

Design and applicability of the OBM process for New Delhi

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Design and applicability of the OBM process for
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Author(s)

Kamal Azrague and Stein W. Osterhus

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Summary

Delhi, the capital city of India is facing many problems of water shortage which are exacerbated by the population growth, urbanization, industrialization and migration. As part of the WP 7.9 of the TECHNEAU integrated project, the feasibility of the OBM process combined with river bank filtration to provide a sustainable and safe drinking water supply for Delhi is investigated within this report. The OBM process belong to the “multi-barrier” concept and represent the characteristics to be a strongly robust process since it comprises of an oxidation step, a biofiltration step and a membrane filtration step. The removal of the different contaminants present in the river bank filtered water is assessed step by step within the OBM. The results of the assessment show that except for fluoride, all the contaminants have the potential of being removed by the OBM. However, with additional treatment such as activated alumina filtration or coagulation, possible treatment improvements could be achieved to provide a safe drinking water.

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1 Introduction

Delhi, the capital city of India is about 1485 km² in area, out of this approximately 63% is urbanized. Problem of water shortage in Delhi has been exacerbated as a result of high natural population growth, urbanization, industrialization and migration. Consequently, Delhi is facing a future of very limited water resources.

Water requirement for drinking purpose of Delhi is mainly met from river Yamuna. However, groundwater is also used for drinking purpose in few areas. The Yamuna River is one of the heavily polluted rivers in India. Pollutants flowing into the river are contributed from the waste of the cities situated along its bank. The Yamuna River passes through Delhi, the capital of India, where it receives large volumes of domestic and industrial effluents. The river is the main source of water supply to Delhi.

As part of work package 5.2 of the TECHNEAU integrated project, an analyze has been done on the function and relevance of managed aquifer recharge (MAR) techniques with a main focus on Riverbank Filtration (RBF) to enable sustainable water resources management, especially in developing and newly industrialized countries (Pekdeger et al. 2008; Sprenger et al. 2008). Within the study investigations on RBF are being performed at different field sites in Delhi, India. Chemical and microbiological parameters were analyzed, in order to detect contaminants and characterize waters from different sources for detailed hydrogeological investigations. Results obtained from the study at the Nizamuddin field site show that positive effects during bankfiltration can be observed only for one heavy metal (Al), while no significant changes can be observed for Pb and Cu and an increase in the concentration can be observed for Fe and Mn. Other substances like As, F and Ammonia increase during the underground passage while no significant changes or a decrease in the concentration can be observed for B and NO₂⁻, respectively (Pekdeger et al. 2008). In addition, a comparison of surface water and bank filtrate shows that the major amount of organics is either strongly reduced or completely removed during infiltration and subsurface passage (Sprenger et al. 2008). Under the given field conditions a 3 log₁₀ removal of somatic bacteriophages after 1 m of travel distance was achieved (Sprenger et al. 2008). At this field site elevated concentrations of several substances like As, Fe, Mn, F and NH₄ will, however, make a post-treatment necessary.

As part of the TECHNEAU's WP2.2, NTNU (Norwegian University of Science and Technology) has been developing a new process concept combining oxidation, biodegradation and membrane filtration (OBM) for drinking water treatment. In the current report a study on the feasibility of the OBM process as a cost-effective multi-barrier RBF post treatment has been investigated.

1.1 Location of the site

Nizamuddin Bridge field site (NI) is situated in the urban central part of the city of Delhi, on the eastern bank of Yamuna River. Within this segment in central Delhi, the Yamuna River is highly contaminated by discharge of sewage and industrial wastewaters. Aquifer recharge takes place under “natural” conditions probably due to groundwater abstraction to the east.

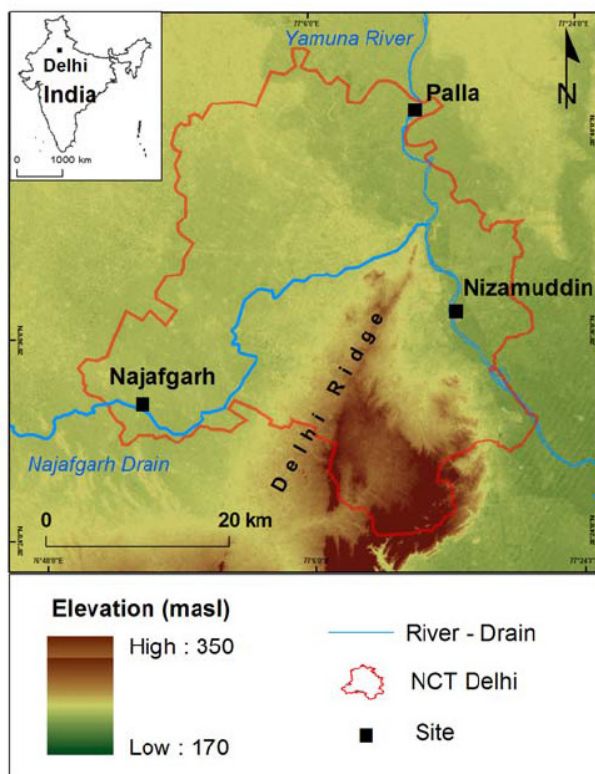


Figure 1. Location map of the selected field site with the geomorphology and the main river / drain.

1.2 Water quality at the Nizamuddin Bridge

The data used for this study are obtained from the deliverables 5.2.6 and 5.2.2. These reports provide overviews on the water analysis performed during the studies on River Bank Filtration in Delhi, India within WP 5.2 of TECHNEAU integrated Project.

1.2.1 Physicochemical Parameters and Inorganic Ions

Table 1 presents a summary of all the water quality values at the site of interest (Pekdeger et al. 2008). According to these data DOC, ammonia, fluoride, arsenic, iron and manganese, need to be treated by a post treatment such as the OBM process.

Table 1. Hydrological and chemical properties of the studied aquifer at Nizamuddin Bridge (EC = electrical conductivity, DOC=dissolved organic carbon, b= below detection limits).

	Shallow Piezometer (PZ-1,4,5)			Deep Piezometer (PZ-3)			Indian desirable limit - Permissible limit
	Min	Mean	Max	Min	Mean	Max	
Temperature (°C)	24.7	26.5	28.6	26.5	26.8	27.7	
pH at 20°C	6.5	7.8	6.9	8.0	8.0	8.0	Between 6.5 and 9.5
EC (µS/cm)	1062	1622	2490	847	877	894	
DOC (mg/L)	1.3	13.7	6.7		1.3	5.2	
HCO ₃ (mg/L)	335	786	1647	482	530	564	200 - 600
Ca ²⁺ (mg/L)	5	140	307	9	16	33	75 - 200
Mg ²⁺ (mg/L)	14	38	63	4	11	35	30 - 100
Na ⁺ (mg/L)	85	136	183	158	197	210	
Chloride (mg/L)	92	166	220	12	50	200	250 - 1000
SO ₄ (mg/L)	b	102	365	43	46	50	2 - 400
NO ₃ (mg/L)	b	0.5	5	b	b	b	45
Al (mg/L)	b	0.06	0.23	b	0.05	0.09	0.03 - 0.2
Cu (mg/L)	b	b	0.01	0.003	b	0.003	0.05 - 1.5
Pb (µg/L)	b	8	28	3	3	3	10- 50
Fe (mg/L)	b	17	38	0.08	0.22	0.28	0.3 - 1
Mn (mg/L)	0.7	3.2	6.5	0.05	0.07	0.09	0.1 - 0.3
F (mg/L)	b	1.68	4	3	3.4	5	1 - 1.5
As (µg/L)	7	37	84	b	2	6	50
NH ₄ (mg/L)	0.1	16	60	b	0.03	0.05	
NO ₂ (mg/L)	b	0.01	0.025	0.005	0.02	0.05	
B	b	0.01	0.08	b	0.07	0.11	1 - 5

1.2.2 Bacteriological analysis

The variation of the fecal coliform concentration in the Yamuna River ranges from 4.5×10^5 - 9.2×10^7 cfu/100 mL, without any significant difference between monsoon and non-monsoon time with an average for both seasons of 1.6×10^7 cfu/100 mL. Literature data from CPCB 2006 reports an annual average concentration of fecal coliforms in the range of $1 - 8 \times 10^6$ cfu/100mL. The SGS laboratory analyzed coliforms in the range of 1.5×10^5 cfu/100mL (Table2) (Sprenger et al. 2008).

Table 2 Microbial parameters analyzed by the SGS laboratory (Sprenger et al. 2008).

Observation point	<i>E.coli</i> (cfu/100 mL)	Coliform (cfu/100 mL)	<i>Enterococci</i> (feacal streptococci) (cfu/100 mL)	<i>Clostridia</i> (cfu/mL)
Yamuna River	1.20E+05	1.50E+05	5000	<1
PZ-2a	6.80E+04	8.00E+04	3600	<1
PZ-2c	5.00E+02	4.00E+02	<100	<1
PZ-2	2.00E+02	1.80E+03	<100	<1

1.2.3 Human pathogenic viruses

High numbers of adenoviruses and noroviruses have been found in the Yamuna River (Table 3) at the Nizamuddin Bridge, but none in the observation well in 50 m distance from the river (Sprenger et al. 2008).

Table 3 Human pathogenic viruses in the Yamuna River and the PZ2 (travel time approx. 50 d) (Sprenger et al. 2008).

Observation point	Adenoviruses	Noroviruses	Hepatitis A*	Hepatitis E*
Yamuna River	$3.6 \cdot 10^4$ (genome copies/100mL)	$5.4 \cdot 10^4$ (genome copies/100mL)	positive in 100 mL	positive in 100 mL
PZ-2	negative in 2000 mL	negative in 2000 mL	negative in 1000 mL	negative in 1000 mL

*detected semi-quantitatively using PCR or nested PCR.

1.2.4 Organic trace substances: non target screening

The organic trace components that accumulate in the Yamuna River in Central Delhi get largely attenuated during RBF. The GC scans from the samples of surface water (NI-SW) and the adjacent piezometer (NI-PZ) shows that most of the peaks from the NI-SW sample (especially from the F3 fraction (non polar) where the number and abundance of identified substances is by far the highest) are removed or at least largely reduced (Sprenger et al. 2008).

Other substances from the F1 (polar) and F2 (medium polar) fractions, including camphor, menthol, cresol as well as the endocrine disruptor and suspected cancerogenic BPA are no more traceable in the bank filtrate. Likewise, none of the drugs that were found in the river (nicotine, caffeine, ibuprofen, CBD) could be detected in the samples from the bank filtered water. The same applies for the biomarkers (i.e. cholesterol), herbicides and most other substances. Only the fatty acids or their methyl esters (FAME) respectively are not completely removed: the most abundant ones, palmitic acid and steric acid, still have traceable peaks in the groundwater sample. Considering the abundance of the peaks, it is assumed that these fats and oils are effectively degraded during infiltration and soil passage, but the travel time and distance are still insufficient for a complete removal of the fats and oils under the given environmental conditions (Sprenger et al. 2008).

One substance is detected in considerable abundance in the groundwater, and was not found in the river: **MCH** (F3 fraction, 14.1 min) is a pesticide that acts as a beetle repellent. The substance, however, is not expected an environmental risk in minor concentrations, because it shows no or only very minor adverse effects on nontarget species and has an overall low toxicity. Anyhow, the exposure to humans should be minimal to non-existent and not acceptable in drinking water (Sprenger et al. 2008).

1.2.5 Organic trace substances: quantitative analysis by SGS

The concentrations of all pesticides and most other substances mentioned in Table 4 remain below the detection limit for all samples from Nizamuddin Bridge. Only anionic detergents (e.g. MBAS) were detected in the Yamuna River in a concentration of 0.18 mg/L, which remains below the desirable limit of 0.2 mg/L, and the permissible maximum of 1.0 mg/L of the Indian Standards. In all the samples of the piezometers, concentrations of these

detergents remain below the detection limit of 0.05 mg/L. This indicates that after only 1.5 m of subsurface passage, a possible pollution with anionic detergents is effectively attenuated. With regard to the apparent absence of the other organic compounds, it should be considered that the samples are only representing snapshots and are not necessarily characteristic values for the system (Sprenger et al. 2008).

Table 4 Selection of organic substances for quantitative analysis, detection limits and drinking water standards (Sprenger et al. 2008).

Parameter / substance	Detection limit	Drinking water standards	
		Indian standard IS: 10500	WHO guideline
Mineral oil	0.01 mg/L	Desirable 0.01 mg/L Permissible 0.03 mg/L	-
Anionic detergents as Methylene Blue Active Substances (MBAS)	0.05 mg/L	Desirable 0.2 Permissible 1.0 mg/L	-
Phenolic compounds (as C ₆ H ₅ OH)	0.001 mg/L	Desirable 0.001; Permissible 0.002 mg/L	-
Benzo(a)pyrene	0.1 mg/L	-	0.7 µg/L
Benzene	0.1 µg/L	-	0.01 mg/L
Epichlorohydrine	0.01 µg/L	-	0.4 µg/L
1,2-Dichloroethane	0.1 µg/L	-	0.03 mg/L
Tetrachloroethane	0.1 µg/L	-	0.04 mg/L
Trichloroethane	0.1 µg/L	-	0.07 mg/L
Vinyl chloride	0.1 µg/L	-	0.3 µg/L
Trihalomethanes (Total)	0.1 µg/L	-	i.e. 0.2 mg/L for Chloroform
Polynuclear aromatic hydrocarbons (PAHs)	0.1 mg/L	-	-
General multiresidue screening = Pesticides (61 different substances)*	0.1 µg/L	Desirable absent; Permissible 0.001 mg/L	-

*Alpha-BHC, Beta-BHC, Lindane (gamma BHC), Delta BHC, o, p-DDD, o, p DDEo, p' DDT, p,p' DDD, p,p' DDE, p,p' DDT, Endodulphan I, Endodulphan II, Endosulphate sulphate, Aldrin, Dieldrin, Heptachlor, Heptachlor epoxide, Endrin, Methoxychlor, Alachlor, Butachlor, Chlordane, Dicofol, Hexachlorobenzene (HCB), Chlopyrifos, Chlopyrifos methyl, Phosphamidon, fenitrothion, Parathion, Malathion, Methyl paraoxon, Methyl parathion, Malaaxon, Dimethoate, Phosalone, Quinalophos, Primiphos methyl, monocrotophos, Fenchion, Phorate, Phorate sulphone, Phorate sulphoxide, Isoproturon, Methidathion, Dichlorvos, Carbofenotion, Clofenvinfos, Diazinon, Azinphos methyl, Tri azophos, Ethion, Cyfluthrin, Cypermethrin, Deltamehrin, Esfenvalerate, Lambda cyhalothrin, Premethrin, Carbaryl, Carbofuran, Aldicarb, Methomyl

2 Applicability and design parameters of the OBM for Delhi

TECHNEAU's partner NTNU (Norwegian University of Science and Technology) has been developing the Oxidation-Biodegradation-Membrane filtration (OBM) process. The OBM process belongs to the "multi-barrier" concept and represents the characteristics of being a strongly robust process since it comprises of an oxidation step, a biofiltration step and a membrane filtration step. This makes the OBM a promising system which could meet the above treatment requirements to provide a good drinking water quality.



Figure 2 Diagram of the OBM drinking water concept.

Concerning the fundamentals and design of the ozonation-biofiltration and of the OBM, different reports already exist (Azrague and Osterhus 2006; Juhna and Melin 2006) and therefore that part will only briefly be described in this report.

2.1 Oxidation

The purpose of the oxidation step could be many, including:

- Oxidative conversion of harmful or undesirable compounds to more easily removable compounds:
 - Oxidize soluble inorganic compounds to less toxic or less soluble compounds that will precipitate and be removed later in the process.
 - Oxidize persistent organic compounds and micro pollutants to easy biodegradable compounds
 - Remove color.
 - Remove taste and odor.
- Provide hygienic barrier towards pathogens.

For the oxidation different methods and combinations are possible, however, the most relevant and promising would here be ozonation. Therefore, oxidation by ozone is the method that will be considered in this report.

2.1.1 Ozone system design principles

There are two primary conditions that has to be determined in the design of the ozone system; 1) The required ozone dose which determines the size of the ozone generators and its input components (compressors, air dryers, oxygen producer, etc), and 2) The required contact time or reaction time (size

of the reaction tanks). However, these two conditions largely depend on the purpose of the ozonation system which would be site specific. For example, depending on the water quality, the half-life of ozone is in the range of seconds to hours. This has to be taken into account when designing the ozone system. The stability of ozone largely depends on the water matrix, especially its pH (high pH promotes ozone decomposition via OH radical), the type and content of the DOC and its alkalinity (Hoigné 1998).

2.1.2 *Potential removal and treatment during the oxidation step*

Among the list of different contaminants present in the water, DOC, biological material, arsenic, iron, manganese, ammonia and bromate could be affected during the oxidation step.

2.1.2.1 *Disinfection and design requirements*

Although RBF has shown to achieve a complete removal of the different virus investigated, it seems like it has performed poor for bacteria. Therefore, an additional disinfection is needed and this could be done by ozonation within the OBM process, representing a first hygienic barrier.

Ozone is a strong and very efficient disinfectant, especially towards bacteria and viruses. It is also very efficient towards *Giardia*, but less efficient towards *Cryptosporidium*. The inactivation of micro organisms by ozone can be described by Chick's law:

$$\text{Log } (N/N_0) = -k C T \quad (1)$$

Where N is the number of organisms that remain viable, N_0 is the initial number of viable organisms, k is the inactivation constant, C is the concentration of disinfectant (mg/L), and T is the exposure time (min). The relative impact of C and T is set to unity. The Chick's law is the basis for design of the disinfection. In order to obtain a certain log inactivation of a certain type of target organism by ozone, a certain C T-value (the concentration of ozone times the contact time) is required. Tables 5, 6 and 7 show the required C T-values at different temperatures in order to obtain different log inactivation credit for the inactivation of *Giardia*, viruses and *Cryptosporidium*, respectively. There is no separate table for bacteria since they are, in general, more sensitive to ozone than i.e. viruses, and consequently inactivation of viruses ensures at least equal degree of inactivation of bacteria. The required C T-value found in tables 5, 6 and 7 (at the desired degree log inactivation and temperature) would then be the design requirement for the disinfection by ozone giving the needed ozone concentration and contact time.

Table 5. CT requirements to achieve inactivation of Giardia by ozone (USEPA 1991, Guidance manual for the compliance with the filtration and disinfection requirements for public water systems using surface water sources).

Log inactivation	Temperature (°C)					
	≤ 1	5	10	15	20	25
0,5	0,48	0,32	0,23	0,16	0,12	0,08
1,0	0,97	0,63	0,48	0,32	0,24	0,16
1,5	1,5	0,95	0,72	0,48	0,36	0,24
2,0	1,9	1,3	0,95	0,63	0,48	0,32
2,5	2,4	1,6	1,2	0,79	0,60	0,40
3,0	2,9	1,9	1,4	0,95	0,72	0,48

Table 6. CT requirements to achieve inactivation of viruses by ozone (USEPA 1991, Guidance manual for the compliance with the filtration and disinfection requirements for public water systems using surface water sources).

Log inactivation	Temperature (°C)					
	≤ 1	5	10	15	20	25
2,0	0,9	0,6	0,5	0,3	0,25	0,15
3,0	1,4	0,9	0,8	0,5	0,4	0,25
4,0	1,9	1,2	1,0	0,6	0,5	0,3

Table 7. CT requirements to achieve inactivation of Cryptosporidium by ozone using the T10 –method. (USEPA, LT2ESWTR).

Log inactivation	Temperature (°C)									
	≤ 0,5	1	2	3	5	7	10	15	20	25
0,5	12	12	10	9,5	7,9	6,5	4,9	3,1	2,0	1,2
1,0	24	23	21	19	16	13	9,9	6,2	3,9	2,5
1,5	36	35	31	29	24	20	15	9,3	5,9	3,7
2,0	48	46	42	38	32	26	20	12	7,8	4,9
2,5	60	58	52	48	40	33	25	16	9,8	6,2
3,0	72	69	63	57	47	39	30	19	12	7,4

However, the design is not straight forward. The main question is which ozone concentration, C, and contact time, T, should be applied in the design. In general, the ozone consumed by other processes has to be accounted for, since it is not available for disinfection. Similarly, short circuiting, dead volumes, etc also has to be taking into the consideration since it reduces the effective contact time.

2.1.2.2 DOC oxidation

Ozonation causes structural changes to DOC and in particularly to the humic fractions. The changes include a strong and rapid decrease in color and UV absorbance due to a loss of aromaticity and depolymerization, a small reduction of TOC (e.g. 10% at 1 mgO₃×mg C⁻¹), a slight decrease in the high apparent molecular weight fractions, a slight increase in the smaller fractions, a significant increase of the carboxylic functions, and the formation of ozonation byproducts (OBP). These byproducts include small molecular weight compounds like aldehydes (Le Lacheur et al. 2002), acetones, ketoacids (Le Lacheur et al. 2002; Xie and Reckhow 1992) and carboxylic acids

(Gagnon et al. 1997). In general, ozonation makes DOC more biologically available for bacteria in biofilters.

Ozone addition not only increases the biodegradability of the dissolved organics, but also introduces large amounts of oxygen to the water, thus creating an excellent environment for biological growth on the filter media.

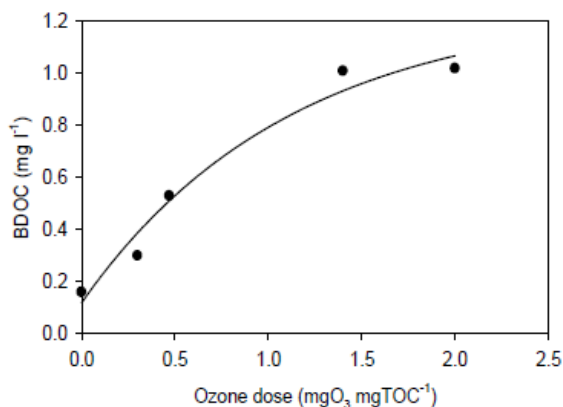


Figure 3. Effect of ozone dose on the formation of BDOC in humic surface water (Melin et al. 2006).

2.1.2.3 *Micro pollutants removal*

Even though the analyses show that most of the micro pollutants are not longer traceable after RBF, possible undetectable and unknown compounds could be present and therefore the OBM should remove them. During the ozonation, a large number of micro pollutants could be oxidized and broken down to more biodegradable compounds. The oxidation can follow two different pathways:

- The direct oxidation involving the ozone.
- The indirect oxidation involving the hydroxyl radical, which is a byproduct of the decomposition of the ozone.

In general the oxidation displays a first order kinetic, and reviews where rate constants are reported can be found elsewhere (von Gunten 2003).

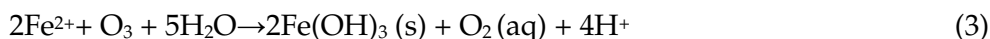
2.1.2.4 *Arsenic removal*

During ozonation, the arsenic is oxidized from As(III) to As (V) with a half-life of only 4 min, which is much shorter than the half-life with pure oxygen ranged from 2 days to 5 days, and with air (half-life of 9 days) (Kim and Nriagu 2000). Therefore, ozone can be used to remove effectively arsenic from groundwater through oxidation, followed by co-precipitation and/or adsorption reactions.

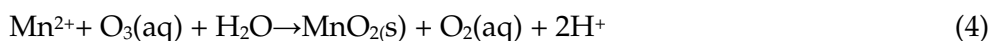
2.1.2.5 *Iron and manganese*

Removal of iron and manganese consists of transforming (oxidizing) the soluble form to a less soluble form that will precipitate and then can be filtered out of the water. Ferrous iron is either oxidized by oxygen (equation 2) or by ozone (equation 3)





Unlike iron oxidation which can be done by aeration, conversion of Mn^{2+} into MnO_2 requires a stronger oxidant like ozone. The oxidation reaction of manganese by ozone takes the following form:



Consequently, iron and manganese should be oxidized and precipitate and removed during the filtration steps.

2.1.2.6 Bromate formation and removal

Since the river bank filtered water might contain bromides, an interest has to be taken on the possible bromate formation. Indeed, great concern has been raised about bromate ions (BrO_3^-), which are formed upon ozonation of bromide (Br^-) containing water. Bromate has been classified as a genotoxic human carcinogen (Kurokawa et al. 1990) and a maximum contaminant level has been set both by the EPA and the European Commission at $10 \mu\text{g/L}$. The formation of bromates by ozonation of bromide-containing water has been studied by several researchers (Buffle et al. 2004; Haag and Hoigne 2002; Pinkernell and von Gunten 2001; von Gunten and Hoigne 1994). Bromate can be formed, during ozonation, via a complicated mechanism in which molecular ozone (O_3) and hydroxyl radical ($\text{OH}\cdot$) reactions are present (figure 4).

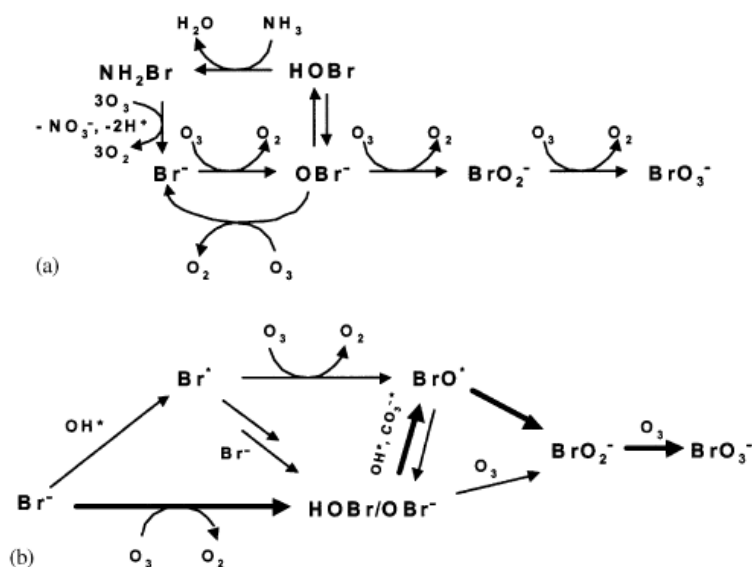


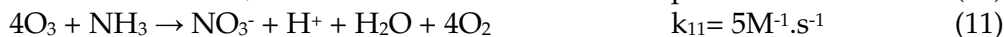
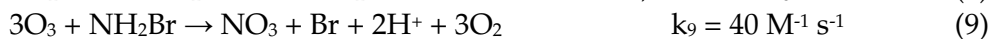
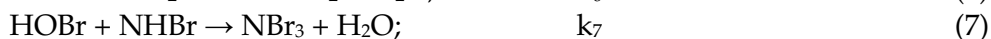
Figure 4. Reaction scheme for bromate formation during ozonation of bromide-containing waters. (a) reactions with ozone and (b) reactions with ozone and OH radicals. The bold lines show the main pathway during the secondary phase of an ozonation process (von Gunten and Hoigne 1994).

To solve the problem of excessive bromate formation, several treatment options have been reported, namely ammonia addition, pH depression, OH radical scavenging and scavenging or reduction of HOBr (Galey et al. 2000; Kruthof et al. 1993; Pinkernell and von Gunten 2001). It turns out that only

the first two options, namely ammonia addition and pH depression, are feasible in drinking water treatment (Pinkernell and von Gunten 2001; Song et al. 1997). Since the RBF treated water contains a high ammonia level, it is likely that the bromate concentration would remain low, however this has to be confirmed experimentally.

2.1.2.7 Ammonia removal

Haag et al. (1984) have indicated that ammonia could be decomposed to N₂ by ozonation in the presence of bromides (Haag et al. 1984). Ammonia removal from water containing bromide during ozonation has been known to proceed as follows (Haag et al. 1984; Wajon and Morris 2002; Yang et al. 1999):



There are three possible pathways for ammonia removal by ozonation with bromide ions. In the first pathway, hypobromous acid reacts with ammonia and is converted to nitrogen gas through a series of reactions (Eq. (5)– Eq. (8)). The second pathway is the formation of nitrate via the bromamide (NH₂Br) intermediate (Eq. (9)). The third pathway is the direct oxidation of ammonia to nitrate by ozone Eq. (11). Therefore there is a potential to remove ammonia during ozonation, however the extent of this removal can not be estimated without any experiments.

2.2 Biofiltration

2.2.1 Biofiltration design

The parameters that could affect the performance of a biofilter are the characteristics of filter media, hydraulic and organic loading rate, and filter backwash techniques. Other factors that could influence the performance of a biofilter are the temperature and the presence of oxidants.

2.2.1.1 Filter Media

The economical and efficient operation of a biofilter highly depends on the characteristics of its filter media. While selecting the filter media, one should also consider the source and concentration of targeted pollutants. For removing offensive organic substances from the drinking water supply line, granulated activated carbon (GAC) or anthracite or filter coal or sand could be a good choice. Previous studies have shown that GAC (an adsorptive media) can be a better choice than anthracite or sand (non-adsorptive media) for the removal of organic substances from surface water (LeChevalier et al. 1992; Wang et al. 1995). A GAC filter might have less specific surface area (surface area per unit volume of filter) available for microbial attachment than a sand filter because the effective size of sand is usually smaller than of GAC.

Furthermore, the size of GAC micropores (1-100 nm) seem to be too small for microorganisms (typically greater than 200 nm in diameter) penetration inside these micropores (AWWA Research and Technical Practice Committee on Organic Contaminants 1981). However, the macroporous structure and irregular surface of GAC offer more appropriate sites for biomass attachment. GAC can adsorb and retain slowly biodegradable components that can be biodegraded by the attached microbial mass leading to continuous bioregeneration of the GAC. It also provides protection from shear loss of biomass. Wang et al. (1995) found the mesoporous GAC surface texture more suitable for biomass attachment than macroporous and microporous GAC. The biofilter media should provide: (i) a suitable surface for quick biomass growth, (ii) larger surface area for biomass growth, and (iii) good surface texture to hold biomass against shear and sloughing. The effect of types of media on the performance of the biofilter is shown in Fig. 5.

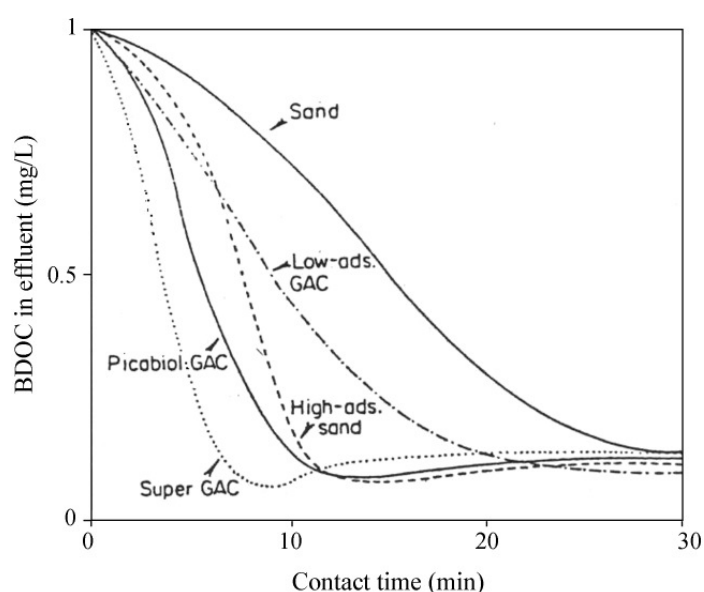


Figure 5. Effect of types of filter media on the performance of biofilter (Billen et al. 1992).

2.2.1.2 Empty Bed Contact Time (EBCT)

The contact time, usually expressed as empty bed contact time (EBCT), is a key design and operating parameter of a biofilter. Usually the percentage removal of organic substances increases with increase in contact time up to an optimum value. Both the filter depth and hydraulic loading can be changed to increase the EBCT. Previous studies have shown that the contact time (and not the hydraulic loading) is the key variable responsible for organic removal. For a given EBCT, organic removal is independent of hydraulic loading in the range typically used in rapid filtration (Carlson and Amy 1995; Servais et al. 1994). Huck et al. (1994) showed that the organic removal efficiency of a biofilter could be approximated by a first-order model (Huck 1994). Servais et al. (1992) reported a linear increase in BDOC removal with the increase in EBCT between 10-30min of the biofilter (Servais 1992). The past studies (Hozalski 1995; Price), which showed very little or no effect of EBCT on

organic removal efficiency of a biofilter, might be due to partial acclimatization of the biofilter.

2.2.1.3 *Filter Backwash*

It is important to select an appropriate filter backwashing technique for successful operation of a biofilter. The biomass attached to the filter media has to be carefully maintained during backwashing (Ahmad 1998; Bablon 1988; Bouwer and Crowe 1988; Graese 1987; Miltner 1995). Ahmad and Amirtharajah (1998) found that biological particles, which are usually hydrophobic in nature, are attached to filter media (GAC) with a greater force than non-biological clay particle (measured as turbidity). The difference in the detachment of these particles during backwashing should be taken into account while selecting or optimizing the backwashing of a biofilter. Previous researches have shown no major loss of biomass during backwash of the biofilter (Ahmad 1998; Lu 1993).

2.2.1.4 *Temperature*

The activities of bacterial community adapted at 10 °C and 20 °C were found to increase with increase in temperature in range of 10- 30 °C.

2.2.2 *Potential treatment and removal during biofiltration*

During biofiltration, DOC, iron, manganese, ammonia, some micro pollutants and arsenic could be removed. This subsection presents the results reported in literature for such treatment.

2.2.2.1 *DOC removal*

When oxidizing organic matter, the biodegradable fraction or the AOC in the water typically increases. This fraction should be removed by a biodegradation step to avoid growth in the network. Biofilter studies on the removal of ozonation by-products (aldehydes and keto-acids) demonstrated that their removal could be described well by a first order model in a plug flow reactor including a term for a minimum substrate concentration:

$$S_{\text{out}} = (S_{\text{in}} - S_{\text{min}}) \cdot e^{-k\theta} + S_{\text{min}} \quad (12)$$

Where S_{in} and S_{out} is the inlet and outlet concentration ($\mu\text{g/L}$), respectively, of substrate from the biofilter, S_{min} is the minimum substrate concentration that can maintain steady state biofilm ($\mu\text{g/L}$), k is the first order rate constant (min^{-1}), and θ is the Empty Bed Contact Time, EBCT (min). Table 8 shows example of first order rate constants for the degradation of ozonation by-products in biofilters, and example of degradation of glyoxal at different starting concentrations and EBCT is shown in Figure 6.

Table 8. Example of first order rate constants for degradation of ozonation by-products in biofilters.

OBP	k (min ⁻¹)	S _{min} (µg l ⁻¹)	OBP	k (min ⁻¹)	S _{min} (µg l ⁻¹)
Formaldehyde	0.39	1.0 – 3.0	Glyozalic acid	0.19	4.8 – 30
Glyoxal	0,21	<1 – 4.6	Pyruvic acid	0.31	2.7 – 27
Methylglyoxal	0.22	<1 – 2.2	Ketomalonic acid	0.10	7.0 - 45
Acetone	0.19	<1 – 1.5			

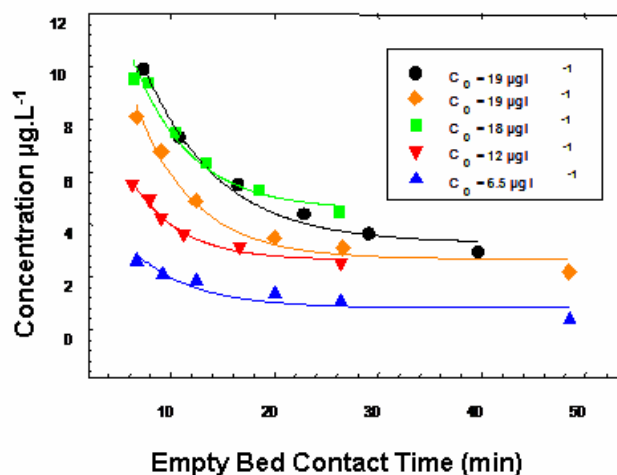


Figure 6. Example of degradation of an ozonation by-product (glyoxal) in a biofilter at different concentration and EBCT.

Bacteria which utilize readily available carbon is located in top layer (Moll et al. 1998) while bacteria able to degrade more recalcitrant carbon are living deeper in the filter.

Griffini *et al.*, (Griffini et al. 1999) observed that in steady-state, the removal of aldehydes and ketoacids most occurred at the top of the biofilter after 2.3 min EBCT. Little additional removal occurred in the rest of the biofilter. In case of BDOC, the samples taken along different biofilters showed also that majority of BDOC removal occurred in the top of the biofilter with some additional removal in the lower levels (Figure 7). The 10-15 min EBCT would have been enough in this case. However, for the more recalcitrant compounds yhat may be present, considerable longer EBCT may be required.

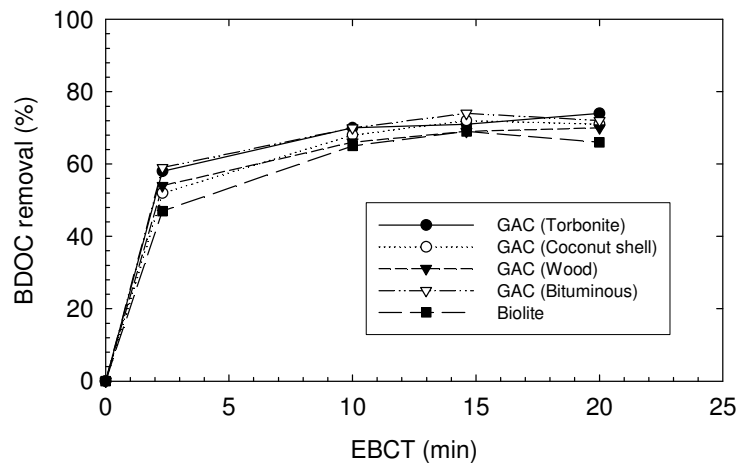


Figure 7. BDOC removal in biofilters with samples taken along the filter column (Griffini et al, 1999).

According to Uhl (2000) such an empirical approach is very well suited to describe removal of DOC and AOC in biofilters (Uhl 2000).

2.2.2.2 Arsenic removal

Iron based biological treatment is relatively efficient in removing As(III) without need of a pre-oxidation step (Katsoyiannis et al. 2002; Katsoyiannis and Zouboulis 2004; Lehimas et al. 2001; Zouboulis and Katsoyiannis 2002). Limitations of this system are that iron must be present in water at sufficient levels. In iron assisted filters, iron bacteria act as a catalyst, consolidate iron oxide in their cell surface and arsenic is adsorbed on bacterial cell surface by a surface complexation mechanism (Katsoyiannis and Zouboulis 2006). Oxidation of Fe(II) to form particulate Fe(III) was thought to take a series of steps, including oxidation, hydrolysis, polymerization, and precipitation. One of the major advantages of biological oxidation of Fe(II) to form particulate Fe(III) was that iron bacteria provided a large surface area (Katsoyiannis and Zouboulis 2004).

Arsenic removal in conjunction with the biological oxidation of iron or manganese or both has been reported (Katsoyiannis et al. 2002; Katsoyiannis and Zouboulis 2004; Lehimas et al. 2001). Lehimas et al. (2001) used natural groundwater and arsenite (As(III)) and ferrous iron was added in pilot scale experiments (Lehimas et al. 2001). They found that As(III) removal was directly related to the concentration of iron added to the system. For an initial As(III) concentration of 400 µg/L, the ferrous iron required to remove 90% of As(III) was 10–12 mg/L and iron oxidizing bacteria *Gallionella* oxidized ferrous to ferric iron. Katsoyiannis and Zouboulis (2004) concluded that biological oxidation of iron (II) by *G. ferruginea* and *Leptothrix ochracea* and subsequent adsorption of arsenic onto the biotic iron oxide was effective in removing arsenic from groundwater. They believed that As(III) was partially oxidized to As(V) by the microorganisms. A recent study found a new bacterial strain, B2, which is dominant in Fe-rich groundwater, responsible

for oxidizing As(III) to As(V), possibly enhancing the arsenic removal process (Casiot et al. 2006). Additional supporting results were also published on biological removal of arsenic (Katsoyiannis and Zouboulis 2004; Katsoyiannis and Zouboulis 2006).

One of the major advantages of biological treatment system is that the adsorbent (iron oxide) is continuously produced because of the biological oxidation of iron present in the natural groundwater. Therefore, there is no need for regeneration of the filter media unlike in most of the adsorption processes. Filter clogging due to iron oxide and bacterial growth is a potential problem arising from the long-term operation of the filter.

2.2.2.3 *Ammonia removal*

Biological nitrification, which employs autotrophic nitrifying bacteria for the oxidization of ammonia to nitrate via nitrite, is the key process in the removal of ammonia. Various parameters influence the nitrification process. Major factors include: dissolved oxygen (DO), temperature, pH, ammonia and nitrite concentrations, organic loading, and hydraulic loading rate (Sharma and Ahlert 1977). Because DO is a chief factor in limiting the nitrification process, its impact becomes even more significant as organic loading increases in the reactor, allowing fast-growing heterotrophic bacteria to compete with nitrifying bacteria for the limited oxygen. It was reported that heterotrophic bacteria have a maximum growth rate of five times and yields of two to three times that of autotrophic nitrifying bacteria (Grady and Lim 1980). In addition to competing for DO, heterotrophic bacteria also compete with nitrifying bacteria for space in fixed-film bioreactors, leading to a decrease in nitrification efficiency or even a failure in the system. As a result, inhibition of heterotrophic bacteria on nitrification is of more concern in systems with high organic material.

Considerable research has been related to the effect of organics on the nitrification process, especially for domestic and industrial wastewater treatment. It has been reported that the bioactivity of nitrifiers were inhibited by an increase in the C/N ratio within fluidized bed reactors, submerged biofilters, and rotating biological contactors (Cheng and Chen 1994; Fdz-Polanco et al. 2000; Gupta and Gupta 2001; Okabe et al. 1996). Ohashi et al. (1995) and Satoh et al. (2000) found that the proportion of nitrifiers decreased with an increasing C/N ratio (Ohashi et al. 1995; Satoh et al. 2000). An exponential decrease of the nitrification rate with an increased influent COD/N ratio was observed in a study on nitrogen removal from high-strength ammonia industrial wastewater (Carrera et al. 2004). The same study also pointed out that the influence of COD/N ratio should be one of the main parameters in the design of biological nitrogen removal process in industrial wastewater treatment. Okabe et al. (1996) discovered that a higher influent C/N ratio retarded accumulation of nitrifying bacteria and resulted in a considerably longer start-up period for nitrification (Okabe et al. 1996). Although the organic impact on nitrification biofilters in aquacultural systems is of high concern, very little investigation has been conducted. Zhu and Chen (2001) determined that the nitrification rate of submerged filters could

decrease by 70% with a C/N ratio of 1 or 2 as compared to a pure nitrification system (Zhu and Chen 2001).

2.2.2.4 Iron and manganese removal

In case that the removal of iron and manganese is not fully achieved during the ozonation, it could be completed in the biofiltration. Fe and Mn removal by biological processes are based on different stages of biofiltration where beds are colonized by Fe-Mn oxidizing bacteria. In nature, iron oxidizing bacteria (IOB) and manganese oxidizing bacteria (MnOB) are widespread. They are prevalent in groundwater, swamps, ponds, hypolimnion of lakes, sediments, soils, wells and water-distribution systems. In the latter they can cause significant clogging problems (Ghiorse 1984). These bacteria which are present in raw water can multiply in sand filters under appropriate conditions and are able to oxidize divalent ions Fe(II), Mn(II) and precipitate them in their oxidized forms Fe(III) and Mn(IV). The bacteria involved in biological Fe and Mn removal need different pH and redox potential (Eh) conditions for each metal, as is shown in Fig. 8, a Pourbaix diagram. IOB may be completely aerobic or microaerophilic, depending on the pH, whereas MnOB require a fully aerobic environment ($DO > 5\text{mg/L}$) to precipitate Mn (Mouchet 1992).

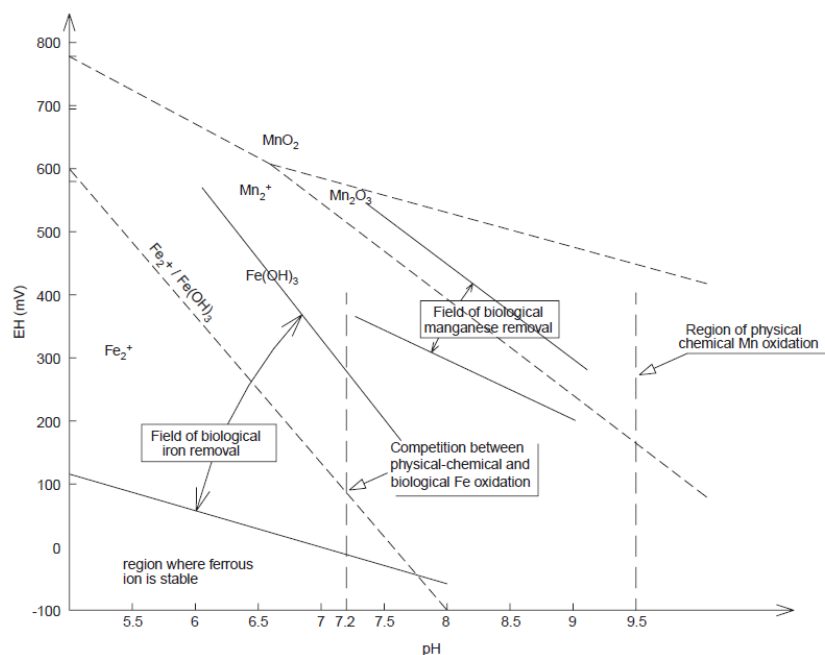


Figure 8. Field of activity of Fe and Mn oxidizing bacteria in a pH-Eh diagram (Mouchet, 1992).

2.2.2.5 Micro pollutant removal

These compounds are removed to different degrees by the commonly used biological processes. While some compounds (e.g. ibuprofen, paracetamol) are effectively removed, others (e.g. carbamazepine, diclofenac) are barely affected by the biological treatment (Onesios et al. 2009). However, when activated carbon is used as media in the biofilter adsorption could occur and

reduce some of the micro pollutants concentration especially the most hydrophobic ones.

2.3 Membrane filtration

The biofiltration step will produce/release sludge, bacteria and particles which should be removed from the water before distribution. It is well known that membrane filtration is an effective method to remove particles, micro-organisms and organic matter from drinking waters. Compared with conventional treatment methods, membrane processes (i) can provide better quality water, (ii) minimize disinfectant demand, (iii) are more compact, (iv) provide easier operational control and less maintenance, and (v) generate less sludge.

Microfiltration (MF) and Ultrafiltration (UF) are the best candidates for this step because they are low pressure or vacuum membrane filtration processes which can produce a constant permeate water quality regardless of feed water conditions because they act as an absolute barrier to suspended solids. In fact, it is more appropriate to express solids removal in terms of log removal. Most membranes are capable of up to 6- log pathogen (protozoan cysts and bacteria) removal, though this will vary depending on the membrane manufacturer. Virus rejection varies significantly between manufacturers with some systems removing virtually no appreciable amount of viruses while others report greater than 6-log removal.

During the RBF water treatment by the OBM, the UF or MF membrane will only remove particles and provide an additional hygienic barrier as described above. However, with additional chemicals the UF or MF membrane could as well provide treatment for DOC, arsenic and fluoride (see subsection 2.5).

2.4 Summary of the predictable OBM treatment for Nizamuddin RBF water.

Table 9 presents the estimated OBM treatment of the RBF water at the Nizamuddin. From this table, except fluoride, all the contaminants have the potential of being removed by the OBM. However, this study is only qualitative and, as said before, experiments should be done to estimate the removal and to design the OBM.

Table 9. Estimated treatment by the OBM of RBF water at the Nizamuddin site

	Oxidation	Biofiltration	Membrane filtration
Turbidity	None	Low	High
Particles	None	Low	High
Viruses	High	None	Low to Good depending of the pore size
Bacteria	High	Low to Good	High
Pathogens	High	None	High
DOC	Low but modification	Low to Good	None
Iron	High	Good	None
Manganese	Good	Good	None
Ammonia	Low to Good	Low to Good	None
Arsenic	Low	Low to Good	None
Fluoride	None	None	None

2.5 OBM possible optimisations

2.5.1 DOC removal

In the case that the ozonation/biofiltration is not able to reduce the DOC at required level a coagulation flocculation could be added prior to membrane filtration. Upstream coagulation can mitigate fouling layer formation on membranes and allows removal of colloidal and soluble compounds smaller than the membrane pores, thus, reducing the DOC concentration.

2.5.2 Arsenic removal

Upstream coagulation/flocculation could as well be used to remove arsenic prior to membrane filtration. The removal of arsenic through coagulation is based on the principle that As^{5+} is absorbed on the flocs formed, which are later separated by means of filtration. The kind of coagulant and the dose used affect the efficiency of the process. It has been thoroughly proved (Cheng et al. 1994; Edwards 1994; Hering et al. 1997; Scott et al. 1995.) that:

- Ferric chloride is more efficient than aluminum sulfate for arsenic removal.
- It is advisable to introduce a previous oxidizing process, since As^{5+} is more easily absorbed than As^{3+} .
- The optimum pH for coagulation-adsorption, when ferric chloride is used, is around 6.90-7.00.
- The addition of a cationic polymer allows for an improvement in arsenic removal and a reduction of the sludge generated.

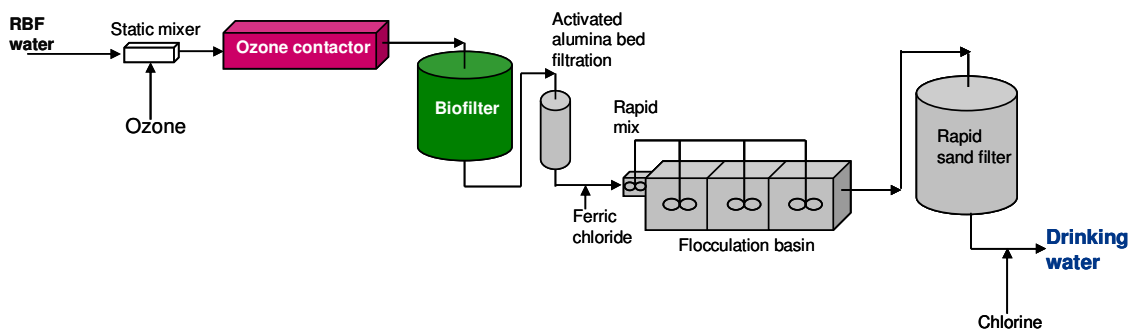
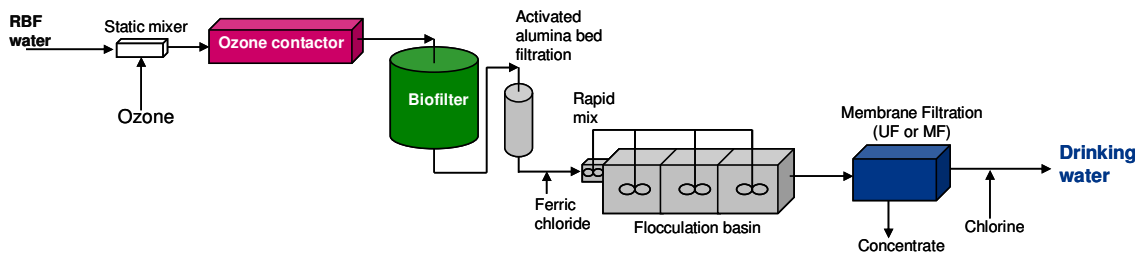
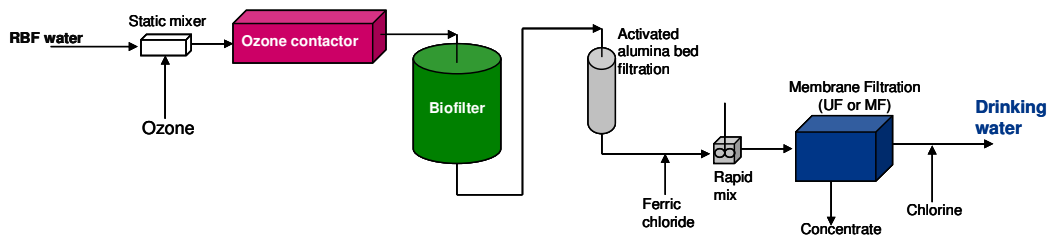
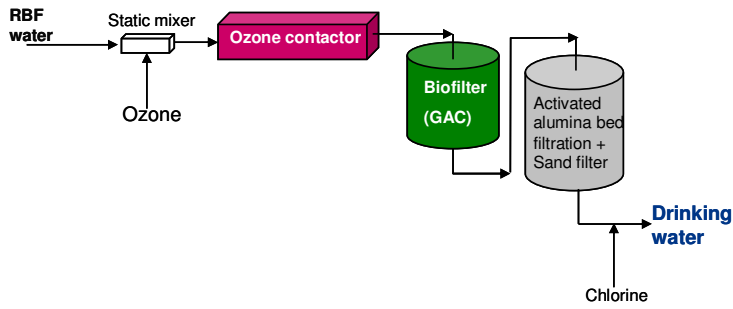
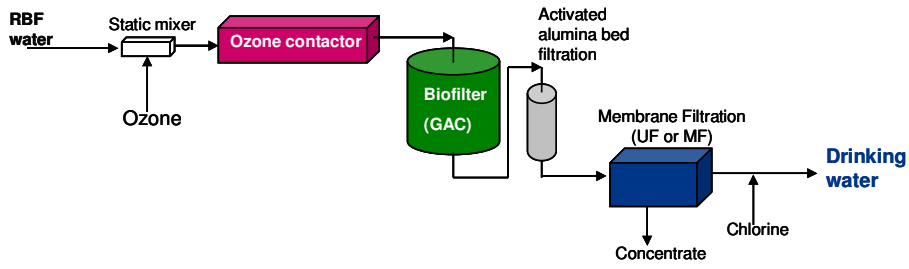
2.5.3 Fluoride removal

Fluoride could be removed by adding an activated alumina filtration. It has been reported that the process can remove fluoride up to 90% and is cost-effective (Meenakshi and Maheshwari 2006). The process is pH specific, so pH of the solution should be between 5.0 and 6.0 because at $pH > 7$, silicate and hydroxide become stronger competitor of the fluoride ions for exchange sites on activated alumina and at pH less than 5, activated alumina gets dissolved in acidic environment leading to loss of adsorbing media (Bishop and Sansoucy 1978).

Coagulation prior to membrane filtration could, as well, be used to remove fluoride. It has been also reported that when proper dosages of aluminum sulfate and sodium hydroxide were added to the reactor, the concentrations of fluoride were reduced from 4.0 mg/L in raw water to less than 1.0 mg/L in finished water, and the concentrations of residual aluminum in finished water was no more than 0.05 mg/L (Zhang et al. 2005).

2.5.4 Examples of possible design for Delhi

Taking into account the possible optimizations (subsection 2.5), different designs (figure 9) could be investigated where the OBM could be either optimized by adding a coagulation step or/and an activated alumina filtration. In addition, for economics reasons, membrane filtration could be replaced by a rapid sand filtration. Indeed, since RBF provides already a good pretreatment and ozonation assures a good barrier for pathogens, the membrane filtration could be replaced.



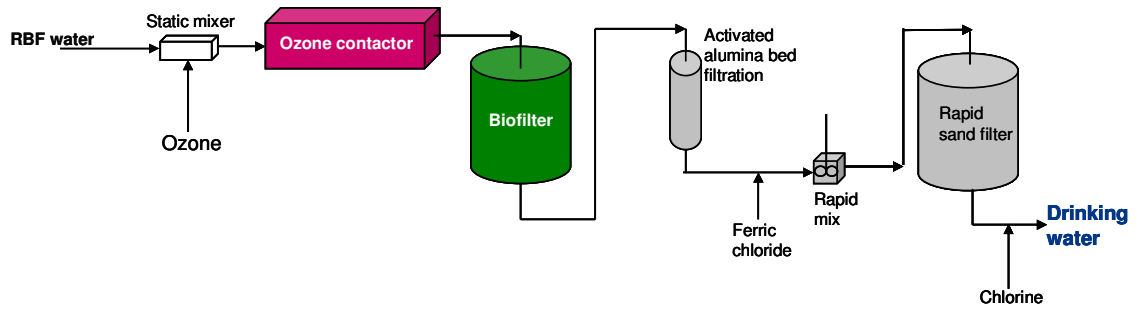


Figure 9. Examples of possible design of OBM or alternatives as post treatments for the RBF at Nizamuddin site, Delhi.

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