Water quality aspects of artificial aquifer recharge with treated waste water

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Abstract

The use of waste water for artificial recharge of aquifers is considered. The physical and chemical processes during percolation of treated waste water to the saturated zone of an aquifer were studied. International experience in this field is summarised in a literature review. Column tests were conducted in order to obtain preliminary information on the loading rates of artificial aquifer recharge under local conditions. Removal of phosphates is found to occur as well as other improvements to the water.

Introduction

Persistant drought conditions in southern Africa during the early 1980's and subsequent water restrictions in the Witwatersrand area brought about an intensified research effort in the field of water resources.

One school of thought is to import water from e.g. the Orange River, the Zambezi River or even the Okavango Swamps to the PWV-area. High costs due to the long distances to be bridged and political vulnerability for this type of scheme might render the idea prohibitive.

Conjunctive use of surface and ground water

The utilisation of ground-water resources is another alternative. If ground water is used in conjunction with surface water resources, not only may deficits be met, but the rate at which river water can be abstracted may even be increased as no allowances have to be made for periods of drought. The efficiency of such a scheme depends on the availability and size of suitable aquifers. Primary aquifers consisting of unconsolidated alluvial deposits are virtually non-existant in the country, but secondary aquifers occur abundantly. The ground water is stored in fractures and fissures. In dolomites these cracks have been widened more in comparison with other formations due to the relatively high solubility of carbonate rocks. Subsequent erosion contributed further to make dolomite formations the most suitable type of secondary aquifer for large-scale water storage. Several vast outcrops of dolomite are to be found in the Republic. One such outcrop of considerable dimensions is located in the direct vicinity of Johannesburg (Figure 1).

The potential contribution of ground water to the water supply system has to be significant in order to facilitate an efficient conjunctive use in the PWV-area. Although the present storage in the dolomites appears to be considerable (Jordaan *et al.*, 1960), high rate abstraction over a number of years with low rainfall could deplete the reserves prematurely. The natural recharge due to percolation from rainfall and streambeds in an average year is, for the dolomites near Johannesburg, approximately 40 to 100 mm of water per year (Enslin and Kriel, 1967). The extent to which ground water could be abstracted can be improved by the introduction of artificial recharge of the dolomite formations. During years of good rainfall the overspill of Vaal River dams could provide the water required for this purpose. However, recycling of partially purified waste water would be a more steady

and reliable scheme. Therefore research within the Water Systems Research Programme has concentrated on this possibility.

Dolomites are well known for subsidence dangers. Even if the recharged water is chemically stable, dissolution could still occur at depth where the pressures increase. Further stresses are also exerted by the ponds on the surface. An extensive tectonic and geohydrological survey will be required.

The study was divided into a quantitative and a qualitative section. The results of a simulation model indicated that under certain assumptions the continuous conjunctive draft could increase the supply by 395 Mℓ/d (12%) over the case where one draws on Vaal River water only. Again under certain assumptions a non-linear optimisation model for the water flows in the southern Witwatersrand showed that even under a continued growing demand water shortage could be prevented well into the next decade and water made available at lower costs by the introduction of a combined water supply (Paling, 1985).

The acceptance of indirect reclamation by means of artificial aquifer recharge with waste water and subsequent ground-water abstraction will strongly depend on the eventual water quality.

This paper evaluates some of the physico-chemical aspects of the infiltration process, based on a review of results from worldwide laboratory experiments and pilot plants as well as simple column tests conducted at the University of the Witwatersrand.

Artificial aquifer recharge can take place either by means of injection wells or infiltration from the surface. Although well injection may be suitable for certain applications water quality requirements for this type of recharge are high. A water quality of less than tertiary effluent may cause a rapid clogging of the wells (Huisman and Oltshoorn, 1981) and may pollute the aquifer to an unacceptable degree, as limited additional purification takes place in the saturated zone. In the following sections only infiltration is considered.

Previous experiments and pilot projects

Montgomery *et al.* (1984) reported on traditional small-scale disposal of crude sewage in well-fractured, fine grained limestone in Hampshire (UK). Ground-water recharge is obviously only a matter of secondary importance here, but remarkable purification results were observed.

Overland treatment of sewage works effluent with incidental ground-water recharge has been practised near many cities in the USA. Since the 1960's high-rate land treatment of sewage works effluent for the specific purpose of recharge has been practised in a number of research and pilot projects of which Flushing
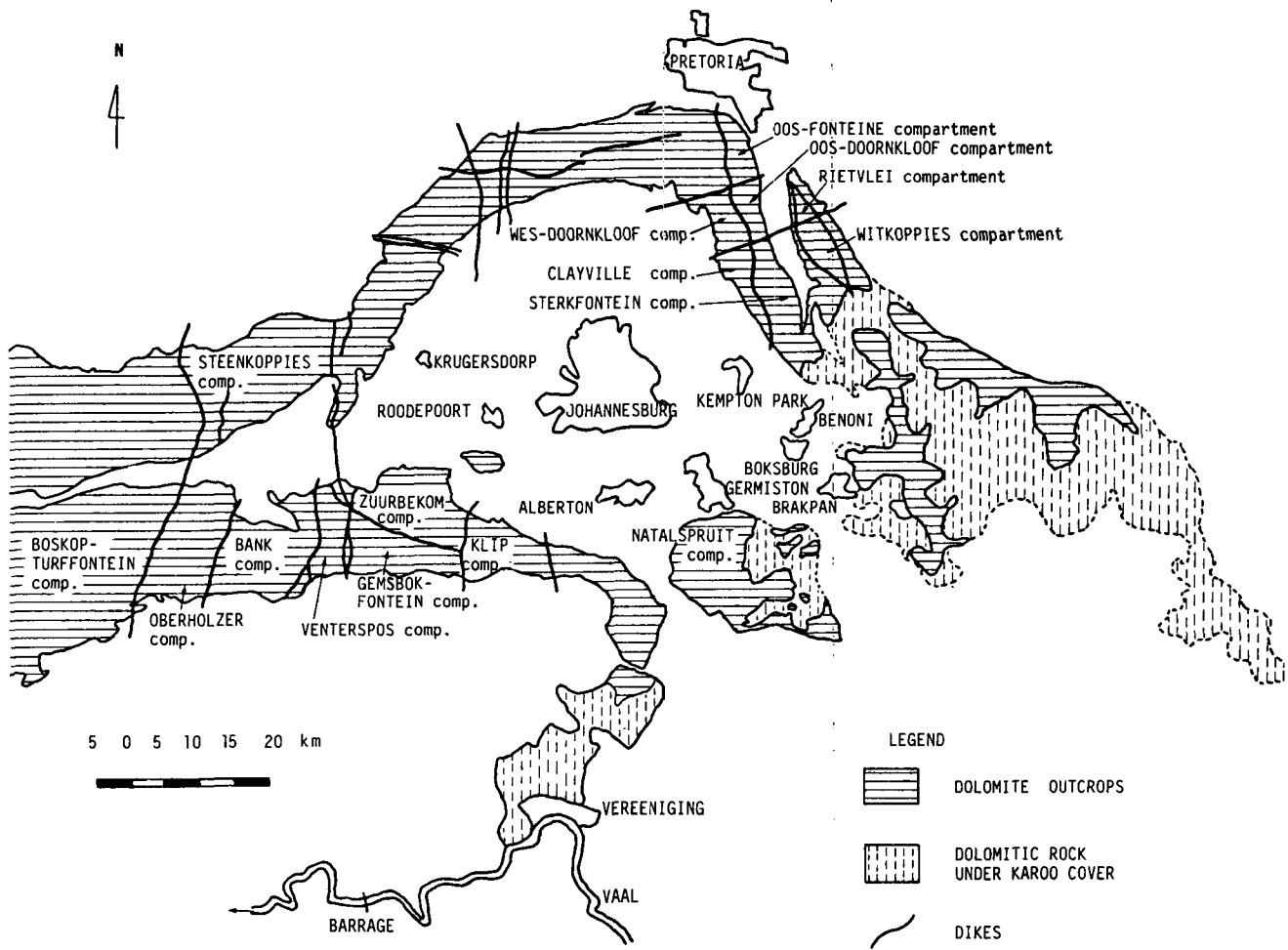
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Figure 1
Dolomite deposits.

PRETORIA

OOS-FONTEINE compartment
OOS-DOORNKLOOF compartment
RIETVLEI compartment

WES-DOORNKLOOF comp.
WITKOPPIES compartment

CLAYVILLE comp.
STERKFONTEIN comp.

STEENKOPPIES comp.

KRUGERSDORP

ROODEPOORT

JOHANNESBURG KEMPTON PARK

BENONI

BOKSBURG
GERMISTON
BRAKPAN

ZUURBEKOM comp. ALBERTON

BOSKOP-
TURFFONTEIN
comp.

BANK comp.

KLIP comp.

NATALSPRUIT comp.

OBERHOLZER comp.

GEMSBOK-
FONTEIN comp.

VENTERSPOS comp.

5 0 5 10 15 20 km

VEREENIGING

VAAL
BARRAGE

LEGEND

DOLOMITE OUTCROPS

DOLOMITIC ROCK
UNDER KAROO COVER

DIKES

Meadows, Phoenix, Arizona is perhaps best known. This project, which ran from 1967 until 1978, is well documented by Bouwer (1970), Bouwer et al. (1974A, 1974B, 1980), Lance (1972) and others. Presently it is being followed up on a larger scale by the so-called 23rd Avenue Project, Phoenix. Tredoux et al. (1980) studied the feasibility of storage and abstraction of reclaimed waste water in the Cape Flats aquifer, Cape Town. Leach and Enfield (1983) reported on nitrogen removal experiments in Pontococ County, Oklahoma and Mathew et al. (1982) on similar tests in the Swan Coastal plain, Australia. Idelovitch (1978) and Idelovitch and Michail (1984) described the Dan Region Project, Israel. In this project the sewage treatment works and the groundwater recharge and abstraction form an integrated recycling process.

All previous studies relate to sandy aquifers. A general conclusion is that most purification takes place in the top few metres of soil during percolation. Limited additional purification was observed during underground flow in the fully saturated zone. Dolomite deposits covered by a few metres of weathered rock or alluvial deposits may therefore provide an equally suitable aquifer.

A theoretical distinction in the purification process during percolation can be made between the interception of suspended matter and the removal of dissolved chemicals. In practice, however, the distinction is not as clear and the various processes interact together.

Interception of suspended matter

Large particles in the waste water with a diameter down to one fifth of the effective grain size (d_{10}) can be intercepted by mechanical straining. The pore size which determines the straining efficiency is, however, not only dependent on the effective grain size but also on the particle size distribution, measured in terms of the Hazen Uniformity coefficient, defined as $Cu = d_{60}/d_{10}$, and the compaction of the soil. The variables d_{10} and d_{60} represent the sieve mesh size through which a 10% respectively 60% (by weight) fraction of the total sample passes.

Removal of suspended matter can, however, be accompanied by other problems. Blocking of the surface layers of the aquifer can decrease the infiltration capacity and reduce the aeration of underlying water.

Suspended matter of finer sizes than the pore openings may be removed by sedimentation and attraction to the surface of the sand grains. The total available surface area is formed by that part of all sand grains that is facing upward, not in contact with other grains and not exposed to scour.

Adsorption causes the retention of finely divided suspended matter with sizes as small as colloidal and molecular dissolved impurities. A distinction is made between active and passive adsorption. Active adsorption can be attributed to the physical attraction between two particles of matter (Van der Waals forces) and the electrostatic attraction between opposite electric charges (Coulomb forces). Passive adsorption refers to the retaining of suspended particles by the sticky gelatinous coating on the sand grains formed by previously deposited bacteria and organic matter. The difference between passive and active adsorption in the meanwhile is small as the forces of attraction exert their influence over a small fraction of the pore space only.

Removal of dissolved chemicals

Removal of suspended solids is complemented by the reduction of a wide range of contaminants by biochemical and bacteriological agents. Organic matter is broken down by the action of micro-organisms into mineral constituents, while inorganic matter may be converted into insoluble compounds, forming a thin coating around the sand grains or a network structure within the pores after which they belong to the solid part of the aquifer.
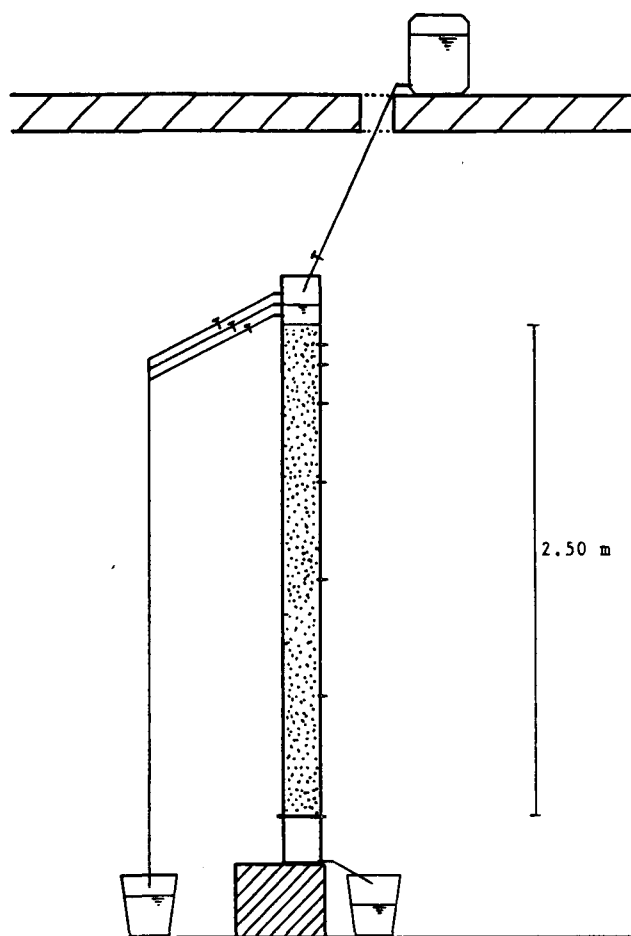

Figure 2
Percolation test set-up.

The transformation and reduction in concentration of dissolved nitrogen has received considerable attention in literature, e.g. Lance (1972), Reeves (1972), Mathew et al. (1982), and Leach and Enfield (1983). Most organic nitrogen (like proteins) in waste water is broken down during secondary treatment into smaller units (e.g. amino acids, amines and amides) and then into ammonia and nitrate.

The most desirable nitrogen removal mechanism is one by which nitrogen is transformed into a relatively stable form that will not contaminate ground water or contribute to eutrophication. The reduction of nitrate to nitrogen gas by biological denitrification meets this requirement. Soils contain an abundance of denitrifying bacteria that can use oxygen as an electron acceptor under aerobic conditions or nitrate in the absence of oxygen. The denitrification rate depends on the temperature and the pH of the system. Denitrification proceeds slowly below pH 5,5 and below 10°C. Most of the gas evolved above pH 6,0 is N_2, while N_2O and NO are predominant at low pH. Ammonium ions in the secondary effluent can be oxidised by nitrifying bacteria via nitrite into nitrate. Nitrification can be accomplished in a ground-water recharge system with an intermittent flooding schedule, creating alternate aerobic (nitrification) and anaerobic (denitrification) conditions.

During flooding residual dissolved nitrogen compounds from the previous flooding period are leached to the ground water. Optimum operating schedules in respect of reduction of nitrogen concentration may vary from 1-day flooding and 1-day drying (Leach and Enfield, 1983), and 2-days flooding and 2-days drying (Lance, 1972), to 10-days flooding and 12-days drying (Mathew et al., 1982). Optimisation of the flooding and drying periods for nitrogen reduction may conflict with the results obtained by optimising the infiltration rate.

The partial or total removal of chemical constituents other than nitrogen is primarily attributed to adsorption on clay and silt lenses and to chemical precipitation and only a very limited extent to bacteriological activity. This implies that eventually a breakthrough of these constituents will occur. According to Idelovitch and Michail (1984) this breakthrough appears to be remote for some chemicals (e.g. phosphorus and detergents), but e.g. the sodium and boron content in the abstraction water in the Dan Region project reached levels equal to those of the infiltration water within a few years.

Insufficient removal of the common minerals from the secondary effluent can form a nuisance insofar as the renovated water has either to be restricted to a limited number of applications, or will require extensive post-treatment. However, the major concern is the possible survival of micro-organisms during soil-aquifer treatment and the breakthrough of trace elements such as heavy metals, insecticides and herbicides.

Viruses and bacteria

According to Bouwer et al. (1974B) most faecal bacteria like faecal streptococci and coliforms die off in the top metre of soil. Virus removal is primarily due to adsorption onto silt and clay particles, and most viruses are adsorbed in the top 5 cm of soil. Deeper penetration of faecal bacteria and viruses was observed after extended drying periods and after heavy rainfall within 24 h after infiltration had stopped.

If faecal bacteria and viruses reach the ground water an underground travel distance in soil of some 100 m seems to be sufficient to reduce their levels to essentially zero. If fissured or fractured substrata are present, longer travel distances may be required. Chlorination of biologically treated effluent has a

negative effect on the removal of coliform bacteria in soil (Montgomery *et al.*, 1984).

Trace elements

Heavy metals and several insecticides, herbicides and chemical wastes can be toxic or possibly carcinogenic even in small concentrations.

The broad range of compounds and the often cumbersome measurement methods, particularly for trace organics, makes any form of water recycling suspect. The potential presence of such substances in treated waste water forms at present the major obstacle for large scale waste-water recycling by infiltration for potable water purposes.

A certain amount of reticence is justified when one is dealing with drinking-water supply where mismanagement can have far-reaching detrimental results. On the other hand one should weigh the advantages and disadvantages of closely monitored, artificially recharged ground water and surface water, the quality of which is often taken for granted.

To miminise the concentration of chlorinated hydrocarbons in the renovated water, waste-water effluent used for groundwater recharge should not be chlorinated prior to infiltration (Bouwer *et al.* 1980, and Idelovitch, 1978).

In case a breakthrough of trace elements and trace organics should occur, post-treatment with activated carbon filters may be introduced to reduce the concentration of these substances.

Experiments

From the previous sections it can be concluded that the performance of an infiltration scheme strongly depends on local conditions and management of the infiltration.

A first attempt was made to collect some indicative data for infiltration of final effluent from biological filters of the Johannesburg Olifantsvlei waste-water purification works. Infiltration was initially considered on alluvial deposits of the Klip River on top of dolomite formations of the Klip River compartment, to the south of Johannesburg. Extensive use of fertilisers on most fields in the area and the complexities involved with collecting groundwater data in the field suggested a different approach. Soil from non-fertilised fields in the same area was collected and used to fill a 200 mm diameter tube. This column was then subjected to infiltration tests.

The percolation test set-up is illustrated in Figure 2. Treated waste water is stored in a 50 ℓ high-level reservoir in a cooled environment. From here a valve-regulated flow is fed onto the 2,5 m long soil column. A constant head is maintained by overflow outlets at 50, 100 and 150 mm above the top of the soil column. For future water quality monitoring at different depths a number of outlets were made along the side of the column. As they were not used for the present tests they were plugged off with rubber stops. The soil is supported at the bottom by a wire mesh. Filtered water is collected from an outlet at the bottom of the tube.

Treated waste water was collected every three or four days and used to refill the high-level reservoir.

A number of soil samples was collected from 0,5 to 1,0 m below surface and tested prior to filling the tube in order to establish some soil characteristics and to enable a reasonable approximation of the field conditions. The unit mass was found to be in the range of 1 730 to 1 780 kg/m^3 at a moisture content of *circa* 12%. Six samples were dried and sieved and the cumulative
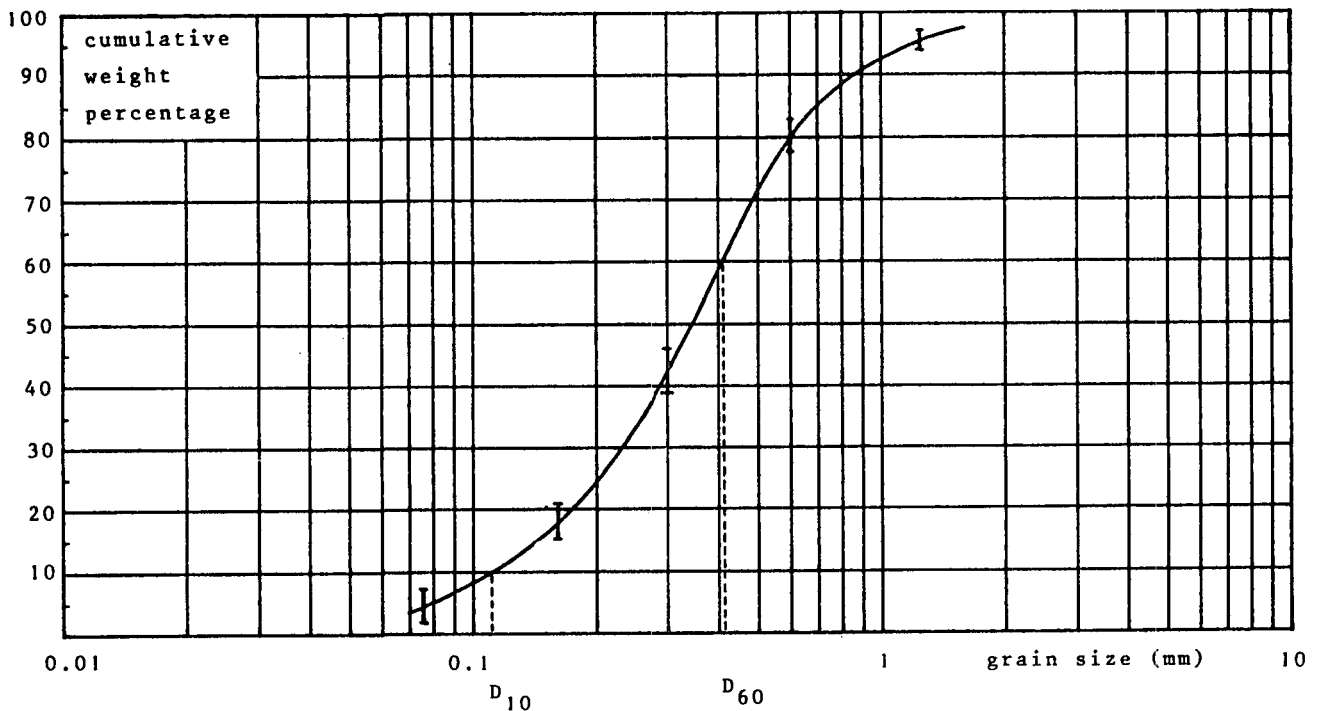
Figure 3
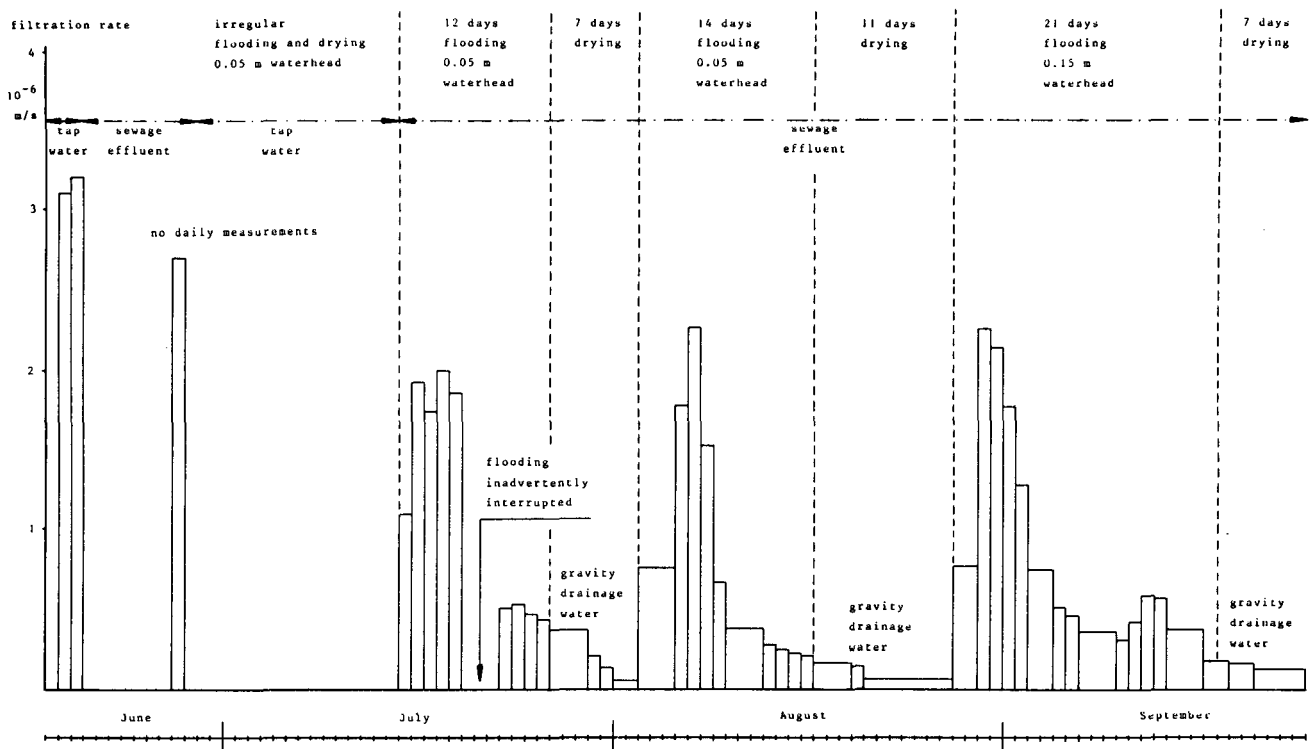*Sieve curve, averaged over six samples.*

Figure 4
Column filtration rates.

weight percentages were plotted against grain size (Figure 3).

With D_{10} = 0,102 mm and D_{60} = 0,411 mm the Hazen uniformity co-efficient becomes D_{60}/D_{10} = 4,1. The average grain size is fairly small. This combined with the rather high uniformity coefficient i.e. with a poor uniformity, may result in a relatively low hydraulic conductivity. The porosity was found to be *circa* 38%.

Results and discussion

Based on the findings of the literature review it was decided to concentrate attention on three aspects of water purification during percolation through soil:

● filtration rate;

● nitrogen processes; and

● breakthrough curves for some other chemical constituents in the treated water.

Filtration rate

The filtration rate is defined here as the volume of the outflow divided by the cross-sectional area of the soil column per unit of time.

Initially the soil column was fed with tap water in order to establish a reference value for the filtration rate. A value of 3,2 × 10^{-6} m/s or 0,28 m/d was found.

This implies that the minimum time required for a water drop to travel through the soil column is 9 d. However, when during a drying period the water level drops below the top of the soil column a contraction of the soil was observed, leaving an opening between the tube and the soil of approximately 0,5 mm. Noting that the average filtration rate during drying is approximately 0,2 × 10^{-6} m/s or 17 mm/d and using a porosity value of 40%, this opening along the wall after a week may theoretically extend up to approximately 0,30 m below surface, which reduces the travel time for some of the water molecules through the column by one day.

During the first flooding-drying cycle using treated waste water the filtration rate was not closely monitored. Prior to the next cycles the column was flushed with tap water again.

Figure 4 shows a considerable improvement of the filtration rate after each drying period followed by a rapid decline. The decline is slower for a greater water head. This may be explained as follows. Suspended solids in the treated waste water are mainly deposited on the surface of the soil column and form a thin layer of low permeability. During drying this layer cracks and forms curled flakes, which re-exposes the original soil in places. The next flooding period supplies new suspended solids that fill those gaps and cause the filtration rate gradually to drop again. For a column filled with soil as used here the low-permeable layer on the surface determines the filtration rate. A slightly higher water pressure as a result of a relatively small increase in water head has therefore a significant influence on the filtration rate.

The filtration rate peaks could possibly decrease over a period of time and necessitate removal of the clogged-up top layer, but the results collected indicated no necessity to do so.
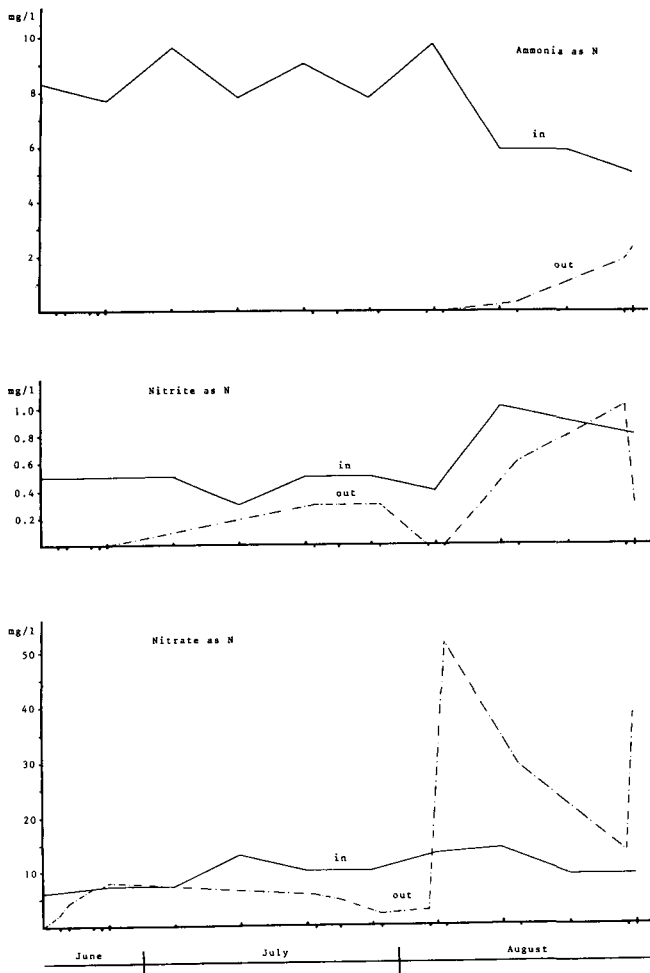
Figure 5
Column test results for nitrogen related compounds.

No provisions were made to mix the content of the high-level reservoir and some settling of suspended solids was observed. However, before every refill operation of the high-level reservoir the surplus was flushed via the top of the tube and part of the settled-out material was deposited on top of the soil. The turbidity of the filtrate varied between 0,3 and 1,2 NTU, which compares well with that of tap water (0,6 to 2,0 NTU).

Chemical constituents

The study of various chemical constituents involved the following procedures. The secondary effluent had to be collected every three to four days to prevent chemical deterioration in the high-level reservoir. Secondary effluent was analysed by the municipal chemical laboratories of the Johannesburg City Council at 8 d intervals. Their test results provided the information on the input water quality for the column test. Samples of the column effluent were collected at irregular intervals of 3 to 7 d, preserved with mercuric chloride and also tested at the same laboratory.

Nitrogen processes

Figure 5 gives the results for nitrogen-related compounds. Initially a 100% reduction in ammonia was found, but in more recent column effluent samples it was on the increase. This may indicate

that the nitrifying process has lost its efficiency. Ammonia may have travelled too far down the column during flooding, so that, when during drying an aerobic environment is created, the nitrifying bacteria cannot reach all ammonia ions. This should indicate that shorter flooding periods are more appropriate.

Nitrite concentrations fluctuate without specific trends at levels below the maximum permissible levels for drinking water (1,0 mg/ℓ). A sharp drop at the end concurs with an increase in ammonia, which confirms the supposition of a reduced nitrifying efficiency. The fact that no excessive increase in nitrite content occurs in the column effluent samples may indicate that sufficient carbon is available in the secondary effluent to not limit the denitrification process in which nitrite is converted into nitrate.

The initial low nitrate levels are followed by two peaks. The start of a new flooding period causes leaching of nitrates, which is reflected in a high concentration in the column effluent samples. The peaks reach fairly high values compared with the concentrations in the secondary effluent, even if the contribution of ammonia converted into nitrates is taken into account. This may be attributed to a storing up prior to the first breakthrough.

The second peak is lower than the first one.

In the present tests the flooding and drying cycles as well as the water head were determined by the progress of the filtration rate. No attempt was made to optimise the nitrogen removal processes.

Chemical constituents other than nitrogen

Both Bouwer *et al.* (1980) and Idelovitch and Michail (1984) found a removal efficiency of 80% or higher for ortho-phosphate. In the present column tests these findings are confirmed as the ortho-phosphate concentration in the column effluent samples was nil during the first three months of testing (Figure 6). Information on the ortho-phosphate concentration in the secondary effluent is confidential and since a 100% removal rate was found it was decided to leave out the scale on the vertical axis.

Idelovitch and Michail (1984) observed a poor removal rate of sodium. The column test effluent shows an increase in sodium concentration almost immediately from the beginning. This may be partly due to release of sodium retained at the beginning of the test. Displacement of sodium ions by ion exchange may contribute to the increase in sodium during the passage through soil. The concentration of soluble organics, measured by the gross parameter COD is, after an initial peak, considerably reduced by infiltration, but shows an increase after some time to values approximating those of the secondary effluent. Reduction is attributed to two major processes: biodegradation and adsorption (Idelovitch and Michail, 1984). Some organic compounds are adsorbed by soil until a breakthrough occurs. It is difficult to tell at this stage whether the biodegradation efficiency in the column test has dropped or a breakthrough of adsorbed material has occurred.

Conclusions

Experience abroad and local small-scale tests indicate a good potential from the quality improvement point of view for artificial recharge of an aquifer with secondary effluent.

The tests indicated an almost total removal of suspended solids down a 2,5 m long soil column. The column effluent showed lower turbidity values than local tap water.

In the early stages of the column tests ammonia reduction was 100%. After a few flooding and drying cycles the ammonia

concentration started to increase, resulting in a reduction of approximately 60%. The degree of ammonia reduction is primarily a function of the length of the flooding and drying periods. A cycle of approximately 11 d flooding and 11 d drying proved to be inappropriate. The ammonia carried down by the infiltration water penetrates too far down the column to be reached by nitrifying bacteria during a subsequent drying period, and is flushed out the column during the next flooding period. A more rapid succession of shorter flooding periods together with adjusted drying periods is likely to provide reduction rates that continue to be high.

Nitrates in the column effluent originate directly from the infiltrate or by bacteriological processes from the infiltration borne ammonia. Based on results in the literature (Bouwer *et al.*, 1980) a reduction in nitrate concentration was anticipated, but none was observed during the column tests. After an initial, partial retention of nitrates in the column, the concentrations in the column effluent increased to values higher than those in the infiltration water. Further experiments under various conditions may establish whether the poor nitrate reduction is caused by the duration of the flooding and drying cycle, the quality of the secondary effluent, or the soil properties.

A complete elimination of ortho-phosphate was observed during the duration of the soil column tests. This observation confirms the results found in the literature, where effective phosphate retention is attributed to precipitation of phosphate as calcium-phosphate compounds.

Sodium on the other hand is a more conservative element and showed a rapid breakthrough. After a few flooding and drying cycles the sodium concentration in the column effluent fluctuated at a level equal to the concentration of the infiltrate.

During the tests the concentration of COD was considerably reduced by infiltration, but showed an increase after some time to values approximating those of the infiltrate.

Recommendations for further research

A three month monitoring period is obviously too short to come to any conclusive results. Therefore, column tests should be continued, shifting the accent from optimising the filtration rates to optimising the removal efficiency of nitrogen and other chemicals. During further tests a wider spectrum of chemicals should be monitored, including e.g. the Total Dissolved Solids, as well as pH values. The initial salinity and the ion exchange capacity of the soil should be established.

The next stage should extend the experiment to field tests, where the consequences of rainfall, semi-permeable base rock and subsequent water flows can be studied. Monitoring methods can then be adjusted in order to safeguard the water quality that is abstracted or escapes uncontrolled from the aquifer. In a final stage the degree of post-treatment should be studied in accordance with the envisaged application of the renovated water.

Acknowledgements
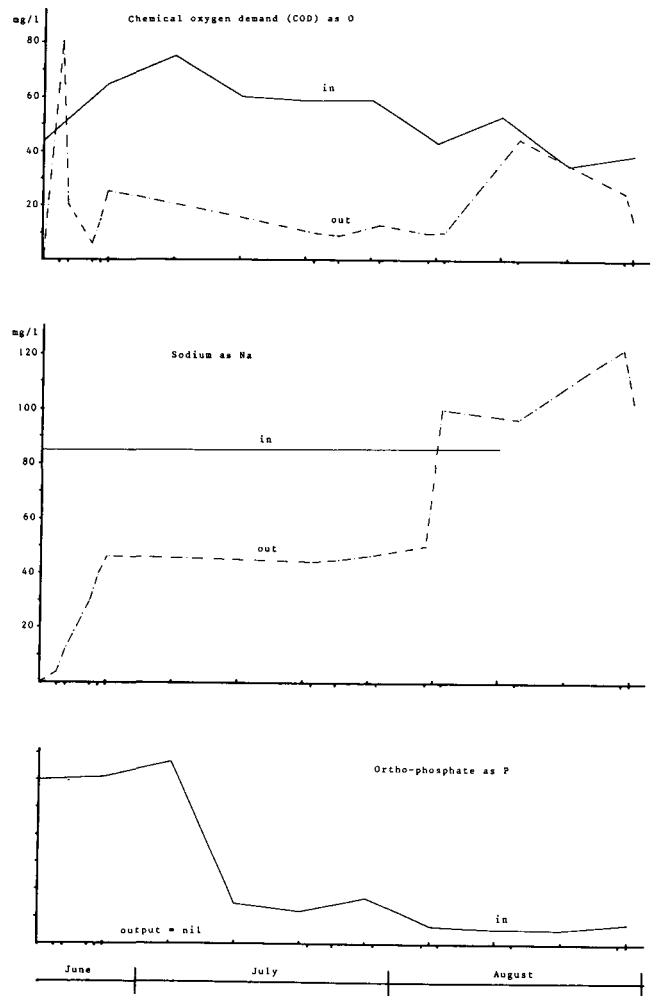
Figure 6
Column test results for COD, sodium and ortho-phosphate.

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