Denitrification of Groundwater for Potable Purposes

Otlanabo Norman Letimela

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DENITRIFICATION OF GROUNDWATER FOR POTABLE PURPOSES

by

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SYNOPSIS

The nitrate content of a large part of Bophutatswana's ground water (34% of boreholes) does not conform to the SABS maximum allowable standard of 10 mg/l NO_3-N.

It is clear from evidence presented that nitrates and related compounds, when consumed in concentrations exceeding 10 mg/l NO_3-N can have a variety of detrimental effects on humans and animals. Some of the documented effects are, methemoglobinemia, cancer, slowing of motor reflexes, hypertension and increased aggression.

The fact that significant effect can be detected in infants consuming water having only slightly more nitrate than the current standard of 10 mg/l NO_3-N, means that potential drinking water with nitrate concentration more than the current standard must be pretreated for nitrate removal before it can be supplied to the consumer.

Three general principles of nitrate removal exist namely, Chemical reduction, Physical-Chemical (Ion-exchange) and biological denitrification. Due to a generally low earning capacity and lack of expertise in the rural population of Bophutatswana, biological denitrification seems to be a more suitable method of nitrate removal for such a population as it is relatively simple to operate.

Although practically any readily available electron donor (carbon source) could be used for biological denitrification in general, this study has shown that for potable water, electron donors are limited to those that will produce an acceptable end product. In this study it was found that methanol and ethanol gave practically similar denitrification results, while molasses added a colour to the water which makes the final water unacceptable for potable and other domestic uses.

The results obtained indicate that virtually complete denitrification occurred at C:N = 1.5:1. Due to the poisonous nature of methanol, it is recommended that for practical purposes a C:N = 1:1 is used. The data obtained shows that denitrification in a Packed bed Reactor (PBR) is nearly completed at a Hydraulic Retention Time (HRT) of 1 hour and more. A relatively small and inexpensive PBR with methanol as carbon source at C:N of 1:1 and HRT of 1 hour is recomended for the rural areas of Bophutatswana.

Comparison of results of inoculation with a mixed and pure culture indicates a higher percentage of nitrate removal with a pure culture than a mixed culture under the same operating conditions. No potential pathogens were observed in the final water where the PBR was inoculated with a pure culture. This finding shows that unlike a mixed culture of activated sludge, a pure culture of denitrifiers, Alcaligenes species in this case, can safely be used for denitrification of potable water.

Biological denitrification has shown to be a cheaper method of nitrate removal than ion exchange. For the treatment of raw water of 60 mg/l NO_3-N and design capacity of 10 m^3/day, the approximate costs are R1.83/m^3 and R1.50/m^3 for ion exchange and biological denitrification respectively. A design capacity of 20 m^3/day results in treatment costs of R1.11/m^3 and R0.95/m^3 for ion exchange and biological denitrification respectively. Biological denitrification is therefore recommended over nitrate removal by ion exchange.

OPSOMMING

'n Groot deel van Bophutatswana se grondwater (34% van al die boorgate) het 'n hoër nitraatinhoud as die maksimum toegelate SABS-standaard van 10 mg/l NO_3^- (as N).

Nitrate wat in hoër konsentrasies as 10 mg/l ingeneem word, het 'n verskeidenheid negatiewe effekte op mens en dier. Voorbeelde van sulke effekte is methemoglobienemia, kanker, vertraging van die motoriese reflekse, hipertensie en toenemende agressie.

Merkbare effekte kan in babas waargeneem word self as die nitraatkonsentrasie net effens hoër is as die toegelate standaard. Dit beteken dat potensiële drinkwater met 'n nitraatkonsentrasie hoër as 10 mg/l eers behandel moet word voordat dit geskik is vir gebruik.

Daar bestaan drie algemene metodes waarvolgens nitrate uit drinkwater verwyder kan word, naamlik chemiese reduksie, fisies-chemies (ioon-uitruiling) en biologiese denitrifikasie. As gevolg van lae per kapita inkomste en gebrek aan kennis by die plattelandse bevolking van Bophutatswana, is biologiese denitrifikasie die beste keuse aangesien dit relatief maklik is om te bedryf.

Alhoewel enige maklik verbruikbare elektronskenker (koolstofbron) vir biologiese denitrifikasie gebruik kan word, het hierdie studie aangetoon dat vir drinkwater, die elektroskenker beperk is tot koolstofbronne wat aanvaarbare produkte lewer. Daar is gevind dat metanol en etanol geskik is en dat beide min of meer dieselfde resultate gee. Molasse het 'n kleur aan die water gegee wat onaanvaarbaar is vir huishoudelike gebruik.

Die resultate wat verkry is dui aan dat byna volledige denitrifikasie verkry word met 'n C:N-verhouding van 1.5:1.

As gevolg van die toksisiteit van metanol word aanbeveel dat 'n praktiese C:N-verhouding van 1:1 gebruik moet word. Die resultate toon aan dat volledige denitrifikasie in 'n gepakte bed reaktor (PBR), plaasvind by 'n hidrouliese retensietyd van een uur of langer. 'n Relatief klein en goedkoop PBR met metanol as koolstofbron in 'n C:N-verhouding van 1:1 en hidrouliese retensietyd van 1 uur word dus aanbeveel vir die plattelandse gebiede van Bophutatswana.

Inokulasie met 'n reinkultuur het beter nitraatverwydering tot gevolg gehad as inokulasie met 'n gemengde kultuur, onder dieselfde bedryfstoestande. Geen potensiële patogene is in die finale uitvloeisel waargeneem as die PBR met 'n reinkultuur geïnokuleer is nie. Hierdie waarneming dui daarop dat, anders as met 'n gemengde kultuur van geaktiveerde slyk, 'n reinkultuur van denitrifiseerders, in hierdie geval 'n *Alcaligenes* spesie, met veiligheid geruik kan word vir die denitrifikasie van drinkwater.

Daar is aangetoon dat biologiese denitrifikasie 'n goedkoper metode is as nitraatverwydering met ioonuitruiling. Vir die behandeling van rou water met 'n konsentrasie van 60 mg/l NO_3-N en ontwerpkapasiteit van 10 m^3 per dag, is die benaderde kostes R1,83 per m^3 en R1,50 per m^3 vir ioonuitruiling en biologiese denitrifikasie onderskeidelik. 'n Ontwerpkapasiteit van 20 m^3 per dag het die volgende kostes tot gevolg: R1,11 per m^3 en R0,95 per m^3 vir ioonuitruiling en biologiese denitrifikasie onderskeidelik. Biologiese denitrifikasie is dus 'n beter keuse as ioonuitruiling.

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CHAPTER 1

NITRATE SITUATION IN BOPHUTATSWANA

1.1 Bophutatswana and its water supply

Bophutatswana consists of seven separate geographical units, totalling 44,055 km^2 in a generally east-west line ranging between the industrial heartland of Southern Africa and the Kalahari desert. The total population of Bophutatswana is about 1.8 million of which about 80% live in rural areas and only 20% in urban areas.

Water resources are very limited as Bophutatswana is a semi-arid state. Since the entire state falls within the boundaries of South Africa, (Figure 1.1) joint use is made of water sources emanating from shared catchments. In certain cases potable surface water is purchased from organizations such as the Rand Water Board, Magalies Water Board and municipalities such as the Brits and Vryburg ones.

Bophutatswana depends on ground water resources for much of the water supplies to the rural areas. One important consideration is that natural conditions often dictate which type of water development is feasible. In many cases the surface rocks are not porous or fissured and contain little water. Occasionally ground water is contaminated by chemicals from geological formations or pathogens from pit-latrines. In these cases it is usually necessary to pump water from elsewhere (Schutte, Letimela and Croucamp, 1989).

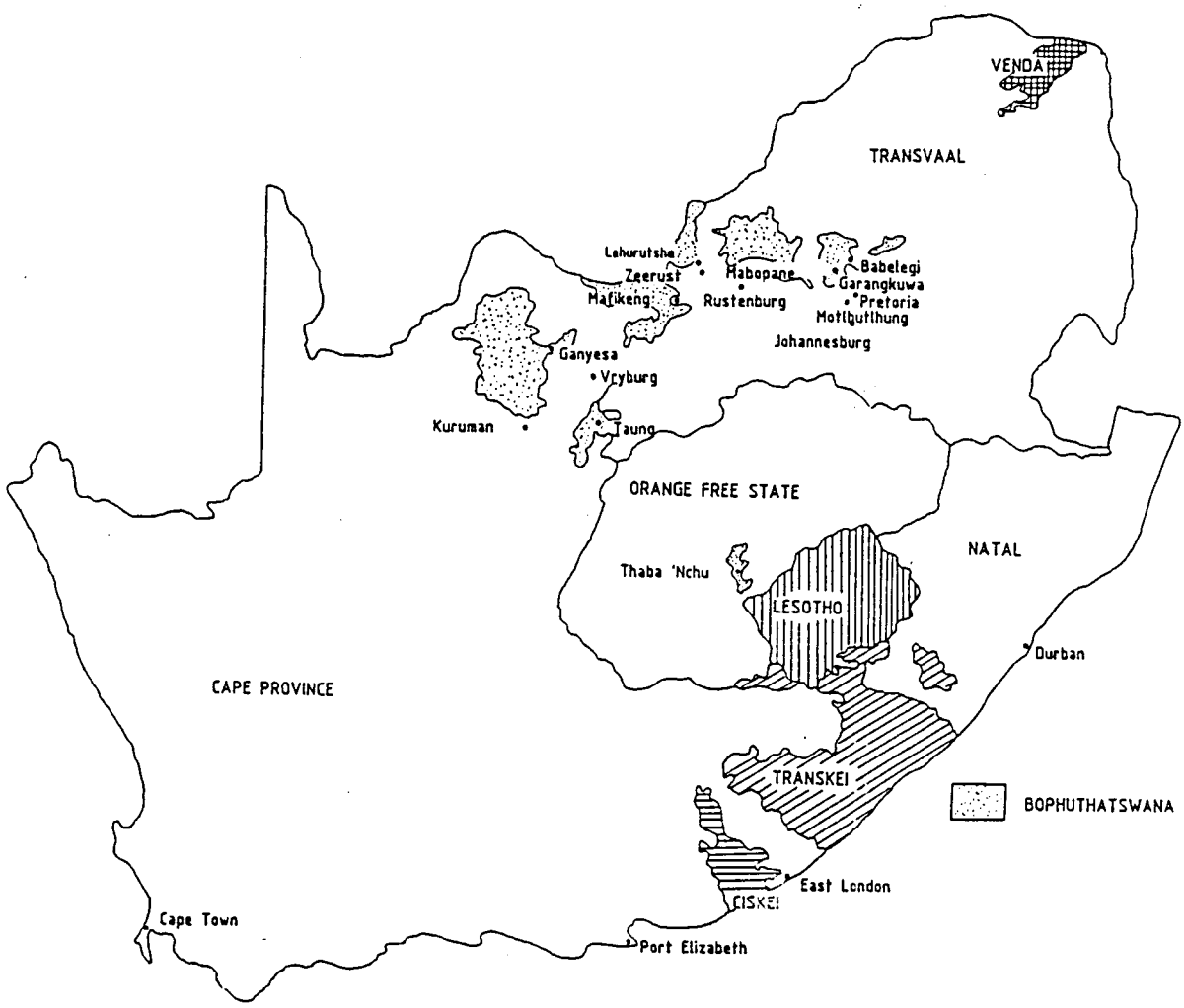

Fig. 1.1 Distribution of different regions of Bophutatswana within South Africa.

In certain areas and in particular most of the rural areas, community or own boreholes are used to meet the water demand.

1.2 Water supply to rural areas

As mentioned above water supply to the population in rural areas is mainly from ground resources. This can be in the form of an engine or windmill driven pump pumping into a small reservoir and a limited reticulation system to a few standpipes. It may, however, also be in the form of regional schemes where water is pumped from a number of boreholes to a central reservoir and from there to the village reticulation system and also for other uses such as irrigation. An example of such a scheme is the Dinokana scheme in the Lehurutshe district. Water is pumped from boreholes in the dolomite compartments to a 5.7 Ml reservoir from where it gravitates to a reticulation system in Lehurutshe, supplying the water needs of about 17 000 inhabitants (Schutte, Letimela and Croucamp, 1989).

1.3 Water quality distribution

A data survey of nitrates (NO_3^-) and sulphates (SO_4^{2-}) composition of groundwater in different areas of Bophutatswana was conducted, the analysis was done by the Pollution Control Laboratory of the Department of Water Affairs, the Council for Scientific and Industrial Research (CSIR) and other consulting laboratories. Water quality distribution with regards to the nitrate levels varies from region to region (Figure 1.2.). A summary of sulphate and nitrate concentrations in borehole water used for drinking purposes is shown in Table 1.1 and 1.2 respectively.

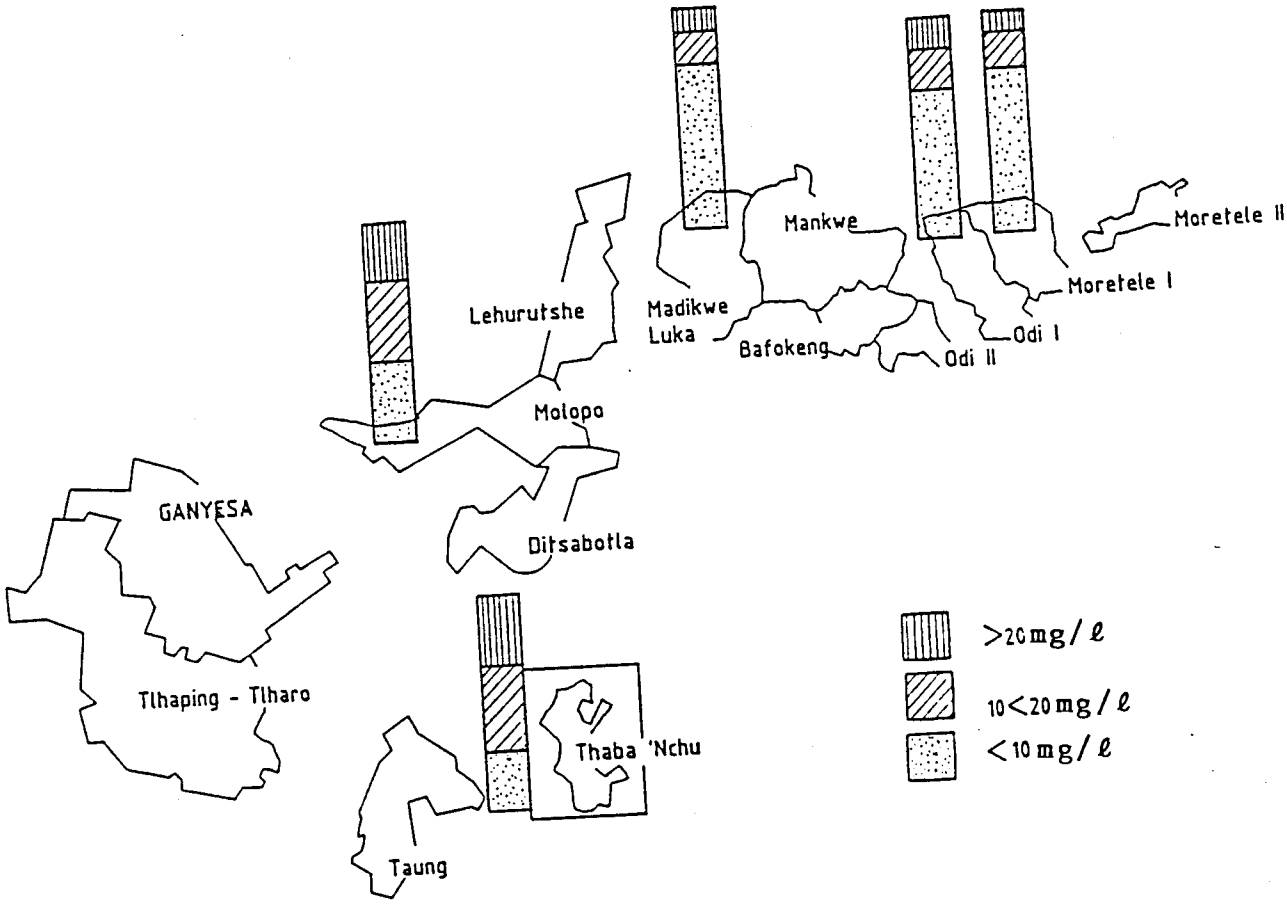

Fig. 1.2 Nitrate pollution in groundwater of different regions of Bophutatswana.

TABLE 1.1 SULPHATE CONCENTRATION IN DIFFERENT AREAS OF BOPHUTATSWANA

District	Total no. boreholes sampled	Percentage borehole water with sulphate concentration shown (mg/l SO_4)				Population served
		<60	60<250	>250	Worst case	
Odi I & II	817	57	34	9	520	340 000
Madikwe/Luka	399	71	18	11	458	73 700
Moretele I & II	198	58	35	7	574	193 000
Molopo	113	67	27	6	359	93 000
Taba'Nchu	57	75	21	4	512	56 600

TABLE 1.2 NITRATE CONCENTRATION IN DIFFERENT AREAS OF BOPHUTATSWANA

District	Total no. boreholes sampled	Percentage boreholewater with nitrate concentration shown (mg/l NO_3-N)				Population served
		<10	10<20	>20	Worst case	
Odi I & II	817	67	18	15	120	340 000
Madikwe/Luka	399	74	14	12	375	73 700
Moretele I & II	198	73	16	11	51	193 000
Molopo	113	36	37	26	46	93 000
Thaba'Nchu	57	28	39	33	40	56 600

The South African Bureau of Standards' (SABS) maximum allowable limit for nitrates in drinking water is 10mg NO_3-N/l. According to Table 1.2, 34% (No. of borehole with nitrate concentration > 10 mg/l NO_3-N/Total No. of boreholes sampled) of the boreholes exceeds the SABS limit. Some of the boreholes like those of the Luka area had relatively high nitrate concentrations even as high as 375 mg NO_3-N/l. It is thus clear that the nitrate content of drinking water for a large part of Bophutatswana's population does not conform to the SABS standard.

This study will address the problem of nitrate in drinking water in general and will evaluate methods for reducing the nitrate content of boreholewater to acceptable concentrations.

CHAPTER 2

THE ORIGIN OF NITRATES IN GROUND WATER AND ITS EFFECT ON HUMANS

2.1 Nitrogen compounds in nature

Different nitrogen compounds originating from different sources reach soil and surface water where it is chemically transformed. A schematic presentation of the sources and more important transformations are given in Figure 2.1.

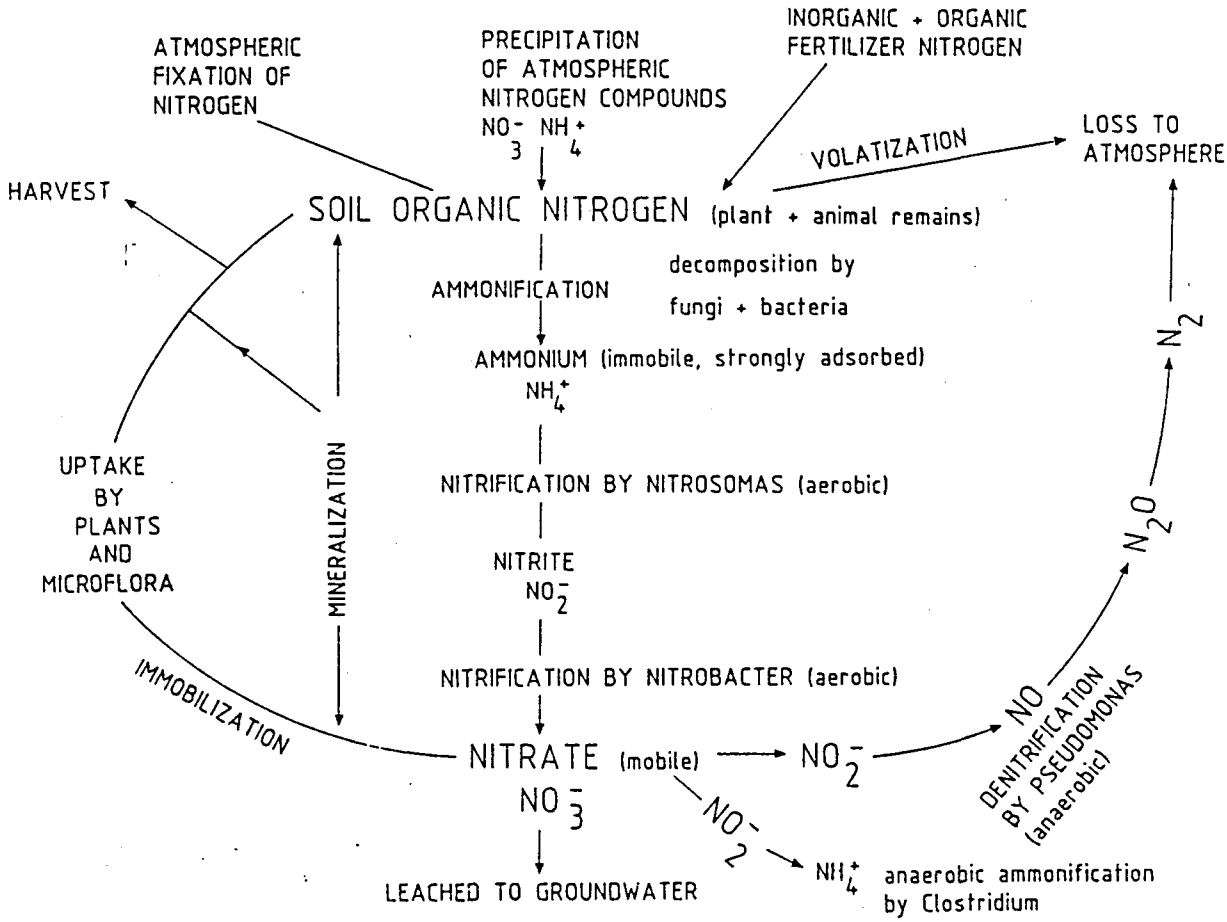

Fig. 2.1 Nitrogen transformations and the nitrogen cycle (Hiscock, Lloyd and Lerner, 1991).

As a result of the electrical discharge in the atmosphere, nitrogen and oxygen combine to form nitrate which is carried to the soil by rain for plant protein production. Herbivorous

animals change plant protein to animal protein. Dead matter on which bacteria act release ammonia, nitrite and nitrate. Nitrogen gas re-enters the atmosphere mainly through the reduction of nitrates. Certain bacteria and algae also fix nitrogen from the atmosphere to produce plant protein (Terblanche, 1991).

According to Terblanche (1991), the means by which nitrates find their way to water supplies include the following:-

Surface waters

- discharge of domestic and industrial water (sewage effluents, food processing wastes, refuse dumps);

- runoff and seepage water from the land containing the nitrified products of decayed vegetation and animal matter (agricultural and urban);

- excess material from oxidation of ammonia and agricultural fertilizers and from nitrates in these materials being washed into surface sources;

- the fixation of atmosphere nitrogen entering surface water from rain or washings from the land; and

- nitrogen fixation by leguminous plants such as peas and the nitrate so formed being washed into surface water.

Ground waters

- seepage of water containing the nitrified products of decayed vegetation and animal matter;

- discharge of domestic and farm effluent into aquifer;

- seepage of nitrification products of special crops grown in the catchment area, e.g. pig farming;

- excess of nitrogenous agricultural fertilizers applied to farm land percolating from the surface into the aquifer; and

- broad irrigation of sewage sludge and effluent on land and its consequent seepage into the ground.

There has been a gradual increase in nitrate concentration levels during the past twenty years in many surface and underground water sources in England and Wales (Royal Commission on Environmental Pollution, 1979). In the Coastal aquifer of Israel, the source of some 20% of the country's water supply, observations made at hundreds of deep wells over the past 15 years indicate nitrate (as NO_3) concentrations have increased at the rate of 1 to 2 mg/l per annum (Soliternik, 1984). The average concentration of nitrate (as NO_3) in ground water in Denmark increased from 3 mg/l to 13.3 mg/l over the period 1940 to 1983 (Moller et al., 1989). Levels in South Africa vary between less than 1 mg/l and 40 mg/l NO_3-N. The average nitrate levels (as NO_3) for 230 drainage regions throughout South Africa falls between 2 and 5 mg/l (Department of Water Affairs, 1990).

According to Pelczar, Chan and Krieg (1986) and Hiscock, Lloyd and Lerner (1991) atmospheric nitrogen is fixed by many microorganisms, e.g., Rhizobium, Clostridium, Azotobacter, etc., this results in organic nitrogen formation. Fixed nitrogen is utilized by plants and converted to plant protein. Plants are consumed by animals to form animal proteins. Excretion products of animals, dead animals, plant tissue and microorganisms are deposited in the soil. Organic nitrogen (protein, nucleic acids etc.) are attacked by a wide variety of microorganisms, complete breakdown thereof yields mixtures of amino acids. Amino acids are de-aminated by many

microorganisms with ammonia (NH_3) as the end product. The production of ammonia is referred to as ammonification. Ammonia is then solubilized (NH_4^+ formed), accumulated and fixed (immobilized) in the soil. This is made mobile by nitrification to the nitrate (NO_3^-) form, in essence that is how NO_3^- reaches ground water.

2.2 Intake of nitrates and its effect on humans and animals

2.2.1 Sources of nitrate intake by humans

Man's major exposure to nitrates comes primarily from vegetables, water supplies that are high in nitrate content or from nitrates used as additives in the meat curing process (Terblanche, 1991). Nitrates are natural constituents of plants. They are present in only minor amounts in fruit, spinach and beets. Radishes, eggplant, celery, lettuce, collards and turnip greens are among the vegetables that generally contain a high nitrate concentration (Terblanche, 1991).

2.2.2 Effects of nitrates on humans

Nitrates are considered relatively non-toxic as they can be readily excreted by the kidneys. The secondary products of it, namely nitrite and nitrosoamines, however, present a health hazard to humans. Nitrites are known to cause methemoglobinemia in infants, while nitrosoamines are known carcinogens - they play a role in the induction of certain gastroentestinal cancers (Fan et al, 1987 and Adam, 1980).

Methemoglobinemia occur when nitrate, which is formed in the stomach from ingested nitrates, reacts with haemoglobin in blood, converting haemoglobin, the oxygen carrier of mammalian blood, into methemoglobin which cannot carry oxygen to the cell tissues (Wild, 1977). It is accepted by many workers that,

under normal circumstances, less than 2% of the total haemoglobin exists as methemoglobin (Shuval and Gruener, 1977). Some evidence that nitrates may directly affect the central nervous system in humans has been published (Petukhof and Ivanor, 1970). The slowing of conditioned motor reflexes in response to auditory and visual stimuli was documented in 39 Russian children whose drinking water contained 105 mg/l nitrate, (not specified as N or NO_3^-,) their reflexes were compared with those children whose drinking water contained 8mg/l nitrate. The concentration of methemoglobin did not exceed normal limits in the school children who drank low nitrate containing water, whereas the children exposed to high nitrate containing water had an average of 5.3% methemoglobin in their blood.

According to Morton (1971), a statewide study of municipal water supplies in Colorado in 1960 suggested that the significantly higher hypertension risk in humans in the eastern plains might be due to higher nitrate concentration in drinking water. Fraser and Chilvers (1981), further investigated this finding and reported that an earlier onset of hypertension exists among residents of communities exposed to nitrate levels of 19 to 123mg/l (as NO_3^-), compared with communities exposed to nitrate free drinking water.

2.2.3 Experiments with nitrate on laboratory animals

Laboratory experiments with rats showed that transplacental passage of nitrates can occur, causing raised methemoglobin levels in the foetus and impaired growth. Rats chronically exposed to nitrate showed thinning and ballooning of cardiac blood vessels. Exposure of mice to nitrates in drinking water caused behavioural effects such as lowered motor activity and increased aggression. Nitrites were used in these studies rather than nitrates, as the animal model does not allow nitrate to nitrite reduction in the gut (Petukhof and Ivanov, 1970). The Department of Water Affairs, Namibia, has recorded

cases where livestock died from waters with too high nitrate concentrations (Henzen and Stander, 1962).

2.3 Conclusion

Nitrogen is fixed from the atmosphere by different micro-organisms. The fixed nitrogen is utilized by plants and animals with the resultant formation of organic nitrogen.

Organic nitrogen compounds are then deaminated with ammonia as the end product. Ammonium ions formed are fixed to the soil. This is made mobile by nitrification and that is how NO_3- reach ground water. Observations made by different researchers indicate that there is clear evidence that nitrate levels in groundwater are gradually increasing. Nitrate levels in South Africa vary between less than 1 and 40 mg/l NO_3-N.

From evidence presented, it is clear that nitrates and related compounds, when consumed in concentrations exceeding 10mg/l NO_3-N can have a variety of detrimental effects on humans and animals. Some of the documented effects are, methemoglobenemia, cancer, slowing of the motor reflexes, hypertension and increased aggression.

The fact that significant effects can be detected in infants consuming water having only slightly more nitrate than the current standard (10mg/l NO_3-N), means that potential drinking water with nitrate concentration more than the current standard must be pretreated for nitrate removal before it can be supplied to the consumer.

CHAPTER 3

THEORY AND METHODS FOR REMOVING NITRATES FROM WATER

3.1 Introduction

Nitrogen exists in many compounds because of the high number of oxidation states it can assume. In ammonia or organic compounds, the form most closely associated with plants and animals, its oxidation state is minus 3 (de Renzo, 1978). At the other extreme its oxidation state is plus 5 when in the nitrate form. In the environment, changes from one oxidation state to another can be brought about biologically by linking organisms. The relationship between various compounds and the transformations which can occur are presented schematically in Figure 2.1.

Three general principles of NO_3^- removal exists namely, chemical reduction, physical-chemical (Ion exchange) and biological reduction.

3.2 Chemical reduction of NO_3^-

Several chemicals have been investigated for reducing nitrate to nitrogen gas. Of the various chemicals studied, only ferrous ion has been determined to be economically attractive. The process requires a catalyst, copper, for denitrification however and the process must take place in an alkaline solution. The drawback to this process is that only 70% of the nitrate is reduced and large amounts of ferrous iron are required, eight moles of iron per mole of nitrate (Sorg, 1979). Neither conventional coagulation nor lime softening are effective removal methods, because nitrate is very soluble in water (Sorg, 1979).

3.3 Physical-chemical: Ion exchange

3.3.1 Theory

Ion exchange is a typical example of a physical-chemical process for nitrate removal. The process is relatively simple and entails passing feed water through a bed of ion exchange resin. The resin bed has a finite capacity for removing nitrate ions and once this capacity has been reached, the bed must be regenerated to restore its exchange capacity (Benefield, Judkins and Weand, 1982).

Ion exchange may be operated in either a batch or a continuous mode. In a batch process, the resin is simply stirred with the water to be treated in a reactor until the reaction is complete. The spent resin is removed by settling and subsequently is regenerated and re-used. In a continuous process the exchange material is placed in a fixed bed or a packed column, and the water to be treated passed through it (Metcalf and Eddy, 1979). The fixed bed system can be operated in the downflow or upflow mode. The operation is referred to as counter-current ion exchange when service is in one direction and regeneration in the opposite direction and co-current ion exchange when both service and regeneration are in the same direction (Benefield, Judkins and Weand, 1982), see Figure 3.1.

3.3.2 Hardware

Different types of resins with different capacities can be used for nitrate removal from drinking water. Examples are the Duolite A101, Duolite A132 (supplied by E.L. Bateman, P.O. Box 565, Boksburg, 1460), Amberlite IRA996 and Imac HP441 and HP555 (supplied by NCP, R & D Department, P.O. Box 677, Germiston). By means of these anion exchange resins nitrate is exchanged for chloride or bicarbonate (Van der Hoek and Klapwijk, 1986).

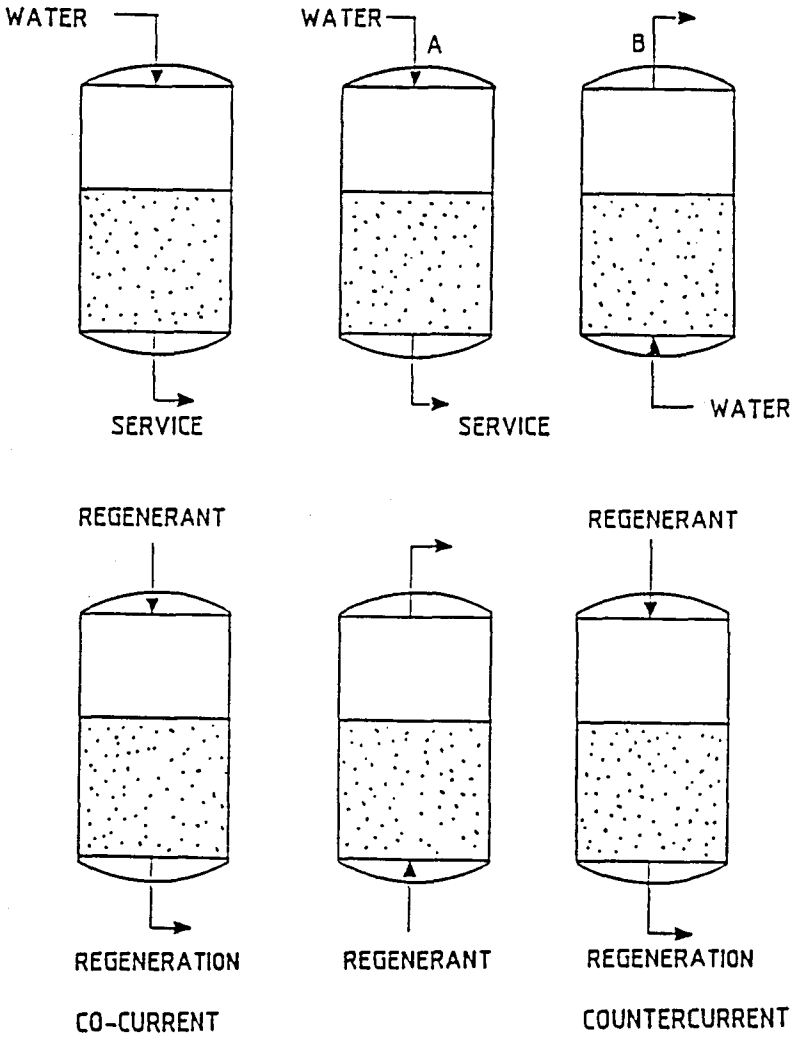

Figure 3.1 Co-current and counter-current ion exchange (Benefield, Judkins and Weand, 1982).

Regeneration of the resin normally involves the use of highly concentrated NaCl solution (50-120g/l resin) at a flowrate of 2-4 bed volumes per hour for a period of 30-45 min (Gauntlett, 1975; Guter, 1982 and Scheltinga, H.M.J. 1985).

3.3.3 Advantages and disadvantages

Nitrate removal by ion exchange is a comparatively simple process amenable to automation. It can be started up at relatively short notice should this be necessary and it only requires a minimum of skilled attention. There is no risk of a bacteriological contamination of groundwater.

The first problem with ion exchange concerns the anion exchange resin's selectivity for sulphate rather than for nitrate. The generally accepted selectivity preference is $SO_4 > NO_3 > Cl > HCO_3$ (Clifford, 1982). This means that the nitrate capacity is low when the sulphate concentration in ground water is high. The second problem is the regeneration of the resin. A large excess of salt is needed, this produces a voluminous brine with high nitrate, sulphate and chloride concentration. Brine disposal can be very difficult as both aspects cause financial and environmental problems.

Ion exchange seems to be a questionable method of nitrate removal for Bophutatswana since ground water with high levels of nitrate and sulphate is consumed by the population in the rural areas (see Table 1.1 and 1.2). The method has high running costs due to large amounts of salt required for regeneration of the resin (50 to 120 g/l NaCl) and also large amounts of brine to be disposed of.

3.4 Biological denitrification

3.4.1 Theory of biological denitrification

Several aerobic heterotrophic bacteria are capable of using nitrate in place of oxygen and converting it to nitrite, ammonia or nitrogen gas. This normally occurs under anaerobic conditions, e.g. in waterlogged soil. The oxygen of the nitrate serves as an acceptor for electrons and hydrogen (Pelczar, Chan and Krieg, 1986). This anaerobic reaction

requires that organic carbon be added to the water to provide the necessary energy for the bacteria. Methyl alcohol, ethanol and acetic acid are generally used as energy sources. Most denitrifying systems use methyl alcohol as the carbon source for economical and operational (low solids production) reasons (Dahab and Lee, 1988). The typical nitrate reduction reaction sequence is,

NO_3^- -------→ NO_2^- ------→ NO ------→ N_2O --------→ N_2

3.4.2 Denitrifiers and Denitrification Stoichiometry

Species of several genera of bacteria are capable of transforming NO_3^- to N_2, e.g., Achromobacter, Agrobacterium, Alcaligenes, Bacillus, Chromobacterium, Flavobacterium, Hyphomicrobium, Pseudomonas, Thiobacillus, and Vibrio (Pelczar, Chan and Krieg, 1986). A carbon source is required for bacterial energy requirements. Using methanol as the carbon source, the energy reaction may be represented by the following equations (Metcalf and Eddy, 1979):

Bacterial Energy Reaction, Step 1:

$$6NO_3^- + 2CH_3OH \longrightarrow 6NO_2^- + 2CO_2 + 4H_2O \quad (1)$$

Step 2:

$$6NO_2^- + 3CH_3OH \longrightarrow 3N_2 + 3CO_2 + 3H_2O + 6OH^- \quad (2)$$

Overall Energy Reaction:

$$6NO_3^- + 5CH_3OH \longrightarrow 5CO_2 + 3N_2 + 7H_2O + 6OH^- \quad (3)$$

A typical synthesis reaction is given by McCarty (1975) as follows:

$3NO_3^- + 14CH_3OH + CO_2 + 3H^+ \longrightarrow 3C_5H_7NO_2 + 19H_2O$ (4) In equation 4 above, nitrate is converted to cell tissue,

represented by the formula $C_5H_7NO_2$. In practice 25 to 30% of methanol required is used for bacterial synthesis (Metcalf and Eddy, 1979). On the basis of experimental laboratory studies, the following empirical equation to describe the overall nitrate removal reaction was developed (McCarty, 1975).

Overall nitrate removal:

$$NO_3 + 1{,}08CH_3OH + H^+ \longrightarrow 0{,}065C_5H_7NO_2 + 0{,}47N_2 + 0{,}76CO_2 + 2{,}44H_2O \quad (5)$$

On the basis of the above reactions one can calculate the amount of methanol required as energy source for denitrifiers to reduce nitrate to nitrogen gas.

If, however, nitrite and dissolved oxygen (DO) are present, the methanol requirement is correspondingly higher, in this case methanol requirement can be computed using the following empirically derived equation (de Renzo, 1978):

$$C_m = 2{,}47NO_3^-\text{-}N + 1{,}53NO_2^-\text{-}N + 0{,}87DO$$

Where

C_m	=	required methanol concentration, mg/l
NO_3^--N	=	nitrate concentration removed, mg/l
NO_2^--N	=	nitrite concentration removed, mg/l and
DO	=	dissolved oxygen removed, mg/l

3.4.3 Hardware

Various types of biological reactors can be used to bring denitrifying bacteria into contact with water to be treated. These include attached growth reactors (e.g. packed columns, rotating disc units and fluidized bed columns), and suspended growth reactors. In the latter category (suspended growth reactors) there is the stirred tank, in which bacterial floc is

kept in suspension by stirring, and the pond, in which the bacteria are not mechanically agitated (Adam and Winter, 1983).

One recent method of biological denitrification is the underground treatment of nitrates. Hiscock (1990) explored possible methods for treating nitrate contaminated groundwater in the U.K. aquifiers, and in particular the potential of developing underground denitrification.

According to Hiscock (1990), in applying denitrification underground, there is the advantage that both denitrification and secondary treatment, which is not normally available at groundwater sources, are performed in situ. In design, plant requirements are small, requiring an injection well for adding nutrients, and a pumping well for abstracting the treated water (Figure 3.2.a). A number of research projects have been completed into the feasibility of the method. However, the results are somewhat conflicting, particularly with respect to the potential for aquifer pore spaces to clog with biological matter.

A modification of the underground method (Figure 3.2.b) which avoids serious problems with clogging of the aquifer, is to perform denitrification in an above ground bioreactor with recirculation of the treated water underground for secondary treatment.

A pilot plant at an alluvial aquifer in France was designed with the central supply well surrounded by three bioreactors with adjacent infiltration pits at a radius of 15m. In this case, the infiltration pits serve to filter organic matter from the denitrified water, with circulation underground (Hiscock, 1990).

Underground denitrification is a potential method for treating ground water nitrate if clogging of the aquifer pore spaces is avoided. If successful, the method has the clear advantage of

producing good quality water for minimal plant investment (Hiscock, 1990). Designs for underground treatment of nitrates according to Hiscock (1990) is outlined in Figure 3.2.

3.4.4 Advantages and Disadvantages

Biological denitrification has the advantage in removing only or mainly the nitrate while the physico-chemical processes are more or less unspecific and remove also other inorganic constituents. Regeneration in the case of biological denitrification is not required, thus there is no problem of brine disposal.

Although biological systems destroy the nitrate definitely, there is a drawback of fluctuation and sensitivity to bacterial toxins. There is a need to develop a large bacterial population that is generally free of pathogens. The system will be out of service if the biological mass is lost.

Since the Bophutatswana rural population has a generally low earning capacity, biological denitrification would be more suitable as it is relatively simple to operate.

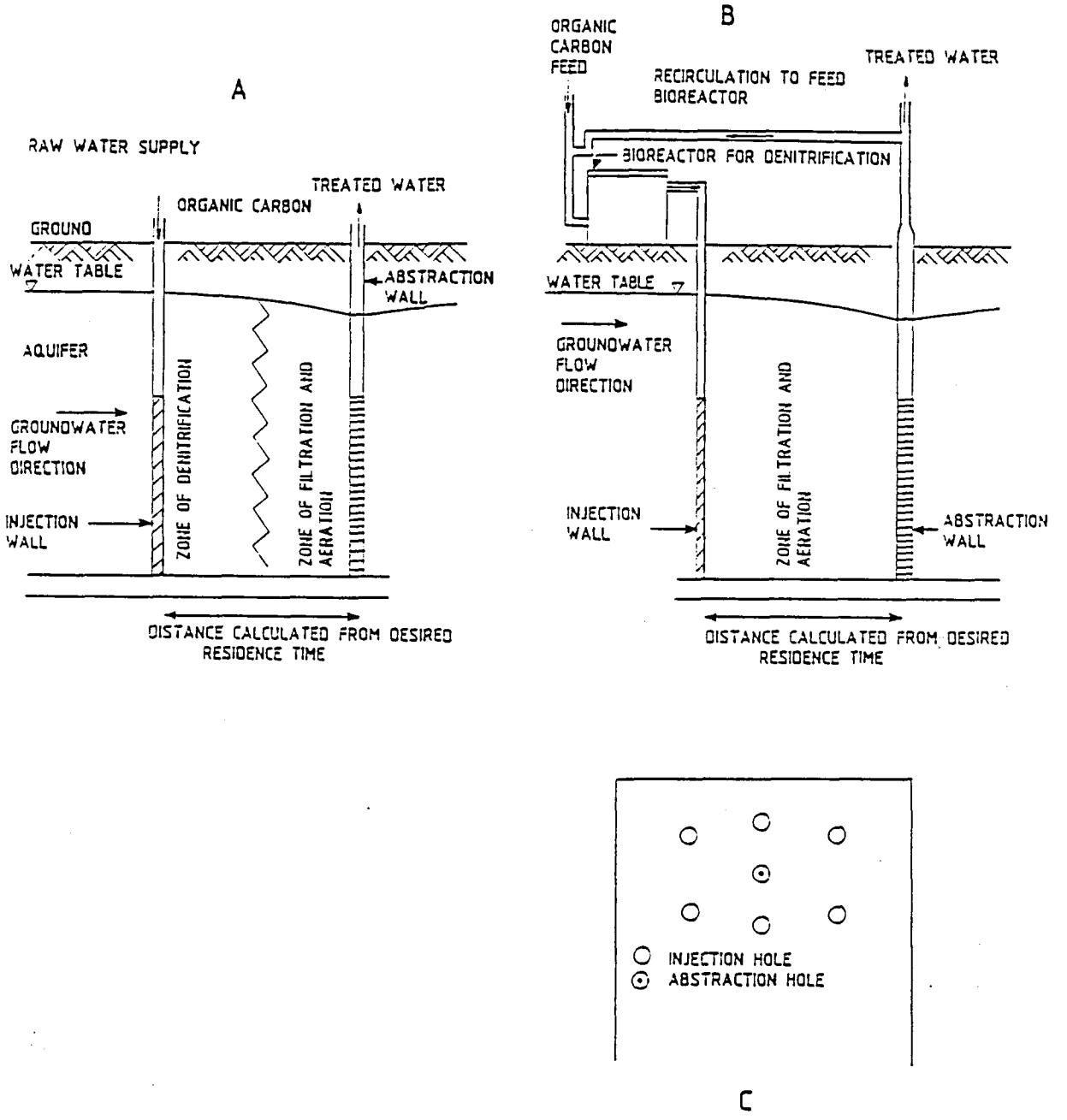

Fig. 3.2(a) Sectional view of underground denitrification.
(b) Sectional view of aboveground denitrification with groundwater recharge.
(c) Plan view of sections.

CHAPTER 4

BIOLOGICAL NITRATE REMOVAL WITH A PACKED BED REACTOR

4.1 Introduction

Bophutatswana is a developing state where approximately 80% of the population is still living in rural areas (Schutte, Letimela, Croucamp, 1989). A significant number of the rural population is also dependant on nitrate contaminated ground water for their potable needs (Table 1.2, Chapter 1). These people have in general a low earning capacity and can therefore not afford and successfully maintain complicated and technologically advanced denitrifying systems.

With literature review (Chapter 3) the basic theory and practical applications of various possible denitrification systems were described. With the available technological knowhow in the rural areas of Bophutatswana as basis, very few of these denitrification systems would be a viable option for the rural areas. From the few denitrification systems that could be considered, the packed bed biological denitrification system would seem to be the most likely to succeed.

In this chapter a laboratory scale packed bed biological denitrification system was evaluated for the purpose of removing nitrates from drinking water.

4.2 Materials and methods

4.2.1 Experimental layout

The experimental system consisted of a feed supply, a packed bed reactor followed by a slow sand filter. The packed bed reactor was provided with sampling ports and a mariotte-type gas meter. The experimental set up is shown schematically in Figure 4.1.

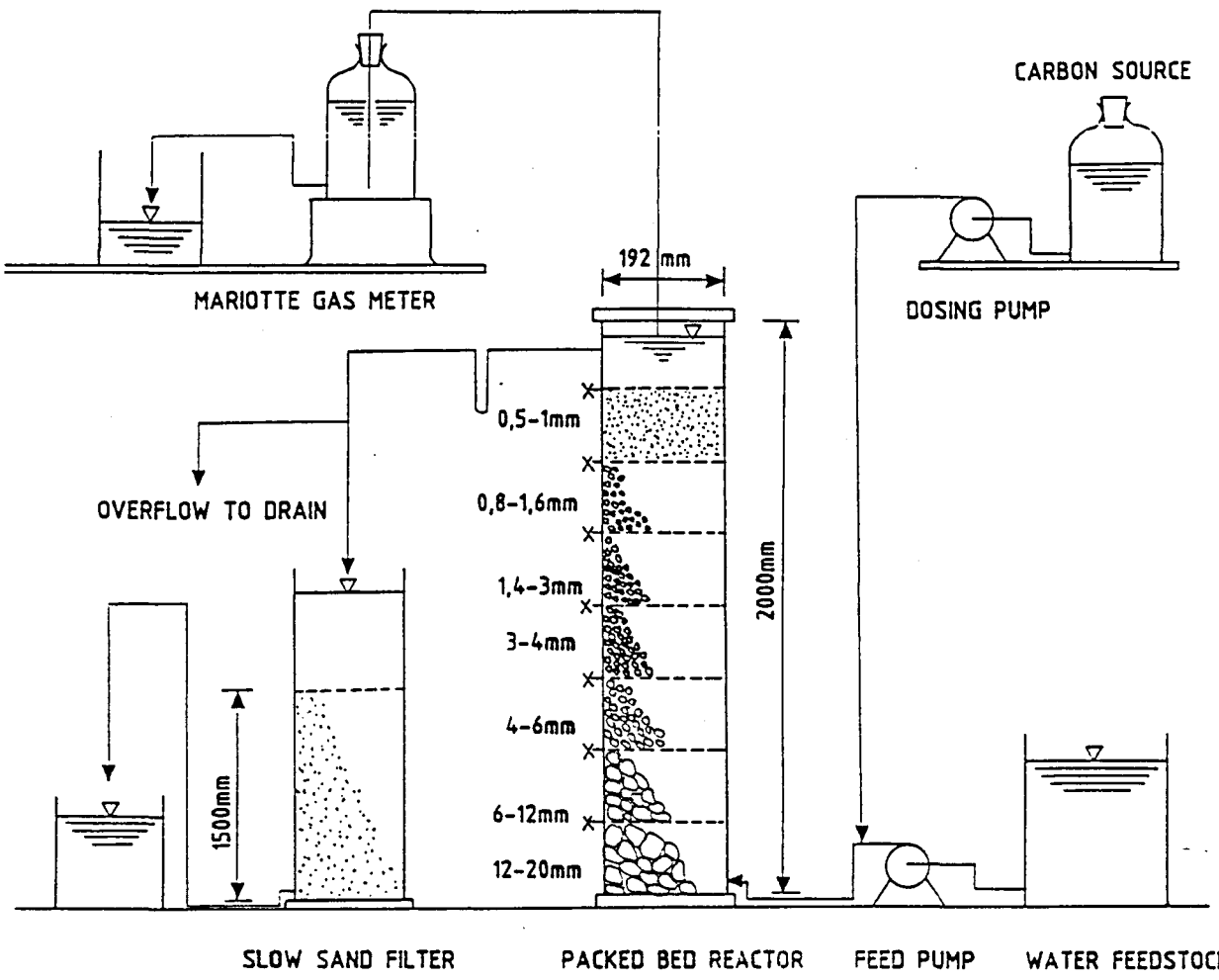

Fig. 4.1 Experimental lay-out.

The reactor was constructed from 'perspex' tubing (192 mm 0 X 2000 mm). Seven layers of quartz stone, gravel and sand (Brits Filter Media), each size range of 250 mm thick with decreasing size from bottom to top were used as packing material (Figure 4.1). This packing resulted in a void ratio of 0.34 with a void volume of 22 litres. Similarly the slow sand filter was constructed from perspex tubing (148 mm x 2000 mm) and filled to a height of 1500 mm with filter sand (effective size 0,250 mm and uniformity coefficient of 1,5).

4.2.2 Experimental borehole water

From the data presented in Table 1.2 (Chapter 1), 34% of the water in boreholes sampled contained more than 10 mg/l NO_3-N and 15% contained more than 20 mg/l NO_3-N, with some boreholes

containing more than 375 mg/l NO_3-N. It was therefore decided to use Pretoria tap water, supplemented with sodium nitrate and sodium sulphate to give a final concentration of 60 mg/l NO_3-N and 60 mg/l sulphate as (SO_4^{2-}) respectively as feedstock. In addition phosphate was added as phosphoric acid to give a final concentration of 2 mg/l as P. Chemical analysis of the artificial borehole water used in the experiments, is given in TABLE 4.1.

TABLE 4.1 TYPICAL CHEMICAL ANALYSIS OF WATER USED AS FEEDSTOCK

Parameter	Unit	Value
pH		7.8
Colour	Hazen	10.0
Turbidity	Jackson	5.8
Conductivity	Ms/m	40.0
Total hardness	(mg/l as $CaCO_3$)	96.0
Calcium	(mg/l as Ca)	18.0
Magnesium	(mg/l as Mg)	13.0
Sodium	(mg/l as Na)	31.0
Potassium	(mg/l as K)	3.0
Alkalinity	(mg/l as $CaHCO_3$)	96.0
Chlorides	(mg/l as Cl)	29.0
Sulphates	(mg/l as SO_4)	60.0
Fluorides	(mg/l as F)	0.3
Nitrates	(mg/l as N)	60.0
Phosphates	(mg/l as P)	2.0
Dissolved oxygen	(mg/l as O)	2.0

4.2.3 Carbon sources

For biological denitrification an electron donor (carbon source) is necessary (see Chapter 3 section 3.4). For this study three readily available carbon sources namely methanol

(CH_3OH), ethanol (CH_3CH_2OH) and molasses ($C_{12}H_{22}O_{11}$), respectively, were used. The carbon source was kept separate from the feedstock water and fed at rates as required by the experimental operation.

4.2.4 Chemical analysis

The following analysis were done: Temperature, pH (glass electrode), Turbidity (HACH turbidimeter 2100 and 2100A), nitrates (Ion selective electrode), nitrites, chemical oxygen demand, sulphates, Total Plate Count and Escherichia coli Counts (Standard methods, 1989).

Nitrogen gas production - the mariotte gas meter was filled with 5N NaOH solution and the resulting gas (mainly nitrogen) was measured by the volumetic displacement of the NaOH solution.

4.2.5 Inoculations

The packed bed reactor (PBR) was filled with water feedstock. Five hundred millilitres of a 4,500 mg/l mixed liquor suspended solids of an active denitrifying activated sludge plant was fed into the bottom of the PBR as inoculum.

4.2.6 Variables evaluated and operation of PBR

Three different electron donors (carbon sources) namely methanol, ethanol and molasses were compared. For each of these electron donors the C:N were stepwise lowered - 2:1; 1.5:1 and 1:1 respectively. For each electron donor and C:N combination an initial nominal hydraulic residence time (HRT) of 12h was chosen. Once steady state was reached, i.e. less than 5% variation in the nitrate concentration of the effluent and the volume of nitrogen gas collected over a period of 2 hydraulic residence times, the hydraulic residence times were

stepwise decreased in the following steps: 6h, 4h, 2h, 1h and 0.5h respectively.

4.2.7 The slow sand filter (SSF)

To start the slow sand filter about 1kg sand (which was constantly submerged) was collected from the nearby Apies river. This sand was placed on top of the sand on the previously prepared sand filter. Effluent from PBR was used as inflow to the SSF. The flow to the SSF was controlled to give a filtration rate of 20cm/h. Excess PBR effluent was wasted.

$$\text{* Nominal retention time} = \frac{\text{Void Volume}}{\text{Flow Rate}}$$

4.2.8 Identification of Pathogenic organisms from PBR Product Water

The API20E was used for the identification of pathogenic organisms contained in the sample of the product water of the PBR. Sampling was done at the effluent point of the PBR. The API20E is a standardized identification system for Enterobacteriaceae and other Gramnegative rods which uses 23 biochemical tests and a data base.

The API20E strip (Supplied by Protea Medical Laboratory) consists of 20 microtubes containing dehydrated substrates. These tests are inoculated with a bacterial suspension and incubated. During incubation, bacterial metabolism produces colour changes that are either spontaneous or develop upon addition of reagents. The reactions are read according to the Interpretation Table and identification is obtained either by referring to the Identification Table, the Analytical Profile Index or the APILAB Software.

4.3 Results

4.3.1 Electron donors

Denitrification in all experiments started within 12 hours and reached steady state within about 3 retention times at a hydraulic residence time of 12h. The water temperature varied between 18 and 21°C.

Although all three electron donors (i.e. methanol, ethanol and molasses) could be used for biological nitrate removal in general, it was soon realized that molasses was not a suitable electron donor for the denitrification of potable water. The reason for this was that unlike the other two electron donors, a yellowish-brown colour remained in the molasses denitrified water which remained even after the slow sand filtration. Furthermore, complete denitrification could only be realized at a minimum retention time of 48 hours.

Apart from molasses very little difference in performance of PBR was observed between methanol and ethanol as electron donors. The actual performance of the PBR on methanol will be given in detail.

4.3.2 Nitrate removal in PBR with methanol as electron donor

The results of different hydraulic retention time (HRT) and different carbon to nitrogen ratios using methanol as electron donor in the removal of nitrate, is summarized in Table 4.2.

TABLE 4.2 THE REMOVAL OF NITRATE IN A PBR AND SSF UNDER DIFFERENT OPERATION CONDITIONS WITH METHANOL AS ELECTRON DONOR

HRT (h)	C:N	COD(mg/l)		NO_3-N(mg/l)		%NO_3-N REMOVED
		PBR	SSF	PBR	SSF	
6	2.0:1	183	155	<1.0	<0.1	≈100
	1.5:1	99	52	<1.0	<0.1	≈100
	1.0:1	17	7	0.9	1.2	≈ 98
4	1.0:1	21	9	3.7	4.0	≈ 93
2	1.0:1	25	11	5.1	5.8	≈ 90
1	1.0:1	32	26	7.1	7.8	≈ 87
0.5	1.0:1	98	81	35.0	36.0	≈ 40

4.3.3 Sulphate removal, turbidity and micro-organisms

After it was established that a C:N of 1:1 give good consistent results, a complete set of analyses were done for varying HRT's and with methanol as electron donor. The results of these experiments are given in Table 4.3.

TABLE 4.3 SULPHATE REDUCTION, TURBIDITY AND BACTERIAL COLONY COUNTS IN THE EFFLUENTS OF PBR AND SSF UPON VARYING HRT

HRT	SO_4-(mg/l)		Turbidity (Jackson) Units		Total plate count		*E. coli* count	
	PBR	SSF	PBR	SSF	PBR	SSF	PBR	SSF
6.0	50	50	1.1	1.5	-	-	-	-
4.0	54	54	2.0	1.5	-	-	-	-
2.0	58	59	2.0	1.5	-	-	-	-
1.0	58	58	3.2	1.9	$102x10^4$	250	0	0
0.5	58	57	5.2	4.8	-	-	-	-

4.3.4 Other observations

The height in the column where denitrification is completed varied according to the HRT. Table 4.4 shows approximate height in the column where steady state nitrate concentrations were observed.

TABLE 4.4 HEIGHTS IN PBR COLUMN WHERE STEADY STATE NITRATE CONCENTRATIONS WERE REACHED UPON VARYING HRT'S AND C:N=1

HRT(Hr)	Height from bottom of PBR column (cm)	Steady state NO_3-N (mg/l)
12	25	1.0
6	100	1.0
4	175	4.0
2	175	5.0
1	175	7.1

4.3.5 Identification of Pathogenic Organisms

The APi20E system was used to identify the potential pathogenic organisms from the PBR product water (section 4.2.8). The results are shown in Table 4.5.

The volume of gas collected as nitrogen in the Mariotte gas meter corresponded very closely (+86%) with the theoretically calculated volumes (section 3.4.2 equation number 5).

Finally, the pH of the final product water varied between pH 7.1 and 7.9 and no nitrite was observed in any of the samples analysed.

TABLE 4.5 POTENTIAL PATHOGENIC SPECIES AND THEIR DESCRIPTIONS

Species	Description (Bergey's Manual of Determinative Bacteriology, 1974)
Klebsiella pneumonia	Widely distributed in nature, in water, grain etc. and is normally found in the intestinal canal of man and animals. It may be isolated in association with certain pathological conditions in man, e.g. infection of the urinary or the respiratory tract.
Klebsiella ozaenae	Occurs in chronic diseases of the respiratory tract.
Enterobacter aerogenes	Found in faeces of man and other animals, sewage, water and dairy products.

4.3.6 Discussion and conclusion

Although practically any readily available electron donor (carbon source) could be used for biological denitrification in general, this study has shown that for potable water, electron donors are limited to those that will produce an acceptable end product. In this study it was found that methanol and ethanol gave practically similar denitrification results (Appendix 1), while molasses added a colour to the water which makes the final water unacceptable for potable and other domestic uses.

The PBR followed by a SSF performed satisfactorily under a wide range of operating conditions. The results in Table 4.2 indicate that virtually complete denitrification, occurred at

C:N=1.5:1. Due to the poisonous nature of methanol (Dahab and Lee, 1988) it is recommended that for practical purposes a C:N=1:1 is used. Although a C:N=1 did not give complete denitrification, especially at relative short HRT's (see Table 4.2), the COD (and thus the electron donor) was reduced to concentrations comparable to COD concentrations found in potable water (Rand Water Board's COD values range between <10 and 24 mg/l). The data in Tables 4.2 and 4.3 shows that denitrification in PBR is nearly completed at HRT of 1h and more. The practical implication of this is that relative small and thus relative inexpensive PBR would suffice for the rural areas of Bophutatswana.

An interesting observation was that very little sulphate reduction (see Table 4.3) occurred under the test conditions. This means that very little tastes and odours could be expected from sulphate reduction. Table 4.3 also shows that both the turbidity and total plate count are significantly reduced by the SSF. Although no E. coli could be detected in the SSF effluent, the API20E systems analysis of bacteria isolated from the PBR and SSF revealed a spectrum of potential pathogenic bacteria. This finding shows that activated sludge cannot be used as an inoculum for a PBR used for the denitrification of potable water. (The question of suitable inoculum is addressed in the next chapter.)

A final observation was that after more than 1,000 bed volumes had been passed a thick attached growth occurred in the lower part of the PBR. No measurable degree of plugging was observed. It furthermore seems as if the graded packed material has no significant effect on the turbidity of the effluent. For practical purposes a homogeneously packed column, packed with stones of 12 - 20 mm in size may suffice.

CHAPTER 5

DENITRIFICATION WITH A PURE CULTURE

5.1 Introduction

Results of the API20E test (section 4.2.8) show that the product water of the PBR inoculated with denitrifying activated sludge, contained potential pathogenic bacteria which survived with denitrifiers in the previous denitrification experiment (Table 4.5). Activated sludge could thus not readily be used as an inoculum for the denitrification of potable water.

In this study denitrifying bacteria were isolated from a source free of human faecal contamination and the denitrifying potential of this non-pathogenic bacteria evaluated.

5.2 Materials and Methods

5.2.1 Enrichment for denitrifying bacteria and the isolation of pure cultures.

About 20g of undisturbed field soil was mixed with 100 ml of sterilized tap water, the sand and debris were then allowed to settle. Serial dilutions were made by transferring 10ml of the supernatant into 90ml sterilized tap water. The process was repeated until the original supernatant was 10-6 times diluted. Ten millilitre aliquots of each of the dilutions were transferred to Erlenmeyer flask containing 250ml denitrification medium (1g glucose, 5g peptone, 1g KNO_3 per litre). The Erlenmeyer flasks were fitted with inverted test tubes which were filled with denitrification media during sterilization. The inoculated Erlenmeyer flasks were incubated at 30°C and daily inspected for any sign of gas collected in the inverted test tubes.

The highest dilution (10^{-3} dilution) which showed visible growth and gas production after 48 hours, was streaked out on nutrient agar and incubated at 30°C. Single colonies were picked and inoculated into test tubes provided with Durham tubes and denitrification media. The bacteria from the test tubes which showed gas production within 24 hours. were tested with the API20E system for pathogenicity. Those denitrifying bacteria which tested negative, were identified and used as inoculum for subsequent denitrification evaluation in PBR.

5.2.2 Denitrification Experiment

5.2.2.1 Reactor Preparation

The PBR used in the first experiment (Chapter 4), was emptied of the media and thoroughly cleaned with tap water. Then new clean media was packed into the clean reactor with gradings the same as in Figure 4.1. The reactor and media were again cleaned by backwashing with tap water and then filled with concentrated solution of calcium hypochlorite [$Ca(OCl)_2$] and allowed to stand overnight in order to disinfect the reactor and media. The PBR was then emptied of the calcium hypochlorite solution and thoroughly rinsed in an upflow manner with tap water.

5.2.2.2 Inoculation of PBR

The inoculation for the PBR was prepared by inoculating one litre sterilized denitrification media with the identified non-pathogenic denitrification culture described above. After 24 hours incubation at 30°C, the culture was transferred to the PBR as inoculum. The PBR was then filled with the nitrate and methanol experimental water (see Table 4.1) and allowed to adapt for 24 hours, after which the feeding programme was commenced.

5.2.2.3 Feeding Programme

The experimental layout was the same as shown in Figure 4.1. Reactor feeding was started at a retention time of 6 hours and a C:N=1.5:1 using methanol as electron donor. After steady state conditions were achieved the C:N was changed to 1:1 and the hydraulic residence time stepwise reduced from 6 hours to 1 hour.

5.3 Results

5.3.1 Denitrifying Bacteria

Applying the APi2ONE system to the isolated denitrifying bacteria showed that the dominating bacteria were from Alcaligenes species. According to the Bergey's Manual of Determinative Bacteriology (1974), members of the genus Alcaligenes occur in dairy products and other foods and in fresh water and terrestrial environments in which they are involved in the decomposition and mineralization processes. Alcaligens are not known to enter into either pathogenic or symbiotic association with plants or animals. Most strains are able to denitrify, by respiring anaerobically in the presence of nitrate or nitrite to produce nitrogen gas.

5.3.2 Denitrifying Experiment

The results of different hydraulic retention times (HRT) and carbon to nitrogen ratios using methanol as electron donor for the removal of nitrate, are summarized in Table 5.1.

TABLE 5.1 THE REMOVAL OF NITRATE IN A PBR AND SSF UNDER DIFFERENT OPERATING CONDITIONS WITH METHANOL AS ELECTRON DONOR

HRT	C:N	COD (mg/l)		NO_3-N(mg/l)		%NO_3-N
		PBR	SSF	PBR	SSF	REMOVED
6	1.5:1	94	49	<1.0	<1.0	≈100
	1:1	16	7	0.5	0.8	≈99
4	1:1	19	11	2.9	3.5	≈95
2	1:1	21	13	4.6	5.1	≈92
1	1:1	29	24	6.2	6.7	≈89

The volume of gas collected as nitrogen in the Mariotte gas meter corresponded very closely (89%) with the theoretically calculated volumes (Metcalf and Eddy, 1979) (Appendix 2). The pH of the final water varied between 7.1 and 7.7. No nitrite could be detected in any of the samples analyzed.

5.4 Discussion and conclusion

Comparison of results in Tables 4.2 and 5.1 indicates a slightly higher percentage of nitrate removal with a pure culture than a mixed culture under the same operating conditions. No potential pathogens were observed in the final water where the PBR was inoculated with a pure culture. This finding shows that unlike a mixed culture of activated sludge, a pure culture of denitrifiers, Alcaligenes species in this case, can safely be used for denitrification of potable water.

CHAPTER 6

NITRATE REMOVAL BY ION-EXCHANGE

6.1 Introduction

An alternative for biological denitrification is nitrate removal by ion-exchange. Since the intention is to remove nitrates from clear groundwater, this process could be a viable alternative due to its simplicity, provided it is also economically comparable with biological denitrification.

In this study the relevant operational parameters of a nitrate removing ion exchange process operating on a typical Bophutatswana groundwater was evaluated for the purpose of a cost comparison between anion exchange and a biological denitrification process.

6.2 Theoretical background

In the ion exchange process unwanted ions (like NO_3^-) are selectively exchanged for the relatively harmless ions (like Cl^-) by means of an anion exchange resin (Benefield, Judkins and Weand, 1982).

Different ion exchange resins are available for different purposes. For NO_3^- exchange, strong anionic resins like Duolite A101, Amberlite IRA996 and Imac HP are available for this purpose.

In this study the relevant operational parameters of a nitrate removing ion exchange process operating on a typical Bophutatswana groundwater were evalutated for the purpose of cost comparison between ion exchange and a biological denitrification processes.

Economical evaluation is broadly a function of the cost of the resin, the amount of salt (NaCl) required for regeneration, the % water wasted during backwashing, regeneration and rinsing and capital costs for hardware.

6.3 Experimental

6.3.1 Experimental layout

The laboratory ion exchange column (I.E.C.) was constructed of perspex tubing with an inner diameter of 48 mm and a height of 1.5 meters giving a void volume of 1.8l (Figure 6.1).

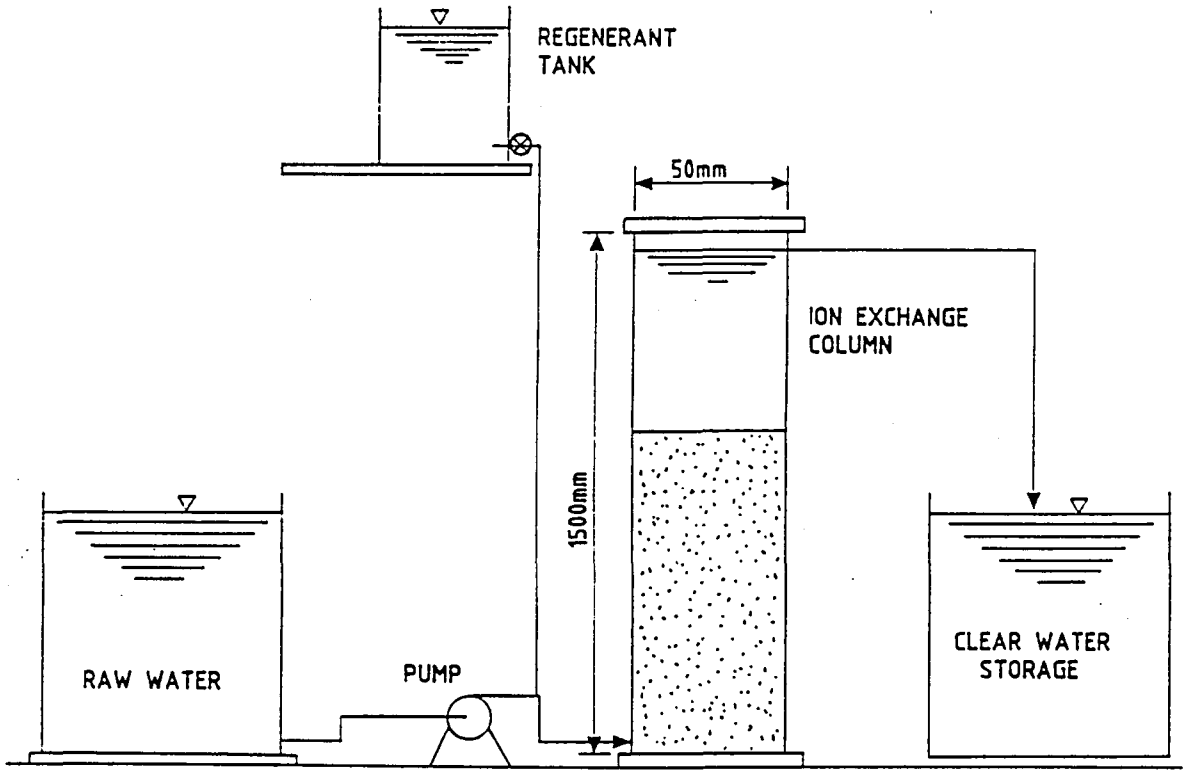

Fig. 6.1 Ion exchange - experimental layout.

The strong base anion exchange resin (Duolite A101) was used for this study. The main physical and chemical properties of the Duolite A101 are listed in Table 6.1.

TABLE 6.1 TYPICAL PHYSICAL AND CHEMICAL PROPERTIES OF DUOLITE A101

Specific gravity	1.07 Cl form
Bulk density	Approx. 700g/l Cl form
Uniformity coefficient	1.3 - 1.6
Ionic form as supplied	Cl
Total capacity	0.09 g NO_3-N/l resin

The resin was soaked in tap water overnight prior to its transfer to the I.E.C. This was to prevent any excess swelling in the column. Test water compositon was as given in Table 4.1 (section 4.1.2). Commercial sodium chloride was used as regenerant.

6.3.2 Operational

Raw water was pumped to the I.E.C. using a magnetic pump. The flow through the reactor was maintained at 6.0 l/h (9 bed volumes/h). The cut-off point for the service mode was fixed at 5mg/l NO_3-N i.e. service was stopped and resin regenerated when the product water reached a concentration of 5mg/l NO_3-N.

6.3.3 Regeneration

The principal variable during this study were the regenerant levels of 120 and 200g NaCl/l resin. Regeneration concentrations were used at a regeneration flow rate of 2 bed volume (bv)/h with contact time of 45 minutes. Displacement of the regenerant was followed by rinsing with 8 bed volumes of treated water.

Regeneration was done in the co-current operation for regenerant levels of 200g NaCl/l resin and in both the co-current and counter-current mode for the regenerant levels of 120g NaCl/l resin. Residual nitrate concentration was measured against bed volumes displaced during service and after regeneration with different regenerant concentration using different regeneration techniques (co-current and counter-current).

6.4 Results

The results of residual nitrate concentration versus bed volumes during service and residual nitrate concentration versus bed volumes after regeneration are shown graphically in Figures 6.2 and 6.3 respectively.

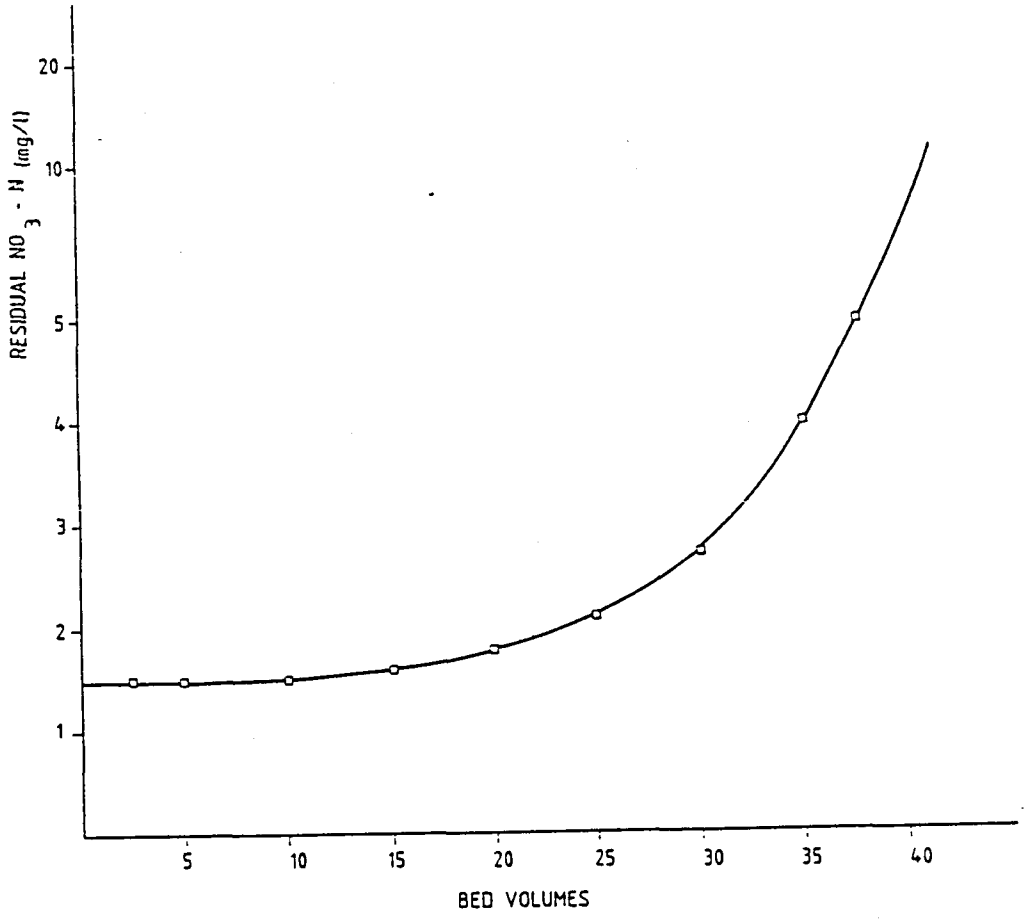

Fig. 6.2 Residual nitrate concentration (mg/l NO_3-N) versus bed volumes.

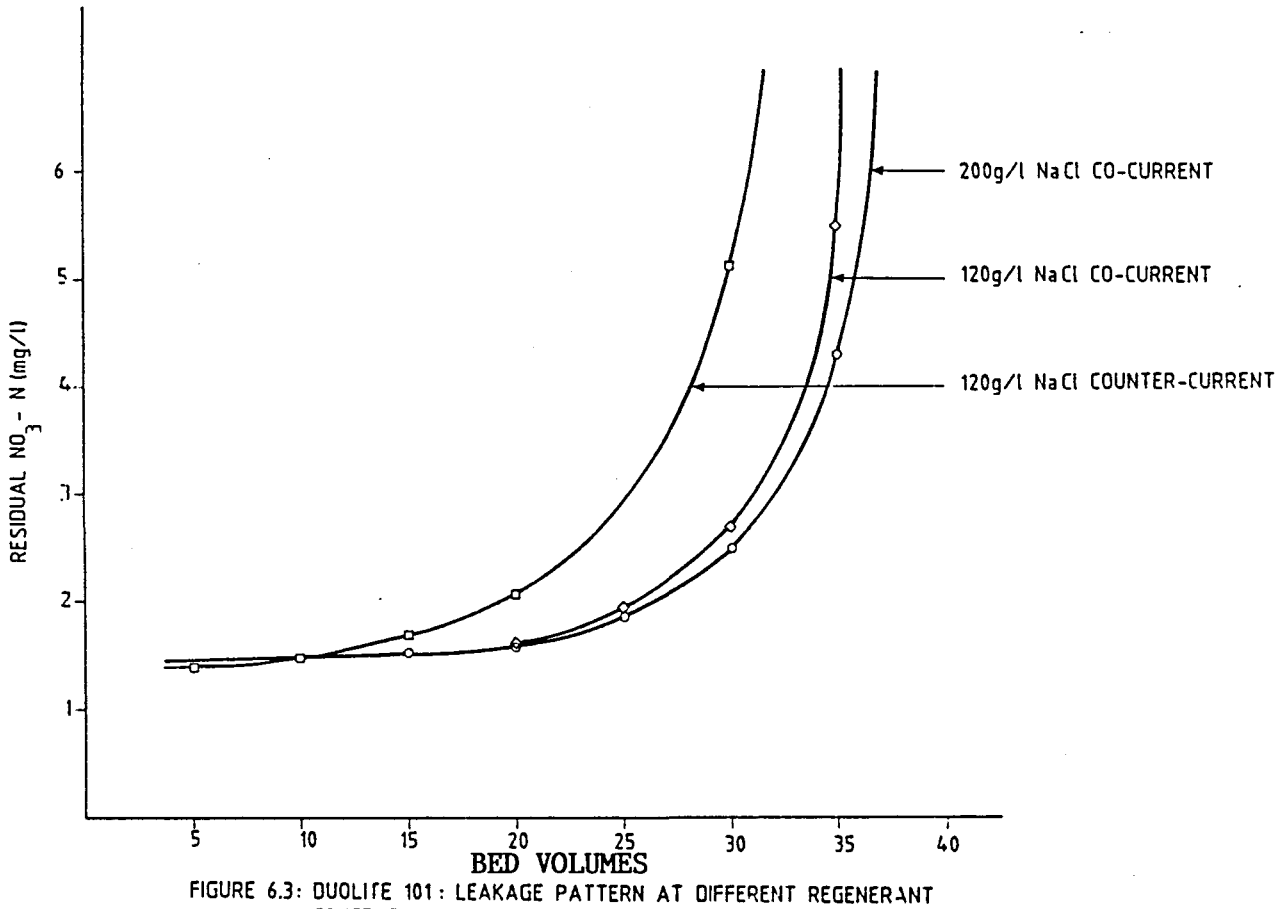

FIGURE 6.3: DUOLITE 101: LEAKAGE PATTERN AT DIFFERENT REGENERANT CONCENTRATION USING DIFFERENT REGENERATION FLOW MODES

Fig. 6.3 Leakage patterns at different regenerant levels using co-current and counter-current operation.

6.5 Discussion and conclusion

The Duolite A101 resin resulted in a nitrate leakage of 1.5 mg/l NO_3-N, when operated at 9bv/h. A nett production of 37 litres of denitrified water was obtained before the nitrate concentration in the product water reached the cut-off value of 5mg/l NO_3-N. This represents an exchange capacity of 0.08g NO_3-N/l resin (the suppliers data is 0.09g/l resin).

Better regeneration efficiency was obtained with regeneration level of 200g NaCl/l resin (average residual nitrate concentration of treated water was 1.6 mg/l NO_3-N with 37 bed volumes before cut-off point of (5 mg/l NO_3-N) was reached. The next better regeneration level was 120 g NaCl/l resin with counter-current mode of operation. An average residual nitrate concentration of treated water was 1.3 mg/l NO_3-N, however only 35 bed volumes could be seen through the resin before cut-off point was reached.

On the average 9% of the treated water was used for regeneration and rinsing of the resin.

CHAPTER 7

COST ESTIMATES AND COST COMPARISON

7.1 Background

Cost estimates and cost comparison is based on design capacities of 10 and 20 m^3/d. The reason for the choice of 10 and 20 m^3/d is that it was found through a survey, that water consumption for communities with groundwater contaminated with nitrate concentrations as high as 375 mg/l NO_3-N varied between 8 and 18 m^3/d.

7.2 Down Payment Calculation

The following parameters were used for the calculation of costs:

Interest rate	20%
Period of payment	10 years
Assumed plant utilization	85% of design capacity
Water quality	60 mg/l NO_3-N

The major Capital Components were the following:

Ion exchange
Ion exchange column.
Resin.
Pumps.
Treated water tower.
Regenerant tank.
Tank stand.

Biological denitrification
Denitrification column.
Media.
Pumps.
Reservoir.
Sandfilter.

Downpayment per cubic metre were then calculated for both design capacities. (10 and 20 m^3/d) for ion exchange and biological denitrification (Kuiper, 1971).

7.3 Results

A summary of the results is given in Table 7.1 and 7.2 below.

TABLE 7.1 COSTS COMPARISON FOR DESIGN CAPACITY OF 10 m^3/d

	Ion exchange	Biological denitrification
Capital cost/m3	R1.48	R1.25
Running cost/m^3	R0.35	R0.25
Total treatment cost	R1.83	R1.50

TABLE 7.2 COST COMPARISON FOR DESIGN CAPACITY OF 20 m^3/d

	Ion exchange	Biological denitrification
Capital cost/m^3	R0.78	R0.70
Running cost/m^3	R0.33	R0.25
Total treatment cost	R1.11	R0.95

7.4 Conclusion

Biological denitrification has shown to be a cheaper method of nitrate removal than ion exchange. For the treatment of raw water of 60 mg/l NO_3-N and design capacity of 10 m^3/day, the

approximate costs are R1.83/m^3 and R1.50/m^3 for ion exchange and biological denitrification respectively. A design capacity of 20 m^3/d results in treatment costs of R1.11/m^3 and R0.95/m^3 for ion exchange and biological denitrification respectively. Biological denitrification is therefore recommended over nitrate removal by ion exchange.

APPENDIX 1

THE REMOVAL OF NITRATE IN A PBR AND SSF UNDER DIFFERENT OPERATION CONDITIONS WITH ETHANOL AS ELECTRON DONOR

HRT (h)	C:N	COD(mg/l) PBR	COD(mg/l) SSF	NO_3-N(mg/l) PBR	NO_3-N(mg/l) SSF	% NO_3-N REMOVED
6	2.0:1	188	160	<1.0	<0.1	100
	1.5:1	99	51	<1.0	<0.1	100
	1.0:1	19	8	<1.0	0.9	98.5
4	1.0:1	23	10	4.0	4.5	92.5
2	1.0:1	28	11	5.9	5.0	91.7
1	1.0:1	30	27	7.5	8.1	86.5
0.5	1.0:1	100	84	40.3	41.4	31.0

APPENDIX 2

COLLECTION OF NITROGEN GAS RELEASED (l/h)

R. T. (HRS)	C:N	N_2 COLLECTED	N_2 EXPECTED	% N_2 COLLECTED
2.4	6:1	0.056	0.06	94
18	6:1	0.067	0.07	95
12	6:1	0.1	0.11	96
12	4:1	0.1	0.11	96
6	4:1	0.21	0.22	97
6	2:1	0.21	0.22	97
6	1.5:1	0.2	0.22	95
6	1:1	0.19	0.22	86
4	1:1	0.29	0.33	87
2	1:1	0.6	0.67	89
1	1:1	1.16	1.34	86
.5	1:1	1.2	2.68	44

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