNO_3 and $NH_4H_2PO_4$), and watered with deionized water as needed. At the end of the growing period, the PM was separated into leaves and roots, and the GG plants were separated into leaves and grains for subsequent chemical analyses. Soil and plant extracts were analyzed by AAS. The influence of stabilizers on the mobility, bioavailability and toxicity of Cr, Mn, Ni, Cu, Zn, Cd and Pb were evaluated using newly developed availability indices such as the modified distribution coefficient (K_d), bioavailablity factor (BF), recalcitrant factor (RF) and transfer factor (TF), all of which give us information about the amount of a metal contaminant that remains in the soil vs. the amount that moves into solution or the food chain.

RESULTS AND DISCUSSION

Tables 1 and 2 show the selected properties and metal contents of the two soil samples used in these experiments. Comparing the soils, it is obvious that the soil sample S1 from Shahadra landfill is more contaminated than S2 from St. John's roadside. The metal concentrations in S1 are in excess of maximum permissible concentrations for an agriculture area (Table 2) (CCME, 1991). In S2, only the total Cd and Pb concentrations were slightly above the limit of the residential area.

Table 1. Selected characteristics of the top soil used in the study

Soil	Sand (%)	Silt (%)	Clay (%)	$CaCO_3$ (%)	Org.C g/kg	pH	CEC meq/100 g
S1 (Shahadra)	68	24	8	18.4	15.8	7.8	12.8
S2 (SJ roadside)	54	38	10	20.6	18.7	7.5	14.7

Table 2. Total metal contents (mg/kg) soils used in the experiments (0–20 cm top layer)

Soils	Cr	Mn	Fe	Ni	Cu	Zn	Cd	Pb
S1	293.3	836.0	89,000.0	251.0	498.7	595.0	5.2	320.0
S2	39.2	43.2	28,600.0	29.3	38.2	236.0	11.2	126.0
MPL*, ppm	100	100	NA	50	100	300	6.0	100

∗ Maximum permissible limits of metal concentration in an agricultural soil.

The immobility in Shahadra landfill soil (S1) with total Zn 595.0, Cu 498.7, Ni 251.0, Fe 89,000.0, Mn 836.0, Cr 293.3, Pb 320.0 and Cd 5.2 mg/kg was found to decrease 60% Zn, 66% Cu, 62% Ni, 57% Fe, 50% Mn, 55% Cr, 70% Pb and 71% Cd after 4 washings of 2 h each at 120 rpm with a fresh lot of the same amout of wash liquid containing 100 ml distilled water with 1g AP at pH 6.8 and temperature 25ºC. The St. John's roadside soil (S2) that was less contaminated, showed greater removal (Table 3).

Table 3. Effect of initial concentration on the metal removal by adsorbent dose 1 g per 100 ml wash liquid containing 20 g soil at pH 6.8, temperature 25°C, agitation time 2 h, rpm 120

Metal	Sample	c_i	% Removal	ce	a = x/m	C/a	Q	b
Zn	S1	595.0	60	249.89	6.902	36.20	20.41	0.0020
	S2	236.0	64	84.96	3.021	28.12		
Cu	S1	498.7	66	179.53	6.383	28.13	36.41	0.0012
	S2	38.2	68	12.22	0.520	23.53		
Ni	S1	251.0	62	95.38	3.112	30.65	22.85	0.0017
	S2	29.3	65	10.26	0.381	26.92		
Fe	S1	89,000.0	57	39,160	996.80	39.28	4,656.95	6.9545
	S2	28,600.0	60	11,440	343.20	33.33		
Mn	S1	836.0	50	418	8.360	50.0	37.25	0.0007
	S2	43.0	56	18.92	0.482	39.28		
Cr	S1	293.3	55	131.98	3.2236	40.91	24.57	0.0011
	S2	39.2	58	16.46	0.4547	36.21		
Pb	S1	320.0	70	96.0	4.480	21.43	30.60	0.0018
	S2	126.0	72	35.28	1.8114	19.44		
Cd	S1	5.2	71	1.352	0.077	17.57	0.66	0.0969
	S2	11.2	74	3.248	0.159	20.42		

Sorption capacity is found to decrease with increase in metal concentration. The higher uptake at lower initial concentration can be attributed to the availability of more isolated metal ions (Singh et al., 2003). Sorption rate was very rapid during initial period of contact and about 80 per cent of sorption was reached within first 60 minutes. However, equilibrium was attained within 180 min. for most of them. The optimum agitation time for iron removal was 90 min whereas for Cr and Ni it was 105 min. The order of metal immoblization with apatite was Cd > Pb > Cu > Ni > Zn > Fe > Cr > Mn (Fig. 1). This is probably due to their ionic size and ionic potential, and this order may be reverse of the their sorption on soil colloids.
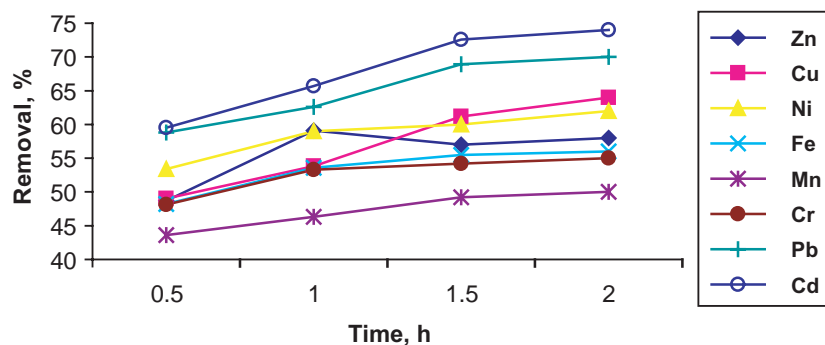

Figure 1. Percentages of metal sorption from soil S1 at different agitation times in 4th washing, pH 6.8, temperature 25°C and apatite dose 1g per 100 ml wash liquid containing 20 g soil

The sorption data well fitted to the Langmuir isotherm. The adsorption capacity of AP was calculated to be 20.41, 36.41, 22.87, 37.25, 24.57, 30.60 and 0.66 mg/g for Zn, Cu, Ni, Mn, Cr, Pb and Cd respectively.

Studies by column process indicate that the sorption by soil without amendment decreased in the order Mn > Fe > Ni > Cr > Zn > Cu > Pb > Cd. This is probably due to several factors such as ionic size, ionic polarity, pH, metal reactivity and retentivity, soil properties and environmental conditions (Davies and Singh, 1995). The leaching or mobility order should be reverse of the sorption order. The sorption of metal molecules goes on increasing as their concentration decreases from 100 to 1 mg/L. The sorption percentage at all levels of Mn^{2+} is more than that of all the cations under this study, and Fe^{3+} is about 10 % more than that Ni^{2+}, and of Pb^{2+} is more than that of the Cd^{2+}. Leaching increases as the flow rate increases. Solution pH significantly affected the extent of contaminant removal (Lombi et al., 2003). The results showed variation of percent removal at various pH values. A specific relation of optimum pH was observed with the nature of metal. Mn showed the highest removal (minimum leachability) at pH 6. Cd had maximum removal at pH 8.0. The maximum removal of Pb was found at pH 8.0. At pH 8.0, Cr had the highest per cent removal. The maximum leachability for Cd was 62.4% and the minimum leachability for Mn was observed for 38.1 % at pH 6.

The R_f value of manganese obtained from the soil TLC was minimum, and so sorption percentage of manganese was more than that of all the metals under this study. Fe^{3+} is about 10 % more than that Ni^{2+}, and of Pb^{2+} is more than that of the Cd^{2+}. The R_f value increased and so sorption by soil without admendment decreased in the order Mn > Fe > Ni > Cr > Zn > Cu > Pb > Cd. The leaching order is directly proportional to the R_f value. The maximum R_f value of Cd indicates that it is the most mobile of all the investigated. The results of batch and column process are similar to that of soil TLC. Stabilizers were able to reduce metal leaching, in most cases, in the order AP > ZT > BT > BG > Fe_2O_3 > AB > ZVI.

A low TF value indicates low metal bioavailability to plants or low metal translocation within the plant. In Table 4, TF values are presented for the roots and leaves of six-week old PM, and for the leaves and grains of mature GG. The highest TF was observed for PM roots and the lowest TF values were in the GG grain. The trends in the data

suggest that although metals accumulated mostly in the roots, substantial Cd and Zn was translocated to the leaves and grains, especially in the untreated soil. For example, the Cd TF value in PM leaves in untreated soil was the highest (1.80). Both apatite and zeolite significantly reduced TF values for each metal, and the highest reduction occurred in PM roots and leaves, and in GG leaves.

Table 4. Amendments like apatite etc. can reduce the movement of contaminants from soils into plants,
as measured in this case by the transfer factor (TF). Cd levels in all soils were 40 ppm.

Soil amendments	PM root	PM leaves	GG leaves	GG grain
Cd only	2.00	1.80	1.00	0.20
Cd + 2.5% ZVI	1.80	1.60	0.80	0.10
Cd + 2.5% AB	1.60	1.40	0.60	0.08
Cd + 2.5% Fe_2O_3	1.50	1.30	0.50	0.07
Cd +2.5% Beringite	1.20	1.00	0.40	0.06
Cd + 2.5% Bentonite	1.00	0.90	0.30	0.05
Cd + 2.5% 2.5% Zeolite	0.90	0.80	0.25	0.03
Cd + 2.5% Apatite	0.60	0.40	0.08	0.00

K_d is relatively low in heavily metal contaminated soil, meaning that metals are fairly mobile. Addition of certain amendments increased the K_d value, reducing the mobile fraction of metals in the soil. Apatite sorbed more Cu^{2+}, Pb^{2+}, Ni^{2+} and Cd^{2+} from the spike solution than zeolite, resulting in K_d values of > 200,000 l kg^{-1}. In contrast, the BF differs for each element, with its value being dependent upon the total concentration of the metal, the source of the metal, and the properties of the soil. While a very mobile element like cadmium can have a BF value of only 2% in uncontaminated soils, the BF for cadmium can increase to as much as 50% in polluted soils. The RF factor typically is lower in soils with high concentrations of contaminants and low soil pH. TF values, which indicate metal bioavailability, are usually high in contaminated soils but decrease upon treatment.

This studies found that the application of increasing amounts of amendments to metal-contaminated soils resulted in increased values for K_d and RF, and decreased values for BF and TF. Such changes among the soil quality indices indicate success of the remediation technique, and may have relevance in risk assessment, monitoring and salty water intrusion into the aquifer.

CONCLUSIONS

The study indicates that adsorption, absorption, ion exchange and desorption take place simultaneously when heavy metals interact with loamy soil with or without amendments. Stabilizers can reduce metal leaching in most cases in the order AP > ZT > BT > BG > Fe_2O_3 > AB > ZVI. The sorption by soil without admendment decreased in the order Mn > Fe > Ni > Cr > Zn > Cu > Pb > Cd as indicated by batch, column and TLC studies. The leaching order should be reverse of it. Cd^{2+} is, thus, the least strongly retained by the soil than the other toxic cations, and hence can pose a more serious problem of polluting groundwater with its extreme toxicity. Zn, Cu, Pb and Cd were found far more mobile than other four metals, and so they are also hazardous for the aquifer. They should, therefore, be properly monitored in soil, aquifer and phytomass at all the contaminated sites in Agra. An increase in metal concentration led to weaker retention. This indicates to possibility of groundwater pollution due to landfilling. Metals are more adsorbed on silty loam than on sandy loam because the former has higher organic matter,

clay, monmorillonite contents, CEC and surface area. Most of Agra soil is sandy loam, and hence it is advised that all wastes containing heavy metals should be treated before disposal into loamy formations.

The order of metal binding capacity of soil components SOM > clay silicate > iron hydroxides indicates SOM to be the most important sorbent. If metal loading is beyond SOM capacity, clay silicate becomes more important binder. ZVI is not very significant binder. Apatite is consistently better than other stabilizers at reducing metal mobility into the aquifer and bioavailability to the plants.

Mineral amendments often contain metal impurities that may exceed environmental limits if applied in excess. The effect of pH changes on the lability of fixed colloidal heavy metals in soils treated with amendments should be considered because most of the soil amendments are alkaline and increase soil pH.

Alkaline additives should be evaluated whether they contain enough Ca to prevent increased DOM leaching from soils as a result of their alkalinity. High Ca reduces DOM leaching, resulting in decreased metal leaching. Both the immediate effectiveness and long-term sustainability of using such soil amendments must be demonstrated to achieve widespread public and regulatory acceptance. Furthermore, it is necessary to evaluate the ability of prospective soil amendments to reduce contaminant mobility to the aquifer, and bioavailability by simple batch and column techniques.

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Mapping groundwater bodies with artificial or induced recharge, by determination of their origin and chemical facies

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Abstract

The mapping of groundwater bodies and their quality has become even more important since the introduction of the European Water Framework Directive in 2000. This especially holds for groundwater bodies with artificial recharge (AR) and river bank filtration (RBF), because they are not natural, neither hydrologically nor hydro-chemically, and are therefore quite vulnerable to pollution and EU verdicts.

A method is proposed to prepare maps with the spatial distribution of all relevant groundwater bodies (hydro-somes), both natural and man-made (or man-induced), and hydrochemical facies (zones) within these hydro-somes. It requires the use of environmental tracers to reveal the origin of each relevant water sample, and the determination of its hydrochemical facies by any combination of the pH-class, redox level, pollution index and base exchange index BEX (or Sodium Adsorption Ratio, SAR).

Examples are given from the coastal dunes (AR by spreading of Rhine River water), and dutch fluvial plain (RBF).

Keywords

Artificial recharge, European Water Framework Directive, mapping, river bank filtration, tracing, water quality.

INTRODUCTION

The European Water Framework Directive and in particular the European Groundwater Directive have or will have a strong impact on how to map, protect and monitor groundwater bodies. This holds in particular for those with Managed Aquifer Recharge and subsurface Storage (MARS). These groundwater bodies are as a matter of fact very special by virtue of their man-made recharge with surface water either by artificial recharge (AR) or induced recharge (River Bank Filtration; RBF).

Good water quality maps need to be made anyhow, for a.o. allocating water resources, optimizing monitoring networks, controling water pollution and the set-up of combined transport-reaction models. In addition, maps form the most effective communication tool for transfer of water quality data in a geographical context, either from expert to expert or from expert to policy-maker or public.

The Hydrochemical Systems Analysis (HSA) was introduced by Stuyfzand (1993, 1999) to provide tools for mapping groundwater quality in a way similar to mapping the geology, pedology or hydrological systems of an area (Fig. 1). The methodology presented here deviates slightly from the original HSA, in order to better address the mapping of man-made groundwater bodies in (semi)natural environments, in the larger variety of hydrogeo-chemical environments in the European Union today.

The methodology presented here results in the mapping of generic water types. For a further characterization a chemical classification of water types can be useful, like the one proposed by Stuyfzand (1986, 1993).

	(SUB)SOIL		GROUNDWATER	
	GEOLOGY	**PEDOLOGY**	**HYDROLOGY**	**CHEMISTRY**
1. GENETICAL UNIT:	**Formation**	**Soil Type**	**Flow System**	**Water body (Hydrosome)**
Example	Westland Formation	Podzol	Recharged Rhine River	Dune water
2. ZONES WITHIN	*Sedimentary Facies*	*Horizon*	*Flow Branch*	*Hydrochemical Facies*
Examples	Younger dune sand	A_0	1st order (local)	Acid, polluted
	Older dune sand	A_1	2nd order (subregional)	Acid, (sub)oxic
	Beach sand	A_2	3rd order (regional)	Neutral, anoxic
	Shallow marine sand	B	4th order (supraregional)	Ditto, deep anoxic
	Lagoonal clay	B_2		Ditto, freshened
	Basal peat	C		

Figure 1. Comparison of the mapping of geological, pedological, hydrological and hydrochemical data, with a first grouping according to the genesis or origin, and subsequent subdivision on the basis of specific characteristics (modified after Stuyfzand, 1993)

HYDROCHEMICAL GROUNDWATER SYSTEMS (WATER BODIES)

Definitions

A hydrochemical groundwater system, or hydrosome (water body; ὕδωρ = water, σωμα = body), was defined by Stuyfzand (1993, 1999) as a coherent, 3-dimensional unit of groundwater with a specific origin. Examples in Fig. 2A are: coastal dune groundwater recharged by local rain water, intruded sea water, recharged river Rhine water and polder water. Within a given hydrosome the chemical composition of water varies in time and space, due to changes in recharge composition and in flow patterns, and due to chemical processes between water and its porous medium. Such variations in chemical character can be used to subdivide a hydrosome into characteristic zones or 'hydrochemical facies', a term introduced by Back (1960). A hydrosome is therefore composed of various facies units.

Groundwater flow systems

It is very important to realize that a hydrosome is not equal to a groundwater flow system (GFS). During its initial stages, a GFS is always occupied by waters of different or another origin (Fig. 2B). The current recharge water will gradually force out the residual (relict) groundwaters or displace the original groundwater. The flow system may be called hydrochemically mature when this process is completed (Fig. 2A), i.e. when it is occupied by one hydrosome. Hydrological maturity is attained when flow has reached the steady state, and therefore generally preceeds hydrochemical maturity. Disturbances by man often lead to quick responses of the GFS by distortion, and slow, long term hydrochemical responses within the GFS, as in case of the coastal area of the Western Netherlands (Fig. 2).

DETERMINING THE ORIGIN

General aspects

The mapping of hydrosomes requires the use of environmental tracers to detect the origin of the groundwater sampled. In areas with groundwaters recharged by AR or RBF the following tracers proved to be extremely powerful, especially when combined: tritium, $\delta^2H/\delta^{18}O$, $\delta^{18}O$-Cl, Cl/Br-ratio and water temperature. Additional information sources, like the geochemical structure of the aquifer system, groundwater flow patterns and position of the groundwater samples, need careful consideration as well. In difficult cases the whole hydrochemical finger print may need to be consulted.
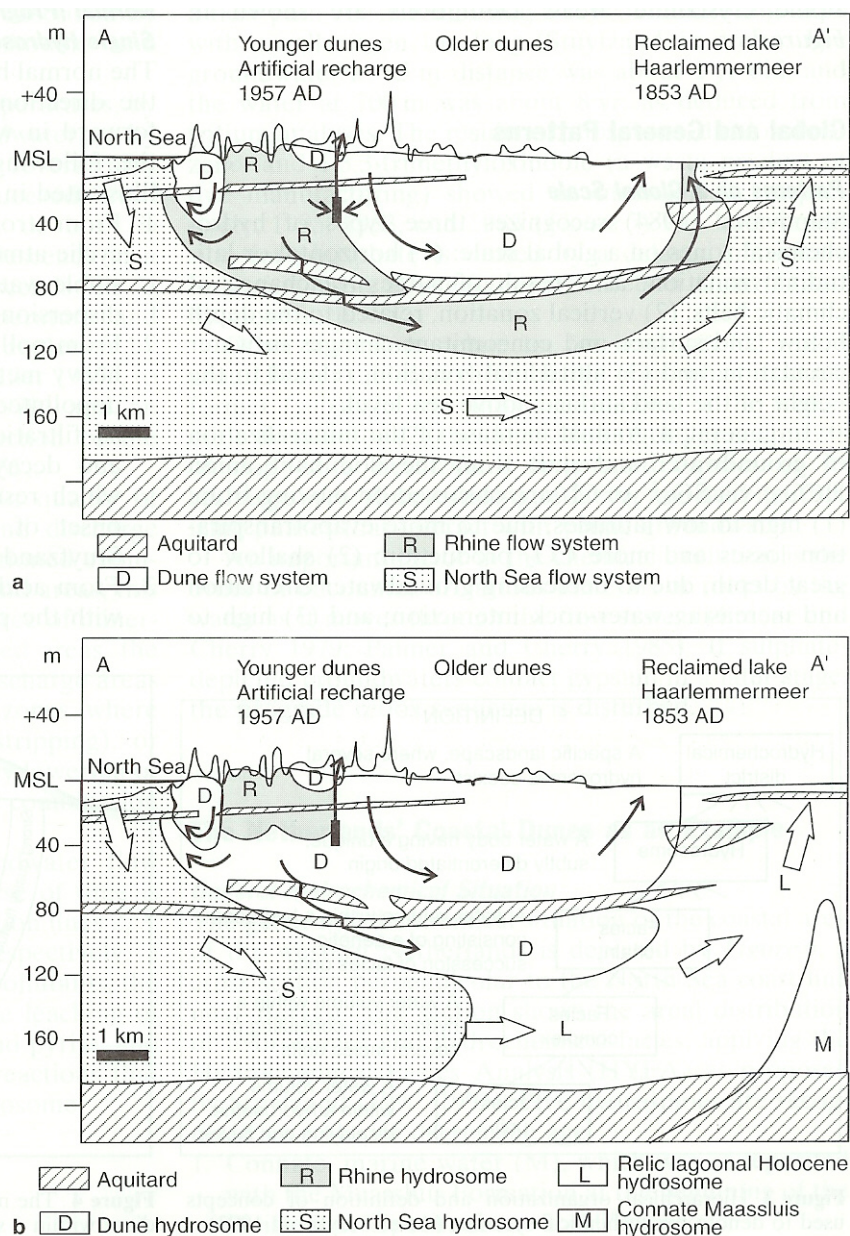
Figure 2. Groundwater flow systems normally attain equilibrium prior
to the hydrochemical system within (compare a with b). After Stuyfzand (1999).

The combi Cl and oxygen-18

The combination of two conservative tracers like chloride and ^{18}O is highly recommended, as the individual tracer may not result in a clear distinction between 2 hydrosomes, especially when the seasonal fluctuations in both overlap (Fig. 3). The anomalously low $\delta^{18}O$ value for Rhine water in the Netherlands is explained by its main provenance in Switzerland and Southern Germany, where precipitation is depleted in ^{18}O.

The Cl/Br-ratio

Many infiltrated surface waters can be recognized from groundwater recharged by local rain water, by way of their anomalously high Cl/Br-ratio, at least in coastal areas (Fig. 4). Their high Cl/Br-ratio is related to the low bromide content of salt waste discharged into many surface waters, either by salt mines or house holds. The Rhine River
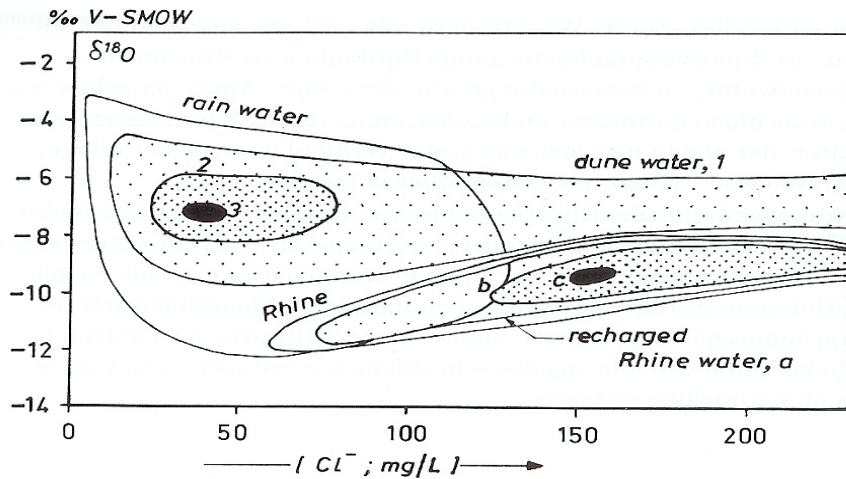
1 = circumference of plot for all 450 pure, dune groundwater samples;

2 = ditto, in vicinity of spreading area south of Zandvoort aan Zee, with subsoil detention times (t_{SUB}) of 5–15 y (100 samples = n);

3 = as 2, with t_{SUB} = 25–100 y (n = 40).

a = circumference of plot for all pure, recharged Rhine River water samples, in spreading area to south of Zandvoort aan Zee (n = 250);

b = ditto, in upper aquifer and Holocene aquitards (t_{SUB} = 3–20 y, n = 140);

c = observed, in the second aquifer (n = 5).

Figure 3.   Recognition of coastal dune water and recharged Rhine River water by their position in a $\delta^{18}O/Cl$ diagram, with variation of focus from a highly variable recharge composition (rain and Rhine River water, respectively) to a well-mixed composition in the second, (semi)confined aquifer (after Stuyfzand, 1993)
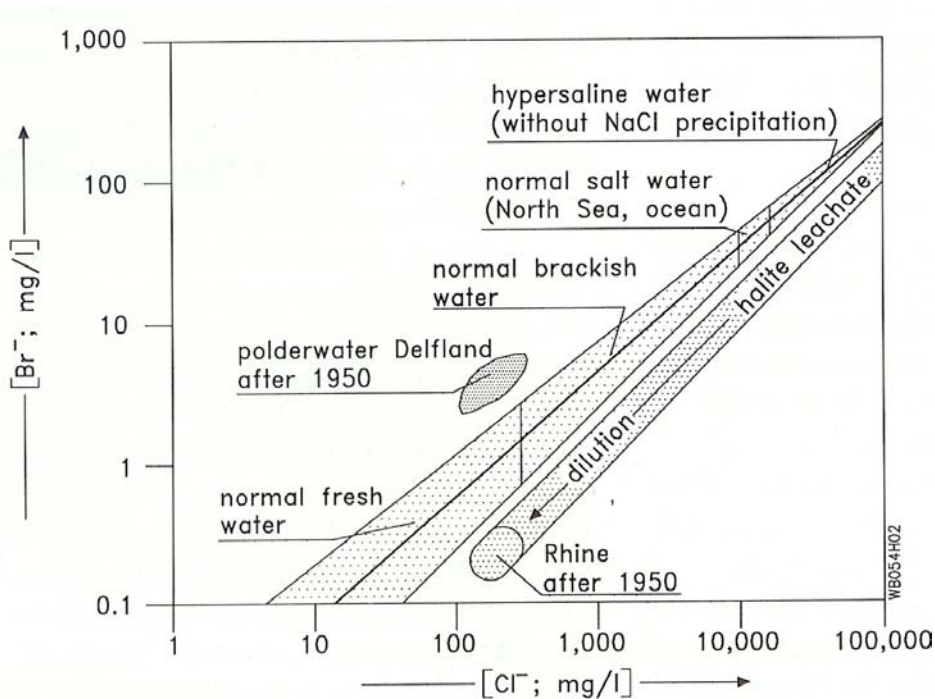

Figure 4. Cl/Br-plot for recognizing authochtonous groundwaters including brackish and salt groundwaters, amidst recharged and bank filtrated Rhine River water, as well as recharged polder water with raised Br concentration (Stuyfzand, 1993)

again poses an excellent example in the Netherlands, due to halite salt dumps into its tributary Mosel. On the other hand, characteristic low Cl/Br-ratios in the Netherlands were observed in surface water deriving from the greenhouse district near Delft. This water was recharged in the period 1970–1988 into the dunes to the south of The Hague (Fig. 5). It had high Br concentrations due to inputs from methylbromide, a soil desinfectant which, after application in the glasshouses, mineralized to bromide and leached.

DETERMINING THE HYDROCHEMICAL FACIES

General procedure

The mapping of hydrochemical zones may require to determine 4 chemical characteristics of groundwater (modified after Stuyfzand, 1999): the pH (3–5 classes), redox level (3–7 classes), pollution index (2–7 classes) and either the base exchange index (3 classes) or SAR (Sodium Adsorption Ratio). The number of classes depends on the desired level of subdivision on the map, and the diversity observed. The number of chemical characteristics (facies parameters) can be lowered or raised if necessary.

For instance, recharged Rhine River water may have the following facies: neutral (pH 6.5–8), deep anoxic (with significant SO_4^- reduction), moderately polluted (pollution index = 10–40) and with negative base exchange index (NaKMg-deficit).

Each of the 4 chemical characteristics receives an appealing code. In order to reduce the number of codes on the map, those codes are not displayed which are considered 'standard state' of the hydrochemical system. The standard state is: neutral, (sub)oxic, unpolluted and without base exchange.

The pH

One of the most important hydrochemical parameters is pH (Appelo and Postma, 2005). It determines a.o. the mobility of pollutants in infiltration water and the leaching rate of aquifer minerals. The classes discerned are listed in Table 1. They correspond with the general acid buffer sequence for multi-mineral soils (Ulrich et al., 1979).

Table 1. PH-classes as part of the hydrochemical facies determination

pH-classes	Facies descriptor	Value	Principal acid buffer	Mobility metals
B	Basic	>8.2	$CaCO_3$, $(H)CO_3$	Oxy-anions #
N	Neutral	6.2 - 8.2	$CaCO_3$, HCO_3	Oxy-anions #
a	Slightly acid	5.0 - 6.2	Al-silicates	Zn, Cd
A	Acid	4.2 - 5.0	Ca + Mg exchange	Cu, Ni
H	Strongly acid	< 4.2	$Al(OH)_3$, $Fe(OH)_3$	Al, Cr, Fe, Pb

= like arsenate, chromate, molybdate, selenate, vanadate, wolframate.

The redox level

The other, most important key variable in AR and RBF systems is the redox environment. It determines the mobility, dissolution, degradation and toxicity of inorganic and organic substances in or in contact with the water phase (Stumm and Morgan, 1981; Stuyfzand, 1998). The direct measurement of the redox potential E_H with electrodes runs into practical problems, however, and is handicapped by unreliable results (Lindberg and Runnells, 1984) or difficulties in quantitative thermodynamic interpretation (Peiffer et al., 1992). Unfortunately the same holds for its calculation from a single redox pair like Fe^{2+}/Fe^{3+} (Lindberg and Runnells, 1984; Barcelona et al., 1989).

Therefore the redox level was deduced from all redox sensitive main components of water, i.e. O_2, NO_3^-, SO_4^{2-}, Fe, Mn, NH_4, CH_4 and eventually H_2S. This led to the semi-empirical redox indexing as outlined in Fig. 5 and Table 2.
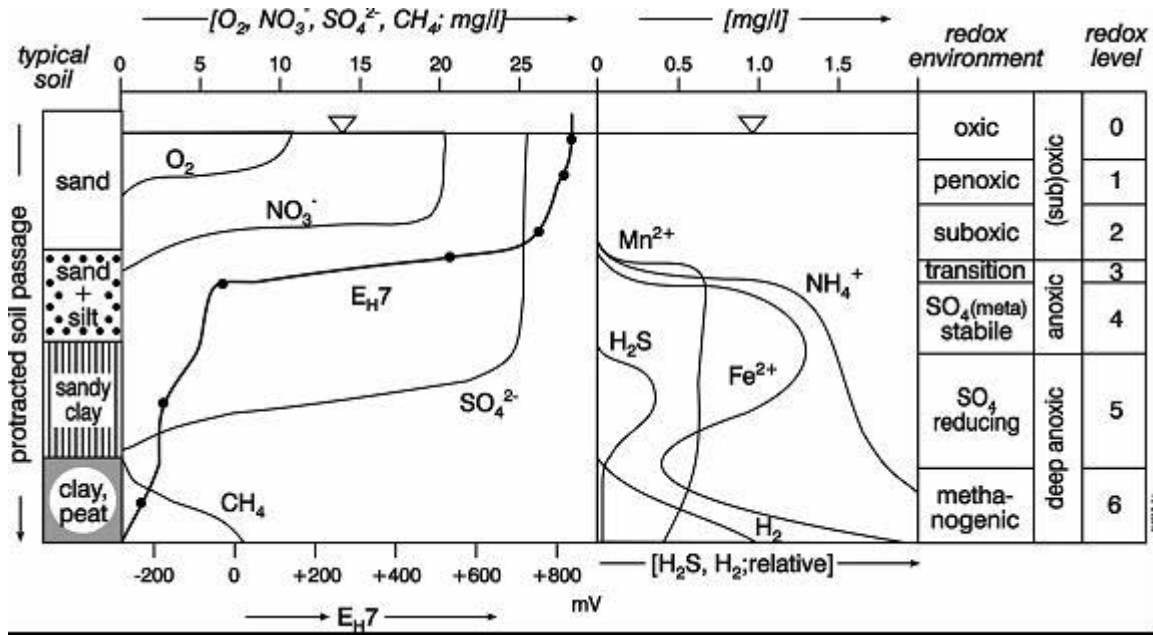

Figure 5. Classification of the natural redox environment according to Stuyfzand (1988, 1993), based on the main redox components of water : O^2, NO^{3-}, SO_4^{2-}, Fe, Mn and CH_4. Subsoil passage is assumed as piston flow in a system, which is closed from the atmosphere and progressively richer in organic carbon. The initial O^2, NO^{3-} and SO_4^{2-} concentrations (at the water table) are set at 10, 20 and 25 mg/l, respectively. The indicative redox potentials at pH = 7 (EH7) derive from Stumm and Morgan (1981).

Table 2. Practical criteria for the determination of the redox index (after Stuyfzand, 1993).
For each level the relevant redox reaction and indicative redox potential at pH = 7 (EH7) are given.
Concentrations in mg/l.

Level	Environment		Criteria		
0	Oxic:	$O_2 \geq 0.9\ (O_2)_{sat}$			
1	Penoxic:	$1 \leq O_2 < 0.9\ (O_2)_{sat}$			
2	Suboxic:	$O_2 < 1$,	$NO_3^- \geq 1$		
3	Transition:	$O_2 < 0.5$,	$NO_3^- < 0.5$,	$Mn^{2+} \geq 0.1$,	Fe^{2+}
4	Sulphate-stable:	$O_2 < 0.5$,	$NO_3^- < 0.5$,	$Fe^{2+} \geq 0.1$,	$SO_4^{2-} \geq 0.9\ (SO_4)_O$
5	Sulphate-reducing:	$O_2 < 0.5$,	$NO_3^- < 0.5$,	$0.1\ (SO_4)_O < SO_4^{2-} < 0.9\ (SO_4)_O$	$CH_4 < 1$.
6	Methanogenic:	$O_2 < 0.5$,	$NO_3^- < 0.5$,	$SO_4^{2-}\ \ 0.1\ (SO_4)_O$	$CH_4 \geq 1$.
Redox clusters:					
0-2	(sub)oxic:	$O_2 \geq 1$ or	$NO_3^- \geq 1$		
3-4	anoxic:	$O_2 < 0.5$ and	$NO_3^- < 0.5$,	$SO_4^{2-} \geq 0.9\ (SO_4)_O$	$CH_4 < 0.5$
5-6	deep anoxic:	$O_2 < 0.5$ and	$NO_3^- < 0.5$,	$SO_4^{2-} < 0.9\ (SO_4)_O$	$CH_4 > 0.5$
0-2	aerobic:	as (sub)oxic (0-2)			
3-6	anaerobic:	as anoxic (3-4) or	deep anoxic (5-6)		

$(O_2)_{sat}$ = oxygen saturation concentration, with t = temp [^{o}C] and Cl^- = chloride [mg/L]:
  = $14.594 - 0.4\ t + 0.0085\ t^2 - 97\ 10^{-6}\ t^3 - 10^{-5}\ (16.35 + 0.008\ t^2 - 5.32/t)\ Cl^-$
$(SO_4)_O$ = sulphate concentration in influent (prior to aquifer passage; can also be calculated by linear regression with chloride).

The pollution index WAPI

The Water Pollution Index WAPI is calculated according to Stuyfzand (2002) with various updates. WAPI is the square root of the kwadratic mean of subindices A-J, each of which covering a specific water quality aspect

(Table 3). Most sub-indices consist of the unweighted mean of scores for related quality parameters, on the basis of the ratio of the measured value and the natural background (NB; see Table 4). The chosen NBs can be compared with target values for groundwater in the Netherlands (LBB, 1998), maximum acceptable concentrations for drinking water (MACs according to WLB 2001, which is based on EU regulations) or MACs issued for surface water to be recharged with the purpose to recollect for drinking water supply (IB 1993).

The use of such ratio's is advantageous thanks to (a) their clear environmental significance, e.g. 3 meaning 3 times more polluted than natural background, and (b) the possibility to plot each subindex in a radar plot. WAPI can be easily adapted to embrace the available data on water quality and to relate to deviating natural backgrounds. WAPI-classes for mapping are specified in Table 5.

The Base EXchange index (BEX)

BEX has been defined by Stuyfzand (1986, 1993) as the sum of the cations Na^+, K^+ and Mg^{2+} (in meq/l), corrected for a contribution of sea salt:

$$BEX = (Na^+ + K^+ + Mg^{2+})_{measured} - 1.0716\ Cl^- \tag{1}$$

In Table 5 the following BEX-classes are discerned with interpretation as follows:

A significantly positive BEX may indicate freshening of the aquifer by displacement of a relative saline NaCl-type groundwater type through a relatively fresh $CaHCO_3$-type groundwater. In that case the cation exchange reaction given below, proceeds from left to right:

$$aCa^{2+} + [bNa, cK, dMg]\text{-}exchanger \leftrightarrow [aCa]\text{-}exchanger + bNa^+ + cK^+ + dMg^{2+} \tag{2}$$

with the meq-balance : $2a = b + c + 2d$.

BEX assumes no significant (!) other sources and sinks for these ions, an assumption which generally holds in the Western Netherlands and many sedimentary environments in temperate climates (dolomites excluded). Although there are distinct deviations from Eq. 2 and silicate weathering may constitute a source of Na, K and Mg, the sign of BEX practically always indicates the right direction of the displacement: a significantly positive value points at freshening (reaction 2 proceeds to the right), a significantly negative value at salinization, and a value close to zero at equilibrium. For a further discussion reference is made to Stuyfzand (1993).

Table 3. Determination of the Water Pollution Index WAPI by averaging quality aspects A − J (modified after Stuyfzand, 2002). Values >1 indicate that natural background levels are exceeded.

WAPI = sqrt [(A² + B² + C² + D² + E² + F² + G² + H² + I² + J²)/10]	
with:	**Example (unit see Table 4)**
A = Esthetics	(suspended solids + colour/20 + taste/2 + odour/2 + turbidity/4)/5
B = Acidity	$(pH - 7)^2$
C = Oxidation or reduction capacity	|MOC|/2.7 [me/L] with: MOC = $4O_2 + 5NO_3^- + 7SO_4^{2-} - 3NH_4^+ - Fe^{2+} - 2Mn^{2+} - 8CH_4$
D = Nutrients (eutrophication potential)	$(PO_4\text{-}P/0.02 + ? [NO_3\text{-}N + NO_2\text{-}N + NH_4\text{-}N]/0.3)/2$
E = Total salt content	$(Cl^-/12 + EC/350)/2$
F = Inorganic micropollutants (Trace elements)	(As/5 + Ba/50 + Cd/0.05 + Cr/5 + Cu/3 + Hg/0.02 + Ni/9 + Pb/4 + Zn/9)/9
G = Org. Micropollutants (excl. Pesticides)	(ΣBTEX + ΣLUMP + Σothers + ΣPAH + ΣVOCl)/5 (see Table 4)
H = Pesticides	(atra + sima + diur + αHCH + γ|HCH + HCB + bent)/0.07
I = Radioactivity	$(rest\beta/10 + {}^3H/8)/2$
J = Microbiology	([Colc₂₂/1000] + [Coli₄₄] + [Fs])/3

Note: If parameters are omitted, then the final division factor should decrease accordingly (C excluded). The natural background of each parameter equals its individual division factor.

Table 4. Preliminary survey of parameters involved in the determination of the Water Pollution Index WAPI, with their natural backgrounds (division factor in WAPI), and various water quality standards applied in the Netherlands

Quality Aspect	Parameter	Unit	WAPI applied background	Target quality Groundwater LBB, 1998	Drinking Water Standard WLB, 2001	Artificial Recharge MACs Neths IB, 1993
A (esthetics)	Colour	mg/l Pt/Co	20		20	
A (esthetics)	Odour	N	2		2	
A (esthetics)	Suspended solids	mg/l	0.5			0.5
A (esthetics)	Taste	N	2		2	
A (esthetics)	Turbidity	FTE	1		1	
B (acidity)	pH	-	7		7-9.5	
C (ox/red cap)	CH4	mg/l	4.8			
C (ox/red cap)	Fe(II)	mg/l	11		0.2 (tot)	
C (ox/red cap)	Mn(II)	mg/l	1.1		0.05 (tot)	
C (ox/red cap)	O2	mg/l	11		>2	
C (ox/red cap)	SO4	mg/l	15	150	150	150
C + D	NH4	mg/l	0.4	2.6-12.9#	0.2	3.23
C + D	NO2	mg/l	0.1		0.1	
C + D	NO3	mg/l	2.5	24.8	50	24.8
D (nutrients)	ortho PO4	mg/l	0.06	1.22–9.2	6.1 (tot)	1.22 (tot)
E (total salt)	Cl	mg/l	12	100	150	200
E (total salt)	Electr. Conductivity (20°C)	uS/cm	350		1,250	
F (IMPs)	As	ug/l	5	10	10	10
F (IMPs)	Ba	ug/l	50	50	500	200
F (IMPs)	Cd	ug/l	0.05	0.4	5	0.4
F (IMPs)	Cr	ug/l	2	1	50	2
F (IMPs)	Cu	ug/l	3	15	2 (plateau)	15
F (IMPs)	F	ug/l	250	500	1,100	1,000
F (IMPs)	Hg	ug/l	0.02	0.05	1	0.05
F (IMPs)	Ni	ug/l	10	15	20	15
F (IMPs)	Pb	ug/l	4	15	10	15
F (IMPs)	Sb	ug/l	0.05	0.01	5	
F (IMPs)	Se	ug/l	0.05		10	
F (IMPs)	Zn	ug/l	10	65	3	65
G (OMPs, BTEX)	benzene	ug/l	0.2	0.2	1 (sum BTEX)	
G (OMPs, BTEX)	ethylbenzene	ug/l	0.2	0.2	1 (sum BTEX)	
G (OMPs, BTEX)	toluene	ug/l	0.2	0.2	1 (sum BTEX)	
G (OMPs, BTEX)	xylene	ug/l	0.2	0.2	1 (sum BTEX)	
G (OMPs, Lump)	an detergents	ug/l	10			
G (OMPs, Lump)	AOX	ug Cl/l	5			30
G (OMPs, Lump)	EOX	ug Cl/l	0.5			
G (OMPs, Lump)	mineral oil	ug/l	50	50	10	200
G (OMPs, Lump)	PCB-total (n=7)	ug/l	0.5	0.01	0.5	
G (OMPs, Lump)	sum chlorophenols	ug/l	0.1		0.1	0.4
G (OMPs, others)	4-nitrophenol	ug/l	0.01			
G (OMPs, others)	benzothiazole	ug/l	0.01			
G (OMPs, others)	EDTA	ug/l	0.1			
G (OMPs, others)	MTBE	ug/l	0.01			
G (OMPs, others)	tributhylphpsphate	ug/l	0.01			
G (OMPs, PAHs)	anthracene	ug/l	0.02	0.02	0.05	0.02
G (OMPs, PAHs)	benzo(a)anthracene	ug/l	0.002	0.002	0.05	
G (OMPs, PAHs)	benzo(a)pyrene (Borneff)	ug/l	0.001	0.001	0.01	0.1(SB)
G (OMPs, PAHs)	benzo(b)fluoranthene (Borneff)	ug/l	0.001		0.05	0.1(SB)
G (OMPs, PAHs)	benzo(ghi)perylene (Borneff)	ug/l	0.002	0.002	0.05	0.1(SB)
G (OMPs, PAHs)	benzo(k)fluoranthene (Borneff)	ug/l	0.001	0.001	0.05	0.1(SB)
G (OMPs, PAHs)	chrysene	ug/l	0.002	0.002	0.05	0.02
G (OMPs, PAHs)	fluoranthene (Borneff)	ug/l	0.005	0.005	0.05	0.1(SB)
G (OMPs, PAHs)	indeno(123cd)pyrene (Borneff)	ug/l	0.0004	0.0004	0.05	0.1(SB)
G (OMPs, PAHs)	naphtalene	ug/l	0.1	0.1		0.1
G (OMPs, PAHs)	phenanthrene	ug/l	0.02	0.02	0.05	0.02
G (OMPs, PAHs)	pyrene	ug/l	0.05		0.05	
G (OMPs, VOCl)	chloroform	ug/l	0.1	0.01	25 (sum THM)	
G (OMPs, VOCl)	sum dichlorobenzenes	ug/l	0.01	0.01	1	
G (OMPs, VOCl)	trichloroethene	ug/l	0.01	0.01	10 (+tetra)	0.5
H (Pesticides)	alpha-HCH	ug/l	0.01	0.01	0.1	0.05
H (Pesticides)	atrazine	ug/l	0.01	0.0075	0.1	0.1
H (Pesticides)	bentazone	ug/l	0.01		0.1	0.1
H (Pesticides)	diurone	ug/l	0.01		0.1	
H (Pesticides)	gamma-HCH (lindane)	ug/l	0.01	0.00002	0.1	0.05
H (Pesticides)	hexachlorobenzene	ug/l	0.01	0.01	0.1	0.05
H (Pesticides)	simazine	ug/l	0.01		0.1	0.1
I (Radioactivity)	total alpha	mBq/l	0.1		0.1	
I (Radioactivity)	rest-beta	mBq/l	10		1	
I (Radioactivity)	tritium	TU #	8		840	
J (Microbiology)	colony counts 22°C	CFU/l	1,000			
J (Microbiology)	faecal streptococs	CFU/l	1			
J (Microbiology)	SO3-reducing clostridia	CFU/l	1			
J (Microbiology)	thermotolerant coli counts	CFU/l	1			

= 1 TU (Tritium Unit) = 0.119 Bq/L

Table 5. The various hydrochemical facies descriptors and their code. If standard state, then the code is omitted for simplicity. In the examples of section 5 the WAPI-classes are combined into unpolluted (WAPI < 10) and polluted (WAPI >10).

Facies Code	Facies descriptor	Value	Standard state
pH-classes			
N	Neutral / basic	>6.2	yes
a	Slightly acid	5.0 - 6.2	no
A	Acid	< 5.0	no
Redox			
o	(sub)oxic	0 - 2	yes
r	reduced (anoxic)	3 - 4	no
d	deep anoxic	5 - 6	no
WAPI			
u	unpolluted	0 - 2	yes
b	bit (slightly) polluted	2 - 10	no
m	moderately polluted	10 - 40	no
p	polluted	>40	no
BEX			
f	freshened	> +(0.5 + 0.02Cl)	no
e	equilibrium	0	yes
s	salinized	< -(0.5 + 0.02Cl)	no

EXAMPLES

The coastal dune aquifer system in the Western Netherlands

Along the North Sea coast, in the dune area in between Hoek van Holland and Zandvoort (W-Netherlands), there are 5 different sites with artificial recharge of surface water (numbering as in Fig. 6):

1. Monster: spreading started in 1970 with untreated polder water. Since 1983 coagulated Meuse River water is used (ca 5 Mm^3/a). The areal extension is limited because of an effective recovery system and an aquitard with high hydraulic resistance at 20 m-MSL (Mean Sea Level).

2. Scheveningen: spreading started in 1955 using filtrated Rhine River water and since 1976 coagulated Meuse River water (52 Mm^3/a). The spatial extension is relatively large due to a high overinfiltration (recharge > recovery), a less effective recovery system and lack of resistant aquitards.

3. Katwijk: the AR operation started in 1940 using untreated polder water, and since 1988 coagulated Meuse River water (20 Mm^3/a). The spatial extension is limited due to underinfiltration (recharge < recovery), and the presence of a resistant aquitard at 4 m-MSL. Meuse River water is not yet present on site 3 in Fig. 6, due to the period chosen for the map (1980–1990).

4. Zandvoort South: spreading strated in 1957 using filtrated Rhine River water and since 1974 coagulated Rhine River water (57 Mm^3/a). The spatial extension is limited due to underinfiltration (recharge < recovery), an effective recovery system and the presence of a resistant aquitard at 16 m-MSL.

5. Zandvoort North: since about 1900 untreated sewage water was spread in local dune depressions. This changed in 1964 into a more controlled spreading of pretreated sewage effluent via basins (1 Mm^3/a). The spatial extension is relatively large due to lack of a nearby recovery system. The recharge area is at about 1.5 km per pendicular to the section in Fig. 6 and the discharge is through pumping wells in the section, which explains the curious shape of this hydrosome.

All water bodies in Fig. 6 are pH-neutral (and calcite saturated) due to a high $CaCO_3$ content of the dune and underlying beach sands. The recharged polder waters deviate from the Rhine and Meuse River waters by their freshened and reduced character (AP_{FR}) already at their starting point in the upper aquifer. This is due to a primary Na+K+Mg-excess of this surface water which receives seepage of dune groundwater displacing relict brackish groundwaters. Their initial reduced facies is caused by very high nutrient loads which led to much sludge formation and thus consumption of all available O_2 and NO_3. Downgradient this AP_{FR} water looses its primary freshened character (positive BEX) by exchange for Ca (site 3).

The general hydrochemical evolution of Rhine River water downgradient, in the period 1980–1990, is: from polluted (thus pH-neutral, (sub)oxic, BEX-equilibrated), to polluted and reduced, to polluted, reduced and salinized (site 4). The salinized facies is due to displacement of fresher dune water, having relatively low Na, K and Mg concentrations.

The evolution of Meuse River water downgradient (area 2), in the period 1980–1990, deviates from the one for Rhine water by a freshened instead of salinized facies. This is due to displacement of saltier Rhine River water, having relatively high Na, K and Mg concentrations.

The improved quality of Rhine and Meuse River water, notably since the early 1980s, has contributed to a significant decrease of their pollution index WAPI. This also holds for the upper infiltrated waters in the dunes since the 1990s, thus with a significant delay. This delay is due to (a) the slow leaching of pollutants that accumulated in both sludges and the aquifer system, (b) abolition of the chlorination for transport (from pretreatment plant to infiltration area) since about 1990, and (c) removal of the older, relatively polluted sludges from the basins mainly in the 1990s.
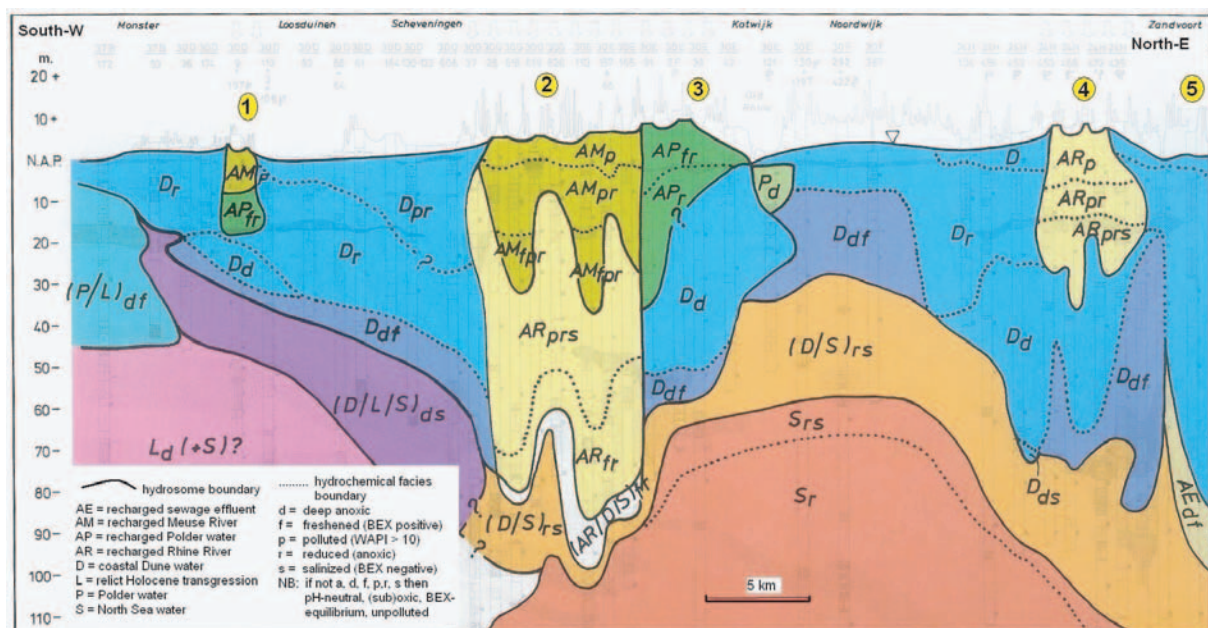

Figure 6. Spatial distribution of hydrosomes (groundwater bodies) and their chemical facies (period 1980–1990) along a section parallel to the North Sea coast, in between Hoek van Holland and Zandvoort (slightly modified after Stuyfzand, 1993). For characteristics of recharge sites 1–5 see text.

The Rhine fluvial plain in the Netherlands

The spatial distribution of Rhine bank filtrate, in its fluvial plain in the Netherlands (Fig. 7), is dictated by the exfiltration of fresh groundwater from ice-pushed hills (in between well fields 25 and 45), the pumping of groundwater
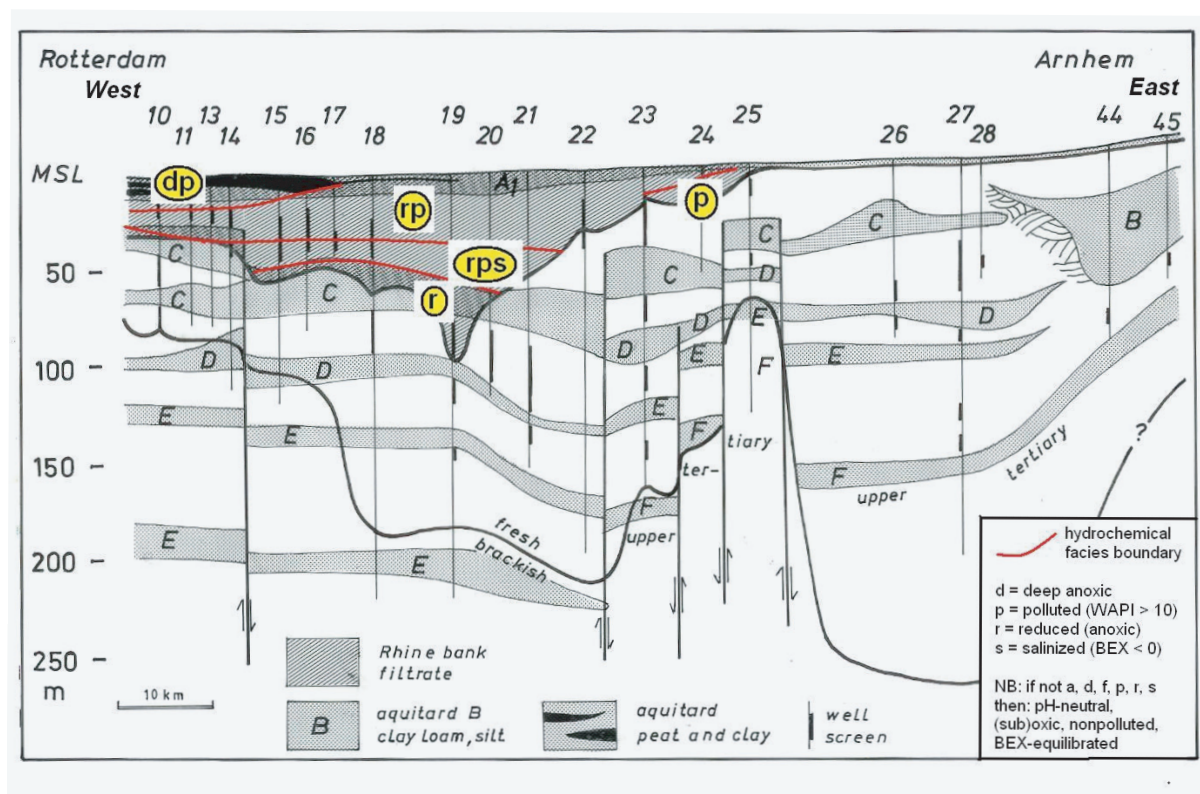
*Figure 7. Longitudinal west-east section of the Rhine fluvial plain in the Netherlands,
along the tributary Lek with the spatial distribution of Rhine River bank filtrate and its hydrochemical facies
anno 1980–1985 (modified after Stuyfzand, 1989). 10–45 = well fields for drinking water supply.*

for drinking water supply (especially in between well fields 25 and 20), and the low polder levels behind the river dikes from well field 19 downgradient to well field 10. Low polder levels force the river there to infiltrate even when pumping wells would be lacking.

The hydrochemical facies evolves more or less as follows. Just below the upper aquitard, the redox index shifts from (sub)oxic in the east to deep anoxic in the west, due to increased contact with a thickening aquitard gaining also in organic material content. Note that nearly all well fields tap, at shallow depth, more reduced river bank filtrate than at greater depth. This is caused by an increased contribution, at greater depth, from the fairway which partly is cutting through the upper aquitard.

At greater depth the facies may change from BEX-equilibrated into salinizing where relatively fresh groundwaters are displaced by relatively salty Rhine River water (if infiltrated after 1910 ca.).

Most Rhine bank filtrates that infiltrated after 1950 are polluted (WAPI > 10).

CONCLUSIONS

Maps can be made with the method presented, to display the spatial distribution of groundwater bodies with a specific origin (the *hydrosomes*, like North Sea, recharged Rhine, Meuse or polder water, and dune water), and characteristic hydrochemical zones (the *facies*) within each hydrosome. Such maps are especially useful when reporting about the impact of MARS related activities and when reports are needed in consequence of the EU Water Framework Directive.

Groundwater monitoring programs for drinking water supply, environmental protection and nature conservation benefit from such maps in at least two ways (Stuyfzand, 1999): (1) by selecting the strategical positions for monitoring wells downgradient of expanding hydrosomes and downgradient of facies boundaries; and (2) by composing the right analytical program (specific trace elements in acidified systems only, and specific organic microcontaminants exclusively in for instance young, (sub)oxic waters or recharged fluvial water).

The areal distribution of hydrosomes with artificial recharge or river bank filtration offers ideal possibilities of independent chemical verification of hydrological models. MARS systems also offer ideal possibilities to study the behaviour of selected pollutants in field conditions (including natural heterogeneities for instance regarding redox environments, and with sufficiently long flow distances or time scales) while the composition of the input is often well known.

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Wastewater reuse and potentialities for agriculture in Nigeria

Anthony Johnson Akpan

INTRODUCTION

Although the awareness of the impact of human activities on the environment, especially on water resources, and the resulting threat for both nature and man is growing, relatively few effective bridges to properly address water problems are being built in developing countries. An effective and important bridge to be built concerns water reuse for food production. The paper draws attention to the growing importance of wastewater reuse as an essential part of the planning of the integrated and sustainable water resources management can be evidenced. It looks at the proposed pilot programme of North South Development (NSD), a Sub Sahara African and Middle East focused development organisation 'Rural Wastewater Treatment' in the South West Region of Nigeria.

Wastewater has been used by mankind in agriculture since the beginning of modern civilization. Treated wastewater has a fundamental role in some sustainable management and planning of water resources as a substitute to the water that would be used in irrigation for agricultural purposes. Thus, if implemented, wastewater reuse in irrigation could increase food production with the purpose of improving living conditions of the inhabitants of several parts of Nigeria and consequently lead to mitigate poverty. At the same time, wastewater reuse plays an important role in environmental and water resources conservation. This practice release good quality water reuse adds an economic dimension to water resources planning. In this perspective, the need for sanitation, wastewater treatment and it's reuse in agriculture for food production are presented in the paper as decisions to be made in Nigeria to build bridges with the objective to promote sustainable development, with reflexes on the country's economic, political and environmental scenes. Water reuse reduces the water demand on water bodies due to the substitution of potable water for water with inferior quality. This practice has been put in evidence and discussed intensively in many countries as well as it is used in several part of the world. It is based in the conception of source substitution. Such substitution is possible in function of the required quality for a specific use. Therefore, Large volumes of potable water could be saved Due to reuse when water with inferior quality is used (generally post-treated effluents) to supply the needs that can renounce water within the potability standards framework. The growing demand for water has made planned wastewater reuse an update issue of growing importance. Thus, wastewater reuse must be considered as part of a wider activity that is the rational or efficiecnt water use. This also concerns the control of water loss ans wastefulness, the reduction of waste production and water consumption. Brazil happens to be a county where nature was very generous as far as water availability is concerned. There is about 8% of all the world fresh water in this country. Although this figure seems very impressive, the relatively abundant Brazilin water resource are facing dangerous threat: population growth, environmental problems, lack of rigid environmental legislation or most times, difficulties even to apply the existing legislaton. The National Congress has been approving modern legislation concerning water resource management policies since 1997 and fortunately many reforms in the water resoures sector are starting to take place. In a country with the Nigerian conditions, so vast and diverse, such reforms may take a long time to be fully implemented .

Agriculture consumption of water

Water conservation is a critical issue throughout the world. In Sub Sahara Africa and Nigeria more specifically, water issues significantly impact the countries efforts in National Food Security and sustainable Livelihood

improvement initiatives. In a country of 130 million plus people and in the size of the State of Texas in the US, Nigeria represents a challenge unlike any other in the world. See Example 1 showing Nigerian water requirements for main food production. This does not even reflect the domestic and drinking consumption volumes required for a nation of this size.

Example 1. Water requirement equivalent of main food production

Product	Unit	Equivalent water in cubic meters
Cattle	Head	4,000
Sheep and Goats	Head	500
Meat Bovine fresh	Kilogram	15
Meat Sheep fresh	Kilogram	10
Meat Poultry fresh	Kilogram	6
Cereals	Kilogram	1.5
Cirtus fruit	Kilogram	1
Palm Oil	Kilogram	2
Pulses, Roots and Tubers	Kilogram	1

Source: FAO, 1997b.

This table gives examples of water required per unit of major food products, including livestock, which consume the most water per unit. Cereals, oil crops and pulses, roots and tubers consume far less water.

Poverty alleviation

For more than 5 decades we have been acutely aware of the importance of sound agro-economic and farm management implications. In early 1957, Professors Goldberg and Davis of Harvard Business School developed the concept of agribusiness as the panacea for addressing the problems of poverty and lack of access to food and basic necessities of life.

There is no better time than now to address this issue but this time in a more sustainable manner. We need our efforts to focus on water treatment facilities, water capturing mechanisms, irrigation systems, training farmers in both farm technologies and economic farm management. Additionally, we need to train people in sustainable environmental development and health awareness to deliver a more rounded socio-economic programme of development. This must be coupled with ongoing research and education in these areas for future generations.

Efforts thus far

Despite the overwhelming efforts being made by key players: National Universities (University of Abeokuta and the University of Ibadan), FAMAN, IFAD, the CBN, Research institutions (International Institute of Tropical Agriculture (IITA), the State Agriculture Development Programmes (ADP's) and the respective State and Local Governments in Farm Management, Agro-Economics, Water Treatment and Irrigation, more must still be done to further develop the sub regions capacity in water resource management for agriculture to survive.

We need to carefully plan and implement low cost and low technology water treatment facilities, improve access to clean water and increase its utilization within farm irrigation.

The rural sector in Nigeria has suffered from decades of neglect in terms of sewage infrastructure and wastewater management, treatment and reuse resulting in declining productivity, growth and competitiveness. This problem is

profoundly exasperated as the demand for domestic water increases due to high urban populations, increasing birth rates and moves to raise living standards.

Nigeria is richly endowed with a wide range of resources including a large, well-educated population of approximately 130 million, land and water resources, oil and gas as well as mineral deposits. The economy depends to a large extent on oil revenue, however agriculture remains the mainstay of the economy as it accounts for 40% of GDP, employs over 70% of the workforce and provides 90% of non-oil export earnings. (UNDP Federal Republic of Nigeria Poverty Reduction Programme)

A way forward

It is in this context that North South Development (NSD), a Sub Sahara African and Middle East focused development organisation initiated this programme 'Rural Wastewater Treatment' in the South West Region of Nigeria.

It is undeniable that sustainable development is interrelated to wastewater treatment and water reuse, especially in Sub Sahara Africa. Policies and practices to develop economic dependency and development can be successfully promoted through regional and international cooperation of which this programme exemplifies.

In Israel, where the scarcity of water effects the entire Middle East region, it has been proven that agricultural diversification into crops, irrigated using treated wastewater, can be more economically attractive and yield higher prices. This invariably benefits the local socio-economic condition, alleviates poverty and increases food security.

TECHNICAL DESCRIPTION OF THE PROPOSED PROJECT

The SCFEER wastewater treatment model – Introduction

The common 'Israeli Model' of rural area WWTP's (Wastewater Treatment Plants) is based upon a series of treatment/ stabilization ponds. This system is simple, yet effective, and is suitable for Sub-Saharan African countries because of several reasons:

- Need for water for agriculture.
- Plenty of sunlight and mild to high temperatures all year round.
- Land area is not a limiting resource.
- Need for WWTP's based upon simple technologies.
- Need for protection of water resources – by preventing potential pollution by non-treated wastewater (WW).

Based upon 15 years of experience in operating a full scale WWTP, of 1,500 m³/day capacity, the SCFEER (Sakhnin Center for Environmental Education and Research), belonging to the Towns Association for Environmental Quality (TAEQ) of the Beit Natufa Basin, had developed an improved and innovative WWT model, consisting of the following series of ponds and processes (see attached scheme of the 'SCFEER WWT Model'):

1. A plastic covered anaerobic settling pond;
2. A pond acting as a Trickling Bio-Filter (TBF);
3. A Constructed Wetland (CWL) pond;
4. Water reservoir (for the period when irrigation is not needed);
5. Chlorination;
6. Irrigation.

The quality of effluent produced by this technology is suitable for non-limited irrigation (e.g. irrigation of non-cooked eaten vegetables).

The use of the effluent resulting from the SCFEER technology of WW treatment (WWT) not only saves water and protects the environment, but also recycles the nutrients contained in the WW (e.g. N, P, K and micro-nutrients). Thus, the use of treated WW is expected to reduce the farmers' expenses for fertilizers.

General objective

North South Development, in a joint venture with the CEER and various Nigerian partners proposes to design the construction and operation of a **demonstration pilot plant** of WWTP in the South West Region of Nigeria.

Specific objectives

Construction and operation of a **demonstration pilot – plant (DPP)** of a WWTP that will be constructed and operated in Nigeria. The DPP will treat about 100 m^3 of WW per day that will be used for irrigation of adjacent agricultural fields.

1. Operation of the DPP and monitoring by both, academy and agronomy local experts, backed by NSD and by SCFEER experts.
2. Determining of optimal design and operation parameters, fitted for local conditions.
3. Design of **full scale WWTP projects** for Nigeria and in other Sub-Saharan countries, with the support of local and of international funding.

Technical specifications, dependencies and land requirements

1. The DPP requires daily supply of about 100 m^3 of piped fresh domestic WW.
2. Land requirement of the DPP is about 6,000 m^3.
3. Irrigated Agricultural fields should be available close by to the DPP for application of the treated effluent.
4. Technical manpower: skilled technician for operation and maintenance of the project, capable of operating field instruments and monitors and capable of reporting.
5. Location: accessible for visits of academic/professional supervisors (e.g. agronomists, engineers etc.).
   (Remark: the project site should be guarded 24 hours a day to prevent theft).

Work plan and timetable

Stage	Period (Months)
General design and detailed design for local conditions	3
DPP construction – soil, civil eng. and piping works	6
DPP construction – pumps and solar system installation	3
Project running in	3
Controlled and monitored operation + reporting	24
Total	24

SCFEER MODEL' OF RURAL AREA
WWTP (WASTEWATER TREATMENT PLANT)
For 100 m³/d in AFRICA SUB-SAHARAN REGION
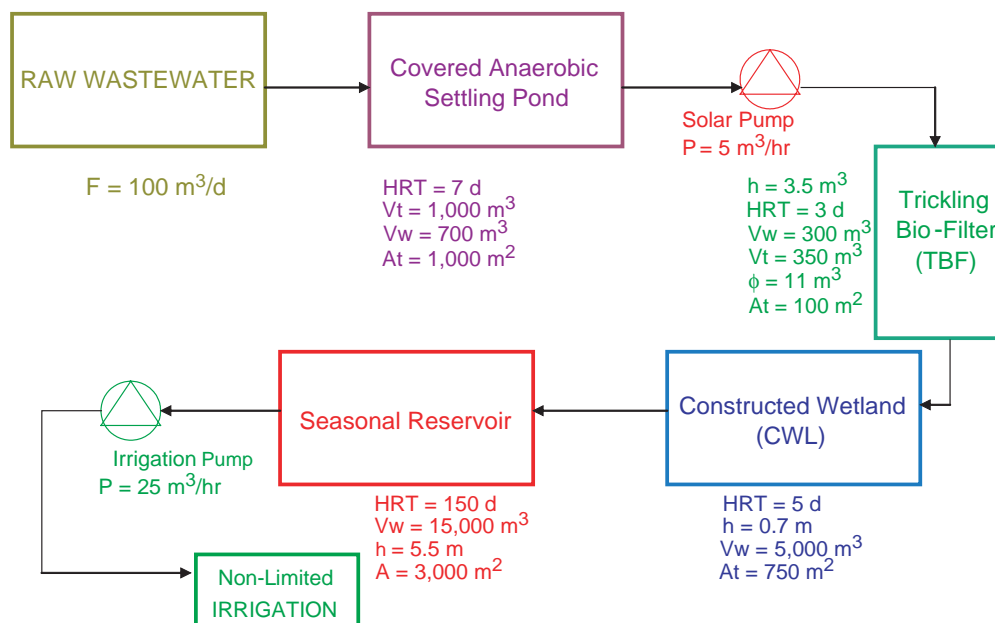
Legend:

At = Area (total)
Φ = Diameter
F = Daily Flow
h = Height of reactor (or depth of pond)
HRT = Hydraulic Retention Time

P = Capacity
SCFEER = Sakhnin Center for Environmental
                 Education and Research
Vt = Volume (total)
Vw = Volume (working)

CONCLUSIONS

The paper demonstrates that attention must be driven to the fact that the use of treated wastewater for food production would promote countless environmental benefits and build bridges to sustainability in Nigeria and other developing countries. It would promote the treatment of wastewater in Nigeria – which now barely accounts for 10% of the total effluent from domestic and industrial discharges. It would save potable freshwater for nobler uses, especially in those regions where water is scarce, as it is the case of the semi-arid region of the Nigerian Northeast. Moreover, it would abate the environmental impacts caused by the discharges of untreated wastewater into water bodies in Nigeria. The tools to start a dialogue would this way be ready. It is of utmost importance that the concerned authorities of this country interact with the affected population in order to build bridges to dialogues with the organized civil society, rural and urban users, food producers and the water managers themselves. At the moment when a new model of water resources management takes place in Nigeria, The Federal Ministry of Water Resources starts its activities full of responsibilities to organize the water sector in the country facing a variety of challenges. Thus it is demonstrated that it is now the moment to start building bridges and coordinating the dialogues. These bridges would lead to listening of the different concerned sectors perspectives, perceptions and expectations for an effective water resources management, efficient water use and a less inequitable society where all have the right to access to proper water, sufficient food and live a life with dignity and less social disparities.

Protecting an island aquifer by using recycled water as a hydraulic barrier – a case study of Bribie Island

Christopher Pipe-Martin

Abstract

Bribie Island is a sand island with a shallow unconfined aquifer that is used as a potable water source. Water abstraction causes localised lowering of the water table with the possibility of seawater intrusion. Recycled water is used to create a hydraulic barrier by discharge to infiltration basins between the water extraction area and the nearby coast. Modelling has demonstrated the creation of an effective barrier, predicted recharge flow paths and aided development of a monitoring program to assess the effect of recharge on groundwater quality. Borehole monitoring traces the recharge water plume by electrical conductivity changes. Apart from small changes in total nitrogen content there is no statistically significant difference in groundwater nutrients and microbiological indicators in the recharge area when compared to a control site.

Keywords

Aquifer; recharge; intrusion; groundwater; modelling.

INTRODUCTION

Bribie Island is in the southeast of Queensland, Australia. The island supports a permanent population of 15,000. It is supplied with reticulated water from a local water treatment plant that uses ground water from the island's aquifer. Bribie Island covers an area of 144 km^2 with the majority of the island having an elevation of less than 5 m. Population centres are confined to the southern third of the island with the remainder being national park and forestry areas. The island has a subtropical climate with mean maximum temperatures of 29°C in summer and 20°C in winter. Rainfall can occur throughout the year but the majority falls during the summer months of December to March. Mean annual rainfall is 1,330 mm.

Bribie Island is composed of Quaternary sand deposits overlying clay or sandstone; there are no sandstone outcrops. The sand deposits in the southern section comprise Holocene accretion ridges and swales. The total thickness of the Holocene sand deposits in the southern section of the island varies from 5 to 25m. Ground water occurs in as unconfined aquifer in the sand deposits which is recharged by rainfall and depleted by evapotranspiration, seepage around the islands perimeter and extraction by residents and the local water authority.

In 1962 a reticulated water system was provided for the Island's residents, which drew water from bores in a water reserve in the south-eastern section of the island. Due to problems with screen clogging in the production bores an artificial lagoon was created in 1971 by digging a trench into the water table. This trench has been has been extended as demand increased from 0.275 million m³/year in 1972 to 1.3 million m³/year in 2004. Due to the proximity of the water extraction trench to the southern coast there has been concern that local lowering of the water table may result in seawater intrusion into the freshwater aquifer. These concerns have restricted the production of potable water.

AQUIFER MANAGEMENT

Creating a hydraulic barrier

In 1974 a sewage scheme was started which now services all the populated areas of the island. Rather than dispose effluent to an ocean outfall, treated effluent has been used to recharge the aquifer between the water extraction trench and the southern coast with the intention of creating a hydraulic barrier to the southward movement of freshwater and any potential northward infiltration into the aquifer by seawater. Initially recharge was through a single infiltration basin located south of the sewage treatment plant. Recharge water for the project is sourced from the Bribie Island Sewage Treatment Plant, located on the western side of the water reserve. In 1994 the sewage treatment plant was upgraded to a Biological Nutrient Removal process with a maximum discharge of 3.5 million m³/year. Four new infiltration basins were constructed in 1995 between the water extraction trench and the coast. In 2001 further upgrades to the treatment plant included a polishing stage using sand filtration and UV disinfection.

Modelling the aquifer

A steady state model was produced in 1983 by Isaacs and Walker in order to better understand the behaviour of the groundwater resource and make the most efficient use of it. As the aquifer is broad and flat – its width is 300 times its thickness, the equation used assumed a two dimensional horizontal flow for groundwater based on Davey's Law and conservation of mass. The steady state model had a number of limitations. Calibration of the model was used to give the best fit between predictions and observed values. An annual recharge of 300 mm/year and hydraulic conductivity of 25 m/day gave a good fit and were close to the middle of the estimated ranges. The head at the southern boundary of the aquifer was adjusted to compensate for local effects and improve the match of model and field contours. The steady state model showed that a significant proportion of the water extracted comes from north of the water reserve and that extraction rates could be increased if recharge was used to maintain water table levels in the south eastern portion of the water reserve

In 1996 Caboolture Shire Council commissioned John Wilson and Partners (JWP) to produce a ground water model and monitoring program to satisfy the requirements of the Queensland Department of Environment. The model used was a modular three-dimensional finite difference ground water flow model (MODFLOW) developed by McDonald and Harbaugh. Grid spacings of 50 m were used. At each grid point the natural surface level above mean sea level and the depth of the impermeable rock base below sea level were entered. Canals on the Island were assigned a top water level of 0.4 m as this is maintained by a lock system. The model assumes a homogeneous unconfined sand layer, which is true for the water reserve area, uniform rainfall distribution with a recharge rate of 26 percent of monthly falls and constant extraction and application rates by the golf course. The transient model used monthly data for water extraction and artificial recharge from the local water authority, CabWater. A variable hydraulic conductivity was used at the water extraction trench. The infiltration basins were modelled as injection wells, taking into account evaporation rates. A steady state run of MODFLOW was conducted and calibrated. A transient simulation from January 1993 to October 1995 was then carried out with a stress period of one-month intervals. Nine bores with the most reliable and continuous readings were used to calibrate the model. Five bores gave good matches between modelled and measured values. Four bores gave shapes matching field data but results that were higher than measured values. As in the earlier model of Isaacs and Walker the seawater wedge was not included due to poor data on its actual extent.

The model indicated a significant and effective hydraulic barrier existed between the coast and the freshwater extraction trench. The hydraulic gradient between the infiltration area and the water trench indicated the potential for movement of recharge water into the potable water extraction area.

Calibration and interpretation of the modelling results led to a number of recommendations including monthly testing of water table height and water quality in existing bores and construction of additional bores in the immediate vicinity of the infiltration basins and at the coastal boundary of the aquifer. Additionally it was recommended that the model is run and calibrated each year and that monthly ground water quality testing of the water reserve is conducted to determine if recharge is adversely affecting the aquifer.

Existing water quality data of the water extraction trench and bore holes assessed by JWP did not indicate contamination of the water extraction trench. Additional test parameters including total nitrogen were recommended as useful for monitoring water quality. Although it was not considered likely that bacterial pathogens would survive or travel far through the aquifer it was recommended that indicator bacteria should be measured at the coastal boundary where the aquifer discharges to the ocean to demonstrate compliance with environmental guidelines.

In 1999 John Wilson and Partners (JWP) were again engaged to report on ground water quality monitoring and whether aquifer recharge contributed to a minor flooding event at the Bribie Island Golf Course. The transient model was recalibrated with data collected between April 1998 and October 1999. Surface and bedrock levels were refined using updated information. Problems with modelling water extraction were addressed by assigning trench cells with an extremely high hydraulic conductivity (5,000 m/day). The eastern most trench cell was assigned as a well with extraction occurring at this point. For modelling purposes the recharge flow was split evenly over the four basins for each month. Seepage from cells was not modelled due to the minimal runoff from the Island. The model area was split into five zones based on geological conditions and the conductivity of each zone calibrated. The calibrated hydraulic conductivities were then used to calibrate for specific yield over the data period. The transient run used an effective porosity of 0.35 and specific yield of 0.08. The calibrated parameters appear to simulate the ground water regime well except for the peak rainfall events that produced flooding. This is most likely due to the lack of a recharge model and surface water effects during flooding.

Modelling results contributed to the conclusion by JWP that the golf course flooding was attributable to prior wetting of the soil profile by rainfall, a localised ground water mound in the vicinity of the infiltration ponds, surface flow effects and reduced evapotranspiration and increased recharge on the golf course caused by clearing and irrigation.

Groundwater quality, JWP report

Groundwater monitoring has been considered essential to the assessment of the recharge project as the aquifer is used as a potable water source and because water is released from the aquifer along the environmentally sensitive south-eastern coast of the island. The JWP study identified flow paths for water entering the aquifer via the disposal ponds. Bores 3, 4, 5, 28, 32 do not lie on flow paths and were considered remote enough to be considered background bores. It was estimated that recharge water from the basins requires four years to reach the ocean by the shortest route and 12 years to reach the water trench by the shortest route. The flow paths indicated that bores GC2 and CB1 could be used to assess water discharging to the ocean. On the basis of 50 percentile values, GC2 was assessed as not being affected by recharge with all parameters similar to background bores. CB1 showed evidence of oxidised nitrogen uptake and ammonia release by vegetation. The conductivity and pH of this bore were not consistent with other bore results probably due to sea spray contamination of soil and subsequent leaching into ground water.

Results from GC1 and TWB1 were used to assess water quality of recharge water moving toward the trench. With the exception of conductivity water quality in GC1 and TWB1 was similar to background levels and considered to be unaffected by recharge. Data from the water extraction trench indicated most parameters were similar to background values. Conductivity and pH values were higher than background bores due to evaporation, atmospheric gas exchange effects and differing water quality from northern areas feeding into the trench. Coliforms are

effectively completely removed from water recharged to the basins however the trench contains significant numbers due to wildlife access.

EFFECT OF RECHARGE ON GROUNDWATER QUALITY 2001 TO 2004

In accordance with the recommendations made by JWP the water quality monitoring program was amended to include new bores to monitor recharge water at the infiltration basins and the coastal boundary and revised parameters. To assess the influence of artificial recharge on groundwater the water quality data collected between 2001 and 2004 was analysed. Flow paths predicted by the transient model suggested BI3, BI4, BI5, BI28, or BI32 could be used as background bores. BI32 was chosen as the best control site due to its remoteness from flow paths. Water quality data was tabulated for BI32 for each year from 2001 to 2004. A two-tailed t test was performed on data from each year using α of 0.05 and assuming unequal variance. The results of these tests showed significant variation between years for electrical conductivity (EC25), making it appropriate to compare sites on a year-to-year basis rather than aggregate results over the whole study period.

The bore sites analysed were the control site BI32, BI 4 which is remote from predicted recharge flow paths, BI3 and BI28 on the edge of predicted flow paths, TWB1adjacent to the infiltration basins, CB1 on coastal boundary of aquifer, GC1 north of the infiltration area on southern boundary of golf course, and BH 15 which is north of the infiltration basins between the golf course and water extraction trench. Data for pH, conductivity, total nitrogen and total phosphorus were compared to the control site for each studied bore as well as water used for recharge for each year between 2001 and 2004. The statistical significance of variation between means was assessed by applying a two-tailed t test with α of 0.05 and assuming unequal variance.

- *Conductivity*

Mean electrical conductivities for the studied bores are contained in Table 1, figures in bold are statistically different to the control site BI32. All the sites on the predicted flow paths showed statistically significant differences to the control site for conductivity for each year studied. A mixing plot is obtained typical for a conserved parameter with concentration decreasing steadily with distance from the basins. Assuming the natural aquifer EC25 to be equal to the remote site BI32, the proportion of recharge water in the aquifer can be calculated for a particular site. BH 15 is on the most direct flow path between the infiltration basins and the water extraction trench being 800 m north of the basins and 200 m south of the south arm of the trench. Calculations indicate that for the years studied between 15 and 39% of the water at BH 15 was from recharge. These figures are subject to confounding effects due to BH 15 being on the golf course boundary and are the maximum recharge water component. The pH of water at this site indicates that golf course management does have a significant effect on water quality at BH 15. Some of the EC25 variation from background levels is likely to be due to golf course effects. A similar analysis of conductivity at the coastal boundary (CB1) indicates that between 51 and 100% of water in the aquifer at the southern boundary is recharge water. Once again these are maximum components, as it would be expected that salinity in this area is affected by leaching of sea spray into the aquifer.

Table 1. Groundwater electrical conductivity (EC25 uS/cm)

	BI32	BI3	BI28	BI15	GC1	recharge	TWB1	CB1
2001	165	217	229	513	701	1,063	863	1,066
2002	170	351	230	513	770	1,186	907	940
2003	219	292	280	473	792	1,555	1,006	951
2004	237	248	285	466	1,106	1,723	1,309	1,008

- *Total nitrogen and phosphorus*

The shallowness of the aquifer and the intrusion of the vegetation root zone are expected to result in significant processing of nutrients within the aquifer. Mean total nitrogen for the studied bores are contained in Table 2, figures in bold are statistically different to the control site BI32. Total nitrogen (TN) was significantly higher than background at TWB1 in 2001, 2002 and at B1 28, CB1 and GC1 in 2001, 2002, 2003. BI15 was not significantly different to background in 2001, 2002, 2003 and was significantly lower in 2004. The inconsistently high results at B1 28 are likely to be due to a nearby low-lying swampy area contributing nitrogen to the aquifer. It appears that recharge increases TN at the southern boundary of the aquifer by 0.7–1.4 mg/L but does not influence groundwater concentrations north of the golf course in the vicinity of the water trench. There was no statistically significant variation in total phosphorus from background values in any site other than a 0.02 mg/L increase at TWB1 in 2003 and at 0.16 mg/L increase at CB1 in 2004. Recharge does not seem to have a significant effect on phosphorus in the aquifer.

Table 2. Groundwater total nitrogen (mg/L N)

	BI32	BI3	BI28	BI15	GC1	Recharge	TWB1	CB1
2001	0.8	1.1	**1.3**	0.9	**0.6**	2.4	**2.1**	**2.2**
2002	0.8	1.0	**1.5**	0.5	**2.0**	2.4	**1.5**	**2.1**
2003	1.0	1.2	**2.3**	0.7	**2.5**	3.5	1.4	**2.4**
2004	1.7	**0.7**	3.0	**0.5**	1.5	2.7	3.0	2.4

- *pH*

Groundwater pH seems to be affected by a number of factors other than recharge. The recharge water pH is between 7.0 and 7.3. At TWB1 the pH is 4.6–5.0, this drops to 4.2 to 4.6 at GC1, however increases to 7.3 to 7.5 at BH 15, which is further along the same flow path. The pH also rises at CB1 to between 6.4 to 6.6. The elevation of pH at sites BI15 and CB1 is most likely due to golf course green and fairway management and sea spray effects respectively.

- *Bacterial indicators*

Total coliforms and E coli were used as indicators of microbiological quality of groundwater at the bore sites. Median values for each year were compared to the control site. The median value for E coli at all sites and in all years was 0. There were no consistent trends in total coliform medians with CB1 having lower results that the control site in 2001 and 2004 but higher in 2002, 2003. BI15 the closest site to the water trench had lower median coliform counts than the control site in all years studied. TWB1 adjacent to the infiltration area had a higher than control median in 2003 only. The results obtained for indicator bacteria indicate efficient removal of bacteria by die off and filtration effects within 200 m of the infiltration basins.

Table 3. Median total coliforms in groundwater

	BI32	BI3	BI28	BI15	GC1	TWB1	CB1
2001	7	62	6	3	1	0	1
2002	7	101	100	5	3	3	21
2003	20	14	200	14	13	111	53
2004	200	3	16	49	8	12	83

CONCLUSIONS

The use of infiltration basins on Bribie Island has successfully protected the aquifer against seawater intrusion despite significant lowering of the water table adjacent to water extraction trench. Modeling has been able to successfully indicate groundwater levels in the recharge area and predict flow paths for the recharge plume. The model has been useful in investigating localised effects of aquifer recharge and establishing an effective monitoring program. The monitoring program has confirmed movement of recharge water towards the potable water extraction area but indicates that recharge water is a minor component of groundwater in the vicinity of the extraction trench. Nitrogen levels in groundwater are variable with a general increase at the southern boundary compared to background but no clear trends north of the recharge area and no increase near the potable water extraction trench. Groundwater phosphorus and bacteria are not significantly affected by recharge. Groundwater pH across the aquifer is affected by a number of factors with the effect of recharge being minor. There appears to be no serious environmental or health related impact resulting from the use of reclaimed water for aquifer recharge at Bribie Island.

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Groundwater explotation in an arid zone in relation to the recharge in a central region of the Argentine Republic

Norberto Gabriel Bucich

Abstract

To obtain a sustainable management of the subsurface hydric resource in the area under study considering the best profitable use of the resource in relation to the recharge. This is an advance based on a regional understanding of the geology, the aquifer system and its relation with the hydrodynamics.

The behavior of the system is analyzed in respect to different exploitation alternatives in order to know which are feasible under a series of physical, technical and economical restrictions. This process analyzes the results of supposed simulation scenarios of the resource exploitation with the purpose of determining the quality of the additional groundwater obtainable from the system with an adequate management of the recharged volumes.

By means of the application of the proposed methodology it is possible to obtain a sustainable meliorated utilization of the resource by balancing the recharge by taking advantage of a reduction of the ground water discharge by evaporation.

The environmental deterioration permits to state that the exploitation of the hydric resource under the present model should be minimun if the conservative extraction requisites proposed in the simulations are fulfilled and provided a correct management of the ground water resource.

The application and control of the proposed methodology will determine an optimization of the recharge of the system.

Keywords

Hydrogeology; balance; model; recharge.

INTRODUCTION

The present paper describes the application of a mathematical model of a subterraean flow system with the purpose of achieving a sustainable management and the best development of the groundwater resource in a basin situated to the west of the city of San Luis, Argentina (Fig. 1). This model provides an advancement in the regional understanding of the geology of the basin, the related aquifer and its hydrodynamics.

METHOD

The simulation code used in the groundwater tridimensional model in finite differences is the one developed by the United States Geologic Survey (Arlen W. Harbaugh and Michael G. Mac Donald) widely known as MODFLOW. In the present work the VISUAL MODFLOW (Nilson Guiger and Thomas Franz) was used in its version 2.8.2., 2000 WATERLOO HYDROGEOLOGIC INC.

The simulation area is included in a basin with two main tectonic elements. To the east, the San Luis Range mountains and their southern prolongation in the subsurface, here with a group of isolated basement outcrops. To the west, the Beazley sedimentary basin, of tectonic origin, with a total column of about 4,000 meters in thickness (Criado Roque et al., 1981). The Bebedero salina is a topographic low situated in the central part of the basin (Fig. 2).

The simulated surface is included in a rectangular area limited by Gauss-Krüger coordinates Y_{min} 6278400; Y_{max} 6360000; X_{min} 3435400; X_{max} 3485000.

The groundwater model was subdivided by using a grill with 102 rows and 64 columns into square shaped cells, each with 800 meters long sides, oriented from north to south. Each cell covers surface of 0.64 km^2, thus generating a total area of 4,047.36 km^2. 3,681 cells are active (2,355.84 km^2) and 2,643 are inactive (1,692.52 km^2). In Fig. 2 is represented the modelled area, with indication of null flow and of constant hydraulic head boundaries. The latter is situated in the Bebedero salt lake rim, which constitutes the regional base level. The null flow boundary was placed parallel to subterranean flow lines in the NW and S edges of the modelled area. The model was simulated in steady state.

The water table depth does not exceed 180 m in the piedmont deposits of the San Luis Ranges (Fig. 2). The water table is shallower to the west, southwest and southeast parts of the modelled area and comes near the surface in the neghbourhood of the Bebedero salina.

The main sources for the recharge of the basin are the brooks draining the western slope of the San Luis ranges, together with the Nogoli and Chorrillo rivers. The recharge from rain on the area has been discarded because it is to low compared to the amounts recharged from these rivers. Moreover the depths of the water table do not allow the rainfall to satisfy the field capacity.

The adjusted model balance indicates that the total water input in the system is 183 Hm^3/year, of which 135 Hm^3/year run out of the area towards the southest, that is, to the Bebedero salina, which, as previously said, behaves as the base level of the system (constant head boundary in the model). In the salina rim, where the water table is vey close to the surface, an evaporation of 47 Hm^3/year takes place. Fig. 3 shows the contours corresponding to the calibration of the model.

For the calculation of the groundwater evapotranspiration in the salt lake marginal zone was applied the calculation modulus included in the 'Visual Modflow' computational code. In this way

Figure 1. Studied area

Topographic contour interval = 50 m
Contour levels (meters amsl)
● Constant head boundary
— No flow boundary
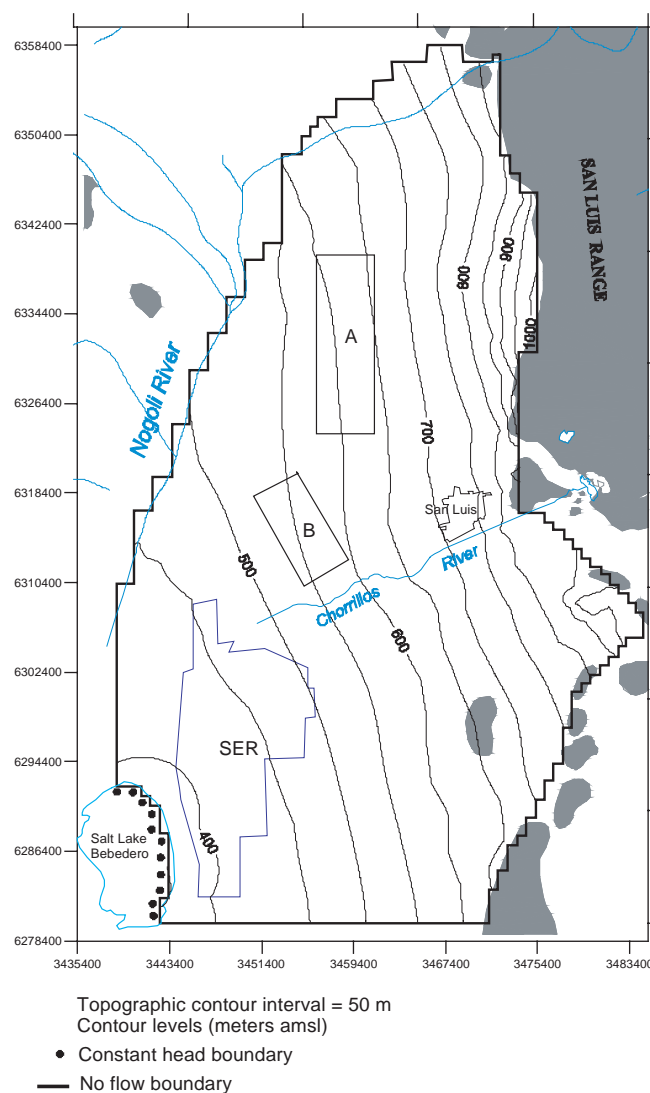
Figure 2. Topographic map

was determined is 13 meters the maximum depths where evapo-transpiration effects can be produced. This value was taken from studies in other basins with similar characteristics, in the same geologic setting, and where detail studies related to root depths were carried out (Durbaum et al., 1976).

Simulation scenarios

With the adjustment of the flow model different development simulations were made in order to determine the exploitation feasibility under a series of physical, technical and economic restrictions. With this process were analyzed different supposed simulation scenarios for the development of the resource, with the purpose of determining the additional volume of groundwater that can be driven from the system Due to space restrictions, in the present paper are explained the two most representative scenarios obtained from this method.

The ubication and extension of the simulated scenarios were determined by a decision support system (Bucich et al., 2002) by which different characteristics of the environment were analyzed: groundwater depth, transmissivity resulting from the model, chemical compsition of the water, soil type, and planned cultivation in function of cost an market performance.

Figure 3. Groundwater levels

- *Scenario I*

The first simulation scenario corresponds to the plant of an existing enterprise which is the only groundwater development of importance in the modelled zone (SER, Fig. 3). This estate has an extension of 18,500 hectares and is situated in the SW sector of the modelled area. This enterprise has to be taken into consideration at the time when the scenarios will be planned, in view of the size and importance of the existing and planned installations.

The detail studies carried out in this estate (Bonini et al. 2000) established that the exploitation flow volume obtainable by drilling can fluctuate from 120 to 200 m^3/h. The irrigation systems planned (central pivot irrigators with 900 meters radius) require 4 wells to satisfy an output of 550 to 600 m^3/h per pivot. In this way four wells
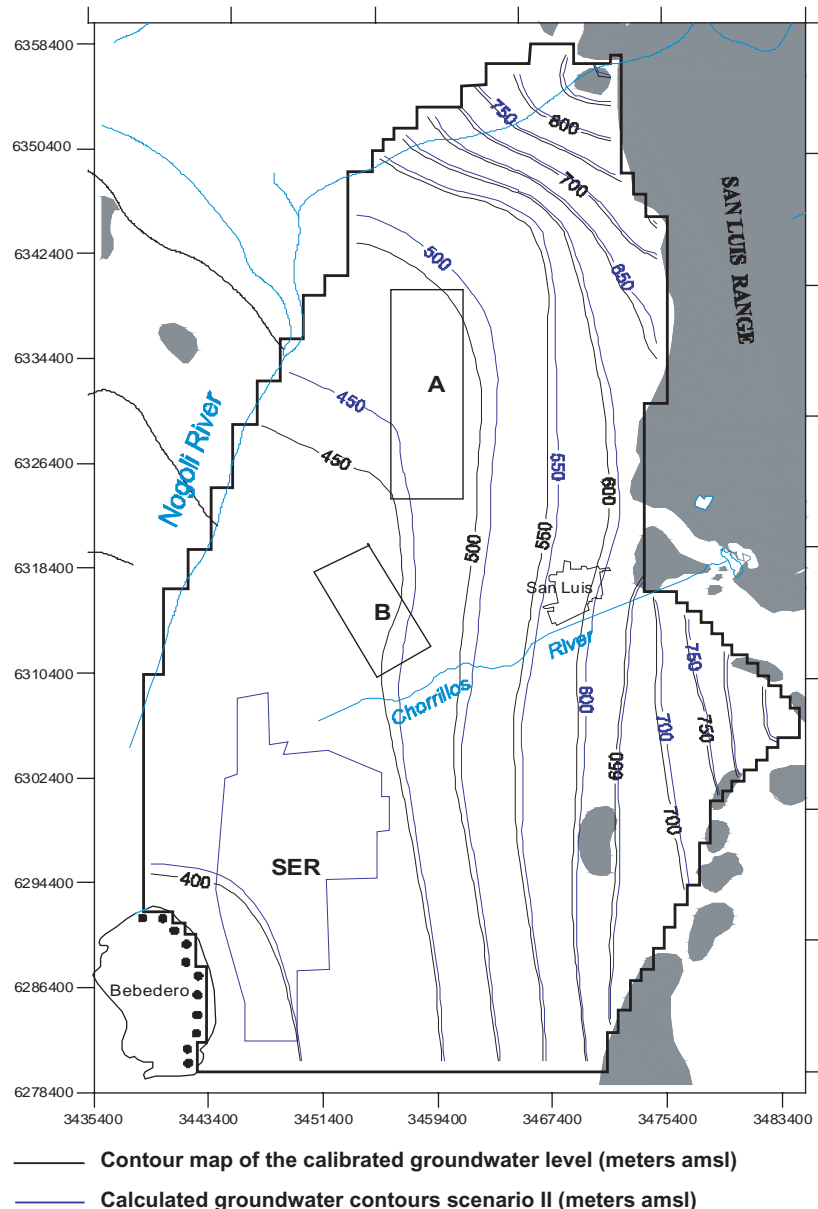
were planned for each pivot, so that with seven pivots 843 m^3/d would be extracted during three months every year.

The simulation considers an extraction output of 8.3 Hm^3/year in the wells, distributed in a 17.28 km^2 surface. In the area next to the salt lake, due to the shallow depth of the groundwater surface an evapotranspiration of 46 Hm^3/year is produced. This means 2% less than in the system without any external action. Thus a subterranean flow of 128 Hm^3/year to the south and southwest takes out of the area 7 Hm^3/year less than in the calibrated model.

The comparison between the piezometry calibrated by the model versus the results of the piezometry with exploitation is SER (first simulated scenario) corroborates a drawdown of the water table up to 2.9 m in the eastern part of the estate, whereas in the central part of the model the descent is only 1 m.

This simulation was run for 27 wells producing the same volume and located according to the detail study. It should be made clear that it is not advised to drill new wells in the western sector of the estate, or to increase the water extraction, because this could facilitate the inversion of the flow lines from the zone of natural discharge of the system (Bebedero salt lake).

The matematical model shows that, with the establishment of the SER estate, the volumes drawn by pumping do not affect the direction of the groundwater flow provided that the current expoitation regime is maintained. It is likely that an incorrect arrangement of new wells or an incorrect pumping could affect this flow direction, which means the risk of invasion of salty water and brine from the near surface levels of the Bebedero salina.

• *Scenario II*

In this resource development scenario two areas, A and B, were added (Fig. 4). These areas are characterized by groundwater available in quality and quantity sufficient to satisfy the irrigation requirements.

Contour map of the deph of calibrated groundwater level

Contour map of the deph of calculated groundwater level scenario II
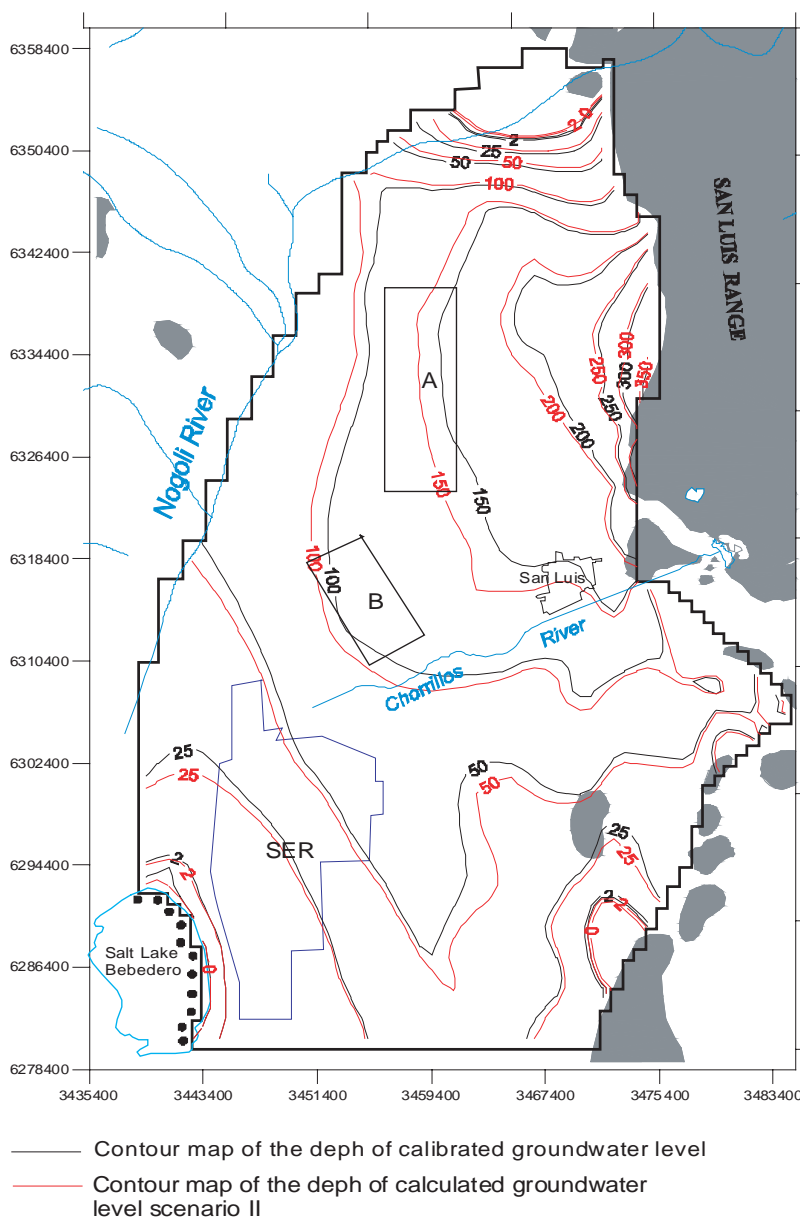
Figure 4. Deph of groundwater levels

The simulated extractions were: for area A, 6 Hm^3/year and for area B, 21 Hm^3/year.

According to the water balance, the subsurface flow towards the Bebedero salina will be reduced to scarcely 104 Hm^3/year, and the evapotranspirable volume to 43 Hm^3/year because of the drawdown of the water table.

During the run of these scenarios the hydraulic heads calculated by the model in each case did not show important changes in their morphometry and gradients. The groundwater flow velocities did not experience notable variations (Fig. 3).

The maximum ground water level drawdown reached 11.65 m in area B, with an increased descent in the SER area, higher than 5 m. The descent was less in the SW extreme of the modelled area where the constant height cells next to the salina rim did not distort the water table drawdown. In Fig. 4 area shown the water table depth isolines for the calibrated model and for the simulation of this scenario.

In Table 1 are synthetized the volumes for the calibrated model and for the described development scenarios. It can be clearly observed that, with a constant recharge value, a reduction of about 4 Hm^3/year is obtained in the ground-water evapotranspiration. This ilustrates that, with a strategic location of the exploitation scenarios, it is possible to have additional flow volumes without changing the recharge values, thus minimizing its effects.

Table 1. Volume relations according to the scenarios

	Const head (Hm^3/year)	Pumping wells (Hm^3/year)	Recharge (Hm^3/year)	Evpt (Hm^3/year)	Total (Hm^3/year)
Calibrated	135	0	180	47	182
SER	128	8.3	180	46	182
A,B,SER	104	35.4	180	43	182

PUMPING FIELDS

To established the possible minimum distances between pumping wells in areas A and B according to their hydro-dynamic characteristics, production rates and a maximum water table fall of 20 m, an analytical model was applied, which permits the calculation of drawdowns at any point of the surface to simulate.

With this method it is possible to know, not only the piezometric surface descent, but also the drawdown in each development well submitted to intensive pumping, including the loss of head due to their efficiency (Bucich et al, 2003).

In Tables 2 and 3 are summarized the simulated drawdowns for different produced volumes and distance, in areas A and B.

Table 2. Determination of distances for area B

B T = 300 m^2/d	Volumes (m^3/h)		
	50	100	150
Distances between wells (m)			
100	9.89		
150		19.27	
4,000			22.78

Table 3. Determination of distances for area A

A T = 240 m^2/d	Volumes (m^3/h)		
	50	100	150
Distances between wells (m)			
100	12.19		
150		20.72	
4,000			27.98

In Table 2 can be observed that, for area A, with volumes higher than 100 m^3/h and distances between wells of 4,000 meters, the water table drawdown is more than 20 m. In Table 3 is observed for area B, that 20 meters drawdown is reached with volumes higher than 100 m^3/h and well distances exceeding 150 m.

CONCLUSIONS

The discharge of the ground water system is produced by evapotranspiration due to vegetation, and evaporation in the Bebedero system, which, as expressed in foregoing pages, is the base level for the discharge of the basin because here converge the underground glow lines.

The available data were sufficient to achieve a satisfactory calibration.

The global mass balance gives an available water volume in the system of 183 Hm^3/year, which makes possible its use for agrarian activities, since the surface hydric resource is used to supply water to villages.

The balance of the adjusted model indicates that the total input to the system is 183 Hm^3/year from which 135 Hm^3/year flow out of the area, to the South and Southwest, towards the Bebedero salt lake, which acts as the base level of the system (constant head boundary in the model). In the perimeter of this salina, where the water table is very near to the surface, an evapotranspiration of 47 Hm^3/year takes place.

In the simulation of scenario II, according to the water balance, the subterranean flow to the Bebedero salt lake is reduced to only 104 Hm^3/year, and the evapotranspirable volume to 43 Hm^3/year, as an effect of the deepening of the water table.

With the results obtained it can clearly be observed that, by keeping constant the recharge value, it is possible a reduction of about 4 Hm^3/year in the groundwater evapotranspiration. This shows that, with a strategic ubication of the development scenarios, additional volumes can be obtained without changes in the recharge values, thus minimizing their effects.

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Preliminary hydrologic investigations of Nubia sandstone and fractured basement aquifers in the area between El Shalateen and Halayeb, Eastern Desert, Egypt

Yehia L. Ismail, Esam El Sayed and Mohammed A. A. Gomaa

Abstract

El Shalateen-Halayeb triangle is a promising district for tourism and agricultural development. Pre-Miocene rocks (fractured basement and Nubia sandstone) represent the main water-bearing formations in the investigated area. Rainfall and occasionally flash floods represent the main sources of recharge. Groundwater occurrence and movement in basement aquifer is mainly controlled by the structural elements reflect a good environment for groundwater entrapment. In fractured basement aquifer, the values of transmissivity have wide variation. Nubia sandstone is detected as a water-bearing and its transmissivity reflects poor potentiality of the aquifer is mainly due to high channel gradient. Regionally, the direction of groundwater flow is mainly restricted by the variable hydraulic gradients from locality to another.

Groundwater salinity of basement aquifer varies from 438 mg/l to 10,409 mg/l, reflecting a wide variation in groundwater quality from fresh to saline. The groundwater quality of Nubia sandstone aquifer varies from fresh to brackish, where the salinity ranges from 459 mg/l to 1,292 mg/l. In basement groundwater, Cl is the most correlated anion with salinity ($R^2 = 0.9615$), and Na is the most correlated cation with salinity ($R^2 = 0.9153$). In Nubia sandstone groundwater, SO_4 is the most correlated anion with salinity ($R^2 = 0.7573$) and Na is the most correlated cation with salinity ($R^2 = 0.8578$). Groundwater quality was evaluated for different uses and some recommendations were given.

INTRODUCTION

The main objective of this study is to evaluate groundwater resources quantavely and qualitatively.

Meteorologically, Eastern Desert lies within the Egyptian arid belt. The investigated area is characterized by scarce rainfall and occasional flash floods, so, floodwater control should be taken into consideration.

Hydrographically, the investigated area is distinguished into three great basins facing Red Sea to the east. The first is Barnis basin (3,000 km²), the second is El Shalateen – Abu Ramad Basin (2,300 km²) and the third is Halayeb Basin (2,500 km²). Geologically (Fig. 1), the investigated area is occupied from the western side by basement rocks. Nubia sandstone outcrops close to water divide. Tertiary volcanic rocks are extruded at the foot slope of Red Sea mountainous shield. Isolated patches of Miocene sediments outcrops due west of Abu Ramad and Halayeb area, (alternating limestone and marl of Gebel EL Rusas Formation). Quaternary deposits form deltas of Wadis as well as Wadi fill of main channels.

GROUNDWATER OCCURRENCES

Twenty-three water points were selected to evaluate groundwater resources in the study area (Fig. 2 and Table 1). Eighteen of them are tapping fractured basement aquifer and five get their water from Nubia sandstone aquifer in the upreaches of Wadi Hodein.
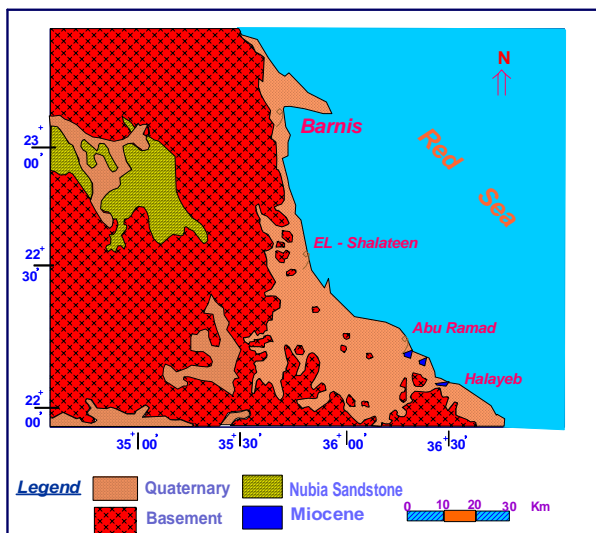
*Figure 1. Geologic map
of Barnis - Halayeb district*
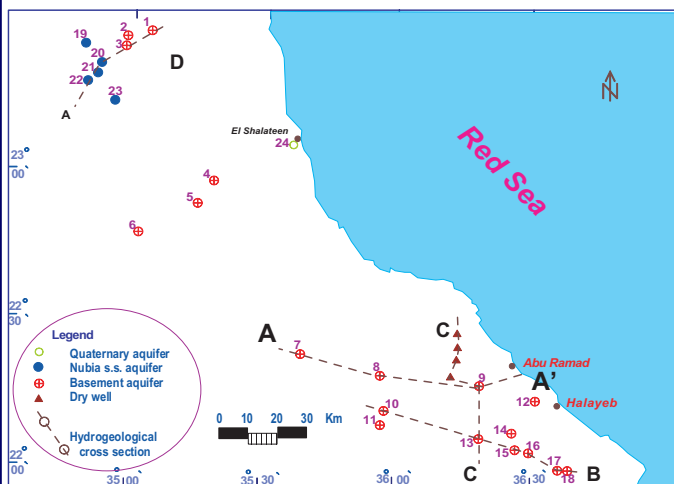
Figure 2. Well location map

Table 1. Hydrogeological data of the investigated aquifers (Jan, 2004)

Well No.	Water Point	Basin	Aquifer Facies	Well Type	Well diameter (m)	Depth to Water (m)	Total Depth (m)	Ground Elevation (m)	Water Level (m)
1	EL Gahelia well	Wadi Hodein		Dug Well	3.30	3.30	9.50	+400	–
2	Eiqat-a well	"		"	2.00	7.00	-	+134	+ 127
3	Eiqat-b well	"		"	1.90	8.50	10.25	+134	+125.5
4	EL Beida- a well	Wadi EL Beida		"	2.00	4.2	4.4	-	
5	EL Beida-b well	""			1.80	3.00	3.8	-	-
6	Madi well	Wadi Madi		"	1.90	6.7	7.85	-	-
7	Meisah well	Wadi Meisah		"	2.10	5.90	-	+45.9	+40
8	Gomidlum well	Wadi EL Deib	Basement	"	2.00	27	28	+83	+56
9	Aquamatra well	Wadi Audeib		"	2.10	4.5	-	+285	+280.5
10	Mahareka well	Wadi EL Deib		"	1.35	1.00	2.00	+85	+84
11	Shoshab well	Wadi EL Deib		"	1.50	18.16	19.00	+94	+75.84
12	Sararah well	Wadi Sematay		"	1.00	17.00	22.38	+112	+95
13	Salalat Osar well	"		"	1.80	13.50	-	+340	+326.5
14	Sararat well	"		"	1.00	21.2	23.00	+220	+198.8
15	Okak well	Wadi Shandodi		"	1.20	5.40	-	+320	+314.6
16	Eremit well	"		"	2.15	11.80	-	+300	+288.2
17	Frukit–a well	Wadi Shalal		"	2.00	17.0	-	+300	+283
18	Frukit – b well	"		"	3.15	15.5	17.7	+300	+284.5
19	Ain Abraq	Wadi Abraq		Spring	-----	Flowing	-	-	-
20	Abu Saafa–a well	Wadi Hodin		Drilled	-----	22	310	+270	+248
21	Abu Saafa–b well	"	N. Sandstone	Drilled	-----	8.17	90	+315	+307
22	Abu Saafa–c well	"		Drilled	------	13.30	87	+294	+281
23	EL Dif well	Wadi EL Dif		Drilled	-----	21.75	134	+241	+219

Hydrogeologic conditions of the basement aquifer

Fractured basement aquifer is composed of older granitoids, younger granites, gneisses, migmatites, schists, metasediments, gabbro, diorites and quartazites. These rocks are highly weathered and strongly fractured, jointed as
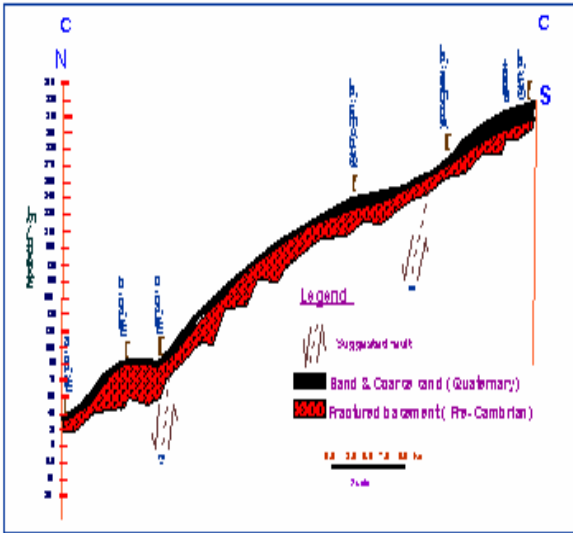
Figure 3. Hydrogeological cross section(C–C')
from north to south (Abu Ramad - Halayed District)

well as faulted, giving a great chance for accumulation and movement of groundwater. Depth to water varies from 1 m to 27 m. Water level ranges between +40 m and +326.5 m. Groundwater occurrence and movement in fractured basement are mainly controlled by the orientation, size and density of the intrusive dykes, which act as a doming factors for percolated water.

The areal extension of the hydrogeological setting encountered in this region was illustrated through the cross section C – C' (Fig. 3). The section revealed the extensions and characteristics of geological boundaries, barriers or structural faults that have a direct impact on the groundwater occurrence and its movement. It is clear, that the structural dykes act as barriers against the groundwater movement towards the north. So, the recent bore holes tests are dry.

Hydraulic parameters

Seven pumping tests were carried out on selected seven dug wells using the following methods :

$T = Q / 4\pi L\ sw * F(Uw, B)$ $S = 4\ Tt / (rw)^2 * (Uw)$ (Papadopolus and Cooper, 1967)

$T = (rc)^2 / t * 1$ $S = (rc)^2 / (rs)^2 * \alpha$ (Papadopolus et al., 1973)

Where,

   T : is the transmissivity (m^2 /day)

   Q : is the rate of discharge (m^3 / hour)

   rw : is the well radius (m)

   sw : is the well drawdown (m)

   rs : is the radius of open hole (m)

   rc : is the radius of casing in interval over which water level fluctuates (m)

   F (μw, B) : is the well function.

The obtained results (Figs. 6-7, inclusive and Table 2), revealed a wide variation in transmissivity due to the strong impact of the structural and lithological setting on the groundwater occurrences. The fractured basement aquifer at Sararat area displays very low transmissivity due to less deformed younger granites. Poor groundwater potentiality is due to weak chance for surface runoff to replenish or feed the concerned aquifer (high velocity of surface runoff as a result of high channel gradient).

Table 2. The calculated hydraulic parameters of the basement aquifer

Well No.	Water Point	T (Transmissivity) m^2/day	S (Storativity) dimensionless
1	El Gahelia well	784	0.08
7	Meisah well	198	7.06×10^{-3}
10	Mahareka well	139	5.95×10^{-3}
17	Forkit–a well	19.23	1.52×10^{-4}
2	Eqat well	12	2.22×10^{-5}
11	Shoshab well	4.58	1.13×10^{-6}
12	Sararat well	2.75	1.27×10^{-6}
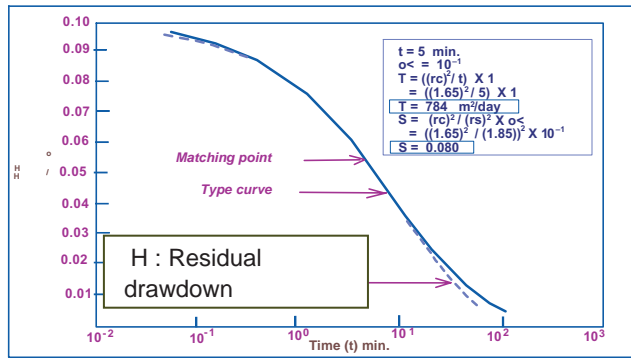
Figure 4. Analysis of pumping test data of El Gahelia dug well using slug test (Papadopulos et al., 1973)
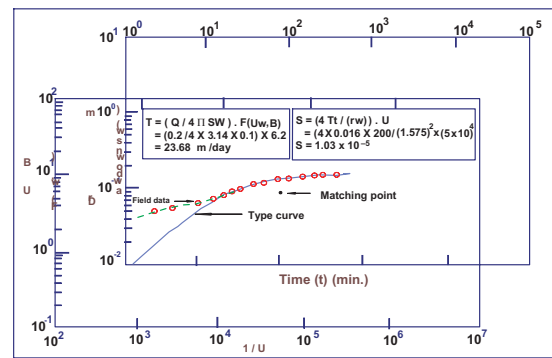
Figure 5. Analysis of pumping test data of Frokeit well (Papadopulos' method).

Hydrogeologic conditions of Nubia sandstone aquifer

Nubia sandstone aquifer has a tremendous importance in the area. It covers the northwestern part of the investigated area. Nubia sandstone rests directly on basement rocks. It is composed of fine to coarse sandstone intercalated with clay. The thickness ranges between 100 m up to 500 m (Aglan, 2001). It is detected as a water-bearing in Wadi Abraq, Abu Saafa and Wadi EL Dif. Five representative water points were selected in the study area. The groundwater occurs under confined conditions (Abraq spring) and semi- confined conditions (drilled wells). Rainfall and flash floods on the sandstone plateau and the surrounding fractured basement rocks are considered to be the main sources of groundwater replenishment. Depth to water varies from 8.17 m to 21.75 m from the ground surface. On the other hand, the water level ranges between + 248 m and + 281 m (Fig. 6, Table 1).

The hydraulic parameters of Nubian sandstone aquifer

Elewa (2000) mentioned that, the aquifer sediments reflect a wide range of transmissivity (from 2.72 m^2/day to 72.4 m^2/day)). This could be attributed to the lateral facies change as well as the impact of the structural setting. To confirm the obtained results, a trial was carried out by the author using raw data of pumping tests of El Dif well (GARPAD, 1996), using Jacob straight line method (1963). The obtained value of the transmissivity is comparable to the previous data (Fig. 7).
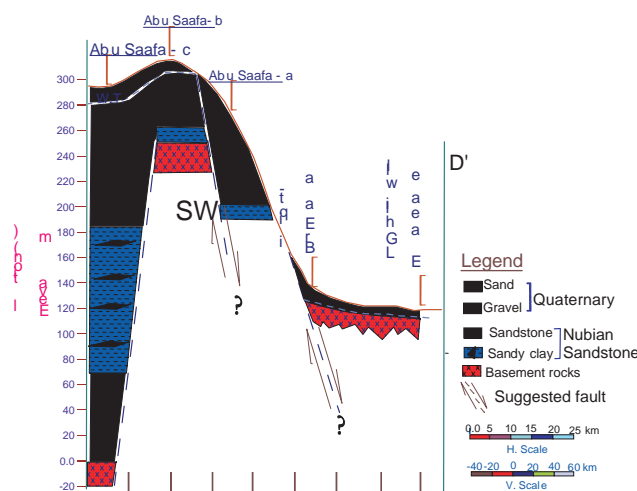
Figure 6. Hydogeological cross section (D–D') from southwest to northeast Abu Saafa - El Gahelia area
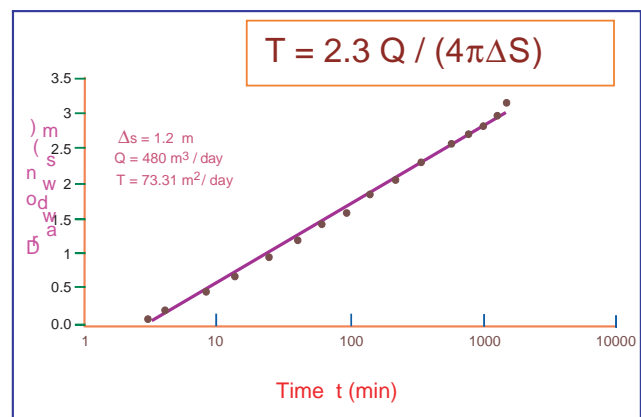
Figure 7. Analysis of pumping test data of Bir El Dif

HYDROGEOCHEMICAL ASPECTS

Hydrogeochemical aspects of parts of the concerned aquifers are discussed through the hydrochemical analyses of twenty three groundwater samples as well as two samples representing rainwater and Red Sea water (Table 3). They reflect the following results :

1. Groundwater salinity : The groundwater salinity ranges from 438 mg/l (No. 6) to 10,409 mg/l (No. 15) indicating a wide variation in quality from fresh to saline in the fractured basement rock aquifer (Hem, 1989). On the other hand, the groundwater salinity varies from 459 mg/l (No. 19) to 1,292 mg/l (Nos. 20 and 21), reflecting fresh to brackish water in the Nubia Sandstone aquifer (as they are closer to the watershed area).

2. Groundwater chemical types : The groundwater in the fractured basement aquifer is characterized by Na > Ca(Mg) > Mg(Ca) / Cl > SO4(HCO3) > HCO3(SO4) ionic proportion. So, the groundwater chemical type is mainly Cl–Na (Fig. 16), On the other hand, the ionic proportion; Na > Mg > Ca / HCO_3 (Cl) > Cl (HCO_3) > SO_4 characterises the Nubia sandstone aquifer. Consequently, two chemical water types are recognized. The first type HCO_3–Na (Nos. 19 and 21), while the second type Cl–Na is detected at Abu Saafa area (Nos. 20 and 22), indicating the final phase of groundwater metasomatism.

3. Salinity-major components relationships : The relation between salinity (TDS) and major components were statistically illustrated (Figs. 8 and 9). According to these diagrams, a positive correlation between salinity and concentration of ions is evident, since the increasing of major constituents leads to the increasing of total

Table 3. The hydrochemical analyses data (2004)

No.	Water point	pH	TDS	Units	Ca	Mg	Na	K	CO3	HCO3	SO4	Cl
1	EL Gahelia well	8.10	438	Ppm	9.80	45.24	95	12	7.62	312.5	15	98
2	Eqat-a well	8.35	2368	Ppm	58.80	107.20	660	27	72.39	418.30	590	644
3	Eqat-b well	7.87	1600	Ppm	47.04	111.90	385	21	22.90	511.30	370	386.40
4	EL Beida-a well	7.18	2244	Ppm	135.20	69.54	600	10	Nil	230.12	200	1115
5	EL Beida-b well	7.13	8586	Ppm	594.10	290.10	2100	13	Nil	275	1100	4352
6	Madi well	7.95	511	Ppm	46.20	18.10	120	9	Nil	335	50	100.50
7	Meisah well	7.83	2750	Ppm	219.50	19.05	680	19	11.43	352.5	600	1024.80
8	Gomidlum well	7.12	4330	Ppm	712.10	160	550	15	Nil	49.70	950	1918
9	Aquamatra well	7.30	1111	Ppm	58.20	40.10	290	15	Nil	296	180	380.11
10	Mahareka well	7.87	1080	Ppm	113.68	38.10	220	10	3.81	143.30	135	448
11	Shoshab well	8.39	2980	Ppm	58.80	71.44	1000	7	15.24	267.30	140	1554
12	Sararah well	7.85	6967	Ppm	529.20	202.40	1700	23	15.24	286.60	1750	2604
13	Salalat Osar well	8.00	1096	Ppm	54.9	38.10	310	12	19.05	367.90	100	378
14	Sararat well	8.00	4776	Ppm	548.8	202.4	975	8	26.67	197.5	495	2422
15	Okak well	7.75	10409	Ppm	901.6	476.30	2400	7	11.43	205.30	800	5712
16	Eremit well	7.85	8537	Ppm	352.8	261.95	2500	6	15.24	344.70	100	5129.40
17	Frokeit-a well	7.30	1246	Ppm	40.50	63.17	350	11	Nil	288.14	150	488.14
18	Frokeit-b well	7.12	4786	Ppm	812.40	200	610	19	Nil	69.80	1100	2010
19	Ain Abraq	8.10	459	Ppm	54.88	35.72	75	8	3.81	351.40	15	91.10
20	Abu Saafa- a	7.10	1223	Ppm	44.10	65.12	310	12	Nil	310	100	537.10
21	Abu Saafa-b	7.30	1292	Ppm	27.25	22.40	430	11	Nil	599.10	230	272.10
22	Abu Saafa-c	7.11	1219	Ppm	88.15	67.14	250	14	Nil	265.10	230	438.10
23	EL Dif well	7.10	591	Ppm	59.20	44.12	100	17	Nil	365	40	148.50
-	Rain water	7.86	155	ppm	10	9	21	2	Nil	63	14	31
-	Sea water	7.96	38120	ppm	439	1343	12000	380	18	137	2300	21568

Note: TDS in mg/l.

salinity. The more significant correlation coeifficient, the more correlated with salinity. In the fractured basement aquifer, Cl, Na, Ca and Mg are significant, SO_4 is little significant, while HCO_3 is not significant with total salinity (Fig. 8). On the other hand, in Nubia sandstone aquifer, Na, SO_4 and Cl are significant, while Ca, Mg and HCO_3 are not significant (Fig. 9).

4. Groundwater classification : The investigated groundwater samples were plotted on semilogarithmic paper suggested by Schoeller (1962). They reflect a general resemblance and similarity among each other. In the fractured basement aquifers, two ionic patterns are recognized, due to the presence of separate local parts of aquifers characterized by the great facies changes. The first is Ca > Mg < Na&K < Cl > SO_4 > HCO_3 (Nos. 4, 5, 6, 7, 8, 10, 12, 14, 15 and 18). The second is Ca < Mg < Na&K < Cl > SO_4 < HCO_3 (Nos 1, 2, 3, 9, 11, 13, 16 and 17). On the other hand, in Nubia sandstone aquifer, the groundwater samples follows the pattern; Ca < Mg < Na&K > Cl > SO_4 < HCO_3, exhibiting fresh water characters except the slightly increase of Mg over Ca due to the presence of sediments rich in Mg in Nubia sandstone aquifer.
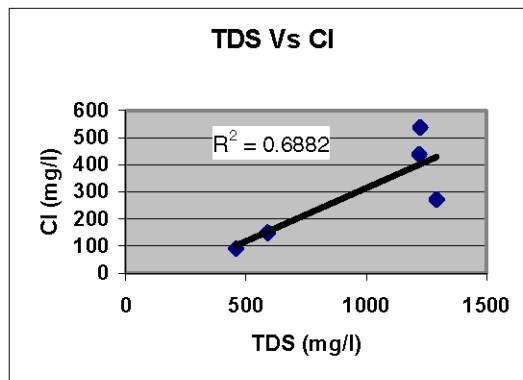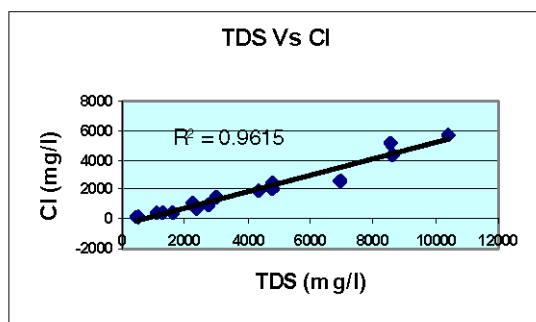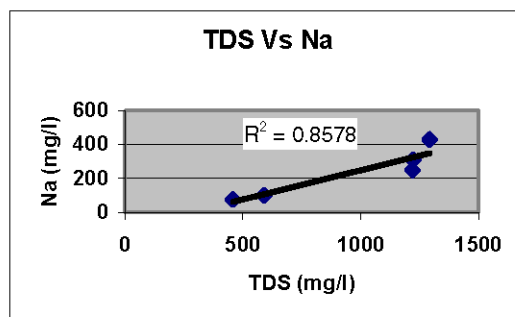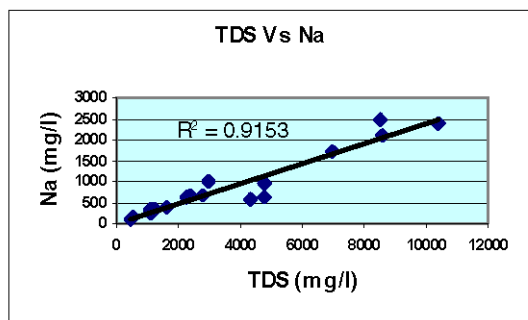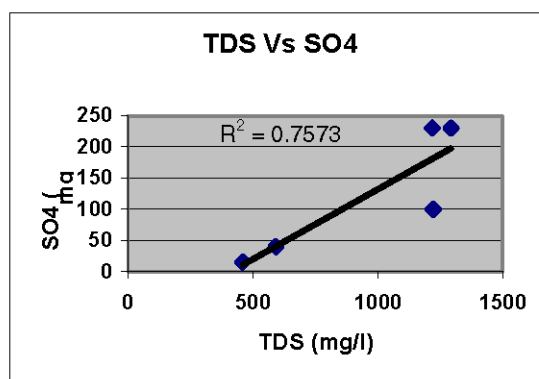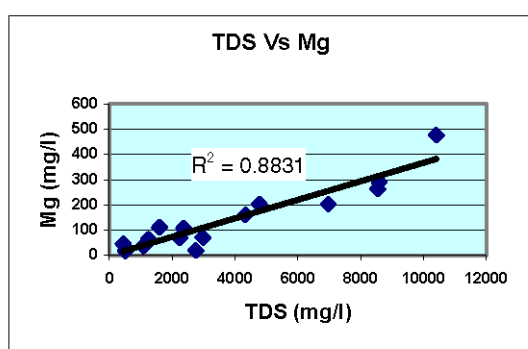
Notes:
  Ca, Mg, Na and Cl *Vs* TDS are significant.
  SO_4 *Vs* TDS is little significant.
  HCO_3 *Vs* TDS is not significant.

Figure 8. Salinity–major components relationship in fractured basementgroundwater

Notes:
  Na, SO_4 and Cl *Vs* TDS are significant.
  Ca, Mg and HCO_3 *Vs* TDS are not significant.

Figure 9. Salinity–major components relationship in Nubia sandstone groundwater

SUITABILITY OF GROUNDWATER QUALITY FOR USES

Comparing the results of the hydrochemical analyses (see Table 3) with World Health Organization standards (WHO, 1984), (Table 4), it is clear that, the groundwater quality of fractured basement aquifer is suitable for drinking at El-Gahlia and Madi areas, where the salinity lies within the range 438 to 511 mg/l (Nos. 1 and 6). In Nubia sandstone aquifer, the groundwater quality is suitable for drinking at Abraq and El-Dif areas, where the groundwater salinity lies within the range 459 to 591 mg/l (Nos. 19 and 23). On the other hand, the proposed approach by the United States Salinity Laboratory staff of agriculture (USSLs, 1954) is applied for determining the suitability of groundwater for irrigation. In this method a nomogram based on specific electrical conductivity as a function of salinity against sodium adsorption ration as a function of sodium hazard is used. Distribution of the groundwater samples within the nomogram (Fig. 10, Table 4), revealed that, groundwater of El-Gahlia, Madi, Abraq and El-Dif is suitable for irrigation. On the other hand, groundwater quality of El-Beida-b, Gomidlum, Sararah, Sararat, Okak, Eremit and Frokeitb is not suitable for irrigation due to high salinity as well as high sodium hazard (only can be used to irrigate tolerant crops as palm trees). The rest of groundwater samples are of intermediate scale and suitable to irrigate certain kinds of crops as alfalfa, tomato, lettuce and cucumber.

Table 4. Drinking water standards (WHO, 1984)

Items	Acceptable (mg/l)	Permissible (mg/l)
PH	7–8.5	6.5–9.5
TDS	500	1,500
Ca	75	200
Mg	50	150
SO_4	200	400
Cl	200	600
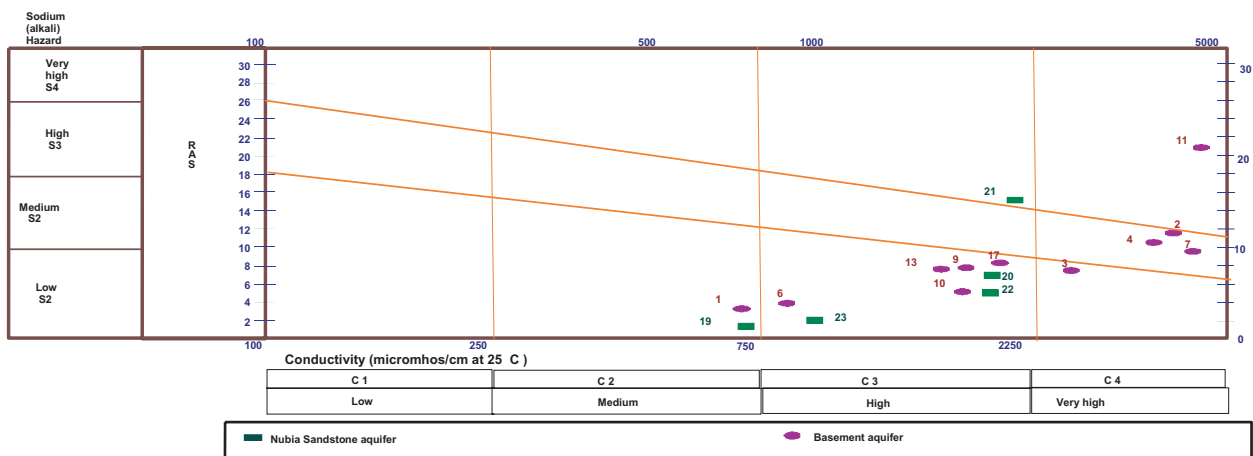
Figure 10. Diagram showing groundwater classification for irrigation (USSL staff, 1948)

CONCLUSIONS AND RECOMMENDATIONS

The aquifers of fractured basement and Nubia sandstone rocks represent the available groundwater resources in the investigated area. The main sources of recharge are the local rainfall and sometimes flash floods after occasional showers. Groundwater occurrence and movement in basement aquifer are mainly controlled by the structural features, where the interaction between fractures and intrusive dykes reflect a good environment for groundwater entrapment. The hydraulic parameters of the fractured basement aquifer revealed wide variation in the trans-

missivity due to the strong impact of the structural and lithological setting on the groundwater occurrences. The poor groundwater potentiality of the fractured basement aquifer is attributed to the weak chance of infiltration of water during surface runoff to replenish the concerned aquifer.

On the other hand, Nubia sandstone aquifer is detected as a water-bearing formation in Wadi Abraq, Abu Saafa and Wadi EL Dif. It rests unconformably directly above the fractured basement rocks. The groundwater occurs under confined conditions (Abraq spring) and semi–confined conditions (drilled wells). It is worth to mention that, the aquifer reflects a wide range of transmissivity (2.72 m2/day to 72.4 m2/day). It can be attributed to the lateral facies changes of the water-bearing rocks as well as the impact of the structural setting. From the hydrochemical point of view, the groundwater quality of the fractured basement aquifer varies from fresh to saline, while in Nubia sandstone aquifer, it varies from fresh to brackish. In basement aquifer, the type Cl–Na is dominant, while in Nubia sandstone aquifer, HCO_3-Na and Cl-Na water types are recognized. In basement aquifer, the groundwater is mostly characterized by permanent hardness (except Nos. 1, 6 and 13 have a perfect temporary hardness). In Nubia sandstone aquifer, the groundwater is mainly characterized by temporary hardness. In basement groundwater, Cl is the most significant correlated anion with salinity (R2 = 0.9615), and Na is the most significant correlated cation with salinity (R2=0.9153). In Nubia sandstone groundwater, SO_4 is the most significant correlated anion with salinity (R2 = 0.7573) and Na is the most significant correlated cation with salinity (R2 = 0.8578).

In view of the present conclusions, aiming to the safety use and development of groundwater resources, the following are recommended :

1. A detailed study of the structural setting should be done to find out the relation betwwen Nubia sandstone aquifer and the underlying fractured basement one.

2. More attention should be focused on flood insurance through the construction of earth and concrete dykes on the upstream portions, specially of Red Sea wadis, which have high gradients to prevent the risk of flash floods and also to increase groundwater recharge.

3. Large diameter wells are recommended to be drilled in the fractured basement rocks.

4. Advanced irrigation system must be applied (drip and sprinkle irrigation).

5. Chemical analyses should be carried out periodically for groundwater samples to ensure that water is valid for different uses (as a long term monitoring program).

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Large scale recharge modeling
in the arid area of the eastern Sahara

W. Gossel, A. Sefelnasr, A. Ebraheem and P. Wycisk

Abstract

A GIS-based (Geographic Information System) hydrogeological database was established for the development of a numerical three-dimensional groundwater flow model for the Nubian Sandstone Aquifer System in Egypt and the adjacent countries in the eastern Sahara. It was calibrated under steady-state and transient conditions. The model was then used to simulate the response of the aquifer to the impact of the climatic changes that took place during the last 25,000 years, to estimate the recharge of the aquifer and to approximate and locate the palaeo-lakes that contributed to the aquifer in past times. From the results of the simulation, it is indicated that the infiltration during the wet periods 20,000 and 5,000 years before present (b.p.) formed the groundwater in this aquifer. The present recharge of groundwater due to regional groundwater flow from the more humid regions in the southern areas of the aquifer is negligible, demonstrating that the groundwater within the Nubian Aquifer System is fossil water, and the aquifer performed unsteady state condition after the last wet period ending 3,000 years b.p.

INTRODUCTION

The sustainable management of water resources especially in the arid regions like eastern Sahara- the most arid region in the world- constitutes a significant issue today due to severe water deficiency, increased demand on water and ecological troubles caused by the non-rational utilization of water storages. The problem of water resources is most critical for the development in this area, as the only available water resource is the groundwater gained from the Nubian Aquifer System (Fig. 1), which consists of continental and shallow marine sediments with a thickness of up to 4,500 m and holds an enormous water reserve. The Nubian Aquifer System in the eastern Sahara, considered the largest groundwater system in Africa, is formed by two major and two minor basins: a) The Kufra Basin, which comprises the southeastern area of Libya, the northeastern area of Chad and the northwestern corner of Sudan, b) The Dakhla Basin of Egypt, c) The southernmost strip of the Northwestern basin of Egypt, d) The Sudan Platform (Klitzsch and Wycisk 1999; Wycisk 1993; Wycisk 1994). The total area is about 2.35 million square kilometers. In the center and north of the System, where a hyperarid climate prevails, the average precipitation ranges from 0–5 mm/year. Obviously, there is no groundwater recharge in most parts of the System since thousands of years. Thus, the aquifer has been under unsteady state conditions for thousands of years before 1960, and the climatic changes including wet periods, 20,000 and 5,000 years ago, supplied plenty of precipitation to meet the requirements for adequate local groundwater formation (Ebraheem et al. 2002; Ebraheem et al. 2003; Ebraheem et al. 2004; El Sayed et al. 2004; Gossel et al. 2004; Heinl and Thorweihe 1993).

In practice, since the beginning of the increasing agricultural development of the Egyptian and Libyan Oases in 1960, there is a common agreement that the aquifer is under an unsteady state condition. The current groundwater exploitation is approximately ten times the maximum present recharge for this aquifer in the Egyptian part (Heinl
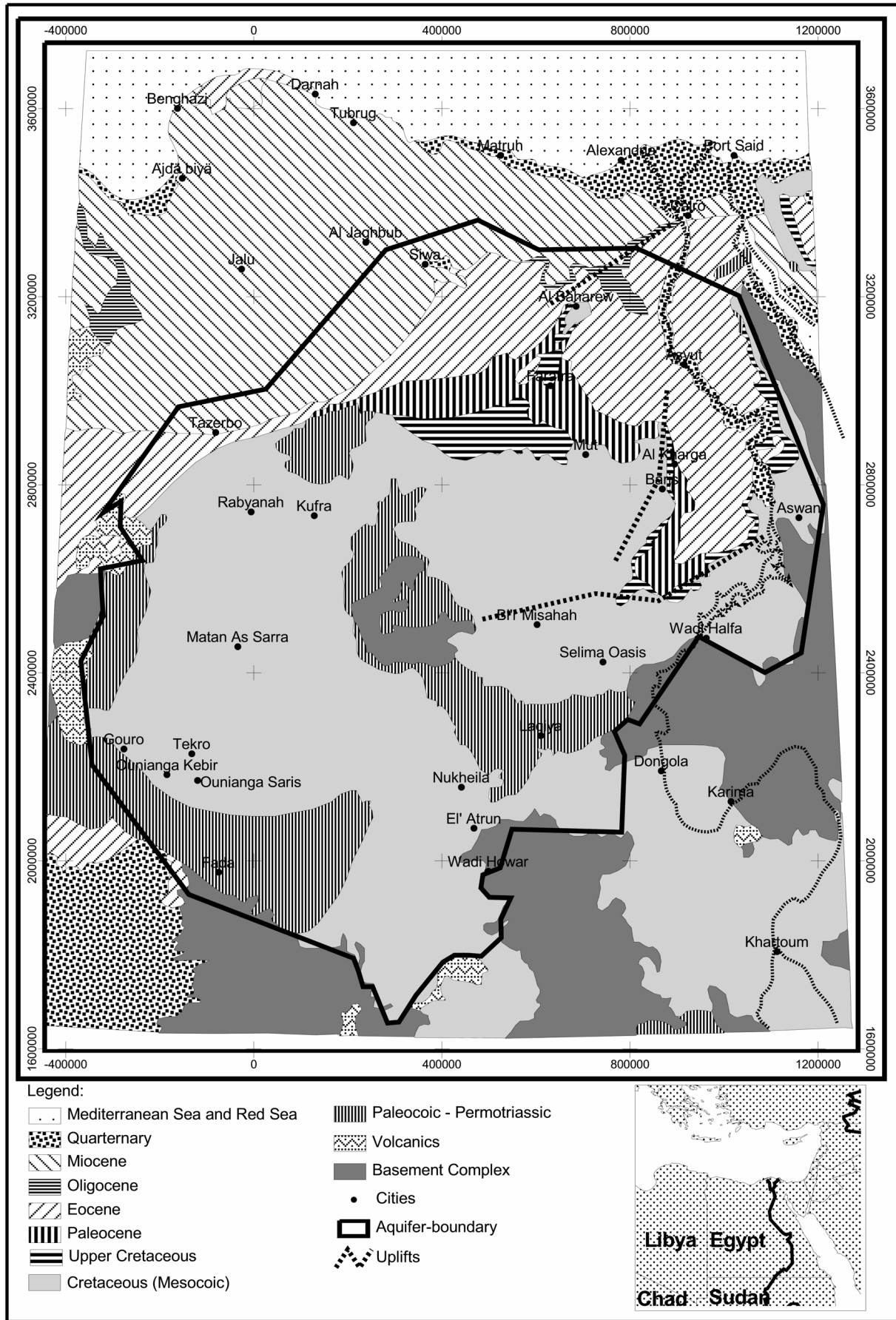
Figure 1. Geological map of the Nubian Sandstone Aquifer System (modified after CEDARE, 2001)

and Thorweihe 1993), even if the local flux from the neighbouring areas of the same system is regarded as a recharge, what leads to progressive decline in the groundwater level. That means, that the aquifer is going to be under overexploitation, if it is further stressed. The increased drawdown leads in turn to an increased cost of the aquifer development, as a reason of more energy consumption, the early replacement and deepening of wells and pumps, and the need to enlarge energy amenities.

This plainly indicates that intellectual careful groundwater resources management of the Nubian Aquifer System on an international scale is fundamental for the sustainable development of the entire area of eastern Sahara.

OBJECTIVES

An integrated GIS-based numerical time dependent three-dimensional transient groundwater flow model for the Nubian Aquifer System in the Western Desert of Egypt and the adjacent countries was developed and used for establishing a spatial and temporal prediction system for groundwater flow and essentially to a) simulate the response of the aquifer to the climatic changes that occurred in the last 25,000 years, b) estimate and calibrate the groundwater recharge of long term time horizons, c) to locate the palaeolakes that existed in the last mentioned period.

METHODS

The fundamental geological, hydrogeological, climatic input-data and newly drilled water wells in Egypt and Libya up to year 2001 as well as the available cross sections of the area of about 2.35 million km^2 were held in a GIS-database that allows an implementation of the numerical groundwater model in different modeling systems. Compiling the data into such coherent and logical GIS-structure supported by a computing environment helps ensure the validity and availability of the data and provides a powerful tool for accomplishing the purposes of the study. GIS also helps in management of hydrogeological data and hydrogeological analysis, vulnerability assessment especially in such huge areas like eastern Sahara and building a hydrogeological database support for numerical-based modeling software.

After constructing the GIS-database, a large-scale finite-element flow model covering the whole Nubian Aquifer System was developed and calibrated under both steady-state and transient simulations (Gossel et al. 2004). In this 3D numerical groundwater model long-term groundwater recharge and aquifer porosity were calibrated by using the results of recent geological research (Pachur, 1999) which indicated that some lakes had existed about 6,000 years b.p. in the southern part of the modeled area as well as sabkha sediments in the northern part.

RESULTS

The simulation results of the last 25,000 years indicated that a recharge rate of 10–50 mm/year was enough to maintain the aquifer in a nearly filled up condition during the two wet periods. The wet time periods can be extracted from Fig. 2 that shows the time dependent distribution of the limnic sediments.

The decline of groundwater surface started about 19,000 years ago, but it was slowed down and completely inter-rupted by regional infiltration during the second wet period (ended 2,500 years ago). It took about 5,500 years after the start of the second wet period to return to the nearly filled up condition and since then the groundwater levels went down for more than 4,000 years. This process is shown at the example of the former 'Lake Ptolemy' in the south of the model area in Fig. 3.
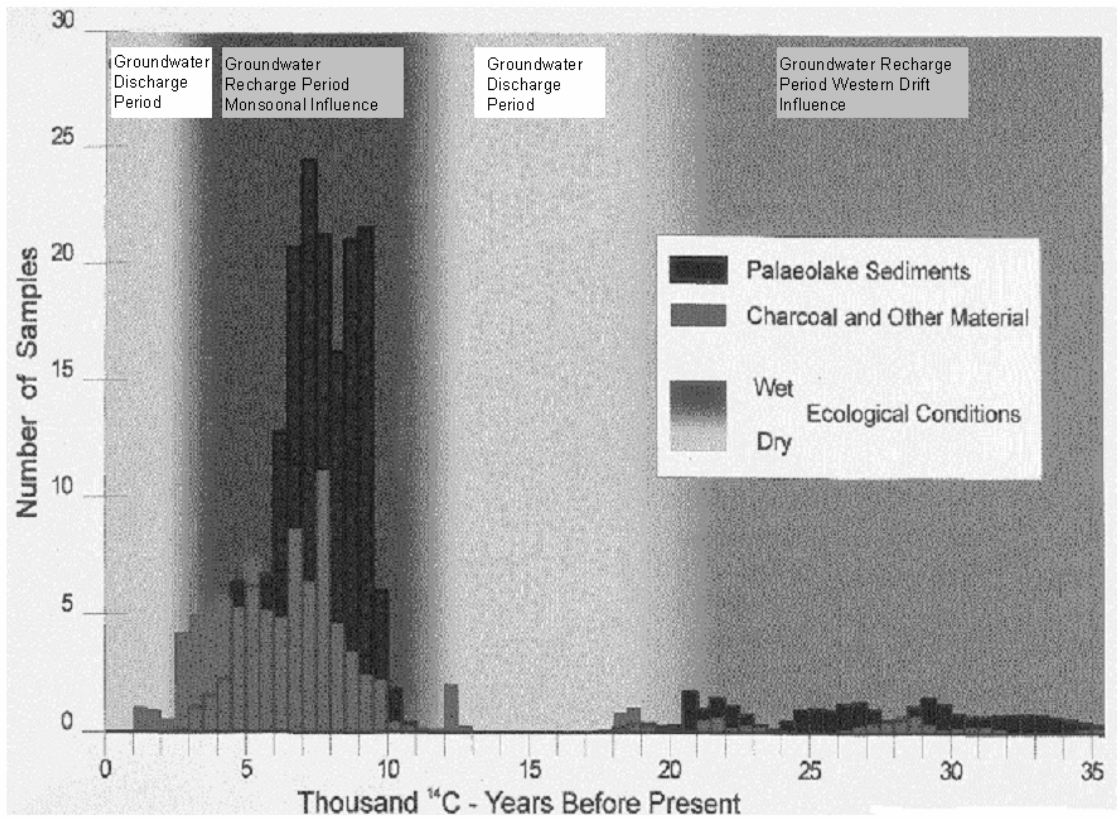
Figure 2. Time scale of wet periods (modified after Pachur, 1999)
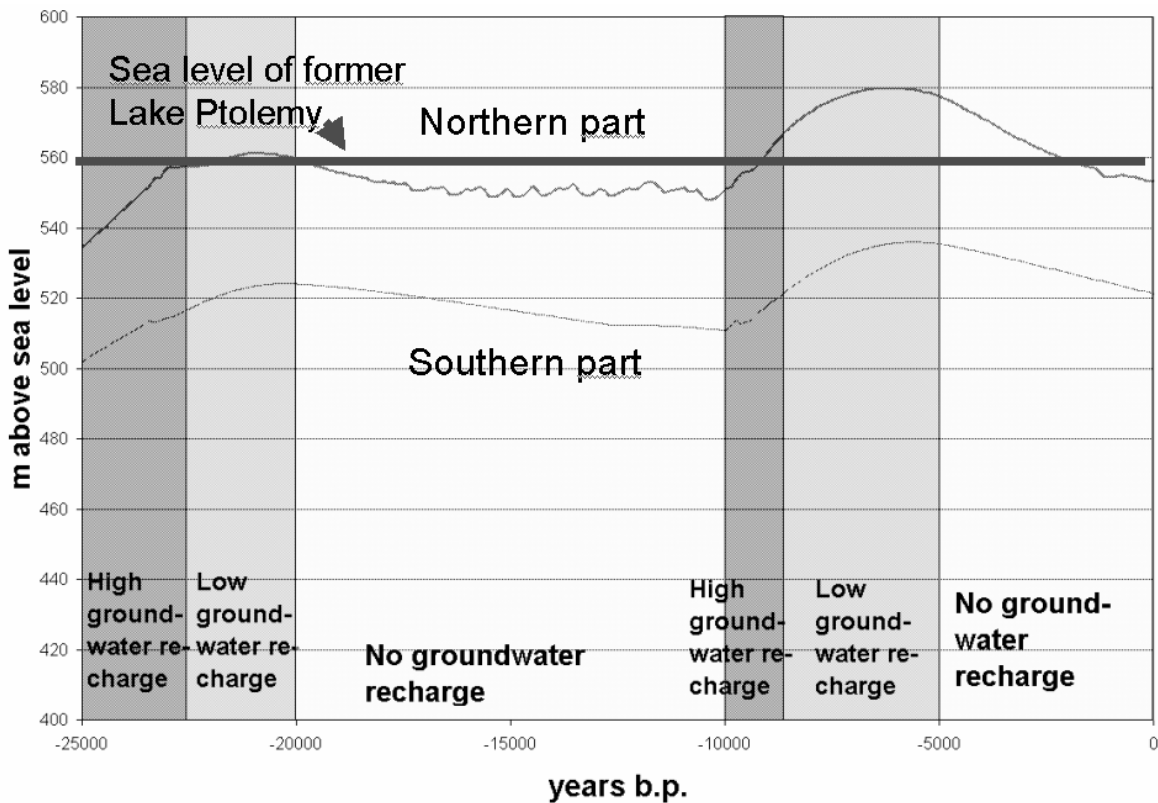
*Figure 3. Recent modelling results for the studied area of the former 'Lake Ptolemy' (NW Sudan)
in the south of the regional numerical groundwater model of the Nubian Aquifer System.
Groundwater recharge in the wet periods leads to groundwater levels at the sea levels reconstructed from sediments.*

In the last 5,000 years discharge was not completely balanced by recharge. Simulation results also indicated that in both wet and dry condition periods groundwater flow from the aquifer to the Nile River has occurred nearly in the whole Nile valley. The amount of Nile water seepage into the aquifer was very low and only possible in the area south of Dongola. After the construction of the Aswan High Dam, rising water level in Lake Nasser increased Nile water seepage into the aquifer. Recharge from the River Nile and percolating rainwater in the southern part of the aquifer are minor if compared to the sum of present natural and artificial discharge of the aquifer. This problem is also discussed in Kim and Sultan (2002) and Dahab et al. (2003).

CONCLUSIONS

The present groundwater management has to take into account, that there is not enough recharge in the southern part to cover the discharge of the oasis in the northern part. The groundwater resource itself was filled up several 10,000 years ago and in the last wet period 5,000 years b.p. The present groundwater extraction must be called groundwater mining and leads to descending groundwater levels to economically unreachable depths. To get a sustainable groundwater resource management of the non-renewable groundwater ressources, the extraction has to be reduced and adapted to highly efficient and optimized long-term cultivation and irrigation strategies. Future modelling work has to be focussed on regional and local management approaches based on the 'carrying capacities' of the entire aquifers and their local conditions respectively.

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Investigation of water spreading effects on water table of aquifer in arid and semi-arid regions

A. Salajegheh and A.R. Keshtkar

Abstract

The central basin of the Iranian Plateau is a semi-arid region due to its precipitation of only 250 mm/year and potential evaporation of over 2 m/y. Most streams in this plateau are ephemeral, while the majority of the permanent rivers are useless because of their high natural salinity. People take advantage of groundwater and groundwater use is continuously increasing. Because of these conditions, artificial recharge by water spreading is considered the most important method to allow stabilizing the yield of the aquifers. This paper analyses the effects of spreading on aquifer yield in four regions of the Iranian Plateau. The spreading increased production of wells and famous Qanats and raised the water table in the aquifer.

Keywords

Precipitation; flood; water; aquifer; recharge; water spreading.

INTRODUCTION

The Iranian Plateau, receiving only 250 mm/y is a semi-arid region, suffering from lack of fresh water for agriculture and domestic uses. Most of the water demand in the regions with little precipitation is met by groundwater extraction, which exceeds natural recharge significantly. Artificial recharge is considered as an approach to remediate aquifer yield and its effects. Objectives are to store river water in the subsoil, prevent salinization of wells and treat wastewater by soil treatment.

Iran applied different methods, i.e. recharge basins, dams, recharge wells and spreading. Spreading over coarse alluvial and aquifer management are the most important of the methods to enhance aquifer yield.

MATERIAL AND METHODS

There are about 35 water spreading research stations in Iran that the current paper reports the investigation results of water spreading in water reservoirs of central plateau four stations. Generally, a floodwater spreading system consists of three parts including intake, off take canal and the area of water spreading. Within the system, intake diverse stream flow into the off take canal which distribute water using diversion weir designed along the canal. Since, the canal is completely leveled, and then water passes the weir as a thin layer. The distributed water on the area joins together and leaves the area. This process is repeated using the other spreading systems to increase water infiltration into the soil.

To study the effect of water spreading on ground water table, the necessary data of Qanats, agricultural as well as peizomethric wells were collected.

For quantitative evaluation of the water spreading project on groundwater table of the study area some statistical comparison were conducted as follow:

1. Rate of flooding and volume of input flood into the system during an event were determined.
2. Rainfall records of the nearest station to the study area were analyzed to compute depth of effective rainfall (runoff)

Qualitative evaluation of floodwater spreading on the groundwater table

To study the effect of floodwater spreading on ground water table in the study area, observed data of Qanats were used for hydro chemical analyses to determine the level of Na^+, Ca^{2+}, Cl^-, SO_4^{2-}, CO_3^{2-}, HCO_3^-, TH, Ec and pH in study area.
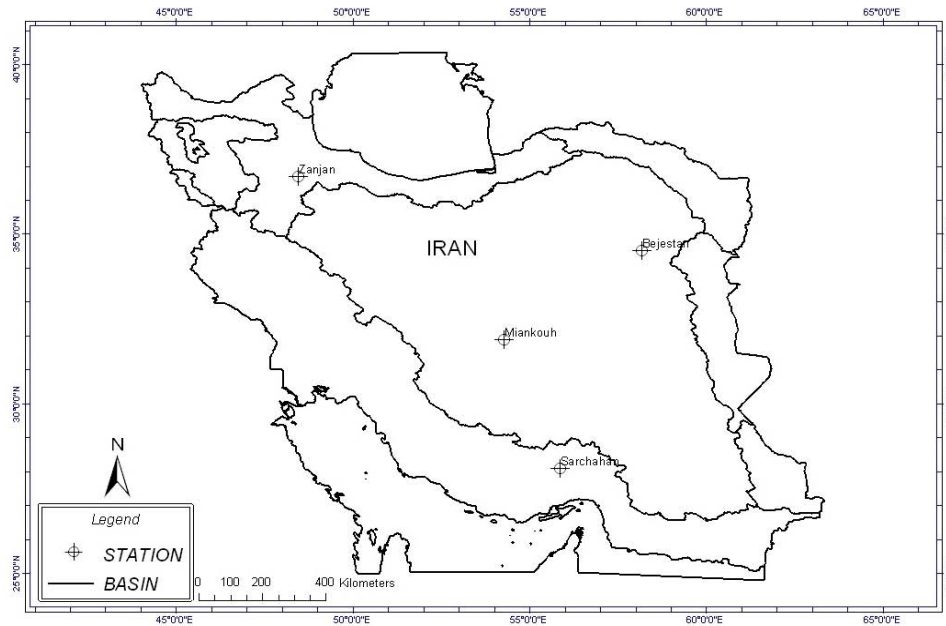
Figure 1. Geographical location of selected stations in Iran
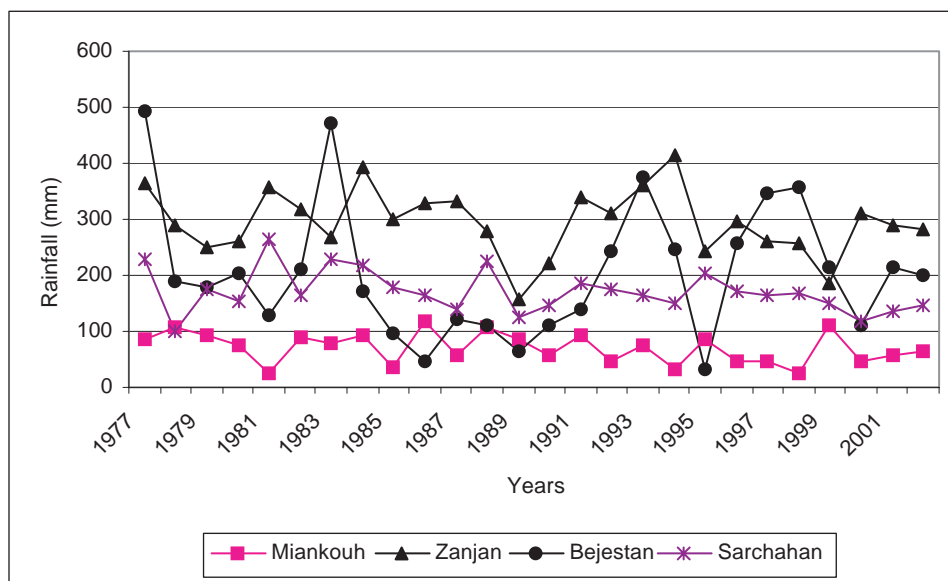
Figure 2. Annual rainfall data of gauging stations

Bejestan desert station

Bejestan is located in south of Khorasan province where the average precipitation in winter is 170 mm The existence of Ghasemabad and Yonesi plains with 2,321 and 2,267 Km^2 area, respectively, provides a preliminary for using surface waters and also flood control Considering the .06 runoff coefficient of both plains, the amount of runoff entered to desert from plain is about 35Mm^3/y. The estimated water consumption in this region is 31 Mm^3/y, all of the drinking water and 95% of the irrigation water being supplied from groundwater. Investigations show that the Annual average water consumption in 2008 will be about 2,900 m^3/d. Hence, due to negative water balance of the region, the recharge of ground waters plays an important role in improving the water condition of the area. Currently 11 Mm^3 of the surface water entering the desert is artificially recharged each year. With the utilization of flood and reclamation of aquifers, water resources of region are expected to change from 29 to 40 Mm^3 in 2008.

Table1. Aquifer characteristics of Bejestan

| Basin | Alluvial width (m) | | Water table (m) | | | Cl$^-$(mg/lit) | |
	Max	Min	Min	Mean	Max	Min	Max
Gh-Abad	200	100	3	30	110	71	2787
Unesi	150	60	4	15	20	—	—

Table 2. Volume of water loss

Basin	Area (km^2)	Annual average of precipitation (mm)	Runoff coefficient	Annual runoff (MCM)	Annual output runoff from the plain (MCM)	Annual input runoff to the plain (MCM)
Gh-Abad	2321	145	0.06	20.193	18.174	5.186
Unesi	2267	140	0.06	19.043	17.139	4.89

Miankouh station

The studied region consists of Ebrahimabad plain and a part of Yazd-Ardakan plain. Both having an average precipitation of 114 mm/y. The alluvial in this region is about 100 m thick. Water spreading in this region is established in three large waterways Fakhrabad, Manshad and Tanglabidly. The capacity of each the water spreading is 600,000 m^3. The total acquired water from this project in waterways during 1998 to 2000 was 10.4 Mm^3. The analysis of information shows that after accomplishment of the project, the maximum discharge of the Qanats in Kamalabad and Arzandazan increased from 67 to 90 l/s and 13 to 36 l/s respectively in the year 2000. Through the accomplishment of this project, the area under plantation of the region increased to 20 acres. Due to the recharge, the maximum discharge of the Qanats of Arzandazan, Saadabad and Mohammadabad increased by 180%, 41% and 7.2% respectively. Regarding the utilization of 10.4 Mm^3 flood in three different stages of 1998, 1999 and 2000 performing the water-spreading Project in the region seems to be effective.

Sarchahan station

Sarchahan plain is located in the south of Iran's plateau. The average precipitation is 204 mm/y. The alluvial fan in this area is suitable for artificial recharge. Currently 130 deep and semi-deep wells and Qanat are in use, yielding 20 Mm^3/y on average. The daily natural recharge of this reservoir was measured about 24000 m^3/d, which leads to annual rate of 10 Mm^3, however, the annual discharge of aquifer has been estimated about 20 Mm^3. Between 1979

and 2,000 a 2,000 ha large area have been submersed to act as an infiltration basin. The recharge into the aquifer during this period was 24,000 m³/day (is < 1 mm/d). We recharged 6 Mm³ in this period and the EC of water reservoirs has been decreased significantly.

Zanjan station

This plain plain is between Sahrein and Garecharion River. Design region area is 250 acres. Depth of alluvial is 80–120 meter, and the depth of ground water level varies from 40 to 60 m. The average precipitation is 350 mm. For evaluation of water spreading process on groundwater flocculation, a Qanat and three peizometric wells were used.

The design was done from 1997 to 2000. Four times of sampling was dined. At the first time of sampling outlet discharge of Qanat increased from 2.1 lit/s to 5 lit/s. At the second time (1998, March) along with maximum melting of snow, amount of discharge of Qanat increased up to 6 lit/s and 9.3 lit/s on 1999, January It reached to 14 lit/s. Maximum amount of discharge of Qanat was in the forth time (1999, April) with 32 lit/s. After this, discharge of Qanat was reduced and become constant at 7.5 lit/s by reason of drought.

CONCLUSION

These studies show that after construction of water spreading stations, affects amount of ground water level reduction, and along with winter precipitation, water table of aquifer and wells and Qanats discharges increased and EC of water resources decreased significantly

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Qanat, a traditional method
for water harvesting
in arid and semi–arid regions of Iran

A.R. Keshtkar and A. Salajegheh

Abstract

Iran is lake of water resources, especially in its arid regions. So the management of water resources in these areas is very important. The arid and semi-arid regions in Iran are mainly located in center and southeast of Iran. This paper investigated reasons of destruction and reclamation methods of Qanats in Dahak River basin. Qanats are a traditional method for water harvesting in arid and semi-arid regions in Iran and some arid regions in world. Qanats play important role to ground water utilization in arid and semi-arid regions of Iran, especially in Dahak River basin. Dahak River basin located in Southeast of Iran and Northeast of Lout plain, with total area 98,000 ha, the annual precipitation is 155 mm, the average evaporation from water surface 1,375 mm.

Keywords

Water resources; destruction; reclamation; Qanats; River basin; Iran.

INTRODUCTION

A traveler flying over Iran can see plainly that the country has an arid climate. The Iranian plateau is largely deserted. Most of Iran (excepting areas in the northwestern provinces and along the southern shores of the Caspian Sea) receives only six to 10 inches of rainfall a year. Other regions of the world with so little rainfall (for example the dry heart of Australia) are barren of attempts at agriculture. Yet Iran is a farming country that not only grows its own food but also manages to produce crops for export, such as cotton, dried fruits, and oilseeds and so on. It has achieved this remarkable accomplishment by developing an ingenious system for tapping underground water. The system, called qanat (from a Semitic word meaning 'to dig'), was invented in Iran thousands of years ago, and it is so simple and effective that it was adopted in many other and regions of the Middle East and around the Mediterranean.

Water has always played a key role in the long history of Iran. Iranians are credited for qanats and the invention of Persian wheel, two ancient irrigation systems which are well known in the world. According to here to the Greek historian, Qanat digging technique was documented and was practiced in the achaemenids era (550–330 BC) 2,500 years ago.

Remains of reservoirs have been discovered along with water intakes, spilwaysand outlets and the sewage systems belonging to pre-achamenids and Assyrians (1500–600 BC). The archaeological surveys suggest that Iranians enjoyed advanced culture and civilization some 7,000 years ago. The civilization of in the western part of Iranian plateau flourished 5,000 years ago and they invented cuneiform writing. Discoveries prove that Iranian were peaceful and ingenious people in third millennium BC who cultivated cultivated land and raised crops and livestock.

The qanat system consists of underground channels that convey water from aquifers in highlands to the surface at lower levels by gravity. The qanat works of Iran were built on a scale that rivaled the great aqueducts of the Roman

Empire. Whereas the Roman aqueducts now are only a historical curiosity, the Iranian system is still in use after 3,000 years and has continually been expanded. There are some 22,000 qanat units in Iran, comprising more than 170,000 miles of underground channels. The system supplies 75 percent of all the water used in that country, providing water not only for irrigation but also for house-hold consumption. Until recently (before the building of the Karaj Dam) the million inhabitants of the city of Tehran depended on a qanat system tapping the foothills of the Elburz Mountains for their entire water supply.

About four-fifths of the water used in the plateau regions of Iran are subsurface and is brought into use in this way. There are literally hundred of miles of qanat in Iran, and many hundreds more throughout the Arab world. The result is a sort of oasis in an otherwise arid area, creating a pleasant oasis of date palms or other crops. Indeed, an oasis could be considered a natural 'qanat', although there is no tunnel, just a spring. At the exit of a typical qanat you see a tunnel adit similar to a mine entrance, which is exactly what it is. The 'mineral' mined is water.
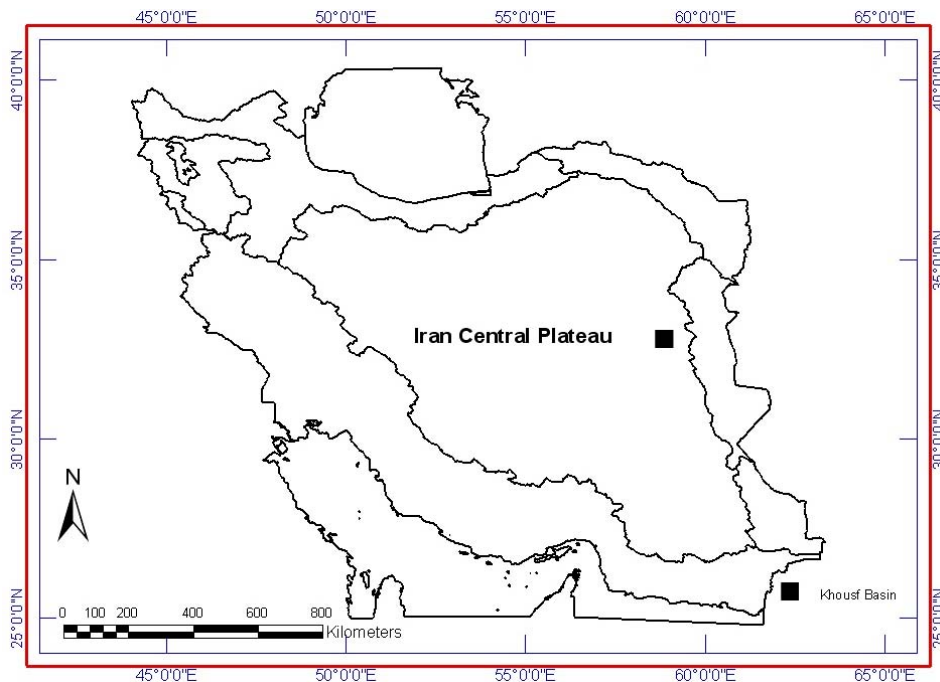
Figure 1. Geographical location of Khousf basin in Iran central plateau

Background

In the early part of the first millennium B.C., Persians started constructing elaborate tunnel systems called qanats for extracting groundwater in the dry mountain basins of present-day Iran (Figure 2). Qanat tunnels were hand-dug, just large enough to fit the person doing the digging. Along the length of a qanat, which can be several kilometers, vertical shafts were sunk at intervals of 20 to 30 meters to remove excavated material and to provide ventilation and access for repairs. The main qanat tunnel sloped gently down from pre-mountainous alluvial fans to an outlet at a village. From there, canals would distribute water to fields for irrigation. These amazing structures allowed Persian farmers to succeed despite long dry periods when there was no surface water to be had. Many qanats are still in use stretching from China on the east to Morocco on the west, and even to the Americas.

There are significant advantages to a qanat water delivery system including: (1) putting the majority of the channel underground reduces water loss from seepage and evaporation; (2) since the system is fed entirely by gravity, the need for pumps is eliminated; and (3) it exploits groundwater as a renewable resource. The third benefit warrants additional discussion.
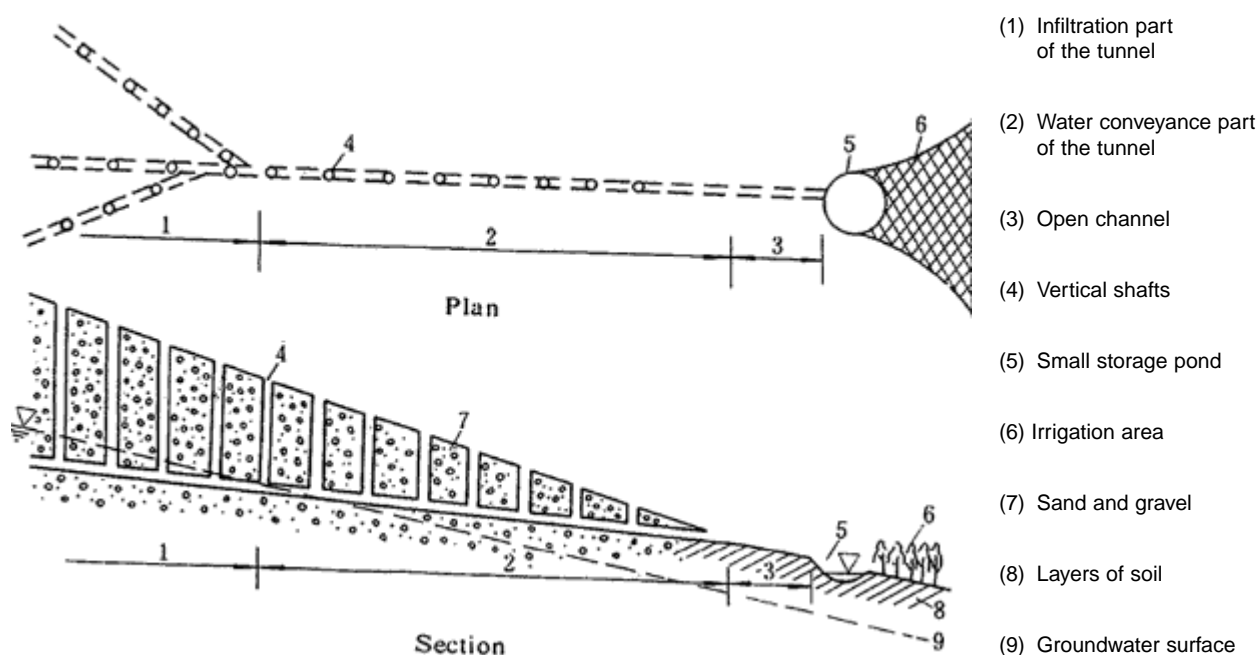
Figure 2. General schematic for a Qanat (Wulff 1968)

(1) Infiltration part of the tunnel
(2) Water conveyance part of the tunnel
(3) Open channel
(4) Vertical shafts
(5) Small storage pond
(6) Irrigation area
(7) Sand and gravel
(8) Layers of soil
(9) Groundwater surface

The rate of flow of water in a qanat is controlled by the level of the underground water table. Thus a qanat cannot cause significant drawdown in an aquifer because its flow varies directly with the subsurface water supply. When properly maintained, a qanat is a sustainable system that provides water indefinitely. The self-limiting feature of a qanat, however, is also its biggest drawback when compared to the range of technologies available today.

Water flows continuously in a qanat, and although some winter water is used for domestic use, much larger amounts of irrigation water are needed during the daylight hours of the spring and summer growing seasons. Although this continuous flow is frequently viewed as wasteful, it can, in fact, be controlled. During periods of low water use in fall and winter, water-tight gates can seal off the qanat opening damming up and conserving groundwater for periods of high demand. In spring and summer, night flow may be stored in small reservoirs at the mouth of the qanat and held there for daytime use.

Construction

A recently discovered book by Mohammed Karaji, a Persian scholar of the 10th Century AD, has a chapter on qanat construction. The techniques he describes are basically the same as those practiced today, eleven centuries later.

Qanats are constructed by specialists. A windlass is set up at the surface and the excavated soil is hauled up in buckets. The spoil is dumped around the opening of the shaft to form a small mound; the latter feature keeps surface runoff from entering the shaft bringing silt and other contamination with it. A vertical shaft 1 meter in diameter is thus dug out. A gently sloping tunnel is then constructed which transports water from groundwater wells to the surface some distance away. If the soil is firm, no lining is required for the tunnel. In loose soil, reinforcing rings are installed at intervals in the tunnel to prevent cave-ins. These rings are usually made of burnt clay. Mineral, salt, and other deposits which accumulate in the channel bed necessitate periodic cleaning and maintenance work.

In countries like Syria, qanats are rapidly drying up. In a recent exercise, three sites were chosen for renovation; each still had significant quantities of flowing water. The selection of these sites was based on a national survey conducted in 2001. The renovation of one the three (Drasiah qanat of Dmeir) was concluded in the spring of 2002.

Lessons learned from pilot projects like the one in Syria led to the development of renovation criteria which included: (i) a stable groundwater level, (ii) a consistent underground tunnel construction; (iii) social cohesion in the community using the qanat; (iv) existing system of water rights and regulation; and (vi) willingness of the water users to contribute. Cleaning of an ancient qanat is not an easy exercise. Not only is the work technically difficult, but also the social organization associated with a qanat has major implications on its future viability (Wessels, 2000).

History

The precise dating of qanats is difficult, unless their construction was accompanied by documentation or, occasionally, by inscriptions. Most of the evidence we have for the age of qanats is circumstantial; a result of their association with the ceramics or ruins of ancient sites whose chronologies have been established through archeological investigation, or the qanat technology being introduced long ago by people whose temporal pattern of diffusion is known.

Written records leave little doubt that ancient Iran (Persia) was the birthplace of the qanat. As early as the 7th century BC, the Assyrian king Sargon II reported that during a campaign in Persia he had found an underground system for tapping water. His son, King Sennacherib, applied the 'secret' of using underground conduits in building an irrigation system around Nineveh.

During the period 550–331 BC, when Persian rule extended from the Indus to the Nile, qanat technology spread throughout the empire. The Achaemenid rulers provided a major incentive for qanat builders and their heirs by allowing them to retain profits from newly-constructed qanats for five generations. As a result, thousands of new settlements were established and others expanded. To the west, qanats were constructed from Mesopotamia to the shores of the Mediterranean, as well as southward into parts of Egypt. To the east of Persia, qanats were constructed in Afghanistan, the Silk Route oases settlements of central Asia, and Chinese Turkistan (i.e. Turpan).

During Roman-Byzantine era (64 BC to 660 AD), many qanats were constructed in Syria and Jordan. From here, the technology appears have to diffuse north and west into Europe. There is evidence of Roman qanats as far away as the Luxembourg area.

The expansion of Islam initiated another major diffusion of qanat technology. The early Arab invasions spread qanats westward across North Africa and into Cyprus, Sicily, Spain, and the Canary Islands. In Spain, the Arabs constructed one system at Crevillente, most likely for agricultural use, and others at Madrid and Cordoba for urban water supply. Evidence of New World qanats can be found in western Mexico, in the Atacama regions of Peru, and Chile at Nazca and Pica. The qanat systems of Mexico came into use after the Spanish conquest.

While the above diffusion model is nice and neat (Figure 3), human activities are rarely so orderly. Qanat technology may have been introduced into the central Sahara and later into Western Sahara by Judaized Berbers fleeing Cyrenaica during Trajan's persecution in 118 AD. Since the systems in South America may predate the Spanish entry into the New World, their development may have occurred independently from any Persian influence. The Chinese, while acknowledging a possible Persian connection, find an antecedent to the qanats of Turpan in the Longshouqu Canal (constructed approximately 100 BC). The Romans used qanats in conjunction with aqueducts to serve urban water supply systems (a qanat-aqueduct system was built in Roman Lyons). A Roman qanat system was also constructed near Murcia in southeastern Spain. The Catalan qanat systems (also in Spain) do not seem to have been related to Islamic activity and are more likely later constructions, based on knowledge of Roman systems in southern France.

Qanats were an important factor in determining where people lived. The largest towns were still located at low ele-

Photograph 1. A windlass is used to bring tunnel spoil to the surface
(display at the Qanat Museum in Turpan, China)
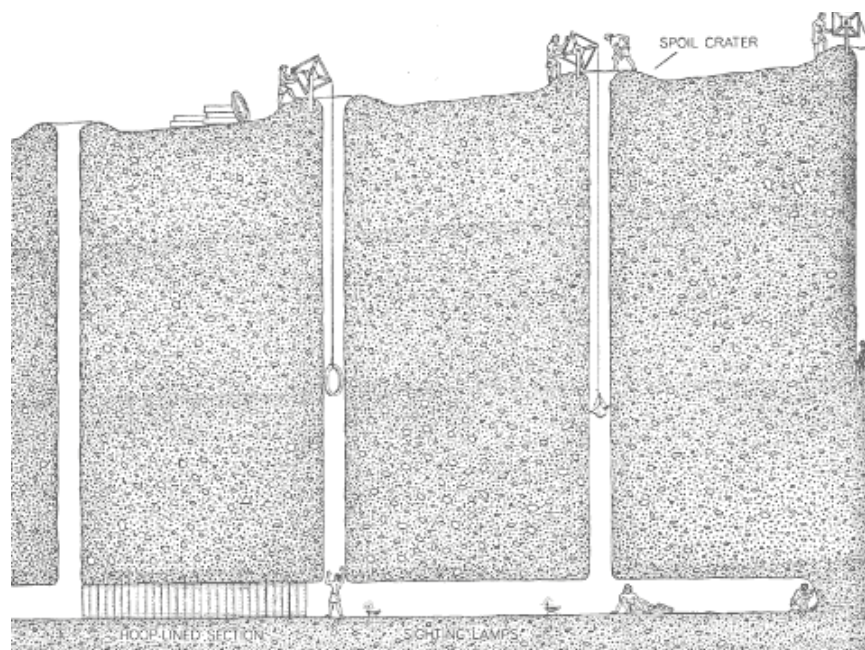
Figure 3. Constructing a qanat using reinforcing rings (from Scientific American)

vations on the floors of intermountain basins and in broad river valleys. Most of these early settlements were defended by a fortress and watered by hand-dug wells sunk into a shallow water table. Qanats enabled these settlements to grow by tapping water-rich aquifers located deep beneath neighboring alluvial fans.

Even more dramatically, qanats made possible the establishment of permanent settlements on the alluvial fans themselves. Earlier settlers had bypassed the areas because water tables there were too deep for hand-dug wells, and the wadis on these slopes were too deeply incised in the fans for simple diversion channels. In these locations, qanats tapped adjacent aquifers with underground tunnels fed with water drawn from upslope alluvial deposits in mountain valleys. For the first time, at these higher elevations, small qanat-watered hamlets appeared.

Qanats and disease

Qanats were frequently used for domestic purposes, as well as irrigation. Because of this, they can transport disease vectors (Afkhami, 1997). A chemical analysis of water, conducted in 1924, from 6 qanats as they entered Tehran revealed water of potable quality in only 2 cases. In 3 other, water purity was questionable and in 1 case the water was definitely unfit for drinking. These results were especially shocking since the samples were taken from closed qanats before they were open to contamination. It has been hypothesized that qanats were a major contributor to the cholera epidemics of the 19th century.

Throughout Iran, even if the qanat water was uninfected before entering the cities, it had ample opportunity to become contaminated while traversing the urban areas in open ditches. With the lack of proper sewage and waste disposal throughout Iranian municipalities, the cholera bacterium easily made its way into drinking water.

Passive cooling systems

Qanats can be used for cooling as well as water supply (Bahadori, p. 149). One technology operates in conjunction with a wind tower. The arid regions of Iran have fairly fixed seasonal and daily wind patterns. The wind tower harnesses the prevailing summer winds to cool and circulate it through a building. A typical wind tower resembles a chimney, with one end in the basement of the building and the other end rising from the roof. Wind tower technologies date back over 1,000 years.

The passive cooling of a wind tower can be enhanced by connecting it to an underground stream or qanat. In the system shown in Figure 4, a shaft (b) connects the qanat to the basement of the building to be cooled. Hot dry air enters the qanat through one of its vertical shafts (a) and is cooled as it flows along the water. Since the underground water is usually cold, the rate of cooling is quite high. The wind tower is placed so that wind flowing through the basement door of the tower passes over the top of the qanat tunnel. When the air flows from a large passage (the tunnel) through a smaller one (the door), its pressure decreases. The pressure of the air from the tower is still diminished when it passes over the top of the tunnel, so that cold moist air from the shaft is entrained by the
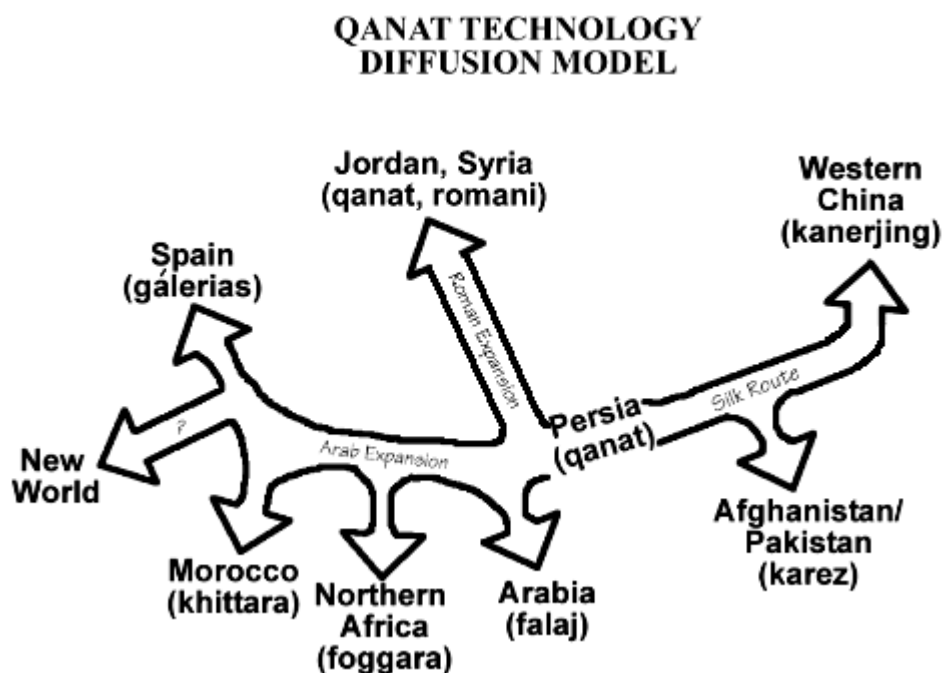
Figure 4. One possibility for the diffusion of qanat technology

flow of cooled air from the tower (c). The mixture of air from the qanat and air from the tower (d) circulates through the basement. A single qanat can serve several wind-tower systems.
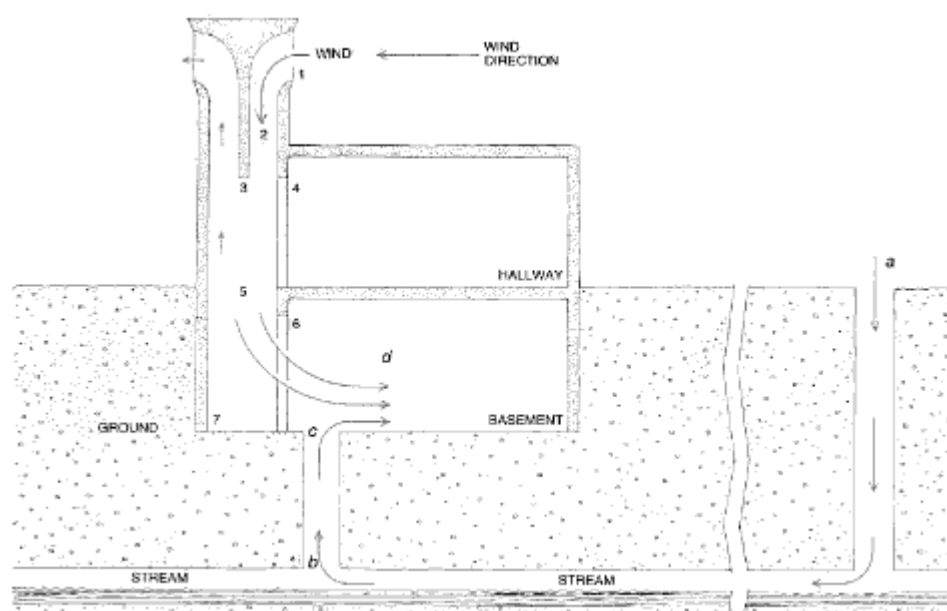
Figure 5. The air flow in a combination wind tower/qanat cooling system (from Scientific American)

MATERIAL AND METHODS

Qanats play important role to ground water utilization in arid and semi-arid regions of Iran, especially in Dahak River basin. Dahak River basin located in Southeast of Iran and Northeast of Lout plain, with total area 98,000 ha, the annual precipitation is 155 mm, the average evaporation from water surface 1,375 mm. Qanats are to this day the major source of irrigation water for the fields that occupy parts of Dahak river basin. There are 33 qanats in Dahak river basin that decrease of rainfall, penetration of Haloxylon seedlings and raises a load of accumulated silt in the process of cleaning a qanat conduit tunnel cause's damage to qanat galleries. For reclamation and raise of discharge these Qantas in Dahak river basin need that do water spreading for artificial recharge, preventing of Haloxylon seedling around qanats and cleaning of qanats tunnel.

Table 1. Annual rainfall data of gauging station

Station Year	Khousf	Station Year	Khousf	Station Year	Khousf	Station Year	Khousf
1975	162.3	1982	264	1989	128.2	1996	219.5
1976	230.3	1983	177	1990	178.5	1997	186.5
1977	145	1984	141	1991	181	1998	184
1978	126.3	1985	83.5	1992	219.1	1999	174.6
1979	216.5	1986	310.5	1993	144.2	2000	62
1980	78.5	1987	156	1994	124.5	2001	115.1
1981	155.5	1988	184	1995	165	2002	147

CONCLUSION

A qanat system has a profound influence on the lives of the water users. It allows those living in a desert environment adjacent to a mountain watershed to create a large oasis in an otherwise stark environment. The United Nations and other organizations are encouraging the revitalization of traditional water harvesting and supply technologies in arid areas because they feel it is important for sustainable water utilization. Investigations shows that in Dahak river basin qanats will reclamation by water spreading for artificial recharge, preventing of Haloxylon seedling around qanats and cleaning of qanats tunnel.

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Recharge Systems for Protecting and Enhancing Groundwater Resources, is the theme of the 5th International Symposium on Management of Aquifer Recharge, Berlin 10–16 June 2005, one of a series of symposia:

Symposium	Proceedings/Theme	Publisher
1. 23–27 Aug 1988 Anaheim, USA	First International Symposium on Artificial Recharge of Groundwater	ASCE
2. 17–22 Jul 1994 Orlando, USA	2nd International Symposium on Artificial Recharge of Groundwater	ASCE
3. 21–25 Sep 1998 Amsterdam, NL	3rd International Symposium on Artificial Recharge of Groundwater (TISAR) *Store or Restore*	Balkema
4. 22–26 Sep 2002 Adelaide, Aus.	4th International Symposium on Artificial Recharge of Groundwater (ISAR4) *Management of Aquifer Recharge for Sustainability*	Balkema
5. 10–16 Jun 2005 Berlin, Germany	5th International Symposium on Management of Aquifer Recharge (ISMAR5) *Recharge Systems for Protecting and Enhancing Groundwater Resources*	UNESCO

Since the turn of the 21st century the term 'Artificial Recharge' has been replaced by 'Management of Aquifer Recharge' to reflect contemporary endeavours to intentionally enhance the quality and quantity of groundwater resources as an integral part of water resources management. Recent conference themes reflect the progress from the initial primary focus on enhancing the volume of water supplies to also addressing water quality and environmental issues.

With climate change, population growth, and increasing urbanisation, the proportion of the world's population living in water stressed areas is increasing. In 2001 20% of the world's population did not have access to safe drinking water supplies. Managed aquifer recharge (MAR) using an increasing variety of methods and sources of water, including reclaimed water, provides potential for replenishing supplies. These extensions also demand a greater understanding of processes occurring in recharge operations to enable improved siting, design, operation and governance of MAR and to develop new measurement methods, models and maintenance procedures to reduce risks and enable sustainable practices. This volume documents progress in all of these areas over the last three years. It is further enriched by new knowledge developed through the NASRI project of the Berlin Water Competence Centre and research and utility partners who have intensively studied bank and pond filtration processes for Berlin's water supply.

This conference series initiated by the American Society of Civil Engineers and now jointly run with the International Association of Hydrogeologists aims to meet the need for generating and disseminating knowledge that will ensure that MAR achieves its potential internationally as a valuable water resources management instrument. The UNESCO International Hydrological Programme (IHP) and the IAH have been supporting the development of an international MAR expert's network since the year 2000. This is the main objective of the IAH Commission on Management of Aquifer Recharge whose web site and email list can be accessed at http://www.iah.org/recharge UNESCO has been a supporter of all conferences in this series, and has taken a publication role with this proceedings in order to extend this knowledge to the widest possible audience, particularly in arid and semi arid regions and communities in developing countries in need of safe water supplies.