13 Treatment Processes for Nitrate Removal from Water Supplies

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13.1 INTRODUCTION

Historically, water treatment for the production of potable supplies was concerned primarily with disinfection and removal of suspended or colloidal material contributing to visible turbidity or colour. The earliest treatment plants were based on the use of slow sand filtration through beds of fine sand to remove turbidity of mineral or algal origin. Such processes were labour intensive (for cleaning accumulated material from the beds) and required large areas of land. However, the ability to produce a high-quality water has led to their continued use, and many slow sand filter plants are still in operation worldwide. In particular, the London area is served by several very large slow sand filter works.

In order to reduce the labour and land requirements for water treatment, more rapid processes were developed. The new treatment plants were still based on filtration through beds of sand but also used chemical coagulation to improve the removal of material through the coarser sand needed to establish the higher rates of filtration. Such plants often include a clarification stage based on settlement or, more recently, flotation, to reduce solids loadings to the filters. The removal of turbidity and colour from the water not only improves its appearance but also helps to increase the efficiency of disinfection necessary to prevent the spread of waterborne disease.

The most widely used and well-established disinfectant for water treatment is chlorine, although concern over the production of chlorination by-products has resulted in a trend towards other disinfectants (for example, ozone or chlorine dioxide). Modern treatment works can involve many process stages, including initial screening, chemical coagulation, clarification, sand filtration, ozonation and chlorination. Concern over organic compounds in water supplies, particularly pesticides, is leading to the widespread inclusion of adsorption processes into the treatment stream, using granular activated carbon. Chlorination by-products and pesticides are among the group of health-related parameters of newly developed concern within the water supply industry. Nitrate is also included within this group.

High-nitrate concentrations in water supplies can lead to a rise in bottle-fed babies of methaemoglobinaema (see Chapters 1 and 15), a condition in which the oxygen-carrying capacity of the blood is impaired. There is also a reported link between high nitrate in drinking

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water and stomach cancer, but the evidence for this is inconclusive (DoE, 1986). The medical implications of high-nitrate concentrations in water supplies have resulted in the EC Drinking Water Directive (EEC, 1980) stipulating a Maximum Admissible Concentration (MAC) of 50 mg $NO_3^-.l^{-1}$ and a Guide Level (GL) of 25 mg $NO_3^-.l^{-1}$. The EC Directive standards are now incorporated into UK legislation by the Water Supply (Water Quality) Regulations 1989 (Statutory Instrument, 1989).

Nitrate in water supplies can be controlled at source through the construction of protection zones or other limitations on nitrate application (see Chapters 11 and 12), but, in addition to the impact on the farming industry of such approaches, the benefits for some aquifers may not become apparent for many years because of the long retention times involved (see Chapter 8). However, protection zones have been established and are under investigation as a long-term solution to the problem of high-nitrate concentrations in some areas.

Another option for control is the use of blending of local low-nitrate sources with the high-nitrate supplies. The low-nitrate sources may have other quality problems, particularly high iron and manganese, but these are cheaper and easier to remove than nitrate. It is also necessary to take into account other chemical changes in the water when introducing blending, which may, for example, lead to corrosion problems because of the alteration in the ionic composition of the water. Blending offers an easy-to-operate solution to nitrate control in water supplies, and is in widespread use. The main drawback is the fact that for some areas the nearest low-nitrate source will be long distances away, and costs for installation of pipes and pumping would be excessive. In such cases it would be necessary to install treatment to remove the nitrate from the supply before distribution. Nitrate is not removed by conventional water-treatment techniques, and a range of nitrate removal process types have been developed in response to rising nitrate concentrations in water sources.

The two types of process currently favoured for nitrate removal from public water supplies are ion exchange and biological denitrification. Membrane processes (reverse osmosis or electrodialysis) can remove nitrate, but are likely to be expensive for all but the very smallest supplies, where they might offer advantages in terms of ease of operation. They may also be chosen for sites where disposal of ion-exchange waste would be expensive.

In the UK, process development started in the early 1970s. Pilot plant trials of biological denitrification led to the development of a biological fluidised bed process (WRc, 1979) and experimental work with ion exchange (Gauntlett, 1975) identified the need for nitrate-selective resins. Similar development work occurred throughout Europe and the USA, leading to the wide range of processes now available, particularly with regard to biological denitrification. Installation of plants commenced in the late 1980s in the UK and other European countries, notably France (Philpot and Larminat, 1988; Richard, 1989) and, to a limited extent, in the USA (Lauch and Guter, 1986).

13.2 ION EXCHANGE

13.2.1 GENERAL PRINCIPLES

In ion exchange, the water to be treated is passed through beds of ion-exchange resin beads, which absorb anions, including nitrate, in exchange for another anion, normally chloride. When the capacity for exchange becomes exhausted the resin is regenerated, normally with sodium chloride solution, to return it to the chloride form ready for the next operational run. The spent regenerant, high in nitrate and chloride, can represent up to 1% of the volume

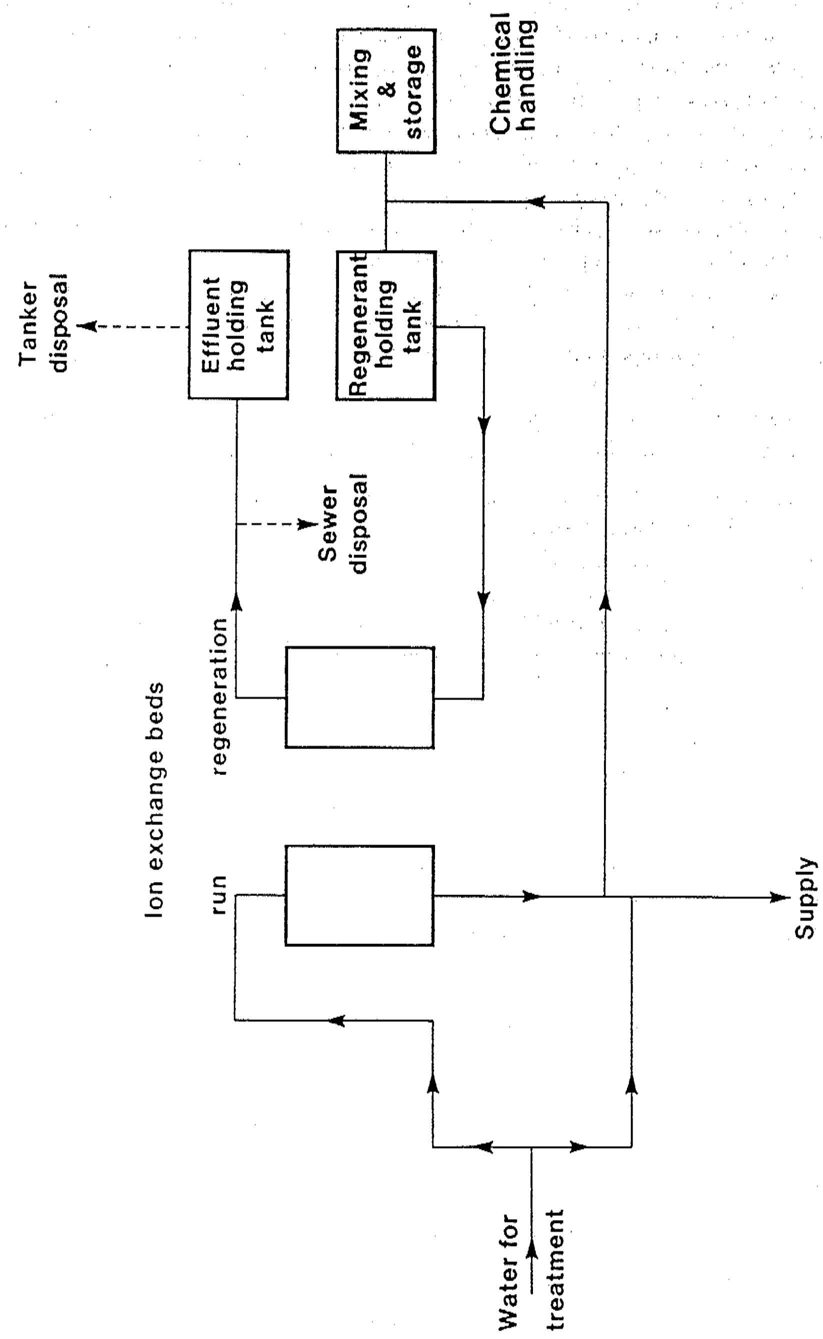

Figure 13.1 Ion-exchange plant schematic

treated; disposal would be by discharge to sewer or tankering to a large river or the sea. The high chloride concentration makes it unsuitable for disposal to agricultural land.

The need for waste disposal can have a major influence on process operating costs. It is possible to reduce the disposal volume using membrane processes, particularly electrodialysis, to treat the waste, but this would also be expensive and the costs must be balanced against those for direct disposal without waste treatment. The use of biological denitrification to treat ion exchange waste has been investigated (van der Hoek, van der Ven and Klapwijk (1988)); removal of nitrate from the waste could make disposal easier and can also allow limited re-use of the regenerant, thus reducing waste volumes.

A layout for a typical ion exchange plant is shown in Figure 13.1. A minimum of two beds would normally be used, operated out of phase with each other to even out the treated water quality variations which occur during a single run. An example of these quality variations for a nitrate selective resin, as discussed below, is shown in Figure 13.2. Counterflow regeneration (regenerant flow in the opposite direction to the treated water flow) is normally used for nitrate removal ion exchange.

The anion exchange resins used for the production of demineralised water (conventional resins) are capable of removing nitrate but suffer from the following disadvantages:

(1) Increased chloride in the treated water as a result of regeneration with sodium chloride, coupled with reduced bicarbonate, a situation which could lead to the corrosion of brass fittings in the distribution system.

(2) Low nitrate removal capacity for water high in sulphate because of the higher selectivity for sulphate over nitrate shown by the conventional resins; lower nitrate capacity results in greater salt requirements and larger volumes of waste regenerant for disposal.

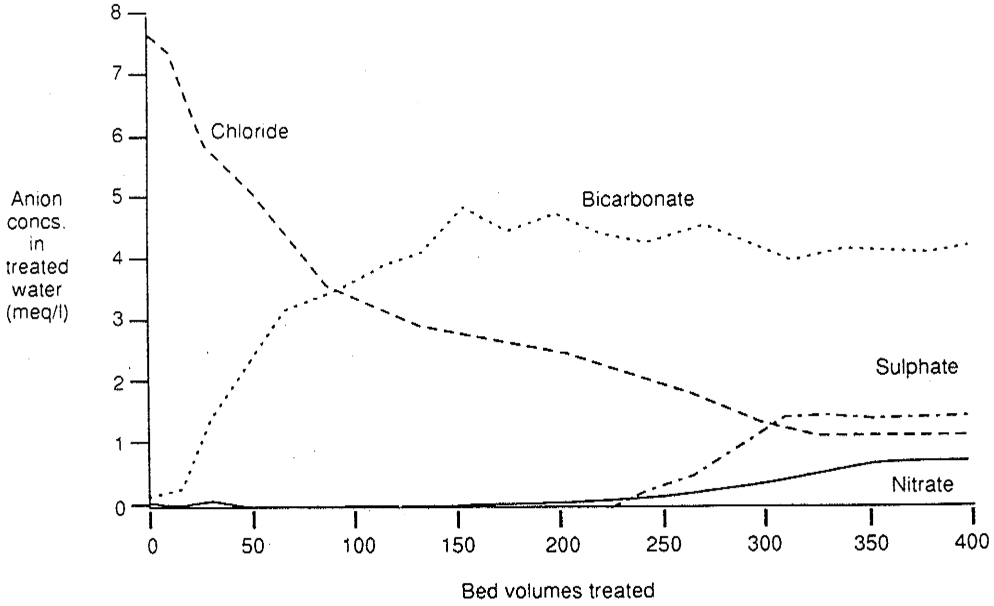

Figure 13.2 Treated water quality variation from a single ion exchange bed over one run (using nitrate selective resin)

(3) Displacement of nitrate from the resin by sulphate (because of the sulphate selectivity) if the run is continued past the point at which nitrate leaving the bed is equal to the inlet concentration.

Nitrate selective ion exchange resins are now available, which have a higher selectivity for nitrate over sulphate. These have a lower total capacity for ion exchange but a higher nitrate removal capacity for high sulphate waters. The lower total capacity results in lower chloride concentrations in the treated water, and the nitrate selectivity prevents the displacement of nitrate by sulphate. The higher nitrate capacity can result in lower costs for regenerant chemicals and waste disposal when treating high-sulphate waters. These advantages of nitrate-selective resins are illustrated in Figure 13.3.

The potential problem of corrosion of brass fittings (particularly dezincification (Croll, 1991) occurs as a result of the increased ratio of chloride to bicarbonate (alkalinity) in the treated water, rather than from the higher chloride concentration itself. In most water supply applications the increase in corrosion potential will be marginal. However, in some cases the corrosion will be unacceptable and measures to reduce the chloride to bicarbonate ratio will be necessary. One way of alleviating this problem is to use bicarbonate during regeneration

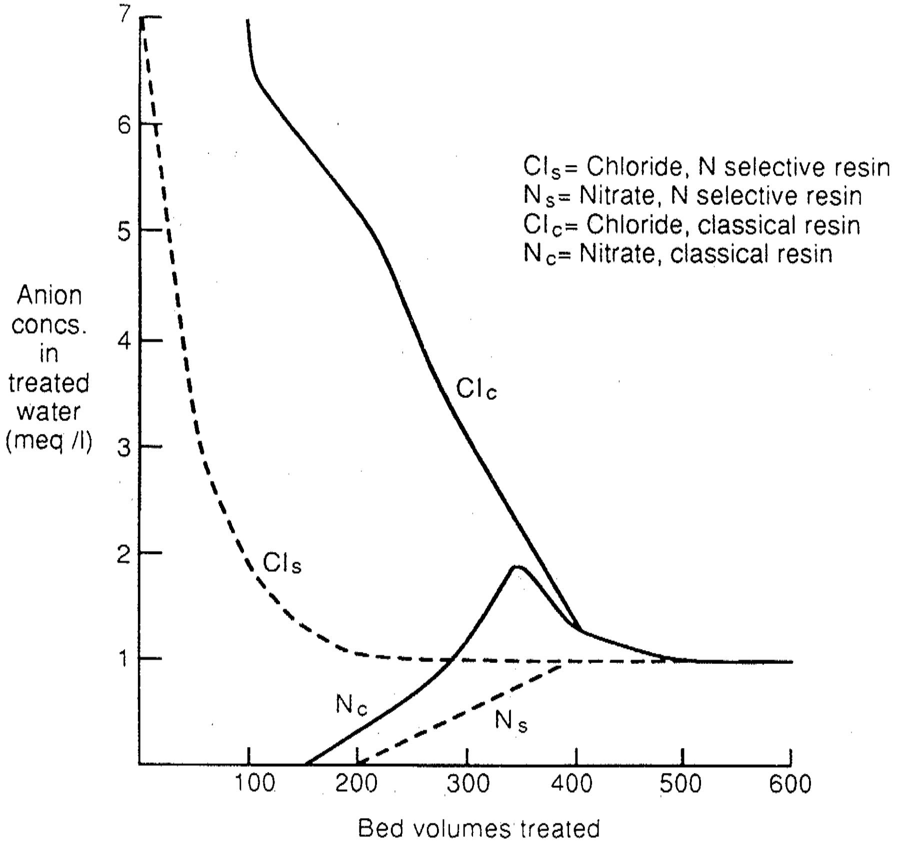

Figure 13.3 Advantages of nitrate selective resins over conventional anion exchange resins

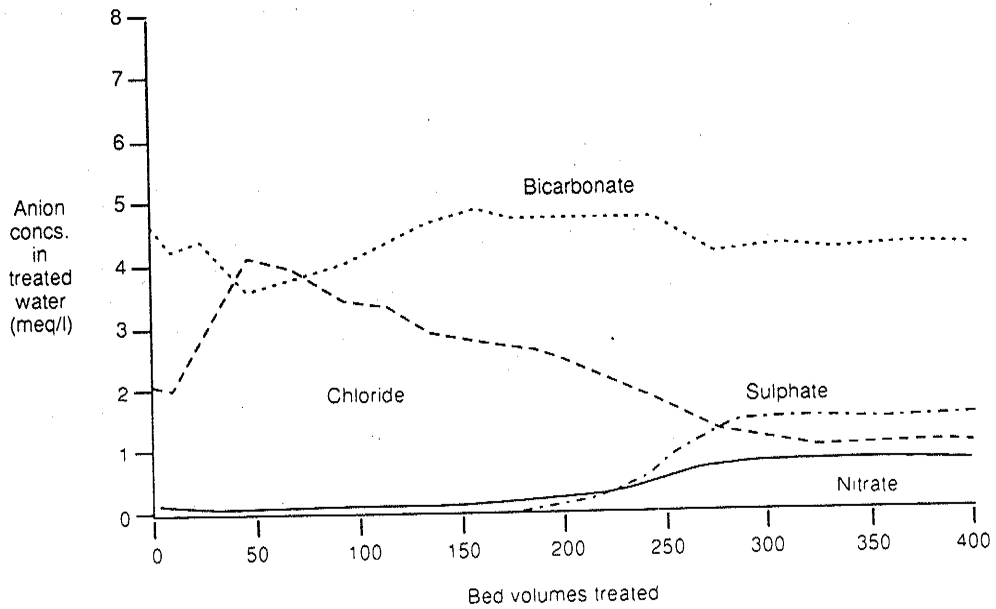

Figure 13.4 The effects of bicarbonate conditioning on treated water quality

to condition the resin after normal counterflow regeneration with sodium chloride. This involves passing a small volume of sodium bicarbonate solution upwards through the bed, such that resin in the lower part of the bed is in the bicarbonate rather than chloride form. The initial high chloride peak passing down the bed is partly exchanged for bicarbonate, reducing the early chloride:bicarbonate ratio in the treated water. An example of this is illustrated in Figure 13.4. Sodium bicarbonate is expensive compared with sodium chloride; the minimum amount needed to achieve a satisfactory chloride: bicarbonate ratio would be used, and in most circumstances the effects on total operating costs would not be large.

One further problem observed in the past has been the deposition of calcium carbonate within the bed (Harries, 1981) probably as a result of the formation of localised conditions of high pH and bicarbonate concentration, which increase the headloss through the bed. Until more information is available on this problem, current advice includes the provision for a facility for an acid wash and/or full bed backwash in any water-treatment plant installed.

13.2.2 PLANT DESIGN AND OPERATION

The design of ion exchange nitrate removal plants is discussed in detail in a WRc report (WRc, 1989). The plant size is based on hydraulic considerations and the degree of nitrate removal required. Ion exchange can achieve nitrate removal of 90% or more, and would be used to treat as small a proportion of the total flow needed to give the necessary overall nitrate removal after blending the treated water with the remainder of the flow. Treatment rates are typically 25 $m^3.m^{-3}$ bed per hour (i.e. 25 bed volumes per hour), which, for most waters, should give operating run lengths of up to 24 hours between regenerations. Regeneration can be initiated on the basis of monitoring the treated water nitrate concentration, although for many groundwater sources, the feed nitrate may be constant enough to allow run length to be based on volume treated, once the nitrate breakthrough curve has been established from initial runs. Small-scale pilot plant trials can give useful information for proposed full-scale plant design and operation, for example, with regard to selection of suitable resins and regeneration conditions.

The factors to be considered when choosing the optimum operating conditions for an ion exchange plant are (1) nitrate removal:salt use ratio and (2) nitrate removal:waste volume ratio. These two factors will vary in importance from site to site. On a site where disposal of the waste regenerant is relatively cheap (for example, to a sewer) the process can be optimised on the salt use, thus minimising chemical costs. If, however, the waste regenerant had to be tankered to a separate disposal site, the transport costs could outweight the chemical costs and the waste volume would need to be minimised. This can be done by:

(1) Using an increased concentration of sodium chloride brine for regeneration (although there is a limit to the maximum concentration that can be used for a particular resin because of the osmotic effects which can physically disrupt the resin structure);
(2) Minimising the volume of wasted rinse water (the water used to displace brine from the resin bed, and the first few bed volumes of treated water which may still contain traces of residual brine).

If no rinse water was needed, then the same regeneration level (in grams NaCl per litre of resin) would be the optimum for both waste volume and salt use. However, as this is not the case, and a fixed volume of rinse, independent of regeneration level, is needed, then the optimum regeneration level for the waste volume can be higher than the optimum volume for salt use. This is illustrated in Table 13.1, which shows examples of typical values for nitrate capacity for a nitrate-selective resin at four regeneration levels using a brine concentration of 5% w/v, with different amounts of rinse water (2 ot 4 bed volumes).

Table 13.1 shows how the optimum regeneration level tends toward the higher values as more rinse water is wasted, and to a fairly low value (100 g NaCl per litre resin) if salt use is the most important factor. The table also illustrates the trend of average chloride to bicarbonate ratio in the treated water with increasing regeneration levels. In areas where dezincification could be a problem the average chloride to bicarbonate ratio of the treated water could be important to minimise blending requirements, and so a higher regeneration level would be used.

Table 13.1 Optimisation of ion exchange

Regenerant level (g NaCl per litre resin)	Bed volumes of regenerant (BV)	NO_3 capacity (meq l^{-1} resin)	NO_3: Salt ratio (meq g^{-1} NaCl)	NO_3: Waste volume (2 BVs rinse) (meq l^{-1})	NO_3: Waste volume (4 BVs rinse) (meq l^{-1})	Average treated water Cl/HCO_3 (meq/meq)
50	1	100	2.0	33.3	20.0	0.9
100	2	250	2.5[a]	62.5	41.7	0.85
150	3	330	2.2	66.0[a]	47.1	0.8
200	4	380	1.9	63.0	47.5[a]	0.75[a]

[a] Optimum values.
Salt concentration = 5% w/v.

13.3 BIOLOGICAL DENITRIFICATION

13.3.1 GENERAL PRINCIPLES

Biological denitrification occurs naturally in sediments of rivers, lakes and reservoirs (see Chapter 5) where, in the absence of dissolved oxygen (anoxic conditions), certain widely occurring bacterial species can use nitrate for respiration, converting it to nitrogen gas:

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \text{ (gas)}$$

To use this natural reaction in a water-treatment process requires the establishment of high bacterial concentrations, achieved by providing a food/energy source for bacterial growth and a large surface area for bacterial attachment in biological reactors. A wide range of biological denitrification processes have been developed throughout Europe. They vary in the type of food/energy source used and the nature of the support bed. The principal variations are:

(1) Processes using heterotrophic bacteria which obtain the energy and carbon required for growth from organic compounds — the carbon sources normally used for denitrification are methanol, ethanol or acetic acid;
(2) Processes using autotrophic bacteria which obtain energy from oxidation of inorganic chemicals (hydrogen (Gahrs, Rutten and Schnoor, 1989) or sulphur (Schippers *et al.*, 1987)) and carbon from CO_2/bicarbonate;
(3) Support beds of gravel, plastic media or fluidised sand.

In operating the denitrification process it is important to minimise the amount of food/energy source entering the water supply system in order to prevent any problems of bacterial growth in the distribution pipework. In heterotrophic processes this is done by controlling the carbon source dose in response to the dissolved oxygen and nitrate concentrations in the feedwater. The autotrophic processes offer advantages in this respect. Excess hydrogen can be easily removed after denitrification by aeration of the water (Gahrs, Rutten and Schnoor, 1989). Autotrophic denitrification by sulphur-oxidising bacteria uses beds of sulphur granules upon which the bacteria grow; the energy source (sulphur) is utilised directly by the bacteria, and does not enter the water being treated. This process also offers simplicity of operation, which could be of benefit for remote rural sites. A disadvantage of the autotrophic processes arises from the relatively low reaction rates compared with heterotrophic denitrification. This results in the need for larger autotrophic plants for the same degree of treatment and, therefore, higher capital costs. Furthermore, nitrate reduction using sulphur increases the sulphate concentration in the water:

$$5S + 6\,NO_3^- + 2H_2O \rightarrow 3N_2 + 5\,SO_4^{2-} + 4H^+$$

For some waters, this could result in the EC Directive MAC for sulphate ($250\ mg.SO_4^{2-}\ l^{-1}$) being exceeded.

The range of heterotrophic processes developed differ with respect to the carbon source and biomass support material. However, the choice of carbon source is primarily economic and/or political rather than technical, and each of the process types is likely to operate satisfactorily with methanol, ethanol or acetic acid as the carbon source. In the UK, methanol is the cheapest carbon source, and the use of ethanol or acetic acid would increase chemical costs by factors of about 2 or 3, respectively.

The biomass support material is in either fixed beds or fluidised beds. The former are

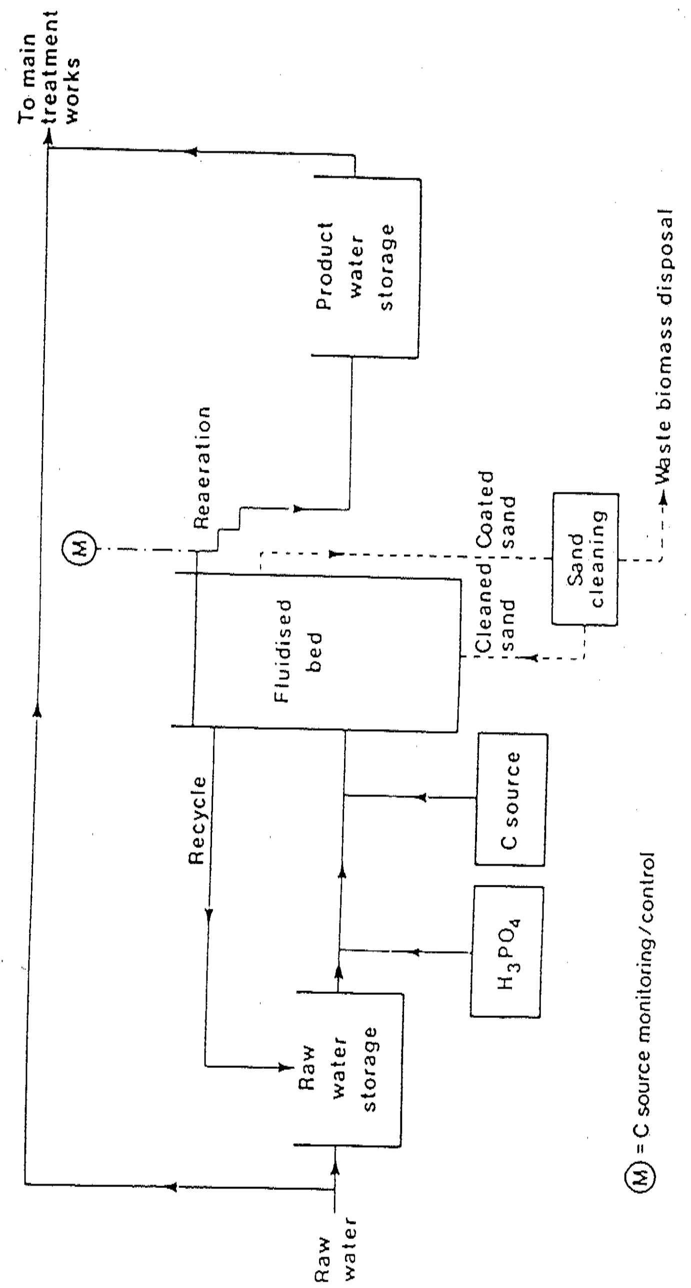

Figure 13.5 Biological fluidised bed schematic

periodically taken out of operation and backwashed to remove excess biomass accumulated as a result of bacterial growth. Biological fluidised bed (BFB) processes are described in more detail below.

'*In-situ*' biological denitrification techniques have been developed (Braester and Martinell, 1988; Mercado, Libhaber and Soares, 1988) which rely on bacterial growth within the aquifer, initiated by injection of a carbon source into the water within the aquifer at strategic points around an abstraction well. *In-situ* denitrification offers potential advantages with regard to capital costs because plant and equipment requirements are minimal. However, its application is likely to be limited to relatively homogeneous aquifers and the area of land required may be substantial, depending upon the properties of the aquifer and the level of denitrification required. The risks associated with failure of *in-situ* systems are much greater than those for treatment plant, in that it may be necessary to abandon boreholes because of aquifer blockage (Kruihof *et al.*, 1985). Treatment plants are, therefore, likely to be the preferred option for most circumstances.

Work in the UK by WRc, Anglian Water and the Department of the Environment has led to the development of a heterotrophic biological fluidised bed (BFB) process (Croll *et al.*, 1985) (Figure 13.5). The use of fluidised sand provides a large surface area for bacterial growth, and consequently high biomass concentrations without the problems of clogging of the bed. The sand is fluidised by upflow of the water being treated. The carbon source (methanol, ethanol or acetic acid) is dosed to the influent water; phosphate dosing may also be necessary to stimulate bacterial growth. Excess biomass is removed, without shutting down the process, by taking sand from the bed through a pump to strip off the attached bacterial film, returning the cleaned sand to the bed and transferring the waste biomass to a suitable point for thickening and storage before disposal. Waste volumes are very small, and the sludge is suitable for disposal to agricultural land.

13.3.2 DESIGN AND OPERATION OF BFB DENITRIFICATION PLANTS

Details of BFB denitrification plant design and operation are given in a WRc report (WRc, 1989). The process uses fine sand (0.3–0.5 mm) as a biomass support, in beds fluidised by upflow of the treated water at rates of 20 m per hour. Removal of dissolved oxygen (DO) occurs at the bottom of the bed, establishing anoxic conditions for nitrate removal in the upper regions of the bed.

The depth of fluidised bed required is a function of the dissolved oxygen and nitrate concentrations in the feedwater, the average biomass concentration in the bed and the rates of DO and nitrate removal at the minimum operating temperature. Other design figures that can be used are as follows:

Average biomass (volatile solids, VS) concentration = 15 kg VS/m^{-3} bed.

Removal rates at 2°C using methanol:

DO removal = 18 g DO/(kg VS.h^{-1})
Nitrate removal = 5.5 g NO_3^-N/(kg VS.h^{-1})

Using these figures, the required fluidised bed depth can be calculated from the raw water DO and nitrate concentration, the total depth required being the sum of the requirements for DO and nitrate. The approximate requirement for DO removal is 0.75 m depth per 10 mg DO l^{-1} feedwater, and for nitrate removal is 0.25 m per mg NO_3^-N l^{-1} removed. As with ion exchange, the plant would be used to treat only a proportion of the total works output to achieve the desired overall nitrate removal. Carbon source requirements for dissolved oxygen and nitrate removal are shown in Table 13.2.

Table 13.2 Carbon source requirements

Carbon source	For DO removal (mg/mg DO)	For nitrate removal (mg/mg NO_3^-N)
Methanol	1.0	2.5
Ethanol	0.5	2.0
Acetic acid	1.2	3.5

Operating experience with all types of biological denitrification process has shown that, at times, elevated nitrate concentrations can occur in the denitrified water, to concentrations well in excess of the EC Directive MAC of 0.1 mg NO_2^- l^{-1} (0.03 mg NO_2^-N l^{-1}). Nitrite is easily removed using chlorine, but the chlorine demand is high (5 mg Cl_2 per mg NO_3^-N) and the nitrite is oxidised back to nitrate, impairing the efficiency of the denitrification process. Biological denitrification processes are normally operated with an underdosing of the carbon source, to leave a small residual nitrate concentration in the treated water, so that virtually complete removal of the carbon source is achieved. Any carbon source remaining in the treated water could lead to problems of bacterial growth in the distribution system. Operation with an underdosing of the carbon source prevents bacterial growth in distribution, but has a tendency to result in nitrite production.

Pilot plant trials have been carried out to investigate the ways of operating the process to maintain nitrite residual concentrations at acceptable levels. This work involved overdosing of the carbon source (methanol) and removal of excess methanol through the development of biological activity in post-denitrification conventional water treatment. The pilot plant was

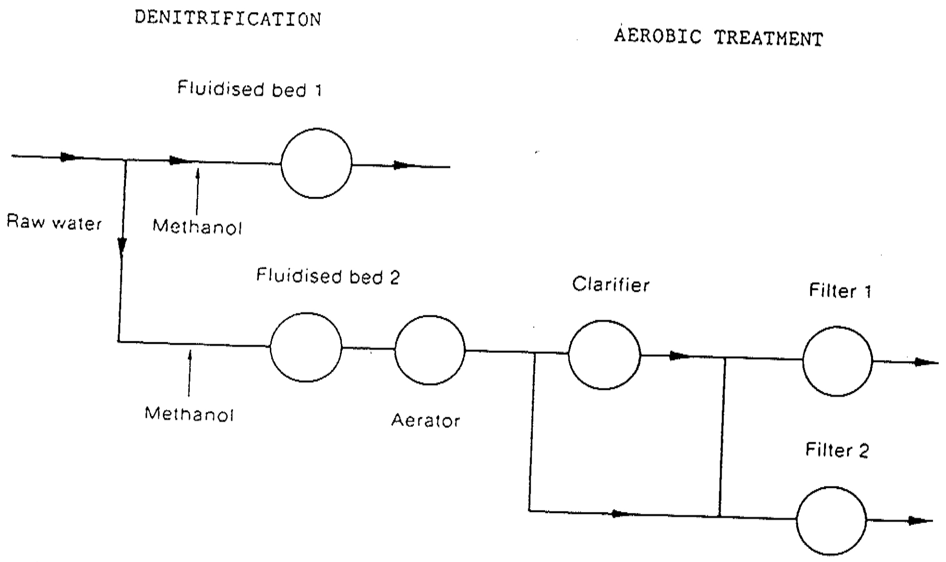

Figure 13.6 BFB pilot plant for investigating control of nitrite residual concentration

designed to compare underdosing and overdosing of carbon source in two fluidised beds operated in parallel. Denitrified water from the overdosed bed was aerated before further treatment using pilot scale floc blanket clarification and/or rapid gravity filtration (both process types being commonly used in conventional water treatment). A schematic of the pilot plant is shown in Figure 13.6.

Overdosing of methanol was successful in controlling nitrite concentrations. This is illustrated in Table 13.3, which shows profile concentrations of nitrate, nitrite and methanol up the fluidised beds. When the methanol is used up in an underdosed bed, high nitrite concentrations remain in the treated water leaving the bed. In the overdosed bed, however, the remaining nitrite is removed, using the excess methanol, once nitrate removal is complete. In principle, the same effect could be achieved by dosing the exact amount of methanol needed for complete denitrification, but problems would be encountered in controlling the dose, which would need to be adjusted as the dissolved oxygen and nitrate concentrations in the feedwater varied.

The results obtained indicated that a methanol excess of up to 5 mg l^{-1} can be removed by conventional treatment processes subsequent to BFB denitrification. This is likely to result from the accumulation of bacteria (carried over from the BFB) within the floc blanket or filter. Regular sampling of the plant over the whole period of operation indicated that good removal could be obtained at water temperatures of below 10°C. Examples of the results are shown in Table 13.4.

Table 13.3 Example of profile results from BFB pilot plant

Height above base of BFB (m)	Underdosed bed (mg l^{-1})			Overdosed bed (mg l^{-1})		
	NO_3N	NO_2N	Methanol	NO_3N	NO_2N	Methanol
0	13	0.12	33	13	0.12	43
1	11	0.37	22	11	0.28	40
2	7	0.49	10	4	0.38	19
3	5	0.48	5	<0.1	0.11	9
4	3	0.60	1	<0.1	<0.1	6
4.5	3	0.42	<0.5	<0.1	<0.001	6

Table 13.4 Methanol removal in conventional treatment processes

Treatment	Mean methanol (mg l^{-1}) in: Feed	Treated
Rapid gravity filtration (coarse sand)	1.7	<0.5
Rapid gravity filtration (fine sand)	1.8	1.0
Chemical coagulation and rapid gravity filtration (coarse sand)	4.8	<0.5
Chemical coagulation and floc blanket clarification	4.8	0.7
Chemical coagulation, floc blanket clarification and rapid gravity filtration (coarse sand)	4.8	<0.5

Chemical coagulation using ferric sulphate

Coarse sand = 8/16 mesh grade
Fine sand = 16/30 mesh grade
Water temperature 16°C

In summary, the pilot plant work indicated that nitrite production could be avoided by using a slight overdose of carbon source (<5 mg l^{-1}) which could be reduced to <0.5 mg l^{-1} by the development of biological activity in conventional water treatment processes.

13.4 MEMBRANE PROCESSES

In reverse osmosis (RO), the hydraulic pressure exerted on one side of a semi-permeable membrane forces water across the membrane, leaving salts, including nitrate, behind (WRc, 1989). This results in the production of a treated water stream and a concentrate waste stream, the relative sizes of each being governed by the pressure differential across the membrane. The principle is illustrated in Figure 13.7. As well as the use of high pressure RO (typically, 15–20 bar) for direct removal of nitrate, it has been proposed that lower pressure (e.g. 5 bar) 'nanofiltration' membranes could be used as a pre-treatment to ion-exchange nitrate removal using conventional resins. These membranes remove sulphate and other divalent ions (e.g. Ca^{2+}) but allow the passage of monovalent ions such as nitrate. Removal of sulphate before ion exchange will increase the nitrate removal capacity of conventional resins (see Section 13.2.1). The economics of such an approach would need to be evaulated for specific circumstances.

In electrodialysis, the membranes allow ions to pass but not water. The driving force is an electrical current which carries the ions through the membranes. By using stacks of alternate anion and cation permeable membranes, it is possible to produce a treated and concentrate stream. Electrodialysis reversal (EDR) is a modification of the process in which the current is reversed on a regular basis to prevent problems of calcium scale formation. The principle of electrodialysis is illustrated in Figure 13.8. Nitrate-selective membranes are now available which make the process more cost-effective for nitrate removal, because the proportion of nitrate removed relative to other ions is increased, compared with conventional EDR. This reduces the overall power requirements per unit nitrate removed.

For both reverse osmosis and electrodialysis the concentrate flows would typically be between 5% and 20% of the throughput; waste volumes are therefore large compared with ion exchange, but the quality of the membrane process waste is higher, and disposal may not present a serious problem. Operating costs for membrane processes are high as a result of power consumption. The economies of scale for membrane processes are low compared with ion exchange or biological denitrification, giving relatively high capital costs for larger-membrane plants.

13.5 PROCESS SELECTION

The main factors influencing the process selection for nitrate removal from public water supplies are:

(1) The need for filtration after biological denitrification, to remove carry-over from the bacterial bed; and
(2) The disposal of ion exchange waste regenerant.

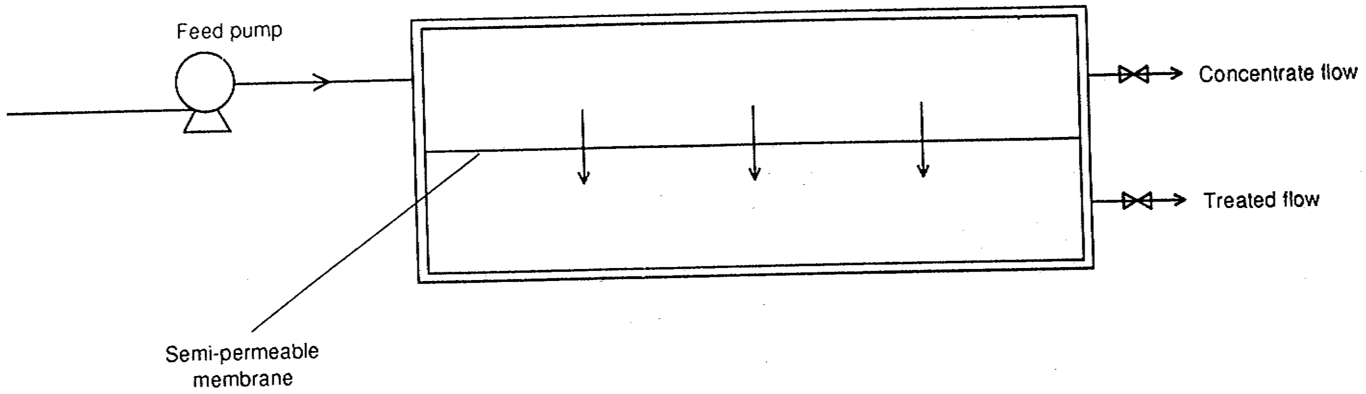

Figure 13.7 Principle of reverse osmosis

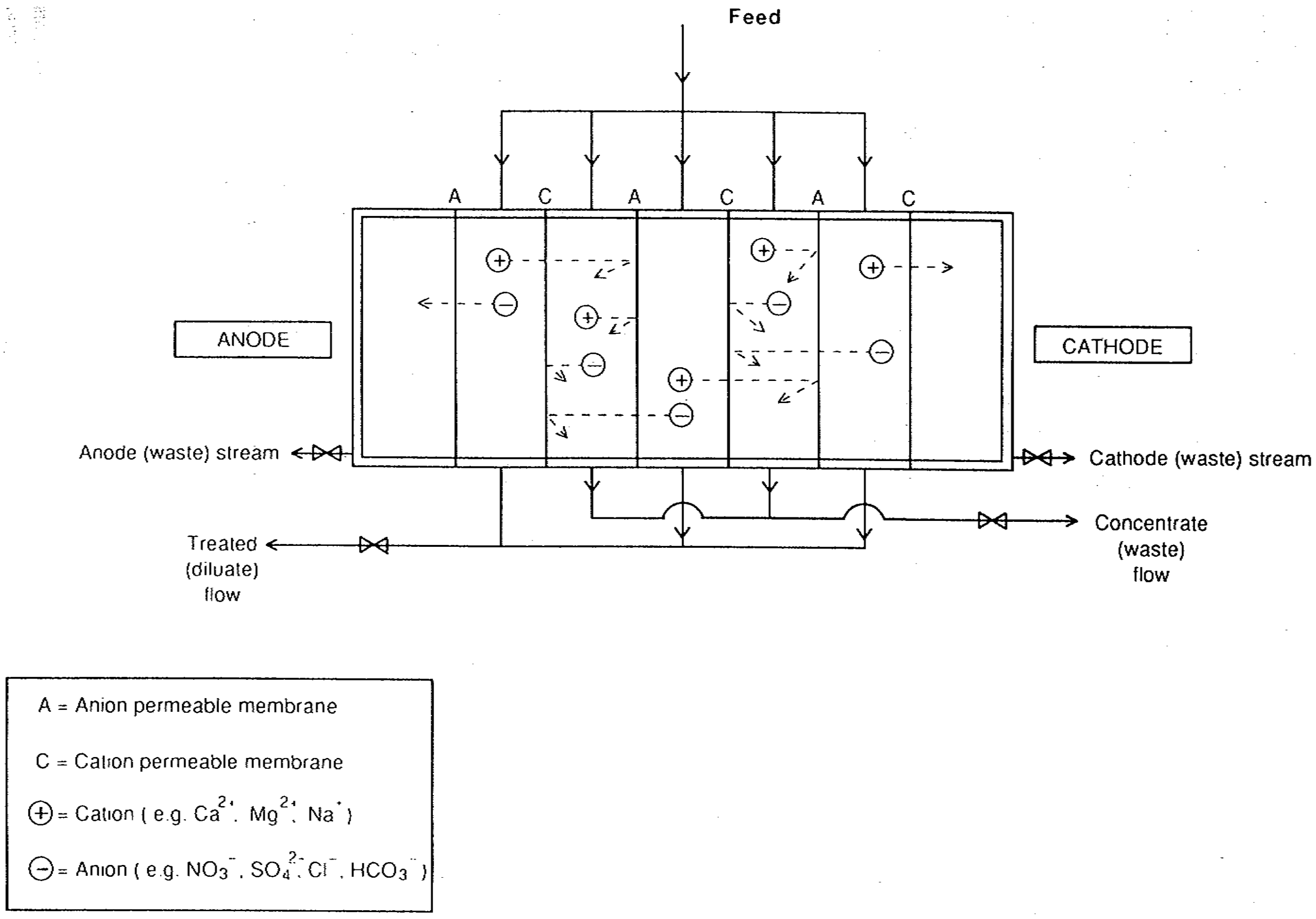

Figure 13.8 Principle of electrodialysis

For most public supplies derived from surface water sources (see Chapter 5) (which are normally relatively large compared with groundwater supplies and are already treated by filtration) with continuously high nitrate concentrations, biological denitrification is likely to be more cost-effective than ion exchange. For groundwater sources (see Chapter 8) (without existing filters), ion exchange would normally be the cheaper process. However, for large groundwater sources (above, say, 20 000 m^3 per day), or where waste disposal by tankering was necessary over distances in excess of 20 km, biological denitrification might offer cost advantages. For very small supplies, the higher costs of membrane processes might be acceptable on the grounds of easier operation or where tankering of ion exchange waste was necessary.

Where nitrate concentrations are intermittently high, the slow start-up characteristics of biological processes may preclude their use in favour of ion-exchange or membrane processes which can be started up rapidly.

The costs for nitrate removal processes are reviewed by Booker, Hall and Hyde (1989), and are discussed in more detail in a WRc report (WRc, 1989). Generally, operating costs plus capital repayment for ion exchange or biological denitrification would be roughly in the range 5–15 p m^{-3} distributed (1991 prices), depending on plant size, nitrate concentration and waste disposal options, among other factors.

13.6 OVERVIEW

The action being taken to maintain nitrate concentrations in water supplies below regulatory levels includes the control of agricultural activity, blending and water treatment. In the long term, the control of agricultural activity through the implementation of nitrate protection zones (Chapters 5, 10 and 12) is likely to offer the most favourable solution to the problem. However, in some areas the full benefits from protection zones may not be derived for many years, and short-term control measures are needed. Blending with local low-nitrate sources is being used, but the availability of low-nitrate sources will limit the potential of this option because of the high costs for pumping over long distances. At many sites, nitrate removal treatment processes will be the only viable control option, at least in the short term and perhaps, in some areas, in the long term.

Treatment by ion exchange offers advantages with respect to operation and costs for smaller groundwater sources. However, the environmental effects of the disposal of ion-exchange waste regenerant, high in nitrate and chloride, are of increasing concern to the water supply industry. As a result, there is likely to be a trend towards the more widespread use of biological denitrification, either to treat the water directly or to treat the ion-exchange waste to reduce its impact on the environment.

REFERENCES

Booker, N.A., Hall, T. and Hyde, R.A. (1989) Removal of industrial and agricultural contaminants from groundwater. *Proceedings of the Conference: World Water '89*, Institution of Civil Engineers, London, November 1989, Thomas Telford, London, pp. 109–15.

Braester, C. and Martinell, R. The Vyredox and Nitredox methods of *in-situ* treatment of groundwater. *Water Science and Technology*, **20**, No. 3, 149–63.

Croll, B.T. (1991) Nitrate removal using ion exchange: brass corrosion considerations. *Institution of Water and Environmental Management Yearbook*, pp. 39–46.

Croll, B.T., Green, L.A., Hall, T., Whitford, C.J. and Zabel, T.F. (1985) Biological fluidised bed denitrification for potable water. In Tebbutt, T.H.Y. (ed.), *Advances in Water Engineering*, Elsevier Science, New York, pp. 180–87.

DoE (1986) *Nitrate in Water*, A report by the nitrate co-ordination group, Pollution Paper No. 26, HMSO, London.

EEC (1980) EC Directive relating to the quality of water intended for human consumption. *Official Journal of the European Community*, No. L229, 80/778/EEC, available in DoE Circular No. 20/82, HMSO, London, 1982.

Gahrs, J.H., Rutten, P. and Schnoor, G. (1989) Drinking water treatment using hydrogen. *Water and Sewage International*, **1**, No. 1, 35–9.

Gauntlett, R. (1975) Nitrate removal from water by ion exchange. *Water Treatment and Examination*, **24**, 172–93.

Harries, R.C. (1981) *Studies of the removal of nitrate from potable surface water by ion exchange*, Unpublished PhD Thesis, Hatfield Polytechnic.

Kruihof, J.C., Van Paassen, J.A.M., Hijnen, W.A.M., Dierx, H.A.L. and Van Bennekom, C.A. (1985) Experience with nitrate removal in the Eastern Netherlands. Paper presented at the Conference 'Nitrates Dans les Eaux', Paris, 1985.

Lauch, P.R. and Guter, G.A. (1986) Ion exchange for the removal of nitrate from well water. *Journal American Water Works Association*, May, 83–8.

Mercado, A., Libhaber, M. and Soares, M.I.M. (1988) *In-situ* biological groundwater denitrification: concepts and preliminary field tests. *Water Science and Technology*, **20**, No. 3, 197–209.

Philpot, J.M. and Larminat, G. (1988) Nitrate removal by ion exchange: the Ecodenit process, an industrial scale facility at Binic (France). *Water Supply*, **6** (Brussels), 45–50.

Richard, Y.R. (1989) Operating experience of full-scale biological and ion exchange plants in France. *Journal Institution of Water and Environmental Management*, **3**, No. 2, 154–67.

Schippers, J.C., Kruithof, J.C., Mulder, F.G. and Van Lieshout, J.W. (1987) Removal of nitrate by slow sulphur/limestone filtration. *Aqua*, No. 5, 274–80.

Statutory Instrument (1989) No: 1147, The Water Supply (Water Quality) Regulations 1989.

Van der Hoek, J.P., Van der Ven, P.J.M. and Klapwijk, A. (1988) Combined ion exchange/biological dentrification for nitrate removal from groundwater under different process conditions. *Water Research*, **22**, (6), 679–84.

WRc (1979) *Biological removal of nitrate from river water*, Technical Report TR98.

WRc (1989) *Nitrate removal from drinking water: a technical and economic review of treatment processes*, Report 856-S.