

## Chapter 4

# Application of bank filtration in aquifers affected by ammonium – The Delhi example

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### 4.1 INTRODUCTION

Bank filtration (BF), sometimes also called riverbank filtration (RBF), is used worldwide for drinking water production (Tufenkji *et al.*, 2002; Doussan *et al.*, 1997; Grünheid *et al.*, 2005). It has two main advantages: (1) Sufficient quantity of water can be produced independent of the usable groundwater capacity as BF is a form of artificial groundwater recharge (Bouwer, 2002; Dillon, 2005). (2) Low cost post-treatment is often sufficient for the raw water as the process of bank filtration takes advantage of the natural filter capacity of the sediments during the soil passage (Kuehn & Mueller, 2000). Usually, there is a significant increase in water quality for the bank filtrate compared to the surface water regarding organic substances, colour, coliform bacteria and faecal contaminants (Singh *et al.*, 2010; Weiss *et al.*, 2005). Additionally, the water quality at bank filtration wells is relatively constant, and, therefore, it is easier to treat than surface water, which often shows high variation in many quality parameters (Ray, 2004; Tufenkji *et al.*, 2002).

Typically, bank filtration sites are planned according to the local hydrogeological conditions and the wells are constructed to achieve both of the above mentioned effects (Ray *et al.*, 2002). In developing countries there is usually an emphasis on securing sufficient water quantity for drinking. Wells are often constructed along rivers and lakes because alluvial or riparian aquifers generally have good hydraulic properties (Rosenshein, 1988). The shallow depths of the sediments make them easy to exploit and help to reduce drilling costs (Doussan *et al.*, 1997). However, when bank filtration is applied at sewage contaminated surface waters, which is often the case in developing countries (Ray, 2008), a range of problems can arise as contaminated water infiltrates into the aquifer in large quantities and the capacity of the soil to filter the contaminants is often exceeded (Heberer, 2002). The resulting contamination of the aquifer can prevail for many decades, making post-treatment and/or remediation measures necessary. Such contamination is often caused by nitrogen, especially the species ammonium (Hiscock & Grischek, 2002).

Increasing ammonium concentrations at one specific well (P3) in a well field located at the Yamuna River in East Delhi are a cause for concern as the well field is used for drinking water production. Elevated ammonium concentrations at the well were reported since 2006 (Sprenger & Lorenzen, 2014) and Groeschke (2013) identified the sewage contaminated river water as the main source of ammonium in the raw water. Ammonium concentrations are already about ten times higher (5.5–8 mg/L) than the Indian guideline value of 0.6 mg/L total ammonia [0.5 mg/L total ammonia-N] as specified in BIS 10500:2012 and are expected to increase further. As water suppliers have to plan several decades ahead in order to be able to develop appropriate water management concepts, it is important to know about the development of future ammonium concentrations at that well field and other well fields along the Yamuna River to be able to choose appropriate remediation and treatment options.

Numerous studies of ammonium contaminations in groundwater have been conducted, mostly focusing on contamination from point sources - such as septic tank effluents (Hinkle *et al.*, 2007), leachate from sewage farms (Hamann, 2009), leachate

from coking plants (Haerens *et al.*, 2002a) or chemical companies (Clark *et al.*, 2008) – and from contaminations resulting from the infiltration of treated sewage water (LeBlanc, 1984; Ceazan *et al.*, 1989; Böhlke *et al.*, 2006; DeSimone & Howes, 1996, 1998). A comprehensive review of published literature on ammonium retardation is given by Buss *et al.* (2004). Doussan *et al.* (1997, 1998) studied the transport of nitrogen species at a RBF site at the Seine (France), where the river water was a main source of nitrogen – in the form of nitrate. Reducing conditions prevailed in the aquifer owing to the decay of organic matter and the nitrate was reduced to ammonium during the soil passage, while the mineralization of organic matter was an additional source of ammonium. In central Delhi, reducing conditions in the aquifer are caused by the infiltration of reducing surface water. It is expected that without infiltration of reducing surface water, a redox sequence from oxidizing conditions to iron-reducing conditions further away from the river would prevail – as reported by Lorenzen *et al.* (2010a) for a field site upstream Delhi.

In order to understand the behaviour of ammonium in aquifers at BF sites at surface waters highly polluted by untreated sewage, field data have been collected and laboratory column studies have been conducted with aquifer material from Delhi. Results of the analyses and experiments are summarized in section 4.3 and were used as the basis for recommendations about the application of BF in nitrogen contaminated aquifers.

## 4.2 NITROGEN

### 4.2.1 Occurrence and effects

Nitrogen is a redox-sensitive parameter which can occur in different species. The most common forms of nitrogen in the water-soil environment are, in order of decreasing oxidation state (Metcalf & Eddy Inc, 2014; Stumm & Morgan, 1996):

- Nitrate ( $\text{NO}_3^-$ , +V)
- Nitrite ( $\text{NO}_2^-$ , +III)
- Nitrogen gas ( $\text{N}_2$ , 0)
- Ammonia and ammonium ( $\text{NH}_3$ , and  $\text{NH}_4^+$ , both –III)
- Organic nitrogen (OrgN, mostly –III)

Whether the reduced form of nitrogen occurs as un-ionized ammonia ( $\text{NH}_3$ ) or in the form of ammonium ions ( $\text{NH}_4^+$ ) depends on the temperature and, to a stronger extent, the pH of the solution (Table 4.1).

**Table 4.1** Proportions of ammonium ( $\text{NH}_4^+$ ) and ammonia ( $\text{NH}_3$ ) at different pH values (Metcalf & Eddy Inc, 2014, p.94).

$\text{NH}_3 + \text{H}_2\text{O}$	$\longleftrightarrow$	$\text{NH}_4^+ + \text{HO}^-$	pH	Temperature
10%		90%	8.3	20°C
50%		50%	9.25	20°C

At pH and temperature conditions commonly found in natural waters, ammonium is the principal species (Hem, 2005). Sometimes the term “total ammonia” is being used referring to the sum of ionized and un-ionized ammonia.

Nitrogen pollution can cause problems such as eutrophication of surface water bodies (Howarth & Marino, 2006), which can lead to toxic algal blooms or decreasing dissolved oxygen concentrations and related issues such as a decrease in animal and plant diversity. Furthermore,  $\text{NH}_3$  is toxic for aquatic species (Randall & Tsui, 2002), but not for humans at low concentrations (GESTIS Substance database, 2014). Nitrite ( $\text{NO}_2^-$ ) is also extremely toxic to fish or other aquatic species (Metcalf & Eddy Inc, 2014). For humans, excessive nitrogen intake in the form of nitrate ( $\text{NO}_3^-$ ) or nitrite through water can result in diarrhoea or methaemoglobinemia (blue-baby syndrome) in infants (Ward *et al.*, 2005). When chlorination is used for the disinfection of drinking water, the presence of ammonium in raw water, even at low concentrations, causes the formation of chloramines (Weil & Morris, 1949). Higher chlorine doses are necessary to achieve required minimum residual chlorine concentration at the outlet of the water treatment plants (WTPs) and in the distribution system (Duong *et al.*, 2003).

### 4.2.2 Guideline values

Guideline values for nitrogen species given in the Indian standard (BIS, 10500, 2012) and the WHO drinking water quality guidelines (WHO, 2011) are compared in Table 4.2. The WHO did not establish a guideline value for total ammonia because it usually occurs in drinking water at concentrations well below those of health concern. Because the WHO includes the

non-ionized form  $\text{NH}_3$  and the ionized form  $\text{NH}_4^+$  in their definition of ammonia, it is assumed that this is also the case in the Indian Standard BIS 10500, although it is not further defined.

**Table 4.2** Guideline values for nitrogen species in drinking water.

Parameter	Unit	BIS 10500:2012 (India)	WHO (2011)
Nitrate (as $\text{NO}_3^-$ )	mg/L	45	50
Nitrite (as $\text{NO}_2^-$ )	mg/L	No guideline value	3
Ammonia (as $\text{NH}_3\text{-N}$ and $\text{NH}_4^+\text{-N}$ )*	mg/L	0.05	No guideline value

\*Definition of WHO, not specified in BIS 10500.

### 4.2.3 Nitrogen in surface water bodies

In surface water bodies, nitrogen concentrations depend on several factors, mainly land use, sewage disposal and the water balance of the water body, as dilution strongly affects the pollutant concentrations. In European rivers agricultural fertilizers are the main source of nitrogen.  $\text{NH}_4^+\text{-N}$  concentrations range between 0.1 and 0.3 mg/L and median  $\text{NO}_3\text{-N}$  concentrations are around 3 mg/L in large rivers (EEA, 2001). In urban agglomerations in developing or newly industrialized countries nitrogen loads in surface water bodies are expected to be substantially higher: Nitrogen contamination of surface water through untreated or partially treated domestic sewage water is a concern in many of these countries. According to Corcoran *et al.* (2010), up to 90% of sewage water in developing or newly industrialized countries is not collected or treated but discharged directly into rivers, lakes and coastal areas or leached into the subsoil. In the Asia-Pacific region alone, this amounts to approximately 150–250 million  $\text{m}^3$  per day of untreated (domestic) wastewater from urban areas released to the environment (WWAP, 2012). However, data on total inorganic nitrogen concentrations ( $\text{NH}_4^+$ ,  $\text{NO}_2^-$ ,  $\text{NO}_3^-$ ) is scarce, as most studies on river water quality in Asian megacities only report concentrations for nitrate and nitrite, e.g. Sikder *et al.* (2013) and Kido *et al.* (2009). But because of the high chemical oxygen demand in sewage contaminated rivers, the prevailing form of inorganic nitrogen is expected to be ammonium.

### 4.2.4 Nitrogen in sewage water

In sewage water about 60–70% of the nitrogen is present in the form of ammonia/ammonium, depending on the pH of the solution, while the remaining 30–40% is mostly found as biodegradable or non-biodegradable organic nitrogen (Metcalf & Eddy Inc, 2014, p. 712). Fresh domestic wastewater usually contains no more than 1% nitrates and nitrites (Eckenfelder & Argaman, 1991) before it is aerated in the nitrification step at wastewater treatment plants (WWTPs). Total nitrogen concentrations in wastewater are highly variable depending on the diet of the population (Pescod, 1992; Patterson, 2003) and the per capita wastewater flow rate (Eckenfelder & Argaman, 1991). Concentrations between 20 and 85 mg/L are reported as an average for typical domestic wastewater in Metcalf & Eddy Inc (2014), and maximum concentrations can be much higher (sometimes above 150 mg/L – e.g. Ammary, 2007). The main source of nitrogen in wastewater is urea, which is contained in urine and is degraded by biological hydrolysis (Mobley & Hausinger, 1989, Table 4.3).

**Table 4.3** Reactions of urea in the environment (Mobley and Hausinger, 1989).

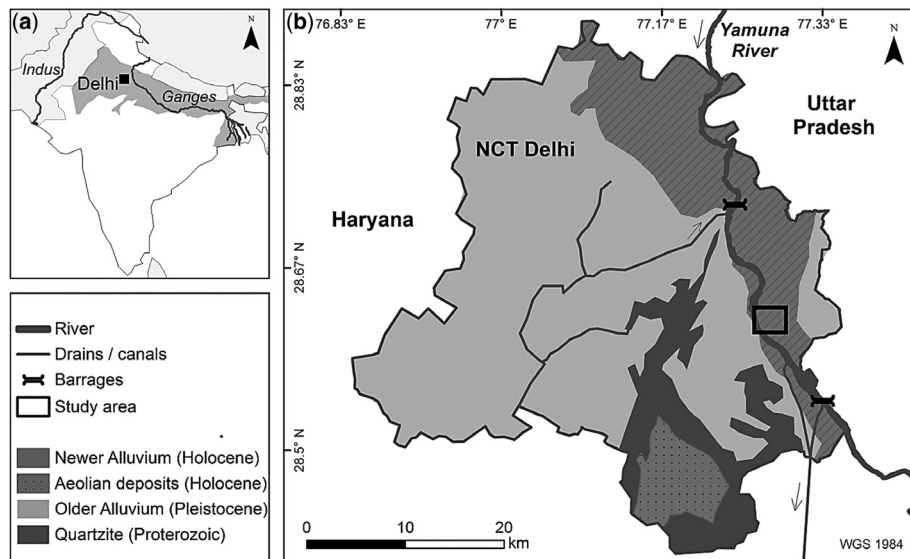
Reactant		Product	Explanation
urea + water		ammonia + carbamate	Hydrolysis of urea
$(\text{NH}_2)_2\text{CO} + \text{H}_2\text{O}$	*→	$\text{NH}_3 + \text{H}_2\text{NCOOH}$	*Naturally occurring enzyme urease catalyzes reaction
carbamate + water		ammonia + carbonic acid	Hydrolysis of carbamate
$\text{H}_2\text{NCOOH} + \text{H}_2\text{O}$	→	$\text{NH}_3 + \text{H}_2\text{CO}_3$	
carbonic acid		hydrogen ion + bicarbonate	Dissociation of carbonic acid
$\text{H}_2\text{CO}_3$	→	$\text{H}^+ + \text{HCO}_3^-$	Increase in pH
ammonia + water		ammonium + hydroxide	
$2\text{NH}_3 + 2\text{H}_2\text{O}$	→	$2\text{NH}_4^+ + 2\text{OH}^-$	Ammonia molecules equilibrate with water

Other sources are faeces (a source of organic nitrogen) and grey water from laundry and personal washing (a source of  $\text{NH}_3/\text{NH}_4^+$ ) (Patterson, 2003). Products like toilet paper had the lowest nitrogen load contribution (Tjandraatmadja *et al.*, 2010). In countries without a regular waste disposal system, garbage disposal in wastewater is another large source of organic nitrogen.

### 4.3 THE DELHI CASE STUDY

#### 4.3.1 Overview

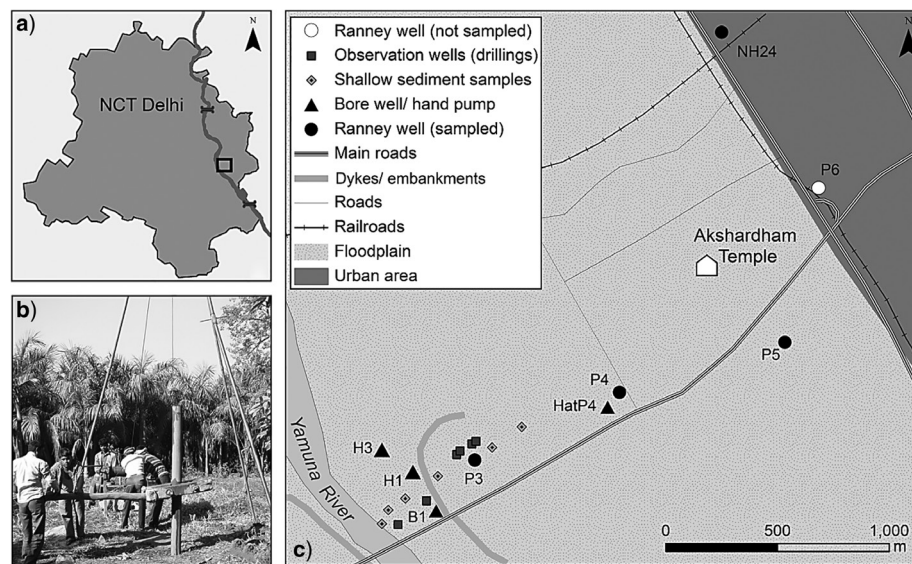
The well fields in East Delhi are an example of unintended contamination by RBF use by placing wells along a sewage contaminated river. Delhi is a megacity with a population of currently 16 million people (Census of India, 2011) located in the centre of the Indo-Gangetic Plain (Figure 4.1). The Yamuna River, the largest tributary of the Ganges River, flows through Delhi in north-southerly direction. As many rivers in Asia, the Yamuna is characterized by high sediment loads during monsoon season (Jha *et al.*, 1988) and a constantly changing riverbed (Khan & Bajpai, 2014). Although numerous dykes and embankments were constructed within the city to control the flow, the river still has the opportunity to shift within certain limits between the embankments (Figure 4.2). Thus, the location of the riverbank frequently changes. The river is dammed up by two barrages within the city area, Wazirabad barrage in the North and Okhla barrage in the South, and the 22 km river stretch between the two barrages is highly polluted by sewage water.



**Figure 4.1** (a) Delhi is located in the Indo-Gangetic plain (Indian part marked in medium grey). Data source: Natural Earth (2011) (b) The study area is located on the East bank of the Yamuna River, where sewage influenced river water infiltrates into the sediments of the Newer Alluvium. Geological Map modified after Geological Survey of India (2006).

The floodplain covers the eastern and western bank along the entire stretch of the river in Delhi. It is mostly undeveloped and mainly used for agriculture. The floodplain sediments are mostly medium grained sands with a layer thickness of up to 70 m in the north of Delhi (Shekhar & Prasad, 2009) and about 20 m in the southern part of the city. They constitute what is known as the floodplain aquifer or Newer Alluvium. Compared to other groundwater sources in Delhi, the groundwater of the Newer Alluvium is found at shallow depth and is characterized by only minor water table fluctuations of about 0–2 m throughout the year and over decades – as opposed to about 4–20 m in other aquifers in Delhi (CGWB, 2012; Shekhar *et al.*, 2009).

Numerous tube wells and about 20 Ranney wells (radial collector wells) were constructed on the floodplain in Delhi, tapping the Newer Alluvium. The wells are not arranged parallel to the riverbank but were constructed across the complete width of the upper floodplain. Owing to losing stream conditions (Lorenzen *et al.*, 2010a) it can be assumed that the wells situated along the river draw a high share of bank filtrate, although they have not been specifically designed for bank filtration.



**Figure 4.2** (a) Location of the study area (b) Drilling of observation well (c) Location of the hand pumps, Ranney wells and of the shallow and deep drillings conducted during the field work (modified after Groeschke, 2013). The river is able to shift between the dykes on the eastern and the western bank.

### 4.3.2 Study area

The study area covers an area of about 2.5 km<sup>2</sup> on the east bank of the Yamuna River in East Delhi near the Akshardham Temple between 720800 and 722800 m E and 3165900 and 3168200 m N (UTM, WGS84 Zone 43 N). Here the undeveloped floodplain is about 2.4 km wide. Four Ranney wells operated by the Delhi water company Delhi Jal Board (DJB – P3, P4, P5, P6) and one Ranney well operated by the Indian Railways (NH24) as well as numerous hand pumps and bore wells used by the local population are located within the area and some of them were used for water sampling. Additionally, four observations wells and two hand pumps were installed in the frame of Saph Pani (see section 4.3.3). The study site with the location and type of the sampling points are shown in Figure 4.2.

#### Previous research

Previous research at this location was conducted 2006 in the frame of the feasibility study IDB India (International Development of Bank Filtration: Case Study India) and 2007–10 in the frame of the TECHNEAU project (Lorenzen *et al.*, 2007; Pekdeger *et al.*, 2008; Sprenger *et al.*, 2008; Lorenzen *et al.*, 2010a; Lorenzen *et al.*, 2010b; Lorenzen, 2011; Sprenger, 2011; Sprenger & Lorenzen, 2014).

#### Geology and hydrogeology

In the study area, like in the entire flood plain in Delhi, the Holocene alluvial sands (Newer Alluvium) are underlain by finer grained Pleistocene sediments of the Older Alluvium. At 38 mbgl Precambrian bedrock was encountered (Sprenger, 2011, p.70). Hydraulic conductivities of the Newer Alluvium are in the range of  $2 \times 10^{-4}$  to  $7 \times 10^{-4}$  m/s (Chatterjee *et al.*, 2009) and Sprenger (2011, p.66) reported an average pore water velocity of 0.9 m/d for this unit at the field site. The hydraulic conductivities of the Old Alluvium are between  $3 \times 10^{-5}$  and  $5 \times 10^{-5}$  m/s (Chatterjee *et al.*, 2009). According to Lorenzen *et al.* (2010a) losing stream conditions prevail on the east bank of the river. Sprenger (2011, p.66) reported infiltration rates of  $6.4 \times 10^{-7}$  m<sup>3</sup>/m<sup>2</sup>/s for monsoon times and  $4.2 \times 10^{-7}$  m<sup>3</sup>/m<sup>2</sup>/s for non-monsoon times. No information is available about the west bank of the river at this location.

#### Description of the production wells

The Ranney wells of the Delhi Jal Board in the study area were constructed in 1973 and commenced operation in 1975. They are about 15 m deep and thus tap the floodplain aquifer. Each well has ten laterals which are each about 30 m long. The



recorded discharge of the wells is about 150–300 m<sup>3</sup>/h (Chatterjee *et al.*, 2009). The wells are typically operated about eight hours every day, except for well number P3, which is sometimes not operating because of elevated ammonium concentrations in the groundwater. The water of well P4 is directed to the Commonwealth Games Village WTP while the water of wells P3, P5, P6 is supplied to the Okhla WTP. Well NH24 of the Indian Railways is constructed similar to the DJB Ranney wells but the water is not used for public water supply.

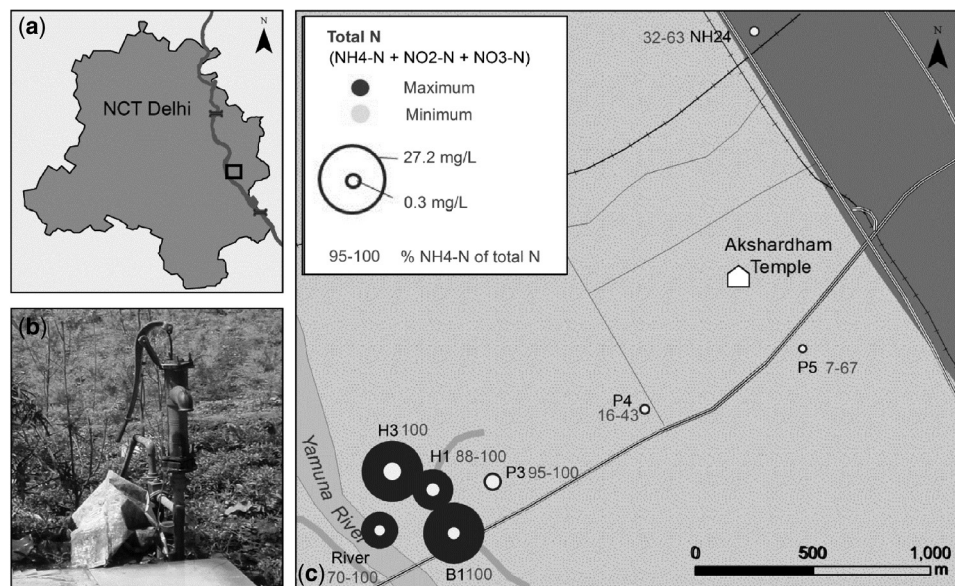
### 4.3.3 Field studies

#### *Water and sediment sampling*

In the frame of Saph Pani, 72 groundwater samples were taken from the sampling points shown in Figure 4.2, including the newly constructed observations wells and hand pumps. In addition, eleven regular river water samples and two samples of the flood event in July 2013 were taken and analyzed (for NO<sub>3</sub><sup>-</sup>, NO<sub>2</sub><sup>-</sup>, NH<sub>4</sub><sup>+</sup>, main cations and anions, pH, oxidation reduction potential, electrical conductivity, dissolved oxygen, trace elements). The four new observation wells were installed at distances between 500–550 m to the riverbank and the two new hand pumps were installed at distances of 35 m and 250 m to the river. The drillings were done by manual auger drilling and have depths between 7.6 and 28 m. Sediment samples were collected of the encountered lithological units. In addition, sediment samples were collected from seven shallow drillings with depths between 2.5 and 4.3 m at distances of 5 m, 75 m, 200 m, 375 m, 500 m, 600 m, and 775 m to the river. This was conducted by using an Eijkelkamp hand drilling device. To sample river bottom sediments and to measure water depths, three profiles were taken across the Yamuna River in December 2013, using a Van-Veen grab sampler.

#### *Results: Ammonium concentrations at the field site*

Varying ammonium concentrations were found in the aquifer close to the river (Figure 4.3). In 2012, a similar trend was observed in ammonium concentrations at the three sampling points B1, H1, and H3, with values between 4.5 mg/L in June 2012 and 26 mg/L in December 2012 (Groeschke, 2013). In 2013, ammonium concentrations still fluctuated (between 6.4 and 35 mg/L), but no trend could be discerned. In the Ranney well P3 at a distance of 500 m from the river, ammonium concentrations varied between 5.5 and 8 mg/L in 2012 and 2013. In wells farther away from the riverbank, ammonium concentrations remained below 1.7 mg/L in both years. In the river water, ammonium concentrations up to 20 mg/L were measured in 2012 and up to 16 mg/L during the field campaigns in 2013. Maximum and minimum concentrations at the field site are summarized in Figure 4.3. Data obtained from water samples were further used to set-up the 1D model (section 4.3.5). A detailed description of the ammonium plume is given in Groeschke *et al.* (2015a).



**Figure 4.3** (a) Location of the study area (b) Hand pump H1 (c) Minimum and maximum total nitrogen concentrations in the water samples taken between March 2012 and December 2013 work (modified after Groeschke, 2013).

#### 4.3.4 Laboratory studies

##### *Sediment analyses*

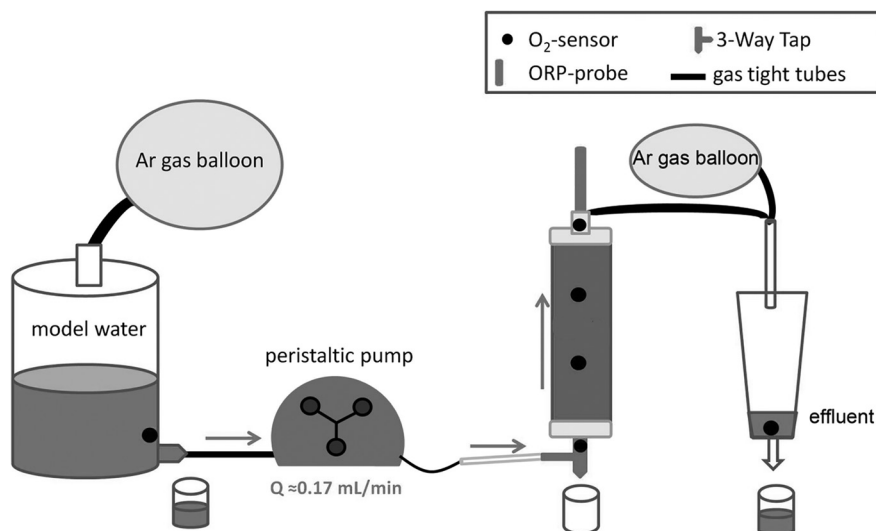
To characterize the alluvial aquifer a total of 25 sediment samples from the unsaturated zone were studied (Groeschke *et al.*, 2015b) and 14 samples from the saturated zone as well as one sediment sample from the river bed were analysed (Groeschke *et al.*, 2015a). Grain size distribution was determined by sieve test and hydrometer method, organic content and carbonate content were measured through loss on ignition. The cation-exchange capacity (CEC) was determined using a barium chloride ( $\text{BaCl}_2$ ) percolation method for sands and gravels and a  $\text{BaCl}_2$  batch method for fine grained sediments.

The sediment's main components range between silt and gravel. The particle sizes increase with increasing sampling depths. Silts and silty fine sands are predominant in the unsaturated zone, while the saturated zone is dominated by well sorted medium sands. Below is a gravelly layer at a depth of about 13.5 m, which consists of real gravel particles (grain size of  $>2$  mm) and of concretions  $>2$  mm made up of clay and silt. It is underlain by silty clays, presumably of the Old Alluvium, at a depth of about 16 m. In the New Alluvium, the hydraulic conductivity ( $k$  value) ranges between  $1.0 \times 10^{-7}$  m/s in the unsaturated zone and  $4.2 \times 10^{-3}$  m/s in the saturated zone if calculated according to Beyer (1964) or – in case of the fine grained sediments – to the U.S. Bureau of Soil Classification (Richter, 1966). The hydraulic conductivities increase with increasing particle size (and thus with depth). The organic content of the sediments ranges between 0.5% and 14.9%. In the unsaturated zone, the organic content was generally higher than in the saturated zone. The carbonate content of the sediments ranges between 0.9% and 18%. The highest carbonate content was found in the gravel layer, which contains the concretions. The CEC ranges between 1.2 meq/100 g sediment in the saturated zone and 37.2 meq/100 g sediment in the unsaturated zone. In the saturated zone, the CEC is slightly higher in the gravelly layer (2.1 meq/100 g sediment) than in the sand (1.6 meq/100 g sediment, Groeschke *et al.*, 2015c). Calcium has the highest share in the CEC, leading to the conclusion that the carbonates consist mainly of calcium carbonate and that the concretions are probably the typical calcite concretions locally known as kankar (Eybing, 2014). The gravel layer will be referred to as kankar in the following text.

A detailed description of the unsaturated zone and an evaluation of its significance for the ammonium contamination is given by Groeschke *et al.* (2015b). The saturated zone, and especially the kankar layer and its significance for flow and transport are described in detail in Groeschke *et al.* (2015a; and 2015c).

##### *Column experiments*

The transport and fate of ammonium in the sand and kankar aquifer materials from the Yamuna floodplain was further investigated in laboratory column experiments at Freie Universität Berlin (Groeschke *et al.*, 2015b; and 2015c). Such experiments are a common method in hydrogeology to determine specific sediment parameters. The goal of these series of experiments was to provide data regarding sorption, degradation and fixation of ammonium under field site conditions. The data was later used to set up a reactive transport model of the field site to predict the future development of the ammonium plume. The set-up of the experiments is shown in Figure 4.4.



**Figure 4.4** Set-up of the column experiments.

The experiments were conducted under suboxic or anoxic conditions, whereby the latter prevail in the aquifer. To achieve laboratory conditions similar to those at the study site, the following parameters were adapted:

- The model water was adjusted for the main cations to be comparable to the concentrations in the groundwater.
- Anoxic conditions were established by using argon balloons to create an oxygen- (and nitrogen-) free atmosphere above the model water container and the effluent sampling flasks. Glass and gas-tight tubing materials were used to inhibit gas exchange. The oxygen concentration was monitored with chemical optical oxygen sensor spots (PreSens) at six points during the entire duration of the experiments. Redox potential was measured using oxidation reduction potential -probes at the outlet of the columns.
- A peristaltic pump maintained a flow rate of  $\sim 0.17$  mL/min, which correlates to a flow velocity at the field site of about 0.9 m/d (Sprenger, 2011, p.66).

The glass columns had an inner diameter of 45 mm and a sediment filled length of 146 mm and were flushed upflow with the model water. Three sets of experiments were conducted with this set-up: pre-tests, adsorption experiments and desorption experiments.

- During pre-tests, the freshly filled columns were flushed with nitrogen-free model water until nitrogen concentrations in the effluent were sufficiently low.
- In the adsorption experiments, the columns were flushed with model water with ammonium concentrations of either 20 mg/L or 10 mg/L until the ammonium concentrations in the effluent were equal to the concentrations in the model water.
- During the subsequent desorption experiments, the columns were again flushed with nitrogen-free model water until nitrogen concentrations in the effluent were low and did not decrease any further.

Each experiment was conducted with two or three columns filled with the same sediment (doubles or triplets). To check for reproducibility, most experiments except for the pre-tests were repeated one or two times.

### *Results of the column experiments*

The column experiments indicate that there is some degradation or fixation of ammonium in the sediments of the unsaturated zone and no or very little natural degradation potential in the sediments of the saturated zone. The transport of ammonium is therefore mainly controlled by cation exchange. In the sand columns, 10–12 pore volumes were necessary to observe ammonium saturation in the sediment and subsequently same ammonium concentrations in the column effluent as in the feed water (100% breakthrough of ammonium) and about 15 pore volumes to flush the ammonium out of the sediment. In the kankar, 30–35 pore volumes were necessary to observe the 100% breakthrough in the adsorption experiments and the flushing of the ammonium in the desorption experiments took about 40 pore volumes (Groeschke *et al.*, 2015c).

### **4.3.5 1D Transport modelling**

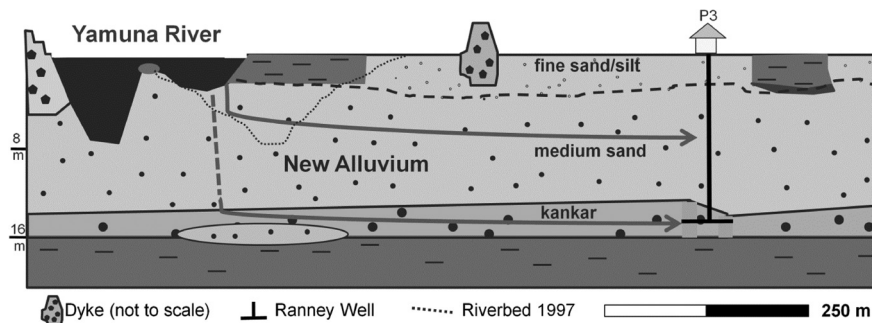
With the aim to predict the future concentrations of ammonium at well P3, 1D reactive transport models (Haerens *et al.*, 2002b) were set up for the field site. Based on the results of the 1D reactive transport column models, of Groeschke *et al.* (2015c), two flow paths in representative aquifer sediments were modelled with PHREEQC v3 (Parkhurst & Appelo, 2013; Figure 4.5). One flow path comprises 500 m distance from the riverbank to Ranney well P3. These 500 m were set up as a column divided into 139 cells with a cell length of 3.6 m each. The time step was set to 4 d, resulting in the average linear velocity of 0.9 m/d as determined by Sprenger (2011). Transport parameters (effective porosities, number of exchange sites, and selectivity coefficients for the cation exchange) were taken from the 1D column modelling without any further adjustments (Table 4.4). Dispersivities were adjusted to the model length. Although dispersion is generally higher at the field scale than at the laboratory scale because of sediment inhomogeneities which are not present in laboratory columns (Gelhar *et al.*, 1992), the dispersivities of the field model were adjusted to represent the magnitude measured in the column experiments: the longitudinal dispersivity was set to 5 m (1/100 of the flow path) in the sand and to 50 m (1/10 of the flow path) in the kankar. Because the sediment is carbonatic (Eybing, 2014) and most water samples at the field site are slightly oversaturated with calcite, calcite was included as an equilibrium phase in the model. To check for numerical errors, the models were also run with 278 cells (1.8 m cell lengths) and 2 d time steps and with 556 cells (0.9 m cell lengths) and 1 d time steps.



**Table 4.4** Transport parameters used in the simplified 1D model.

Parameter	Unit	Sand	Kankar
Effective Porosity ( $n_e$ )*	—	0.24	0.175
Number of exchange sites	meq/1L water	0.054	0.21
$\log_{10} K_{Na/K}$	—	0.67	0.98
$\log_{10} K_{Na/Ca}$	—	0.1	0.18
$\log_{10} K_{Na/Mg}$	—	-0.28	-0.09
$\log_{10} K_{Na/NH_4}$	—	0.55	0.81

\*Effective porosities are not explicitly included in PHREEQC models. They are incorporated through the number of exchange sites.



**Figure 4.5** Flow paths from the river to well P3 in the 1D reactive transport models. The vertical flow from the river to the kankar layer was not considered and modelled. Cross section after geological information given in Groeschke *et al.* (2015a).

To keep the model minimal and straightforward, the following assumptions and simplifications were applied:

- Source water composition (displacing solution) was kept constant, although in reality there is a seasonal variability in the river water due to monsoon–non monsoon compositions.
- Ammonium was decoupled from the nitrogen cycle, meaning it cannot be oxidized to nitrate in the model. This would be representative of anoxic conditions in the aquifer, which by no means must prevail after an improvement of source water quality.
- An average linear flow velocity of 0.9 m/d (Sprenger, 2011) was assumed for both flow paths. It is very likely that flow velocities are much higher in the kankar layer, but real data for this layer are not available.

### Adsorption modelling

To estimate the increase of ammonium concentrations at well P3, the infiltration of sewage influenced river water into the aquifer was modelled. The cells were equilibrated with water samples taken at sampling points still uninfluenced by the ammonium plume. A sample taken from HatP4 in December was used for equilibrating the sand layer and a sample taken at P4 in December 2013 was used to equilibrate the kankar layer. The cells were then flushed with a displacing solution with the composition of a sewage influenced river water sample taken at the field site in December 2012 with an ammonium concentration of 20 mg/L. The compositions of the water samples are summarized in Table 4.5. In the models, it took about 15 years to reach the 100% ammonium breakthrough in the sand layer and 62 years to reach the 100% ammonium breakthrough in the kankar layer (Figure 4.6).

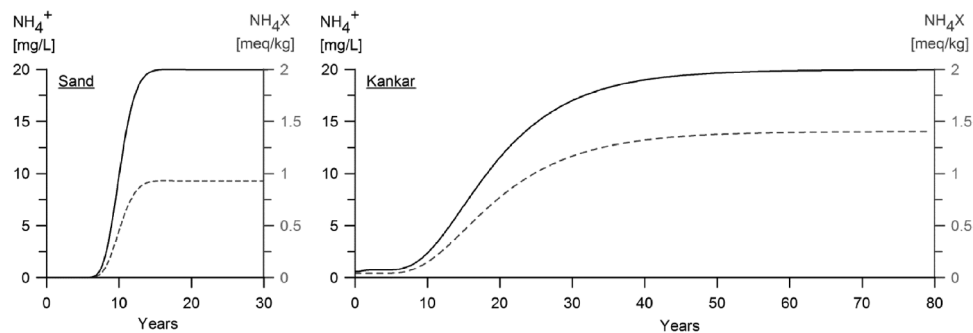
### Desorption modelling

How long the ammonium contamination will prevail in the aquifer after source water quality improves depends not only on the sediment properties, but also on the extent of the ammonium plume. To model ammonium desorption, it was assumed that

- the ammonium plume has completely reached the well P3 and
- the ammonium distribution within the plume is homogenous at 35 mg/L  $NH_4^+$  in the groundwater in the sand layer and 26 mg/L in the kankar layer.

**Table 4.5** Composition of the equilibrating and displacing solutions. Water samples were charge-balanced with alkalinity as  $\text{HCO}_3^-$ . Groundwater samples were taken in December 2013. River water sample was taken in December 2012.

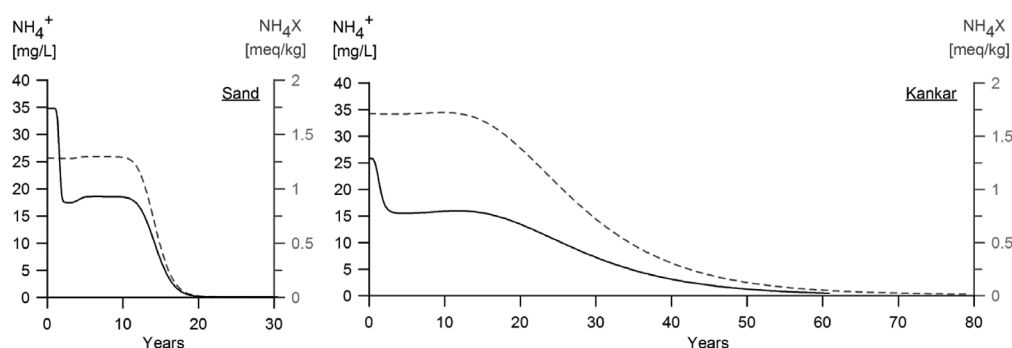
Parameter	Unit	Equilibrating Solution Sand (HatP4)	Equilibrating Solution Kankar (P4)	Displacing Solution (River Water Field Site)
Temperature (T)	°C	26.2	26.4	20.5
pH	pH	7.58	7.4	7.6
Reduction potential ( $E_h$ )	mV	160	175	82
Electrical conductivity (EC)	$\mu\text{S/cm}$	495	893	1588
Sodium (Na)	mg/L	19.9	67.5	171
Potassium (K)	mg/L	5.4	6.8	15.4
Magnesium (Mg)	mg/L	14	23	33.7
Calcium (Ca)	mg/L	63.7	80	65.4
Iron (Fe)	mg/L	0.09	0.1	0.07
Manganese (Mn)	mg/L	0.09	0.3	0.3
Hydrogencarbonate ( $\text{HCO}_3^-$ )	mmol/L	5.2	5.9	6.5
Chloride (Cl)	mg/L	6	78	218
Sulphate ( $\text{SO}_4^{2-}$ )	mg/L	2	53	125
Sulfide ( $\text{S}^{2-}$ )	mg/L	0	0	0
Ammonium ( $\text{NH}_4^+$ )	mg/L	0	0.6	20
Nitrite ( $\text{NO}_2^-$ )	mg/L	0.005	0.03	0.02
Nitrate ( $\text{NO}_3^-$ )	mg/L	0	3.5	0

**Figure 4.6** Results of adsorption modelling. Solid black line: Ammonium concentration in the water in mg/L (shown on the primary y-axis), dashed grey line: Ammonium concentrations on the exchanger in meq/kg sediment (shown on the secondary y-axis). The amount of ammonium sorbed on the sediment depends on the number of exchange sites available, on the solute composition and on the selectivity coefficients, which are sediment-dependent. It can be clearly seen that more ammonium can be adsorbed on the kankar material than on the sand.

The cells of the sand flow path were equilibrated with a water sample from hand pump B1 taken in December 2013 and the cells of the kankar flow path were equilibrated with water composition of sample H250 taken in December 2013 (Table 4.6). After equilibration, the column was flushed with a displacing solution with the composition of the river water upstream Delhi at Palla, where the Yamuna is still uninfluenced by sewage water. Assuming the same average linear velocity of 0.9 m/s in the sand and the kankar, Ammonium concentrations were below the drinking water limit value of 0.5 mg/L after about 19 years in the sand layer and after about 61 years in the kankar layer (Figure 4.7). This is due to the higher number of exchange sites in the kankar and to different selectivity coefficients in both materials. Because degradation of ammonium was not implemented in the models, the results can only be seen as conservative estimates. Furthermore, the average linear velocity in the kankar is probably higher than in the sand and flushing the ammonium out of the kankar layer might therefore be faster than 61 years.

**Table 4.6** Composition of the equilibrating and displacing solutions. Water samples were charge-balanced with alkalinity as  $\text{HCO}_3^-$ . Groundwater samples were taken in December 2013. River water sample was taken in March 2007 in the frame of the TECHNEAU project.

Parameter	Unit	Equilibrating Solution Sand (B1)	Equilibrating Solution Kankar (H250)	Displacing Solution (River Water Upstream)
Temperature (T)	°C	25.2	24.3	22.3
pH	pH	6.93	7.23	8.56
Reduction potential ( $E_h$ )	mV	105	84	268
Electrical conductivity (EC)	$\mu\text{S/cm}$	1615	1153	457
Sodium (Na)	mg/L	97	79.7	35
Potassium (K)	mg/L	17.3	13.2	9
Magnesium (Mg)	mg/L	38.7	24.8	14
Calcium (Ca)	mg/L	126.5	89.1	44
Iron (Fe)	mg/L	16.9	5.2	0.62
Manganese (Mn)	mg/L	0.42	0.27	0.05
Hydrogencarbonate ( $\text{HCO}_3^-$ )	mmol/L	11.9	8.3	2.7
Chloride (Cl)	mg/L	141	115	38
Sulphate ( $\text{SO}_4^{2-}$ )	mg/L	5	4	46
Sulfide ( $\text{S}^{2-}$ )	mg/L	0.04	0	0
Ammonium ( $\text{NH}_4^+$ )	mg/L	35	26	0.1
Nitrite ( $\text{NO}_2^-$ )	mg/L	0.005	0.005	0.2
Nitrate ( $\text{NO}_3^-$ )	mg/L	0	0.05	6



**Figure 4.7** Results of the desorption modelling. Solid black line: Ammonium concentration in water in mg/L (shown on the primary y-axis), dashed grey line: Ammonium concentrations on the exchanger in meq/kg sediment (shown on the secondary y-axis). Like in the adsorption-model runs, more ammonium is adsorbed on the kankar material than on the sand. However, the actual amounts adsorbed in the beginning of the desorption-model run are higher than the amounts adsorbed in the adsorption-models (Figure 4.6). The reason for this discrepancy is that the desorption-model was equilibrated with higher concentrations of ammonium (see Table 4.6).

#### 4.4 OVERVIEW OF REMEDIATION AND POST-TREATMENT OPTIONS

To make ammonium-contaminated groundwater suitable for drinking purposes, either remediation measures have to be applied to lower ammonium concentrations in the groundwater, or the raw water has to be treated before distribution. For remediation, there are generally three different approaches: (1) source control, meaning the removal or control of all known and suspected sources of contamination, (2) in-situ treatment methods, where the remediation takes place in the aquifer itself, or (3) pump and treat measures, which involve treating the water above-ground and re-injecting it into the aquifer. In Table 4.7 a short overview of possible remediation measures is given and the use of the measures in the Indian context is briefly discussed. Post-treatment options, which include physico-chemical treatment options (ammonia stripping, ion exchange, breakpoint chlorination and reverse osmosis) and biological filters, are summarized in Table 4.8 together with a brief assessment of the advantages and disadvantages regarding the application in India.

Table 4.7 Overview of remediation options for aquifers contaminated with ammonium ( $\text{NH}_4^+$ ).

Method	Principle	Advantages	Disadvantages	Use in Delhi
<b>Source Control</b>				
<b>Improvement of source water quality<sup>1</sup></b>	Stopping discharge of untreated sewage into rivers would allow for infiltration of aerated, $\text{NH}_4^+$ -free water into the aquifer.	<ul style="list-style-type: none"> <li>– Might decrease arsenic concentrations in bank filtrate</li> <li>– Decrease in odour pollution, improvement of environment</li> </ul>	<ul style="list-style-type: none"> <li>– Long-term solution: WWTPs need to be planned and built</li> <li>– <math>\text{NH}_4^+</math> contamination will prevail decades after river quality improves</li> </ul>	<ul style="list-style-type: none"> <li>– Likely to change redox conditions in aquifer (arsenic demobilization)</li> <li>– Indian guideline limits for WWTP effluents<sup>1</sup> permit high nitrogen discharge into rivers</li> </ul>
<b>Aeration basins</b>	Construction of aeration basins along riverbanks would lead to infiltration of $\text{NH}_4^+$ -free water.	<ul style="list-style-type: none"> <li>– Short-term improvement of river water quality</li> <li>– Low-tech</li> </ul>	<ul style="list-style-type: none"> <li>– Sufficient distribution of <math>\text{O}_2</math> is difficult</li> <li>– Might not be suitable for variable flow</li> <li>– Clogging of diffusers might occur</li> </ul>	<ul style="list-style-type: none"> <li>– High COD and BOD in Yamuna in Delhi</li> <li>– Shifting riverbed: adjustment of facilities</li> <li>– Bacterial contamination: safety measures</li> </ul>
<b>In-Situ</b>				
<b>Oxygen injection – Bioxwand<sup>*2</sup></b>	In-situ bioremediation through oxygen gas injections into the aquifer using lances.	<ul style="list-style-type: none"> <li>– No addition of further chemicals. Suitable for drinking water protection zones.</li> </ul>	<ul style="list-style-type: none"> <li>– <math>\text{O}_2</math> consumed by oxidation of <math>\text{Fe}^{2+}</math>, <math>\text{Mn}^{2+}</math> and organic material present in the aquifer.</li> <li>– Increase in <math>\text{SO}_4</math> and hardness</li> </ul>	<ul style="list-style-type: none"> <li>– High GW temperatures: fast nitrification process</li> <li>– Post-treatment for increased hardness of water at the WTPs might be necessary</li> </ul>
<b>Sequential permeable reactive barrier using polymer mats<sup>3</sup></b>	Use of in-situ polymer mats. Up-gradient mat delivers $\text{O}_2$ to induce nitrification as groundwater moves past. Down-gradient mat delivers ethanol to induce denitrification.	<ul style="list-style-type: none"> <li>– 90% reduction in total N in field experiments</li> <li>– In-situ remediation over short time frame or GW flow distance.</li> <li>– Cost effective</li> </ul>	<ul style="list-style-type: none"> <li>– Measure changes local groundwater flow regimes.</li> <li>– Competing chemicals might exist in the aquifer.</li> <li>– Limited studies available.</li> </ul>	<ul style="list-style-type: none"> <li>– Thinkable to construct shield around wells with high <math>\text{NH}_4^+</math> concentrations</li> <li>– Mats sensitive to fluctuating GW tables (private wells cause drawdown zones)</li> </ul>



<b>Groundwater circulation wells (GCW) and virtually permeable reactive barrier<sup>*4</sup></b>	GCWs induce circulating flow. Increasing pH in flow cell converts $\text{NH}_4^+$ to $\text{NH}_3$ ; removal by negative pressure air stripping. Down gradient further GCWs induce aerobic and anaerobic flow cells.	<ul style="list-style-type: none"> <li>Less negative impact on land use compared with pump and treat methods<sup>5</sup></li> <li>Simultaneous treatment of unsaturated zone and capillary fringe by vapour extraction</li> </ul>	<ul style="list-style-type: none"> <li>Anisotropy of aquifer must be within range that allows circulation cells to develop<sup>6</sup></li> <li>Limited effectiveness in shallow aquifers<sup>6</sup></li> <li>Wells may become clogged<sup>6</sup>.</li> </ul>	<ul style="list-style-type: none"> <li>20 mg/L is target concentration for the negative pressure air stripping. Peak <math>\text{NH}_4^+</math> concentrations in plume in Delhi are 20–35 mg/L: Sufficient to install the VPR barrier only. This requires the installations of numerous GCWs.</li> </ul>
<b>Pump and Treat IVEY-sol aided sorbital filtration method<sup>*7</sup></b>	Installation of injection and abstraction wells across plume. Injection of Ivey-sol surfactant to desorb $\text{NH}_4^+$ from soil. Water is abstracted and treated above ground using ion exchange. Treated water is re-injected.	<ul style="list-style-type: none"> <li>&gt;96% reduction in the dissolved <math>\text{NH}_4^+</math> at the subject site.</li> <li>Addition of Ivey sol enhances bioremediation by 40 to 60%. Ivey sol does not negatively affect water treatment systems.</li> <li>Works well in fine-grained soils</li> </ul>	<ul style="list-style-type: none"> <li>High operation and maintenance costs</li> <li>Complex operation and maintenance</li> <li>Injection of additives into the aquifer</li> </ul>	<ul style="list-style-type: none"> <li>Land use would have to be changed in order to install 5-spot-pattern of wells across the ammonium plume.</li> <li>Injection of chemicals problematic in drinking water protection zone</li> </ul>
<b>Nitrification and denitrification remediation<sup>*8</sup></b>	To trigger in-situ nitrification–denitrification reactions, GW is extracted, mixed with $\text{O}_2$ + nutrients or carbon + nutrients and re-injected. Separate injection and abstraction wells needed for nitrification and denitrification.	<ul style="list-style-type: none"> <li>Especially applicable in sand and sandy gravel aquifers</li> <li>In aquifers with high hydraulic conductivities reaction cell size of injection wells can be maximized and number of injection wells minimized</li> </ul>	<ul style="list-style-type: none"> <li>Not suitable for low hydraulic conductivities</li> <li>Four drillings required in one line parallel to GW flow to complete nitrification and denitrification</li> <li>High maintenance costs and space requirement</li> </ul>	<ul style="list-style-type: none"> <li>Method is costly and requires much space and thus might conflict with agricultural land use</li> <li>Method would involve two extraction-injection cycles</li> </ul>

<sup>\*4</sup>Patented technology; COD = chemical oxygen demand; BOD = biological oxygen demand.

<sup>1</sup> Government of India, 1986; <sup>2</sup>BWB, 2007; Horner *et al.*, 2009; <sup>3</sup>Patterson *et al.*, 2002, 2004; <sup>4</sup>IEG Technology, 2008; <sup>5</sup>Elmore & Graff, 2002; <sup>6</sup>OST, 2002; <sup>7</sup>IVEY, 2012a, 2012b;

<sup>8</sup>Mallath & Chu, 2005

Table 4.8 Overview of post-treatment options for raw water with elevated ammonium ( $\text{NH}_4^+$ ) concentrations.

Method	Principle	Advantages	Disadvantages	Use in Delhi
<b>Physico-Chemical</b>				
<b>Air stripping of ammonia</b>	Adding lime increases pH of water to $\sim 11$ ; $\text{NH}_4^+$ converts to $\text{NH}_3$ ; in contact with air, a gradient exists across the gas/liquid interface and $\text{NH}_3$ will be stripped to the air in a stripping tower. <sup>2</sup> Applied mostly in wastewater treatment. <sup>3</sup>	<ul style="list-style-type: none"> <li>– Increase in pH leads to killing of some pathogens and micro-organisms.<sup>4</sup></li> </ul>	<ul style="list-style-type: none"> <li>– To make water suitable for drinking purposes re-carbonation would have to be adopted.<sup>4</sup></li> <li>– A volumetric air: water ratio of about 3000:1 is required to achieve effective <math>\text{NH}_3</math> removal.<sup>1</sup></li> <li>– Space for stripping towers.</li> </ul>	<ul style="list-style-type: none"> <li>– Suitable for Indian climate: high temperatures increase efficiency.<sup>2</sup></li> <li>– Neutralization before distribution is necessary and WTP effluent has to be tightly monitored.</li> </ul>
<b>Ion exchange</b>	Zeolites (e.g. Clinoptilolite and Chabazite) are selective for $\text{NH}_4^+$ ions. Water is passed through a bed of zeolites to achieve 86–99% ammonium removal, depending on concentration and water composition. Method successful for up to 200 mg/L $\text{NH}_4^+$ . <sup>3</sup>	<ul style="list-style-type: none"> <li>– Zeolites can be regenerated using <math>\text{NaCl}</math><sup>5</sup> or biological regeneration.<sup>6</sup></li> <li>– Method has no sensitivity to fluctuation in <math>\text{NH}_4^+</math> influent concentration.<sup>6</sup></li> </ul>	<ul style="list-style-type: none"> <li>– Inconvenient for WTPs with a capacity <math>&gt; 80,000 \text{ m}^3/\text{d}</math> because of space required for ion exchange columns.<sup>7</sup></li> <li>– Presence of Ca reduces <math>\text{NH}_4^+</math> adsorption onto the zeolite.<sup>8</sup></li> <li>– High input concentrations require large volumes of zeolites.<sup>9</sup></li> </ul>	<ul style="list-style-type: none"> <li>– WTPs in Delhi have a capacity <math>&gt; 180,000 \text{ m}^3/\text{d}</math><sup>10</sup>. Therefore the method is not suitable to retrofit into existing WTPs. Could be used for treating <math>\text{NH}_4^+</math> contaminated GW from Ranney wells by clustering a few wells and installing a small treatment facility for the <math>\text{NH}_4^+</math> treatment prior to conventional treatment.</li> </ul>
<b>Breakpoint chlorination</b>	By adding chlorine to water, a stepwise reaction takes place with the $\text{NH}_4^+\text{-N}$ , first forming mono- and di-chloramines and then, at the breakpoint, $\text{N}_2$ , $\text{NO}_3^-$ and free residual chlorine. Method best suitable for $\text{NH}_4^+$ $< 1 \text{ mg/L}$ <sup>1</sup> .	<ul style="list-style-type: none"> <li>– The mono-chlor-amine and di-chloramine formed, act as a potential disinfectant.<sup>11</sup></li> <li>– Can be combined with As removal.<sup>12</sup></li> </ul>	<ul style="list-style-type: none"> <li>– Cl reacts with organic material and by-products are formed. Activated carbon adsorber needs to be installed too, making treatment expensive.<sup>12,13</sup></li> <li>– Requires frequent monitoring of <math>\text{NH}_4^+</math> and chlorine concentrations.<sup>3</sup></li> <li>– High ammonia-N:chlorine ratios.<sup>14</sup></li> </ul>	<ul style="list-style-type: none"> <li>– Frequent variations in raw water quality: chlorine dosage has to be continuously adjusted to reach breakpoint.</li> <li>– Treatment presumably has to be followed by carbon adsorbers because of organic compounds in the raw water.</li> <li>– <math>\text{NH}_4^+</math> in raw water often <math>&gt; 1 \text{ mg/L}</math>.</li> </ul>

### Reverse Osmosis (RO)

- |   |   |  |  |
|---|---|--|--|
| <p>Water is forced across a semi permeable membrane and molecules and ions, (<math>\text{NH}_4^+</math>), are retained.<sup>3</sup> 94% <math>\text{NH}_3</math> removal in full scale tests with feed concentrations of 33 mg/L.<sup>15</sup> In other studies &gt;98%<sup>16</sup> and &gt;96%.<sup>7</sup></p> | <ul style="list-style-type: none"> <li>– Small space required.</li> <li>– Low start-up time and continuous operation.<sup>7</sup> Initial water parameters don't have major effect on treatment process.</li> </ul> | <ul style="list-style-type: none"> <li>– Often pre-filtration for particle removal + other pre-treatment steps.<sup>3</sup></li> <li>– High investment costs, but comparatively low operating costs.<sup>7</sup></li> <li>– Mineral imbalances can increase corrosive nature of the effluent and post-treatment might be necessary.<sup>3</sup></li> </ul> | <ul style="list-style-type: none"> <li>– Membrane technique would be best to use as an intermediate solution to treat peak concentrations. Too expensive for regular use.</li> </ul> |
|---|---|--|--|

### Biological

#### Biological filters

- |   |  |   |   |
|---|--|---|---|
| <p>Biofilms form through accumulation of nitrifying bacteria on filter material. <math>\text{NH}_4^+</math> is oxidized to <math>\text{NO}_2^-</math> and then to <math>\text{NO}_3^-</math> by different bacteria.<sup>17,18</sup> Sufficient <math>\text{O}_2</math> has to be supplied through aeration step prior to filtration.<sup>19</sup> Removal rates &lt;95% with feed concentrations below 4.5 mg <math>\text{NH}_4\text{-N/L}^3</math></p> | <ul style="list-style-type: none"> <li>– Low construction and maintenance costs.<sup>17</sup></li> <li>– Simplicity in operation.<sup>17</sup></li> <li>– Biodegradable matter (BOM) removed simultaneously.<sup>17</sup></li> </ul> | <ul style="list-style-type: none"> <li>– Colonization takes 2–3 months.<sup>19</sup></li> <li>– Increases nitrate levels, may release bacteria into the treated water.<sup>3</sup></li> <li>– Incomplete nitrification can occur due to: occurrence of elevated BOM concentrations,<sup>20</sup> competition for phosphate with other bacteria,<sup>21</sup> fluctuations in feed concentration,<sup>22</sup> if <math>\text{O}_2</math> concentrations are low<sup>19</sup></li> </ul> | <ul style="list-style-type: none"> <li>– Filters are sensitive to changes in raw water concentrations, which are common in Delhi due to mixing of different groundwater sources and surface water. Microbiology is sensitive to this and reliable functioning is not guaranteed.</li> <li>– Malfunctions detected only through regular, very accurate sampling of influents and effluents.</li> </ul> |
|---|--|---|---|

<sup>1</sup>Gauntlett, 1980; <sup>2</sup>Huang & Shang, 2006; <sup>3</sup>Health Canada, 2013; <sup>4</sup>Jones *et al.*, 2005; <sup>5</sup>Abd El-Hady *et al.*, 2001; <sup>6</sup>Rahmani & Mahvi, 2006; <sup>7</sup>Kurama *et al.*, 2002; <sup>8</sup>Weatherley & Miladinovic, 2004; <sup>9</sup>Li *et al.*, 2011; <sup>10</sup>Govt. of Delhi, 2011; <sup>11</sup>Donnermair & Blatchley III, 2003; <sup>12</sup>Takó and Laky, 2012; <sup>13</sup>Janda & Rudovsky, 1994; <sup>14</sup>Griffin & Chamberlin, 1941; <sup>15</sup>Bellona *et al.*, 2008; <sup>16</sup>Bodalo *et al.*, 2005; <sup>17</sup>Yu *et al.*, 2007; <sup>18</sup>Andersson *et al.*, 2001; <sup>19</sup>Lytle *et al.*, 2013; <sup>20</sup>Manem & Rittmann, 1992; <sup>21</sup>De Vet *et al.*, 2010; <sup>22</sup>Rittmann, 1990.

## 4.5 CONCLUSION AND RECOMMENDATIONS

The use of BF in the Yamuna floodplain in Delhi and in similar hydrogeological settings is basically recommended. However, in these locations, bank filtration should not be seen as a treatment option, but as an option to adapt and improve water management measures. The two main advantages are (1) temporary water storage in the aquifer and (2) a relatively constant raw water composition, improving operating conditions for WTPs. In addition, it is advised to set up a post-treatment concept that is designed specifically for the groundwater parameters at the particular BF location. Such an adapted site-specific post-treatment concept would have the advantage that it would not only reduce elevated ammonium concentrations caused by the infiltration of sewage water, but it would also allow treating other (geogenic) parameters of concern, for example arsenic and fluoride.

In the long term, it is essential to improve the river water quality by implementing sufficient sewage treatment capacity. As this has been widely recognized several new wastewater treatment plants are planned or under construction, e.g. five WWTPs with a designed capacity of 360,000 m<sup>3</sup>/d are likely to be commissioned in 2014–5 (Government of India, 2014). However, elevated ammonium concentrations will prevail long after source water quality has improved.

According to the laboratory column experiments (section 4.3.4) and a simplified 1D reactive transport model that was set up for the field site (section 4.3.5), ammonium desorption in the kankar layer – where the laterals of the contaminated well (P3) are presumably located – will take decades. With the assumptions described in section 4.3, the period of ammonium desorption to concentrations <0.5 mg/L will last for about 61 years in the 500 m strip along the river. This result of the simple 1D model is a conservative estimate, as the average linear flow velocity in the kankar layer is probably much higher than the literature value (Sprenger, 2011) suggests. In general, due to the continuing accumulation of ammonium on the aquifer matrix, desorption times will increase with prolonged infiltration of contaminated surface water. Thus, a short and medium term solution such as post-treatment remains a necessity for the investigated site, even if river water quality improves fast. More detailed 2D models are strongly recommended to make more precise and realistic predictions.

### 4.5.1 Recommended remediation

In general, ammonium remediation options are complex and expensive. At BF sites such as Delhi remediation would often be further complicated by the fact that wells are spread along a long stretch of the river. In-situ or pump and treat remediation measures would, therefore, have to be installed over large areas making remediation extremely costly. If it is decided to develop such a remediation concept, it is of utmost importance to implement the following recommendations:

- *Installation of multi-level observation wells at the well field including levelling survey:* Regular measurements of the water level when wells are operating and when they are switched off, and creation of groundwater contour maps for the different seasons.
- Development of a groundwater flow model based on the water level data. The hydraulic conditions at the well field have to be well known in order to be able to decide on a concept; it is especially important to understand, how the wells influence the flow regime.
- Modelling of scenarios of possible remediation concepts.
- Implementation of groundwater monitoring by regularly taking groundwater samples from the observation wells. To be able to evaluate remediation measures, groundwater quality has to be known and documented before the start of remediation measures.
- Implementation of accompanying groundwater monitoring during remediation.

In case a remediation option is wanted, it is not recommended to use any option involving the injection of chemicals or additives other than oxygen into the aquifer. Although those methods usually are characterized by a faster removal of ammonium, there is a risk of unwanted secondary reactions and formation of by-products, which might not get degraded on the short flow paths to the production wells. BF sites are always water protection zones and therefore special precaution should be taken.

### 4.5.2 Recommended post-treatment

Two options were identified as the most applicable for the given context of high ammonium concentrations in raw water: (1) raw water from affected wells can be mixed with raw water from other sources before treatment or (2) raw water from wells can be treated separately in independent WTPs.

In Delhi, the first option is generally chosen. Raw water from the Ranney wells is usually mixed with surface water before treatment. This has the advantage that the groundwater from the Ranney wells is diluted and parameters such as arsenic remain below the guideline values and do not need to be treated. A major disadvantage of this method is that water



quality is not constant. Nitrogen concentrations in the Yamuna River upstream Delhi show high variations. As the treatment plants are not designed to cope with peak concentrations, it is not always possible to remain below the guideline values for ammonium and/or nitrate. Furthermore, the mixing of ammonium contaminated groundwater with surface water might increase ammonium concentrations in the WTP influents to a level where chlorination is negatively affected.

The second option is therefore recommended. It is better to treat the raw water from the floodplain aquifer separately, e.g. by further pursuing the concept started with the Commonwealth Games Village WTP, a 4.5 million litres per day (MLD) WTP for the water from P4 and nearby bore wells, and the 27 MLD nitrification plant in Okhla.

Although microbial filters (nitrification filters, Table 4.5), as for example used in the Okhla nitrification plant, are a common and cost-efficient option to treat ammonium in raw water, most studies about biological filters for drinking water treatment were not conducted under conditions met in India and the results cannot directly be transferred. Challenges to be met with this technique in locations like Delhi include:

- Supply enough oxygen to cope with the high ammonium concentrations.
- Monitor both inlet and outlet concentrations closely and adapt the hydraulic loading to stabilize nitrogen loading and thus achieve complete nitrification.

Lee *et al.* (2014) reported stable ammonium removal in new, less concentration-sensitive biological filters. However, the reported range of ammonium concentrations is an order of magnitude lower than in the raw water of the Ranney wells in Delhi. Therefore, pilot and full scale studies to find optimum operating conditions for the specific local requirements are recommended if the application of this technique should be further enhanced in India. As an alternative to biological filters, a method with more robustness towards fluctuating input parameters and less downtime in case of failures should be considered, such as ion exchange using zeolites.

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